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Top-down estimate of methane emissions in California using a mesoscale inverse modeling technique: The South Coast Air Basin

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RESEARCH ARTICLE

10.1002/2014JD023002

Key Points:

- A mesoscale inversion ensemble is used to estimate CH₄ emissions in the LA basin
- Emission estimates are consistent with published observation-based estimates
- Oil and gas industries explain the differences between posterior and prior inventories

Supporting Information:

- Data Set S1
- Figures S1 and S2 and Table S1

Correspondence to:

J. Brioude,
jerome.brioude@noaa.gov

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Top-down estimate of methane emissions in California using a mesoscale inverse modeling technique: The South Coast Air Basin

Yu Yan Cui^{1,2}, Jerome Brioude^{1,2,3}, Stuart A. McKeen^{1,2}, Wayne M. Angevine^{1,2}, Si-Wan Kim^{1,2}, Gregory J. Frost^{1,2}, Ravan Ahmadov^{1,2}, Jeff Peischl^{1,2}, Nicolas Bousserrez⁴, Zhen Liu^{5,6}, Thomas B. Ryerson¹, Steve C. Wofsy⁷, Gregory W. Santoni⁷, Eric A. Kort⁸, Marc L. Fischer⁹, and Michael Trainer¹

¹Chemical Sciences Division, Earth System Research Laboratory, NOAA, Boulder, Colorado, USA, ²Cooperative Institute for Research in Environmental Sciences, University of Colorado Boulder, Boulder, Colorado, USA, ³Laboratoire de l'Atmosphère et des Cyclones, UMR8105, CNRS-Meteo France-Universite La Reunion, La Reunion, France, ⁴Department of Mechanical Engineering, University of Colorado Boulder, Boulder, Colorado, USA, ⁵Sandia National Laboratories, Livermore, California, USA, ⁶Now at Ramboll Environ US Corporation, Novato, California, USA, ⁷Department of Earth and Planetary Sciences, Harvard University, Cambridge, Massachusetts, USA, ⁸Department of Atmospheric, Oceanic, and Space Sciences, University of Michigan, Ann Arbor, Michigan, USA, ⁹Environmental Energy Technologies Division, Lawrence Berkeley National Laboratory, Berkeley, California, USA

Abstract Methane (CH₄) is the primary component of natural gas and has a larger global warming potential than CO₂. Recent top-down studies based on observations showed CH₄ emissions in California's South Coast Air Basin (SoCAB) were greater than those expected from population-apportioned bottom-up state inventories. In this study, we quantify CH₄ emissions with an advanced mesoscale inverse modeling system at a resolution of 8 km × 8 km, using aircraft measurements in the SoCAB during the 2010 Nexus of Air Quality and Climate Change campaign to constrain the inversion. To simulate atmospheric transport, we use the FLEXible PARTicle-Weather Research and Forecasting (FLEXPART-WRF) Lagrangian particle dispersion model driven by three configurations of the Weather Research and Forecasting (WRF) mesoscale model. We determine surface fluxes of CH₄ using a Bayesian least squares method in a four-dimensional inversion. Simulated CH₄ concentrations with the posterior emission inventory achieve much better correlations with the measurements ($R^2 = 0.7$) than using the prior inventory (U.S. Environmental Protection Agency's National Emission Inventory 2005, $R^2 = 0.5$). The emission estimates for CH₄ in the posterior, 46.3 ± 9.2 Mg CH₄/h, are consistent with published observation-based estimates. Changes in the spatial distribution of CH₄ emissions in the SoCAB between the prior and posterior inventories are discussed. Missing or underestimated emissions from dairies, the oil/gas system, and landfills in the SoCAB seem to explain the differences between the prior and posterior inventories. We estimate that dairies contributed 5.9 ± 1.7 Mg CH₄/h and the two sectors of oil and gas industries (production and downstream) and landfills together contributed 39.6 ± 8.1 Mg CH₄/h in the SoCAB.

1. Introduction

The average atmospheric mixing ratio of the greenhouse gas methane (CH₄) has increased by at least a factor of 2.5 since preindustrial times [Myhre et al., 2013]. At current global concentrations, CH₄ is the second most important greenhouse gas behind CO₂ for anthropogenic radiative forcing. The atmospheric lifetime of CH₄ of ~12.4 years is much shorter than that of CO₂, but CH₄ has a significantly higher global warming potential than CO₂ (~86 times higher over 20 years and ~34 times higher over 100 years) [Myhre et al., 2013, Table 8.7]. It has been argued that reducing emissions of CH₄ may be an important component of an initial strategy for avoiding the most severe impacts of global warming [Alvarez et al., 2012; Brandt et al., 2014].

Recently, many studies [Pétron et al., 2012; Brandt et al., 2014] found discrepancies in CH₄ inventories derived from bottom-up methods, based on an accounting of process-level activity and emission factors, and top-down approaches, which rely on atmospheric observations to infer emissions from a geographic region and often from a particular type of emissions source. Differences between bottom-up and top-down estimates suggest that the understanding of CH₄ sources is incomplete, leading to uncertainty in the application of regulations to mitigate CH₄ emissions.

Recent top-down approaches have shown that CH₄ emissions in the U.S. were underestimated in existing surface inventories [Kort *et al.*, 2008; Miller *et al.*, 2013; Brandt *et al.*, 2014], especially in the south-central U.S. and California. Greenhouse gas emissions in California have been regulated by the state legislature, which required that statewide greenhouse gas emissions be reduced to 1990 levels or lower by the year 2020. Recent studies have estimated CH₄ emissions both across California [Jeong *et al.*, 2012, 2013, 2014] and within the South Coast Air Basin (SoCAB) [Wunch *et al.*, 2009; Wennberg *et al.*, 2012; Peischl *et al.*, 2013] using either bottom-up inventory or top-down atmospheric approaches. The SoCAB region is particularly interesting because of the large population and known CH₄ sources from a densely populated urban area (Los Angeles) containing a large oil and natural gas production and distribution network, large landfills, and dairy livestock agriculture.

Bottom-up estimates of CH₄ emission in the SoCAB range from 26.2 Mg CH₄/h for NEI 2005 to 34.3 Mg CH₄/h for California Air Resources Board (CARB) 2009 (available at <http://www.arb.ca.gov/cc/inventory/data/data.htm>) and 61.6 Mg CH₄/h for Emission Database for Global Atmospheric Research version 4.2 (EDGARv4.2) (available at <http://edgar.jrc.ec.europa.eu> [Rogelj *et al.*, 2014]). One top-down technique to estimate CH₄ emissions uses observations of mixing ratios of coemitted or well-mixed species (CO and CO₂) and existing inventories of those species to infer CH₄ emissions. Wunch *et al.* [2009] estimated CH₄ emissions in the SoCAB using observed ratios of CH₄ mixing ratio with those of CO and CO₂ from a ground-based Fourier transform spectrometer and found that CH₄ emissions in this area were 45.6 to 68.4 ± 11.4 Mg CH₄/h. Wennberg *et al.* [2012] and Peischl *et al.* [2013] used measurements on board the NOAA P-3 aircraft from the California Nexus, Research at the Nexus of Air Quality and Climate Change (CalNex) campaign in 2010 [Ryerson *et al.*, 2013] to infer SoCAB CH₄ emissions, taking advantage of P-3 flight plans designed for good spatial coverage over the basin. They used observed ratios between CH₄ and CO₂ or CO mixing ratios and their emission estimates from California Air Resources Board (CARB) to derive total CH₄ emissions from the SoCAB of 50.2 ± 11.4 and 46.7 ± 4.6 Mg CH₄/yr, respectively. The accuracy of this method is limited by (i) the accuracy of the surface emission estimates of the coemitted or well-mixed species (CO₂ and CO), and (ii) differences between the spatiotemporal emission patterns of CH₄ and the reference species. Peischl *et al.* [2013] also characterized the source attribution of CH₄ using CH₄ and light alkane measurements in the SoCAB and found that natural gas pipelines, geologic seeps, and the oil and gas industries accounted for most of the underestimation in current bottom-up CH₄ inventories.

Recently, an inversion method was applied to SoCAB CH₄ emissions by Wecht *et al.* [2014], using the adjoint of the Goddard Earth Observing System-Chemical (GEOS-Chem) model at 2/3° × 1/3° resolution and prior emissions estimates from the EDGAR inventory. Wecht *et al.* [2014] estimated CH₄ emissions of 35.3 ± 9.1 and 47.9 ± 9.1 Mg CH₄/h using either GOSAT satellite or aircraft observations during the CalNex campaign, respectively. While the Wecht *et al.* [2014] CH₄ emission estimates agree with those of Wennberg *et al.* [2012] and Peischl *et al.* [2013], large uncertainties may arise from the coarse resolution of the GEOS-Chem model. The SoCAB region is associated with atmospheric mesoscale transport, such as land-sea breezes, and complex terrain, leading to impacts on the planetary boundary layer (PBL) and upslope transport, which influence pollution transport [Angevine *et al.*, 2013]. Hence, an inverse model at mesoscale should enable more confident estimates of the strength and spatial distributions of CH₄ emissions in the SoCAB region.

To tackle this problem, we use the off-line Lagrangian particle dispersion model FLEXible PARTicle-Weather Research and Forecasting (FLEXPART-WRF) coupled to three different Weather Research and Forecasting (WRF) model configurations to estimate source-receptor relationships and apply an inverse model at mesoscale. The mesoscale inverse system uses a Bayesian least squares method assuming lognormal distributions for the observations and the surface fluxes. Inverse models minimize the mismatch between measured and simulated observations of CH₄ concentrations by modifying surface flux densities from individual grid cells. The U.S. Environmental Protection Agency's (EPA's) National Emission Inventory (NEI) 2005 is used as a prior estimate of the CH₄ surface flux. Major sources of CH₄ in NEI 2005 include landfills and livestock but do not include oil and gas emissions. This inverse system has been used successfully to estimate CO, NO_y, and CO₂ in the SoCAB [Brioude *et al.*, 2013a]. Here we use the same inversion method to optimize surface emission fluxes of CH₄ with observations from six P-3 flights in May and June 2010 during the CalNex intensive field campaign. Section 2 presents the details of the methods used. The results are presented and discussed in section 3, followed by the conclusions of this work.

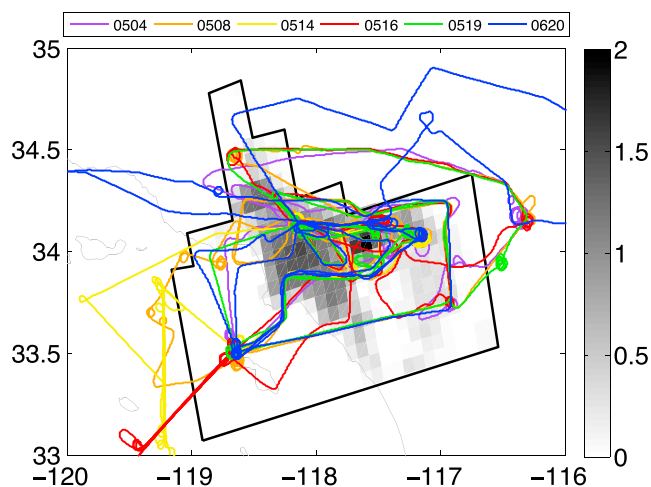


Figure 1. The prior inventory of CH₄ emissions from the U.S. EPA NEI 2005 (unit: $\mu\text{g s}^{-1} \text{m}^{-2}$) averaged at $8 \text{ km} \times 8 \text{ km}$ spatial resolution. Colored lines are flight tracks of six aircraft flights. The black line denotes the SoCAB domain.

nighttime data are not used in this study. Moreover, only observations in the boundary layer are used in the inversion, and we exclude the overwater segmentations of the 16 May flight because the models did not handle the boundary layer over the water properly [Angevine *et al.*, 2012]. No precipitation events were associated with these six flights.

On the research aircraft, CH₄ was measured once per second using wavelength-scanned cavity ring-down spectroscopy (WS-CRDS) by a Picarro 1301 m instrument [Peischl *et al.*, 2012, 2013]. The detailed information about WS-CRDS calibrations is described in section 2.1 of Peischl *et al.*, 2012. They estimated a total inaccuracy of CH₄ measurements of ± 1.2 ppbv and reported a 1 s precision of ± 1.4 ppbv during smooth flight and ± 2.0 ppbv during turbulent flight. For 8 May, we use quantum cascade laser direct absorption spectroscopy (QCLS) measurements [Kort *et al.*, 2011], with an uncertainty of ± 1 ppbv, as the Picarro instrument was not operating on this flight. The median difference between 1 s averaged QCLS and WS-CRDS was 4.5 (± 5.1) ppb shown in Figure 13 of Santoni *et al.* [2014], which correspond to an error of 0.25% for measured CH₄ using 1800 ppb background.

The average flight speed was 100 m/s, and we use 30 s averaged values, equivalent to a horizontal resolution of ~ 3 km. The P-3 measurements in the SoCAB were collected close to the surface sources, and hence, chemical loss and soil sinks of CH₄ can be neglected. The uncertainty on simulated concentrations introduced by assuming CH₄ to behave as a passive tracer is lower than other sources of uncertainty pertaining to the simulation of atmospheric transport, such as those related to the errors in modeled wind field, PBL height, linear interpolations of the data, and turbulent mixing, estimated to be 20 to 30% in Angevine *et al.* [2014]. Therefore, we treat CH₄ as a passive tracer throughout the paper, regardless of the position of the measurements relative to the sources.

2.2. The Prior Emission Inventory

The $4 \text{ km} \times 4 \text{ km}$ EPA NEI 2005 v4 [U.S. Environmental Protection Agency (EPA), 2015] is used to provide the prior emission inventory in the inverse system. CH₄ surface emissions from mobile on-road and off-road sources, nonpoint area sources, and point sources (negligibly small) are processed following EPA recommendations in the EPA SPECIATE 4.1 database [Simon *et al.*, 2010]. The $4 \text{ km} \times 4 \text{ km}$ NEI data are averaged onto the $8 \text{ km} \times 8 \text{ km}$ spatial resolution grid (Figure 1) used for the inverse modeling to relatively reduce the uncertainty with a loss of resolution. We do not include wetland and fire emissions because we consider them to be negligible contributors over the time period of our study. Emissions from oil and gas industries are very small in the NEI 2005.

It is expected that the limitations of the NEI 2005 inventory (underestimation of emissions from oil and gas industries and spatial distribution of sectors) will be compensated by the inversion method. Changes in spatial distribution in the posterior are discussed in section 3.2.

2. Methods

2.1. Observations

The NOAA P-3 aircraft measured CH₄ mixing ratios over the SoCAB during CalNex 2010 [Ryerson *et al.*, 2013]. In our inverse modeling analysis of the CalNex campaign, we use six flights (4 May, 8 May, 14 May, 16 May, 19 May, and 20 June) to estimate CH₄ emissions in the SoCAB (Figure 1). These flights were dedicated to characterizing daytime emissions and chemistry in the area, focusing on late morning and afternoon periods when a well-mixed boundary layer was established. Vertical mixing at night is very uncertain in models and measurements, and therefore,

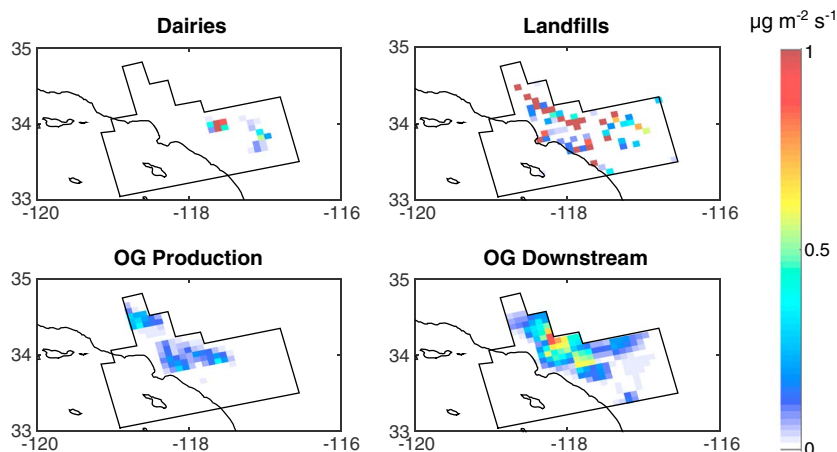


Figure 2. Methane emissions (unit: $\mu\text{g m}^{-2} \text{s}^{-1}$) in the SoCAB area (red box) from four sectors at $8 \text{ km} \times 8 \text{ km}$ spatial resolution: dairy, landfill, oil/gas production, and oil/gas downstream (storage, transmission, and distributions), with unit $\mu\text{g m}^{-2} \text{s}^{-1}$.

2.3. Spatial Distributions of Dairy, Landfills, and the Oil/Gas Industries

Dairy farms, landfills, and the oil and gas industries (production and distribution system) are known to be important sources in the SoCAB area [e.g., Peischl *et al.*, 2013]. Figure 2 shows their spatial distributions at $8 \text{ km} \times 8 \text{ km}$ resolution. Dairies and landfills are mapped using the same data as those shown in Figure 1c of Peischl *et al.* [2013]. The oil/gas production and downstream (including storage, transmission, and distribution) sectors are mapped using a hybrid inventory from Liu *et al.* (Mapping methane emissions from oil and natural gas systems in the contiguous United States, submitted to *Environmental Science and Technology*, 2015). Specifically, the spatially resolved emissions from oil and gas production are generated based on the 12 km gridded inventory of nonmethane hydrocarbons (NMHCs) for nonpoint oil and gas production sources from NEI-2011 and state-specific CH_4 -to-NMHCs emissions ratios from Eastern Research Group, Inc. [2013]. Downstream oil and gas emissions from processing, storage, transmission, and distribution were from a spatial allocation of the national total emission from the EPA Greenhouse Gas Inventory [EPA, 2014] based on geospatial data of natural gas processing facilities, interstate and intrastate pipelines, compressor stations, and population density. The spatial distribution of those sectors in Figure 2 will be used to test independently the spatial distribution in the posterior and discuss CH_4 emissions per sectors. More discussion about these sectors is found in section 3.2.2.

2.4. Modeling

2.4.1. Trajectory and Meteorology

A three-dimensional Lagrangian dispersion model (FLEXPART) [Stohl *et al.*, 2005] was used to simulate hourly back trajectories along the flight tracks. High-resolution meteorological simulations were performed using the Weather Research and Forecasting (WRF) model [Skamarock *et al.*, 2005] to drive FLEXPART [Brioude *et al.*, 2013b]. In this study, FLEXPART-WRF version 3.1 served as our mesoscale atmospheric transport model to simulate contributions to airborne CH_4 mixing ratios from surface CH_4 fluxes, which has been used to constrain species such as CO, NO_x , and CO_2 in airborne measurement-based inversions [Brioude *et al.*, 2013a].

Following Brioude *et al.* [2013a], we also consider three transport models. Uncertainties introduced by the transport models are, in part, taken into account in the inversion method and estimated in our results. The three transport models were designed using three different WRF meteorology configurations (Table 1). The first WRF configuration [Angevine *et al.*, 2012, 2013] (hereafter referred to as “WRF1”) and the second WRF-Chem 3.1 configuration [Kim *et al.*, 2011; Lee *et al.*, 2011] (hereafter called “WRF2”) have been previously evaluated and showed reasonable performance for scientific studies in the SoCAB [Angevine *et al.*, 2013]. WRF1 and WRF2 were used in Brioude *et al.* [2013a]. The third WRF configuration (“WRF3”) is based on WRF-Chem version 3.4, with a nested horizontal grid at 4 km over California and 60 vertical levels. Global Forecast System/National Centers for Environmental Prediction (GFS/NCEP) data are used for the initial and boundary conditions to the WRF3 model. The Noah land surface model with the U.S. Geological

Table 1. Names and Configurations of the Three WRF Runs Used in This Study

Name	Version	Initialization	PBL Scheme	Grid Spacing (km)	Vertical Levels	LSM, Data	Wind Field
WRF1 ^a	WRF 3.3	ERA-Interim	MYJ	4	60	Noah, UCM, MODIS	Time-averaged winds
WRF2 ^a	WRF-Chem 3.1	ERA-Interim	YSU	4	60	Noah, UCM, USGS	Sigma dot ($\dot{\sigma}$) Winds
WRF3 ^b	WRF-Chem 3.4	NCEP-GFS	YSU	4	60	Noah, USGS	Time-averaged winds

^aThe two configurations were used in *Brioude et al.* [2013a].

^bThe detailed description of WRF3.4 parameterizations can be found in http://www.mmm.ucar.edu/wrf/users/docs/user_guide_V3/contents.html.

Survey (USGS) land cover-land use and the Yonsei University (YSU) boundary layer scheme are used. WRF2 only provides the sigma dot (the derivative of WRF vertical coordinate with respect to time) instantaneous wind field while the other two provide time-averaged wind fields. We estimate correlations between the three WRF configurations and observations (Table S1 in the supporting information) of the entire CH₄ time series for the six flights analyzed during CalNex. The correlation coefficients between the three FLEXPART-WRF CH₄ time series are not much larger than the correlation coefficients between the measured CH₄ time series and simulated time series of each FLEXPART-WRF model. Therefore, each model can be considered as independent of the others in the sense that there is no correlation bias.

The Hanna scheme [*Stohl et al.*, 2005] is used in FLEXPART-WRF to prescribe a turbulent profile including both horizontal and vertical turbulences in the PBL. Time-averaged winds generated by the WRF model are used instead of instantaneous wind to reduce uncertainties in the Lagrangian trajectories in complex terrain [*Brioude et al.*, 2012]. PBL height, surface sensible heat flux, and friction velocity were taken directly from the WRF simulation. A total of 10,000 computational particles are released either every 30 s along the aircraft flight tracks or every 100 m during vertical profiles. No convection, chemical transformation, and deposition were simulated. FLEXPART simulates the trajectories over 24 h to focus on the local transport within the study area (i.e., the SoCAB). The trajectories are also limited to 24 h for computational feasibilities. Hence, the influence of previous-day transport and emissions is ignored, which will increase the uncertainty in the flux estimates. *Brioude et al.* [2013a] estimated that restricting the trajectories to 24 h added an uncertainty of ~7% to the surface emission estimates.

The FLEXPART-WRF output has a horizontal resolution of 8 km × 8 km, with 1 h output time interval. The output, also referred to footprint (in $\text{s m}^3 \text{kg}^{-1}$), consists of a residence time within a surface layer (in meter) weighted by the atmospheric density, which represents the sensitivity of the airborne measurements to surface emissions. When combined with a surface emission inventory (in $\text{kg m}^{-2} \text{s}^{-1}$), one can calculate a mixing ratio for each set of trajectories along the aircraft flight track. In this way, FLEXPART-WRF linearizes the transport processes between the surface and the aircraft. This approach alleviates the need for an adjoint model of WRF-Chem (not currently available) in the application of the inversion technique [*Brioude et al.*, 2013a; *de Foy et al.*, 2015].

2.4.2. Inverse Modeling

We apply the inverse modeling approach described in *Brioude et al.* [2011, 2013a] to the CH₄ data collected in the SoCAB during CalNex. The observations used in the inverse modeling are the CH₄ mixing ratio enhancements above background. We define the background CH₄ for each flight as the lowest mixing ratio found in the atmospheric boundary layer below 2 km upwind of the SoCAB, which is also close to the value of the mode in the distribution of all mixing ratios measured below 2 km during the campaign. The enhancements in CH₄ are ~10% of background concentrations on average. The background concentration estimates were associated with an uncertainty of 10 ppb due to the variability in the low concentrations found in the PBL for each flight. This uncertainty was added to the measurement uncertainties in the inversion. Negative deviations from the background are removed from the observation data set and a small value of 10⁻⁵ ppb was used instead.

To calculate the best estimates of CH₄ surface flux emissions, we use an iterative method to minimize a lognormal Bayesian least squares cost function with solutions in a median of the distribution, similar to the one used in *Brioude et al.* [2011]:

$$J = \frac{1}{2} (\ln(y_0) - \ln(Hx))^T R^{-1} (\ln(y_0) - \ln(Hx)) + \frac{1}{2} \alpha (\ln(x) - \ln(x_b))^T B^{-1} (\ln(x) - \ln(x_b)) \quad (1)$$

where y_0 is the observation enhancement above the background value owing to surface emissions with the $p \times 1$ dimension and p is the number of observations. Parameter x is a $m \times 1$ vector of the posterior solutions and m is the number of gridded fluxes in space and time. H are the FLEXPART-WRF outputs, which construct a linear linkage between surface emissions and mixing ratios at given receptors, and the dimension of H is $p \times m$. Parameter x_b is a $m \times 1$ vector of prior fluxes, and R (dimension: $p \times p$) and B (dimension: $m \times m$) are error covariance matrices of observations and prior fluxes in the lognormal distribution space. Parameter α as a regularization parameter that comes from the L-shape criterion [Henze *et al.*, 2009], which is used to balance the errors in both covariance matrices to obtain the smallest sensitivity to the error in either the observations or the prior.

The advantage of using a lognormal transformation in the cost function is that no negative fluxes are found in the posterior so that we obtain a minimum variance solution (equivalent to the mean of the distribution) and not a likelihood, and the lognormal distribution follows more closely the observed CH₄ mixing ratios (after subtracting the background value). This approach avoids the drawbacks of other techniques used to prevent negative fluxes, such as modifying covariance matrices or bounded inverse methods [e.g., Stohl *et al.*, 2009; Miller *et al.*, 2014].

The observation errors, which include uncertainties from the measurements and from the background definition for each flight, typically, are assumed to be uncorrelated. Likewise, we assume that errors in the prior surface inventory are uncorrelated. Hence, the error covariance matrices of the observations (R) and prior (B) are modeled as diagonal matrices.

We use four time intervals to construct a four-dimensional (three spatial dimensions plus time) inversion following Brioude *et al.* [2013a]: (i) morning rush hours between 13:00 UTC and 16:00 UTC (06:00–09:00 local time), (ii) midday hours between 17:00 UTC and 21:00 UTC (10:00–14:00 local time), (iii) evening rush hours between 22:00 UTC and 01:00 UTC (15:00–18:00 local time), and (iv) nighttime hours between 02:00 UTC and 12:00 UTC (19:00–5:00 local time). The four time intervals are applied to both trajectories and prior emissions. The values reported in section 3 are the averages for the two time intervals between 17:00 UTC and 01:00 UTC (10:00–18:00 local time) when we have the strongest confidence in the transport models and therefore in the inversion.

Over the SoCAB domain, the Jacobian matrix H , representing the transport model in the inversion, has a dimension of 768 to 1109 rows (the total number of 30 s averaged observations in each individual flight) by 23280 columns (the total number of surface grid cells with four time intervals). To reduce the dimension of the matrix in order to obtain inversion solutions efficiently and to reduce cross correlations between surface fluxes in the posterior, we cluster surface grid cells in the domain using an optimality criterion based on the Fisher information matrix, which is different than Brioude *et al.* [2013a] that limited the inversion on grid cells with significant emission in the prior. Here we use a similar criterion as Bocquet *et al.* [2011] to construct adaptive grids in the control space:

$$\mathcal{J} = \text{Tr}(BW^T R^{-1}W) \quad (2)$$

where \mathcal{J} is the criterion and W is the derivative of the Jacobian matrix H assuming a lognormal distribution. We classify the surface grid cells into clusters according to the neighbor method: namely, neighboring grid cells having similar values are aggregated as a cluster. In addition, grid cells with high values (e.g., equal to and above -5) of \mathcal{J} are not aggregated. After applying this criterion, the domain in Figure 3, showing the control space of the inversion, uses 1201 clusters for each time interval, reducing the total grid cells in the inversion by a factor of ~ 5 (from 23,280) while mitigating information loss on surface fluxes.

For each flight and transport model, we used 100 iterations of the inversions with a random term (equivalent to a standard deviation of 5% of a lognormal distribution of prior flux estimates) to minimize cross correlation in the prior and hence in the posterior. The uncertainty in the posterior emission estimate from the ensemble of single-flight-based inversions is then estimated by using the posterior covariance matrix to estimate the posterior uncertainty for each realization and as the standard deviation of the distribution represented by the 1800 realizations from the ensemble of six flights and three transport models.

The uncertainty from the FLEXPART Lagrangian model cannot be assessed fully with this approach, but we assume that this uncertainty is small compared to the uncertainties in the meteorological fields [Hegarty *et al.*, 2013]. Angevine *et al.* [2014] evaluated the uncertainty in atmospheric transport to be 20 to 40% in a

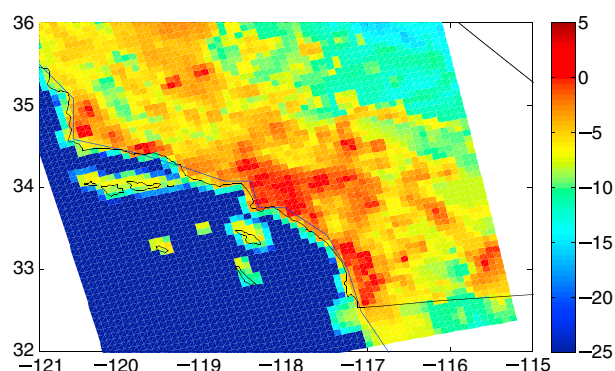


Figure 3. A map (log scale) of the Fisher information matrix criterion in the SoCAB based on the WRF1 transport model at a spatial resolution of $8 \text{ km} \times 8 \text{ km}$. See the text in section 2.4.2 for the details. Grid cells are aggregated into clusters when values of the criterion in adjacent cells are below -5 . Cells with values equal to and above -5 are not aggregated. After aggregating, the original 5820 grid cells are aggregated into 1201, 1199, and 1200 clusters for the WRF1, WRF2, and WRF3 transport models, respectively.

The spatial distributions of footprints have similar patterns and emphasize the impact of northerly and northwesterly background airflow on the SoCAB domain. The six flights sampled predominantly downwind of the surface emissions in the SoCAB area.

The total posterior emissions for the SoCAB domain are calculated for each flight and for each transport model over the SoCAB area (Table 2 and Figure S1 in the supporting information). According to a recent report [United States Energy Information Administration, 2015], the seasonality of natural gas consumption in California had an amplitude of 15% in 2010. As seasonal variation in natural gas distribution leak is small [McKain et al., 2015], one can assume a seasonality of 15% from NG leaks in Los Angeles. Peischl et al. [2013] found that 47% of total CH_4 emission in SoCAB was due to NG leaks. As the seasonality from Landfills and Dairies in the Los Angeles basin should be small, one can expect a seasonality of $\sim 10\%$ in the CH_4 emission.

Diurnal and day to day variations in methane emission have been observed from feedlots [Gao et al., 2011] and landfills. One can expect smaller emission variation from oil and gas facilities as their industrial activities have no particular diurnal variation. Finally, mobile sources, which have a strong diurnal profile, contribute only 1% of the total methane emission and should have a small impact. The six flights used in this study are not enough to fully cover the diurnal and day to day emission variations from landfills and feedlots, sources that contribute 44% to the total methane emissions in Los Angeles (LA) basin [Peischl et al., 2013]. Hence, our posterior emission uncertainty estimates should be considered as a lower limit.

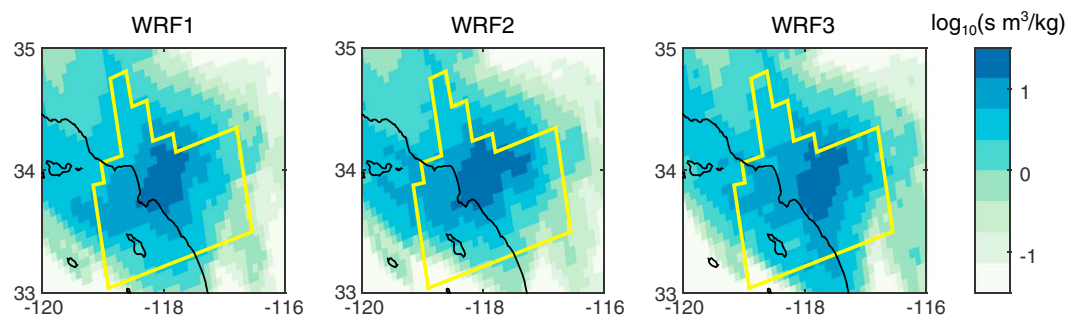


Figure 4. Averaged 1 day backward trajectory footprints with log scale for the six aircraft flights, based on each of the three WRF configurations, respectively. The yellow line shows the SoCAB domain. Footprints related to the overwater segments of the 16 May flight are excluded here.

FLEXPART-WRF ensemble of six WRF configurations due to uncertainties in meteorological fields. It is still unclear how this uncertainty propagates in the inversion system. However, our ensemble of only three WRF configurations might underestimate the uncertainty introduced by the FLEXPART-WRF simulated transport in the posterior estimates.

3. Results and Discussion

3.1. Posterior-Prior Differences in Emission Strengths

3.1.1. Total Emissions of CH_4

Back-trajectory footprints of the six flights from the three transport models are shown in Figure 4. The footprints provide a sense of the ability of the aircraft observations to constrain surface sources across the SoCAB.

Table 2. Total Emissions of CH₄ (Mg/h) Over SoCAB Derived From Each of the P-3 Flights With the Ensemble of Three Different Transport Models and an Averaged Estimate of CH₄ Emissions Based on Six Flights and Three Transport Models

Fight	4 May	8 May	14 May	16 May	19 May	20 June	Mean
Total emissions (Mg CH ₄ /h)	46.6 ± 10.6	46.9 ± 5.5	47.9 ± 10.1	47.8 ± 6.4	48.6 ± 11.2	39.9 ± 7.1	46.3 ± 9.2

The posterior values for the three transport models range from 45.1 to 48.2 Mg CH₄/h. The mean values over the six flights for each transport model are also shown in Figure S1. The three transport models' posterior estimates have a mean value 46.3 Mg CH₄/h with uncertainties of 21%, 19%, and 19%, respectively. Combining the three transport models, we estimate an optimized value of 46.3 ± 9.2 Mg CH₄/h for the SoCAB area (Figure 5 and Table 2). The uncertainty here includes the posteriors' uncertainty and the transport models' uncertainty.

Our estimate is higher by factor of ~1.8 than NEI 2005 (26.2 Mg CH₄/h). Our estimate is in agreement with previous estimates derived from observations taken over the same time period (Figure 5; 50.2 ± 11.4 Mg CH₄/h [Wennberg *et al.*, 2012], 46.7 ± 4.6 Mg CH₄/h [Peischl *et al.*, 2013], 31.9–44.5 Mg CH₄/h (Santoni *et al.*, California's methane budget derived from CalNex P-3 Aircraft Observations and a Lagrangian transport model, submitted to *Journal of Geophysical Research*, 2014), and 47.9 ± 9.1 Mg CH₄/h [Wecht *et al.*, 2014]).

The precision and the accuracy of the surface flux estimates based on a single flight can also be evaluated. The total posterior emission estimates vary by ~6% between the six flights shown in Table 2. However, as mentioned above, the variation in posterior emission across the meteorological models is closer to 20% [Angevine *et al.*, 2014]. Hence, a given day surface emission estimates in the SoCAB can likely be estimated to within ~20% from a single flight's data and a single meteorological model for this summer period.

3.1.2. Time Series of CH₄ Mixing Ratio

Figure 6 presents the comparisons between observed and simulated time series of CH₄ mixing ratio enhancements over background using the prior and posterior estimates for each flight. The simulated time series using the posterior estimates optimized by the inversion shows better agreement with observations than the simulated time series using the NEI 2005 prior inventory. We calculate the mean bias between observations with the simulations using prior estimates and those using the posterior inventory resulting from the inversion. Mean biases with the posterior estimates distinctly decrease in these studies by 50% to 85% compared with prior estimates. We exclude the overwater segments of the 16 May flight because models did not handle the boundary layer over the water properly [Angevine *et al.*, 2012]. The correlations and the correlation coefficients, R^2 , between the observations and the simulated time series are shown in Figure 7 for each flight and each transport model. The slope of the correlations is closer to 1, and R^2 is higher using the posterior estimates compared with the prior estimates for all flights.

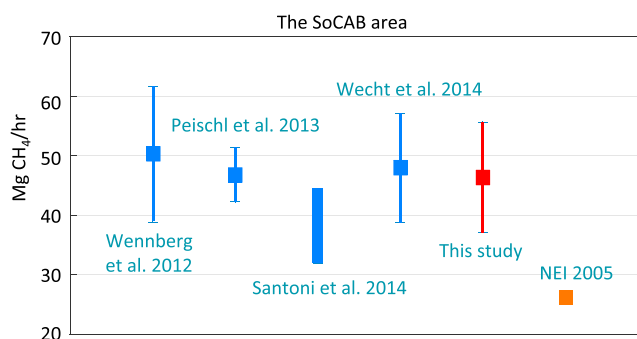


Figure 5. Previous estimates of total emissions of CH₄ in the SoCAB during the CalNex campaign (blue) compared with the estimates from this study. The error bars (1 sigma) present the uncertainties of CH₄ annual emission estimates. The prior estimate of total emissions of CH₄ in the SoCAB from NEI 2005 is shown as well (orange).

3.2. Spatial Distribution of Posterior-Prior Differences

3.2.1. CH₄ and CO Ratios

We use the inversion system and the same six flights as for CH₄ to calculate the posterior inventory of CO for the SoCAB using the updated FLEXPART-WRF version. Our results (~65 kg/s in the SoCAB) are consistent with the CO emissions estimates in Brioude *et al.* [2013a] (68 kg/s). This negligible difference comes from the fact that (i) we clustered grid cells that have small influence on the inversion, and hence, all the grid cells

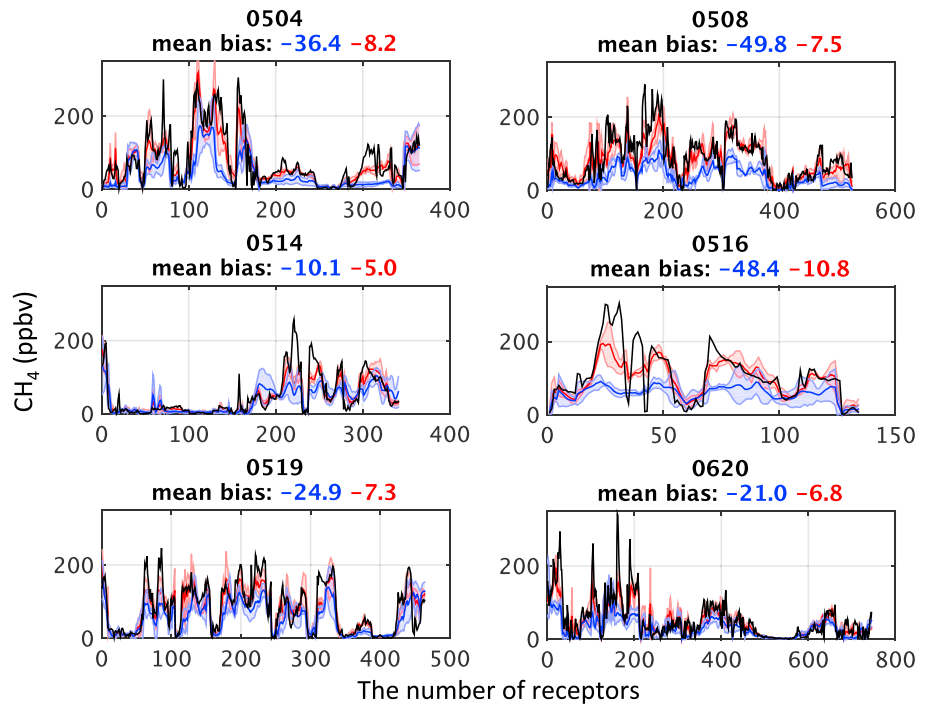


Figure 6. The comparison of observations and models for the time series of the CH₄ mixing ratio for NOAA P-3 flights on 4, 8, 14, 16, and 19 May and 20 June 2010. The plot includes the observations (black line), estimates from FLEXPART based on the prior inventory (blue line), and inversion estimates based on the posterior inventory (red line). The shaded areas indicate 1σ variability for the three WRF configurations. Mean biases were calculated between observations and simulations using the prior inventory (blue text) and between the observations and simulations using the posterior (red text). For 16 May flight, we exclude the flight overwater segments because the models did not handle the boundary layer over the water properly.

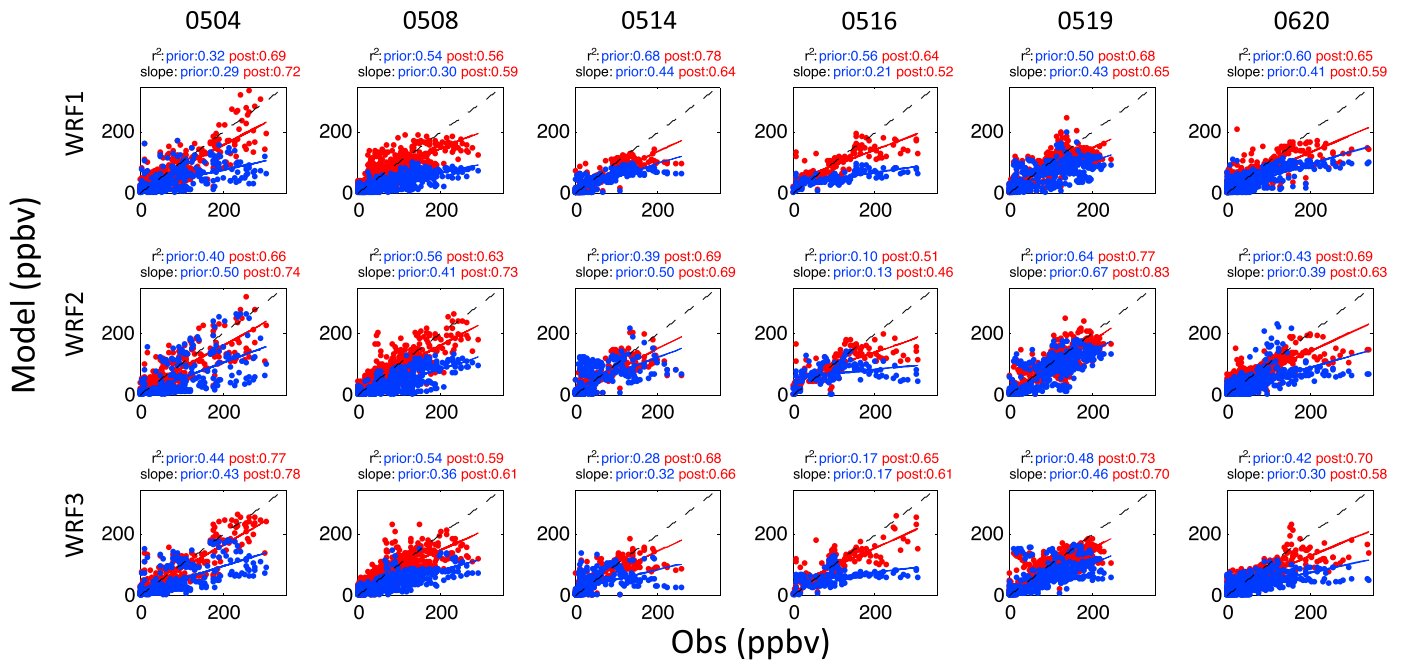


Figure 7. Correlations of the simulated and observed above background CH₄ mixing ratio for each flight using the three transport models. The lines indicate the least squares fits to the correlations between observations and simulations with either the posterior inventory (red) or prior inventory (blue). All correlations in the column are significant with $P < 0.05$.

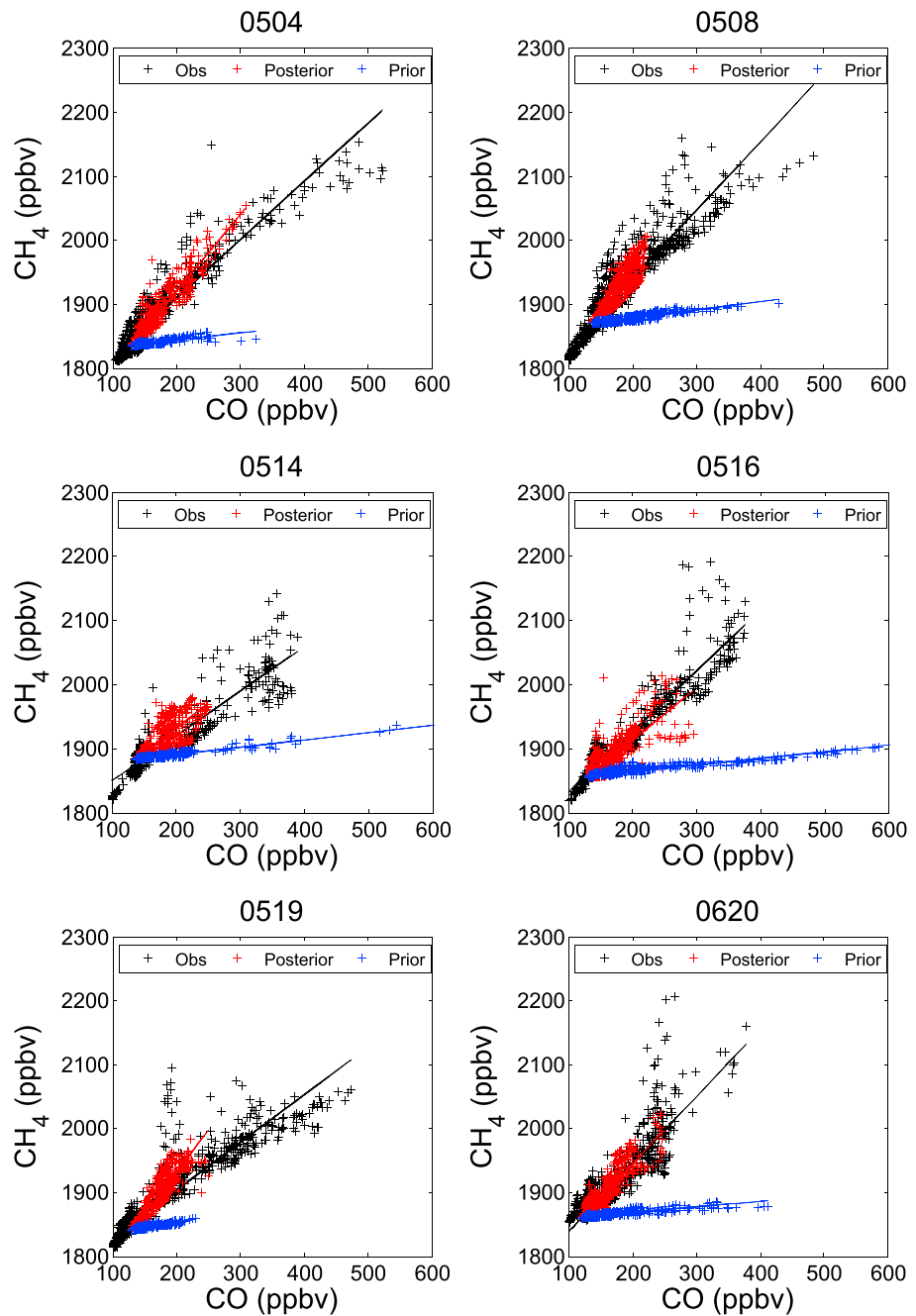


Figure 8. Scatterplot of CH₄ versus CO mixing ratios for measurements (black) and simulations, using prior (blue) and posterior (red) emissions for the flights on 4, 8, 14, 16, and 19 May and 20 June. Simulations include results from all data in three WRF configurations. The black dots represent observations of CH₄ and CO mixing ratios, the blue dots show estimates from FLEXPART-WRF based on the NEI 2005 prior inventories of CO and CH₄, and the red dots show FLEXPART-WRF data based on the posteriors from the inversion in this study. The solid lines represent the least squares fit to each set of data, and the dash lines represent 1 standard deviation differences in the slope of the fits.

within the domain are used, while Brioude *et al.* [2013a] limited the inversion on grid cells with significant emission in the prior, (ii) FLEXPART-WRF version has changed with notably a bug fix on the sensible heat flux. The ratio between CH₄ and CO, equivalent to the slope of the correlation between the atmospheric mixing ratios of these species, is used to evaluate changes in the spatial distributions of CH₄ relative to those of CO between the prior and posterior. Figure 8 presents CH₄ and CO correlations and their least

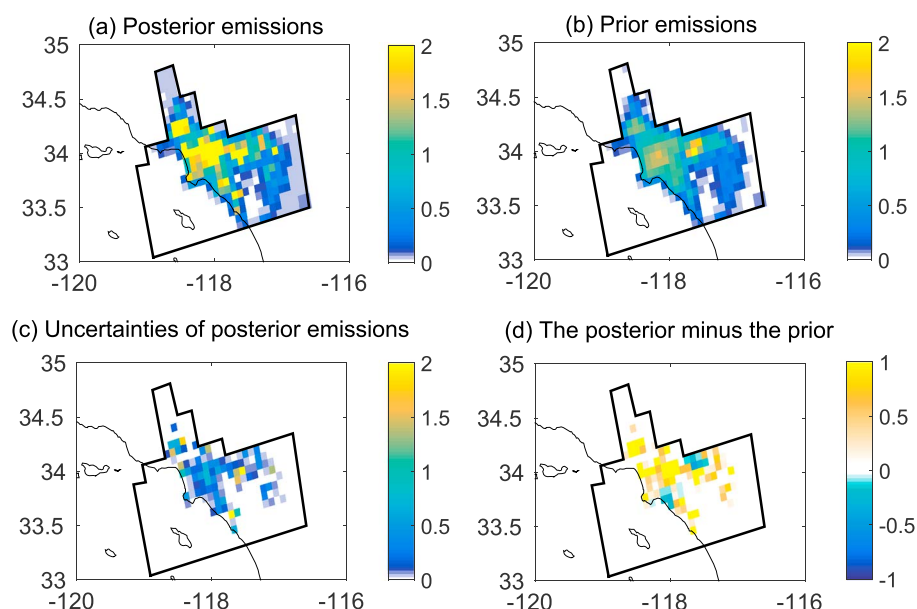


Figure 9. CH₄ surface fluxes (unit: $\mu\text{g s}^{-1} \text{m}^{-2}$) from (a) the posterior inventory of our inversion, (b) the NEI 2005 prior inventory, (c) the uncertainties in the posterior inventory, and (d) the difference between the posterior and the prior inventory.

squares fits from the six flights for the observations and simulated by the transport models using the prior and posterior estimates of CH₄ and CO emissions. The slopes between CH₄ and CO are systematically in better agreement with observations using the posterior surface inventory compared with simulations using the prior surface inventory. Slopes between CH₄ and CO from the observations and posterior estimates are in very good agreement.

Figure S2 shows the spatial distribution of surface emission ratio between CH₄ and CO in the prior and posterior inventories. The prior and posterior inventories of CH₄ are shown in Figure 9 and discussed further in section 3.2.2. As shown in Figure 8, CH₄-CO ratios are underestimated in the prior inventory. The CH₄-CO ratio in the prior inventory (Figure S2) is rather homogeneously distributed in the SoCAB, probably due to the missing oil and gas industry sector. In the posterior inventory, the CH₄-CO ratio is heterogeneously distributed, with large ratios where dairies, landfills, and the oil/gas sources are located. The details of these source types are discussed below.

3.2.2. Changes in Spatial Distribution

The spatial distribution of surface fluxes in the posterior and prior inventories and their differences shown in Figure 9 can be used to help quantify emissions from different source sectors. The posterior inventory (Figure 9a) is based on the averaged posterior inventories from each flight and the three transport models. In the SoCAB area, the prior inventory from NEI 2005 (Figure 9b) shows high emissions of CH₄ over Los Angeles County, Orange County, and the Chino area (34°1'N, 117°41'W) where most dairies are located. The uncertainties of the posterior inventory are shown in Figure 9c. There are mostly low values of uncertainties for many grid cells, but relatively high values in the Chino area and the coastal urban area. The posterior inventory presents the same emission patterns as the prior inventory, but with a particularly large increase in CH₄ emissions in parts of the basin dominated by the oil/gas sources and dairies (Figure 9d and Figure 2).

As shown in Figure 2, CH₄ emissions from the dairy sector predominate in the eastern part of the SoCAB and CH₄ emissions of oil/gas sources predominate in the western part of the basin along the coastal urban area. We cannot differentiate the overlapping grids between oil/gas production and distribution sources and landfills. However, the large increase in CH₄ emissions in the western part of the basin (Figure 9d) of CH₄ emissions in the posterior compared to the prior is located in a region where oil/gas production and distribution dominates. This difference is consistent with Peischl *et al.* [2013] who pointed out that those emissions from natural gas distribution and geological seeps could account for the differences between

the bottom-up and top-down results. We estimate the total emissions from oil/gas production, distribution, and landfill sources contribute 39.6 ± 8.1 Mg CH₄/h in the SoCAB, a factor of 1.8 higher than the value in the prior inventory (22.5 Mg CH₄/h), but consistent with estimates (39.0 Mg CH₄/h) in *Peischl et al.* [2013].

The number of overlapping grids between the dairy sources and the other three source types is negligible (Figure 2), which allows us to estimate the total emissions from the dairy sector. We estimate that dairies contribute 5.9 ± 1.7 Mg CH₄/h in the SoCAB, twice the value in the prior inventory (3.1 Mg CH₄/h). This value is higher than the bottom-up estimate from *Peischl et al.* [2013] of 3.6 Mg CH₄/h, but consistent with their top-down estimates of 5.6 ± 2.9 Mg CH₄/h using a mass balance approach on three P-3 transects downwind of the Chino area during CalNex.

4. Summary and Conclusions

We present the first top-down estimates of CH₄ emissions in the South Coast Air Basin (SoCAB) based on an inverse modeling approach at mesoscale. Annual CH₄ emissions are constrained using in situ measurements from the NOAA P-3 aircraft during the 2010 CalNex campaign over the SoCAB. FLEXPART Lagrangian dispersion model simulated atmospheric transports when coupling with three configurations of WRF runs, the equivalent of three mesoscale transport models. Unlike *Brioude et al.* [2013a], aggregating surface grid cells based on a clustering method is applied to exploit the information content of the inversion to reduce spatial cross correlation between posterior fluxes while allowing computation of the optimal fluxes at a reasonable cost. Simulated CH₄ mixing ratios using the derived CH₄ emissions are in better agreement and better correlation with the measurements ($R^2=0.7$) than using the prior inventory ($R^2=0.5$). The derived CH₄ emission estimates are 46.3 ± 9.2 Mg CH₄/h, in agreement with other recent published observation-based studies.

The inversions are applied to data from individual flights and using the output of the three independent transport models, so our estimated uncertainty includes both the posterior uncertainty and part of the uncertainty from the transport models. However, our uncertainty estimates should be considered as a lower limit. Additionally, we estimate that the total emissions for a given day can be evaluated with data from a single flight with an uncertainty of ~20%. The six flights used in this study are not enough to fully cover the diurnal and day to day emission variations from landfills and feedlots, sources that contribute 44% to the total methane emissions in LA basin [*Peischl et al.*, 2013]. Hence, our posterior emission uncertainty estimates should be considered as a lower limit.

We apply the same inversion with CO measurements to estimate CO surface emissions and analyze CH₄ to CO atmospheric enhancement ratios and surface inventories. The simulated CH₄-CO slopes using the posterior are in good agreement with the observed slopes. The spatial distribution of the CH₄-CO ratios in the posterior inventory is heterogeneous, with higher slopes found in the areas of the basin where most of the oil/gas, landfill, and dairy sources are located. We estimate that dairies in the Chino area contributed 5.9 ± 1.7 Mg CH₄/h, twice the value in the prior inventory (NEI 2005, 3.1 Mg CH₄/h), and consistent with top-down values from *Peischl et al.* [2013] based on a mass balance approach. The oil/gas production and distribution system and landfill sectors together contribute 39.6 ± 8.1 Mg CH₄/h in the SoCAB, a factor of 1.8 higher than the prior inventory, consistent with results in *Peischl et al.* [2013].

We have shown that the mesoscale inverse technique is successful in improving bottom-up CH₄ inventories in terms of flux and spatial distribution in a region with complex terrain like the SoCAB area. Our study supports the view that emissions of CH₄ in the SoCAB area are greater than expected from population-apportioned bottom-up state inventories and missing or underestimated emissions from dairies, the oil/gas system, and landfills in the SoCAB. As shown in this study, airborne measurements used in a mesoscale inversion could be used to monitor greenhouse gas emissions in a region like the SoCAB. The same technique will be applied to CalNex aircraft measurements over the Central Valley to further improve our knowledge of the statewide emissions of CH₄ and the contributions from different source types.

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FLEXPART-WRF model is available at the official FLEXPART website (<http://flexpart.eu>). NOAA P-3 observation data are available and can be downloaded at <http://www.esrl.noaa.gov/csd/projects/calnex>. The posterior emission inventory is available online as supporting information in NetCDF format. The lognormal Bayesian inverse software was developed at CSD/NOAA and CIRES. The WRF initial/boundary data were provided by NOAA/NCEP and NCAR. U.S. EPA NEI 2005 provided the prior emission data. We thank the High Performance Computing Program for their support in running FLEXPART-WRF. We thank Sandia National Laboratories who supported NG production and distribution data. The lead author would like to thank in particular the generous support of a National Research Council Research Associateship Award. This work was supported in part by NOAA's Atmospheric Chemistry, Carbon Cycle, and Climate Program. M.L.F. acknowledges support from the California Energy Commission Public Interest Environmental Research Program to LBNL under contract no. DE-AC02-05CH11231. Z.L. was supported under the Laboratory Directed Research and Development program at Sandia National Laboratories. Sandia is a multiprogram laboratory operated by Sandia Corporation, a Lockheed Martin Company, for the United States Department of Energy's National Nuclear Security Administration under contract DEAC04-94AL85000. Our thanks are given to the three anonymous reviewers who helped to substantially improve the quality of this paper.

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