

Assessment of Methane and VOC Emissions from Select Upstream Oil and Gas Production Operations Using Remote Measurements, Interim Report on Recent Survey Studies

Control # 2012-A-21-AWMA

Eben D. Thoma and Bill C. Squier

U.S. EPA, Office of Research and Development, National Risk Management Research Laboratory, 109 TW Alexander Drive, E343-02, Research Triangle Park, NC 27711, USA

David Olson

U.S. EPA, Office of Research and Development, National Exposure Research Laboratory, 109 TW Alexander Drive, E205-03, Research Triangle Park, NC 27711, USA

Adam P. Eisele

U.S. EPA Region 8, 1595 Wynkoop St. 8P-AR, Denver, CO 80202, USA

Jason M. DeWees and Robin R. Segall

U.S. EPA, Office of Air Quality Planning and Standards, 109 TW Alexander Drive, E143-02, Research Triangle Park, NC 27711, USA,

M. Shahrooz Amin and Mark T. Modrak

ARCADIS Inc., 4915 Prospectus Drive, Suite F, Research Triangle Park, NC 27713, USA

ABSTRACT

Environmentally responsible development of oil and gas assets in the United States is facilitated by advancement of sector-specific air pollution emission measurement and modeling tools. Emissions from upstream oil and gas production are complex in nature due to the variety of equipment designs, differences in maintenance states, and variable product composition. Since component-level emission measurements require site access and are somewhat burdensome, cost-effective approaches to locate and assess emissions using off-site observations are attractive from both a source understanding and routine inspection perspective. A new mobile remote assessment approach was developed, tested and is described herein. The approach was utilized on five upstream natural gas field studies in CO, TX and WY in 2010 and 2011. Preliminary results show median CH₄ emission rates of 0.21 g/s, 0.43 g/s and 0.79 g/s and volatile organic compound emission rates of 0.16 g/s, 0.04 g/s and 0.30 g/s for areas studied in CO, TX, and WY respectively. The distributions were positive skew (mean > 2*median) with the presence of high values in part ascribed to maintenance-related issues such as open thief hatches and failed pressure relief valves that can be mitigated. The difference in volatile organic compound emissions in select areas of TX compared to CO and WY is primarily due to the dry gas nature of the former. A review of acquired summa canister results substantiates this point. The positive and negative attributes and use limitations of the new mobile remote assessment approach are described and next steps in method development are discussed.

INTRODUCTION

Improved understanding of the amount and type of air pollution emitted during oil and gas production operations is important for several reasons. With steady increases in production activity in many areas of the United States, the potential impact of the emitted volatile organic

compounds (VOCs) on regional ozone must be sufficiently assessed.¹⁻³ In addition, a better understanding and local air quality impacts including organic hazardous air pollutant (HAP) emissions is important because oil and gas production operations can exist in close proximity to populations.⁴ Finally, it is important to improve knowledge of greenhouse gas (GHG) emissions from this sector to support updates of national GHG emission inventories.⁵

To inform emission and exposure estimates, model development, and mitigation options for this and related sectors, the United States Environmental Protection Agency (EPA) is developing and applying new measurement methods for both on-site leak quantification and off-site remote assessment of emissions. This interim report discusses progress on these efforts by presenting results from emission survey campaigns in CO, TX, and WY conducted in 2010 and 2011. This paper describes a new remote assessment approach and presents methane (CH₄) and VOC emissions and near source concentration data from over 200 sites. These off-site results are compared with on-site measurement data from several studies. The presentation will include infrared camera footage of emission points, computational fluid dynamic visualization of the remote measurement, and a description of a geospatial database currently under development.

BACKGROUND

In-field oil and gas production units (well pads) separate extracted product into raw natural gas, oil/condensate, and produced water. The natural gas is put into field gathering pipelines for transport to a local gas processing plant for further refinement. The condensate and waste water are stored in tanks at the production site for later truck transport. The composition of the raw product is field-dependent and can range from >95% CH₄ with little condensate (called dry gas) to < 85% methane with significant produced condensate (wet gas). Well pad emission sources can be vented or fugitive (leaks) in origin and since the product streams change with progressive levels of processing, air emissions from different points in the process can differ in composition. Emission profiles can also change over time as the well ages. Due to this variability and differences in production equipment designs and maintenance, there exists considerable uncertainty in emissions. Several approaches have been used for on-site, direct measurement of emissions, but routine application of these are complicated by compositional differences, encountered maintenance states, and site access requirements.⁶⁻⁹

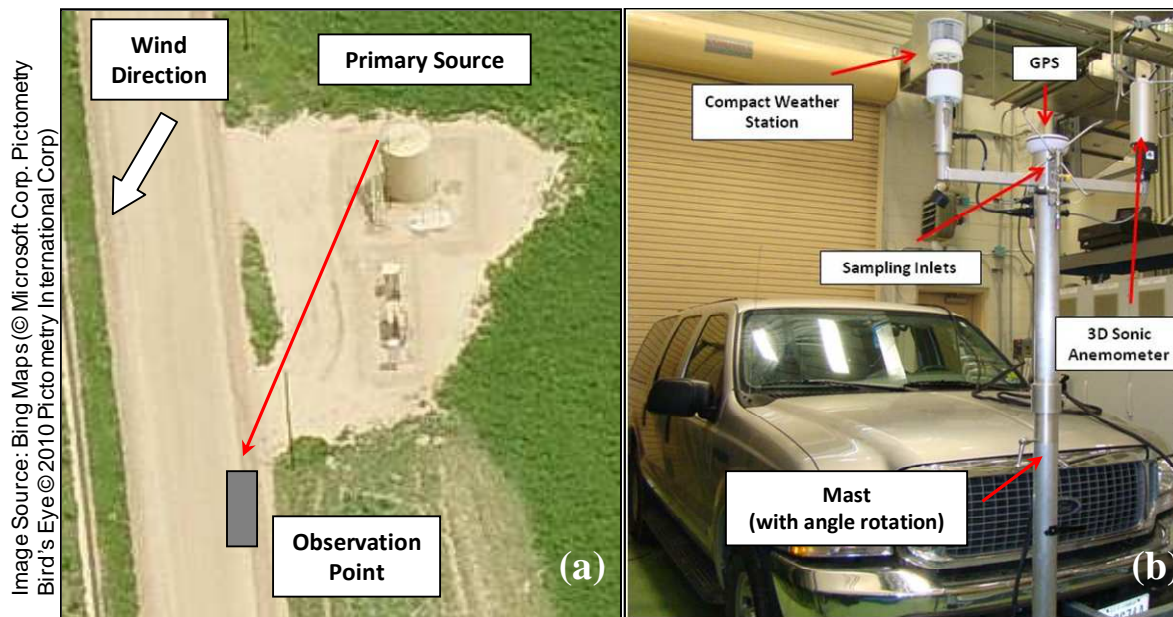
To complement evolving on-site leak measurement approaches, EPA's Geospatial Measurement of Air Pollution (GMAP) program is developing mobile emission measurement techniques for oil and gas and other fence-line applications. The GMAP Remote Emission Quantification (REQ) approach described here utilizes time-resolved instruments, evacuated canisters, and wind measurements to locate and estimate emissions from remote vantage points without need for site access. As with any remote measurement approach, factors such as plume to measurement overlap and wind flow obstructions can complicate downwind emission assessments and limit accuracies. Some improvements in remote measurement performance can be obtained through use of site-specific configurations (i.e. flux plane techniques), released tracers, or advanced computational models, but these come with greatly increased implementation complexity and access requirements. The near-field GMAP REQ approach is designed to be a rapidly deployed inspection method that uses field acquisition and data quality indicators to eliminate measurements with high error potential instead of site-specific configurations or computations. In its current form, the technique produces a 20 minute "snap shot" measure of emissions from near ground level point sources at observation distances of approximately 20 to 200 m. Unlike direct measurements, GMAP-REQ requires wind flow to transport the plume from the source to

the observation location so it can only be utilized under certain conditions. With strict application and favorable conditions, this type of point sensor-based remote measurement is believed capable of measurement accuracies in the $\pm 30\%$ range with ensemble averages achieving accuracies within $\pm 15\%$ by reducing random error effects. Measurement of larger sources at longer distances using metered tracer gas release techniques are also part of the GMAP REQ development effort but are not discussed in this report.

Experimental Methods

For upstream oil and gas applications, the GMAP-REQ platform is a full size sport utility vehicle fitted with lead acid or lithium polymer batteries for operation of measurement equipment. The primary instrument is a model G1301-fc cavity ring-down spectrometer (CRDS) measuring CH_4 , as a surrogate for emissions (Picarro Inc. Santa Clara, CA, USA). To assist in spatial averaging of the plume, sampling is performed through a four-point probe consisting of a 0.95 cm input tube split at the point of sampling into four 0.64 cm dia. inlets set 30 cm apart and mounted to a 2.7 m rotatable mast. The sample flow is nominally 8 slm. Additional equipment includes a high-resolution differential GPS (Hemisphere GPS Calgary, Alberta, Canada), a model AIO compact auto-north weather station (Climatronics Corp., Bohemia, NY, USA), a model 81000 3-D sonic anemometer (R.M. Young, Traverse City, Michigan, USA), a custom canister acquisition system, and a control computer. Figure 1 illustrates a typical measurement configuration near a well pad and provides a close-up view of equipment placement on the sampling mast. The canister (not shown) is attached to a software triggered solenoid at the center of the four-point sampling port just below the GPS. The measurement approach consists

Figure 1: Typical measurement configuration (a), and sampling equipment on mast (b).

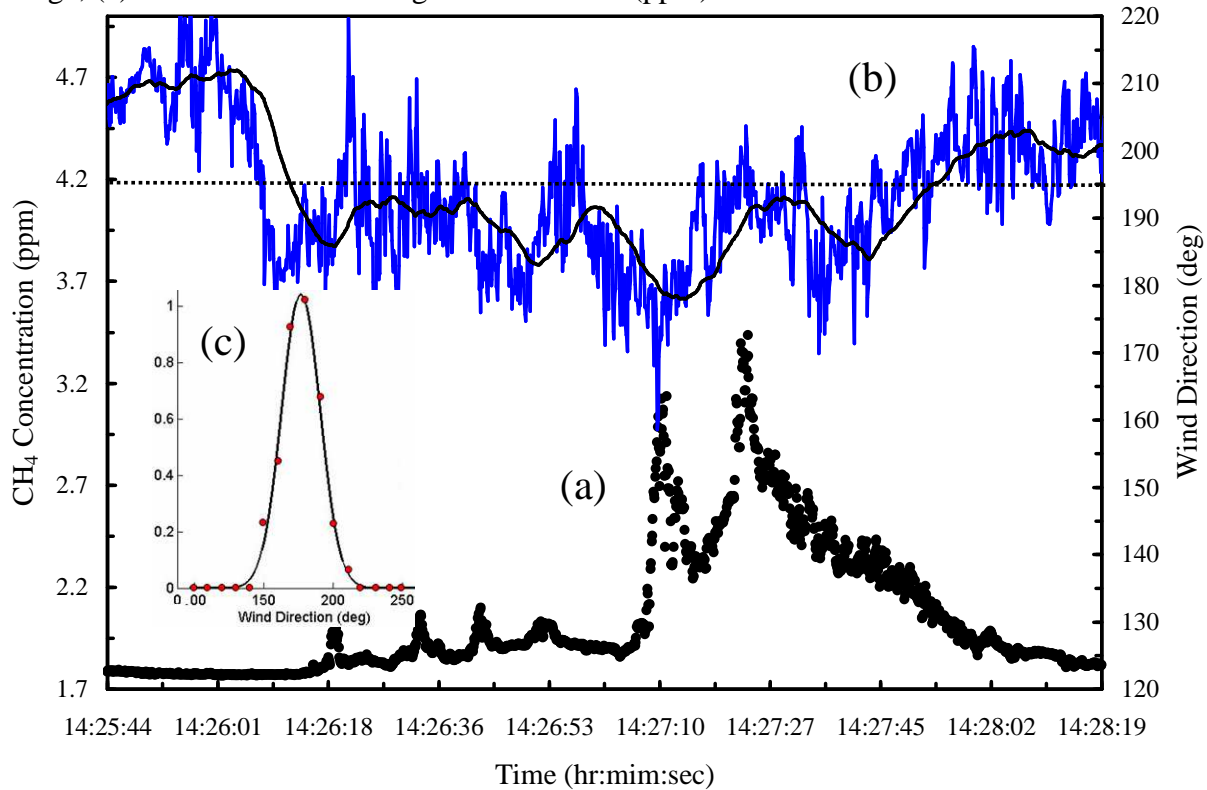


of three primary steps: (1) locate emissions through down-wind, drive-by inspection, (2) determine CH_4 emissions rate by combining time-resolved concentration and wind measurements, and (3) estimate emissions rate of canister-measured compounds by CH_4 ratio calculation.^{10,11} Once measurable emissions are identified, the operator positions the vehicle at an appropriate and safe location near the highest observed CH_4 concentrations facing the source and the engine is turned off to prevent contamination of the measurement from vehicle exhaust.

After placement of traffic cones, the operator obtains off-site infrared video information (if possible) and combines this observation with real-time wind direction and concentration information to identify primary source location(s). The mast is rotated to point in the direction of the source and distance and bearing measurements are taken using a laser range finder and mast-mounted optics. During a 20 minute observation time, data are synchronously acquired at 10 Hz from the CRDS and 3-D sonic anemometer and at 1 Hz from the weather station and GPS using a custom LabView™ control program (National Instruments, Austin TX USA). During the observation, the operator waits for an acceptably high CH₄ concentration with wind direction from the observed source and then triggers a 30 second canister draw for later lab analysis.¹² A post analysis of wind direction and concentration is combined with satellite images and field photographs to refine source identification and observation distance estimates.

The primary assumption of the stationary near-field GMAP-REQ approach is that the fixed-position point sensor is able to obtain representative concentration profiles useful for inverse emission estimation. Representativeness implies sufficient sampling time and spatial overlap of the plume and the probe and the lack of significant symmetry breaking processes such as concentration enhancement by channeling effects. Figure 2 provides an example of time and angle-resolved concentration measurements 82 m away from a 3 m elevated simulated tank emission (0.6 g/s CH₄). As wind direction shifts below $\approx 195^\circ$, the plume begins to be registered as a combination of high and low frequency events (related to vertical overlap and eddy effects).

Figure 2: Example of (a) 10 Hz CH₄ concentration, (b) 10 Hz wind direction with 10second moving average, (c) 20-minute time average concentration (ppm) vs. wind direction with Gaussian fit.



The concentration returns to background levels as wind direction trends above 195 deg. If the observation point is well-centered on the emission plume, a 20 minute observation can produce

numerous such events like those shown in Figure 2a and 2b. Combining these events over the entire observation time allows an average concentration vs. wind direction histogram (in ten degree bins) to be constructed and analyzed (Figure 2c). The character of the time-resolved profiles (mix of high and low frequency components) change in complex ways based on distance to source, atmospheric dispersion, degree of wake induced mixing, and number sources along the observation direction. Regardless of time-resolved form, with sufficient sampling fidelity, the plume centric, time-averaged concentration is believed to carry source strength information useful for the inverse estimates. The REQ approach assumes these measures can be used to produce reasonable estimates of emissions in a variety of scenarios without evoking site-dependent calculations (i.e. to yield a technique useful for rapid deployment).

Significant use limitations are related to spatial overlap of the plume to the observation point, uncertainties in source distance, and heavy obstructions affecting wind flow (trees, fences, etc.). If the height difference between the source and the observation point is too great and/or if too much plume rise exists, the measurement can lead to significant underestimation of emissions through insufficient plume overlap. If the source cannot be identified with confidence or if multiple sources (separated by distance) are present in the angular observation window, the distance utilized in the inverse calculation becomes a key driver of uncertainty. Distance limitations (around 200m) are related to approach assumptions and the necessity to have angular wind sweep generally greater than the plume size. As source size and distance increase, the use of metered tracer gas becomes a preferred approach but at an increase in implementation burden.

Emission estimates using the near-field GMAP REQ approach are determined with two primary algorithms referred to as point source Gaussian (PSG) and backwards Lagrangian stochastic (bLs). A third approach¹¹ was found to overestimate emissions in some cases and is now used to support the assessments. An analysis program, written in MATLAB (MathWorks, Natick MA, USA), time-aligns the measurements to correct for sampling line delay, rotates the 3-D sonic anemometer data to streamlined coordinates, and bins the CH₄ concentration data in ten degree increments by wind direction. The binned values are fitted to a Gaussian function to determine the variation of CH₄ concentration in the crosswind direction and the peak concentration. The program calculates a local atmospheric stability indicator (ASI) used in the PSG estimate that is determined from an average of the turbulence intensity (TI), measured by the 3D-sonic anemometer and the standard deviation in 2-D wind direction ($\sigma\theta$), acquired by the compact met station. The ASI ranges from 1 (TI > 0.205, $\sigma\theta > 27.5^\circ$) to 7 (TI < 0.08, $\sigma\theta < 7.5^\circ$), roughly corresponding to Pasquill stability classes A-D, in steps of one unit with equal increments (TI = 0.025, $\sigma\theta = 4.0^\circ$) defining each step. The program also prepares the CH₄ concentration and 3-D sonic anemometer data for input to the bLs model.

For the PSG emission estimate, the values of horizontal (σ_y) and vertical (σ_z) dispersion are determined from an interpolated version of point source dispersion tables¹² using the measured source distance and the ASI. The PSG emission estimate (q) is a simple 2-D Gaussian integration (no reflection term) multiplied by mean wind speed (u) and the peak concentration (c) determined by the Gaussian fit: ($q = 2\pi \cdot \sigma_y \cdot \sigma_z \cdot u \cdot c$). The bLs approach utilizes the same peak concentration along with 3-D sonic anemometer data in a bLs model called WindTrax.¹³ The data used for the PSG and bLs approaches are pre-processed using a wind acceptance angle filter (+/- 60 degrees) to improve estimation performance by focusing on data originating from the remote source location. The bLs application using more powerful open-path measurements is well-validated.¹³ The use of the angle filtered, plume-oriented coordinates and concentration

data in WindTrax is a nonstandard application of the model developed for this point measurement application to help reduce uncertainty due to atmospheric trending and off-axis source placement that are less of an issue when using open-path measurements with bLs.

The performance of the remote emission estimation algorithms was investigated with a series of 27 CH₄ release and recovery experiments (REQ tests) conducted under a variety of atmospheric conditions (ASI 1-6, wind speed 1 m/s-7 m/s), observation distances (18 m-103 m), and release geometries designed to simulate near-field obstructions and wake flow effects from condensate tanks. For nominal CH₄ release rates of 0.6 g/s ($\pm 10\%$), the PSG and bLs estimates yield averages of 0.56 g/s, ($\sigma = 0.17$ g/s) and 0.57 g/s ($\sigma = 0.23$ g/s) respectively. Since individual PSG and bLs estimates can differ, the current approach employs an average of the two to help protect against method-specific errors through comparison of results. Figure 3a shows the PSG-bLs combined results for the REQ tests as function of distance between the release and observation points with the error bars representing the individual results (PSG in the high estimate position in 67% of the cases) and the closed circles the average of the results. At location 10 m (open circle), the group average (0.57 g/s) with $\pm 1 \sigma$ error bars ($\sigma = 0.18$ g/s) are shown. As evidenced by σ values approaching 30% of the mean, individual measurements can depart significantly from actual; however, repeats can reduce measurement error significantly. The REQ test results do not show significant trends with varying atmospheric conditions although unstable, low wind speed conditions (< 1 m/s) produce little usable data due to plume rise. Measurements beyond about 100 m require favorable atmospheric conditions to transport the plume to the observation location and the largest underestimates in REQ tests occur as distance increases. The largest overestimate (1.03 g/s) occurred in a series of releases where obstructions near the observation point were present so channeling effects were a possible contributing factor.

Figure 3: (a) PSG-bLs combined emission estimate results for release experiments (N=27) and (b) comparison of PSG and bLs results for release and field data (N=321).

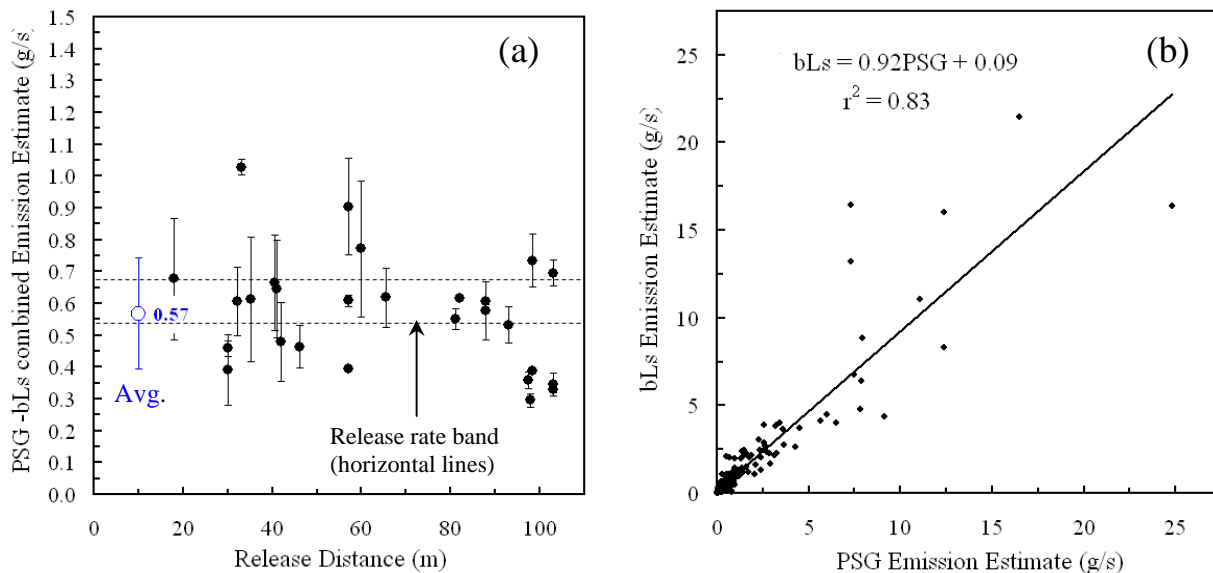


Figure 3b compares the PSG and bLs results for a combination of the REQ tests and the subsequently presented field data. Over this expanded range, the PSG and bLs estimation approaches provide similar results ($bLs = 0.92 PSG + 0.09$, $r^2 = 0.83$). Regarding estimate uncertainty, a one step change in the ASI index in the PSG estimate can change the result by $\approx 25\%$. For the bLs approach with current settings, single measurement standard deviations are on

the order of 30%. Since the peak concentration input is the same for each approach, differences in individual estimates are due primarily to differences in each method's dispersive factor for the emission estimate. As general sources of error, each method shares uncertainty associated with the representativeness of the peak concentration as well as the source identification and distance. Near-field obstructions affect both techniques, likely in somewhat different ways, through both concentration and wind field errors. Further technique development will focus on understanding these factors, differences in model performance, and uncertainty.

The data presented in this interim report were processed with January 2011 versions of the data GMAP REQ data analysis software using data quality filters that remove measurements with average wind speed less than 1 m/s, CH₄ peak concentrations values < 50 ppb over background, and Gaussian fit correlations < 0.7. As technique development is ongoing, these interim results may be revised based on refinements to emission estimate approaches or data screening procedures. A measurement method package with a complete description of the analysis, software, engineering design and operational protocols will be submitted in 2012 to the EPA Emissions Measurement Center for posting consideration as a Category C preliminary method.

Results and Discussion

This paper presents preliminary results from five field campaigns, each approximately 15 days in duration, conducted in the Greeley, CO area in July 2010 and July 2011, the Fort Worth, TX area in Sept. 2010 and Sept. 2011, and the Pinedale, WY area in June 2011. The 2011 studies used a refined version of the technique reflecting both hardware and software improvements based on learning from the 2010 studies.^{10,11} Since conditions were similar, data from the 2010 and 2011 studies are combined by location yielding three primary groups (CO, TX, and WY). In addition to data acquired in the Fort Worth, TX area (Tarrant, Denton, and Wise counties), 27 measurements were conducted in southern TX near La Salle and Carrizo Springs but are not presented here. Approximately 300 remote CH₄ measurements and infrared camera videos and 200 canister samples were collected during these surveys. A Google Earth™ database with custom data viewing interface is being developed to facilitate visualization of these results and will be described in the presentation.

Table 1: Summary of key atmospheric data and background CH₄ concentration data.

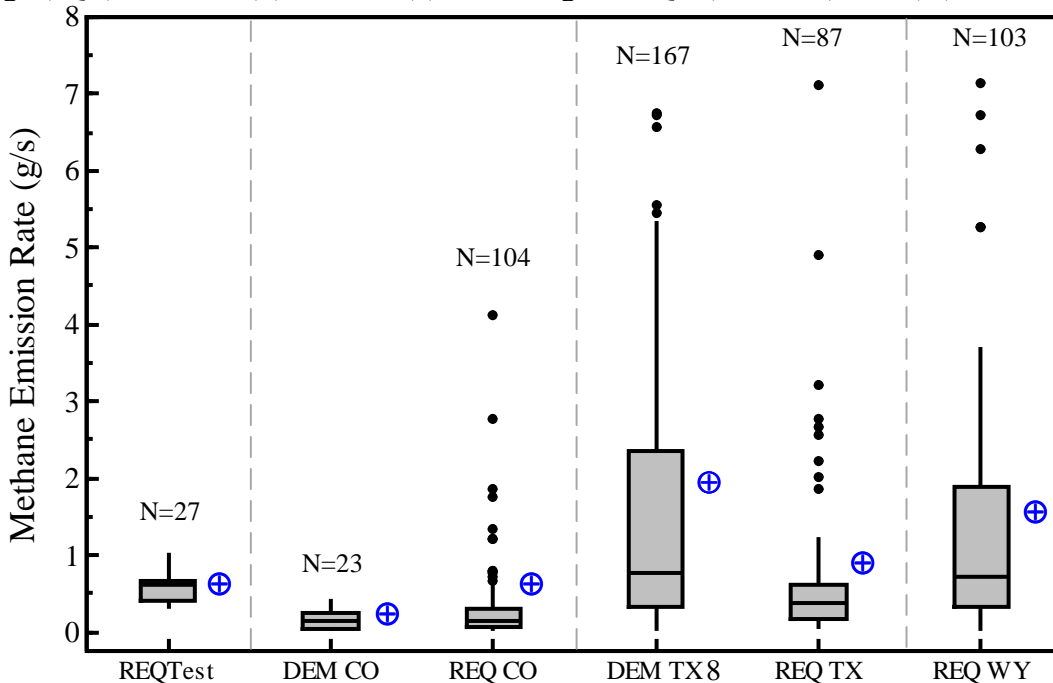
	Colorado (N = 104)					Texas (N = 87)					Wyoming (N = 103)				
	Wind Speed (m/s)	Temp. (deg. C)	ASI (unit)	Dist. (m)	Bkg. CH ₄ (ppm)	Wind Speed (m/s)	Temp. (deg. C)	ASI (unit)	Dist. (m)	Bkg. CH ₄ (ppm)	Wind Speed (m/s)	Temp. (deg. C)	ASI (unit)	Dist. (m)	Bkg. CH ₄ (ppm)
Mean	2.7	30.7	3.7	39	1.79	2.9	31.6	3.7	72	1.85	4.5	18.5	4.2	58	1.77
Median	2.3	30.9	4.0	33	1.77	2.8	31.3	4.0	63	1.83	4.1	17.9	4.0	60	1.76
Stddev.	1.4	3.3	1.5	21	0.07	1.2	3.4	1.7	42	0.07	1.6	4.2	1.3	24	0.04
Min.	1.0	22.3	1.0	18	1.71	1.3	20.8	1.0	17	1.76	1.9	9.8	1.0	17	1.72
Max.	7.6	35.3	7.0	152	2.17	6.0	39.8	7.0	200	2.32	9.4	27.9	7.0	150	1.96

Data from the CO, TX, and WY studies are summarized in Tables 1 and 2 and compared to existing results in Figures 4 through 6. All measurements were conducted in daylight hours on days without significant rainfall. Table 1 summarizes the atmospheric conditions and background CH₄ values for the studies. These data represent a compilation of conditions recorded during each 20 minute remote measurement and are reported after the ± 60 deg analysis filter. Of the three studies, CO shows the lowest mean wind speed and ASI values but also

possesses the closest off-site access for measurement due to the locations of the well pads in relation to public roads. The TX studies had slightly higher wind speeds but also longer observation distances challenging efficient application of the approach. The WY studies possessed the most favorable atmospheric and observation conditions and measurements were easily conducted as a result. The robustness and precision of the CH₄ measurement is key to the measurement approach and the CRDS unit was exemplary in this regard with no calibration adjustments required over the entire measurement set. This is evidenced in the low variance in background values (average of the lowest 100 data points for each observation). CRDS calibration was checked four times each field study and was within 2.5% (on average) of 2.0 ppm and 20 ppm certified standards. No bias correction was utilized in the analysis.

CH₄ emission data acquired by off-site observations using the GMAP-REQ approach are presented in Figure 4 along with results from two on-site direct emissions measurement (DEM) studies. The REQ CO, REQ TX, and REQ WY entries show remote emission measurement results from Colorado, Texas, and Wyoming, respectively (N = number of sites). DEM CO represents preliminary data from a July 2011 EPA direct measurement study in Greeley CO⁹ and DEM TX8 provides results from the City of Fort Worth Natural Gas Study (including only sites with emission measurements).⁸ Also shown are results from the controlled CH₄ release and recovery tests of Figure 2a (REQ Test). The ordinate scale of Figure 4 is limited for ease of viewing with the following values (in g/s) off scale: REQ CO (11.9, 14.2), REQ TX (10.3, 20.6), REQ WY (8.4, 10.3, 11.1, 19.0), DEM TX8, 12 values ranging from 8.5 to 33.1. The direct measurements in DEM CO and DEM TX8 were performed by the same measurement team using the same methodology but the former focused more on condensate tank emissions.

Figure 4: Comparison of CH₄ emission measurements data from several studies, (□) interquartile range (IQR) box with (—) median, (•) exceeding 1.5*IQR (whiskers), and (⊕) mean.



Similarities are evident in the REQ and DEM results with TX and WY showing somewhat larger CH₄ emissions compared to CO. The positive skew distribution (mean > 2*median), a reflection of outlier values, is driven by multiple factors including variations in source size (enhanced

number of smaller production pads vs. larger production units). This is especially true for the DEM TX8 results which contain a wide range of facility sizes. A factor producing a low bias potential in REQ results is related to the underestimation of emissions due to insufficient plume overlap. There are also factors that can lead to high bias in the REQ results (discussed in presentation). The highest observed REQ values are believed to be primarily related to shorter time duration flash emissions from condensate tanks. Because many production pad emissions are short-term in nature, instantaneous emission assessments should not be extrapolated to tons per year values. As evidenced by infrared camera videos, some of the observed emissions are more sustained in nature originating from equipment and pipeline leaks, open thief hatches, failed pressure relief valves, and possible stuck separator dump valves. The mobile, off-site nature of the GMAP-REQ method provides particular utility in locating and assessing maintenance-related emissions which are difficult to capture with DEM approaches requiring prearranged site access.

Table 2 summarizes a subset of emission measurements and concentration data from the REQ studies that include both CH₄ emission and VOC canister data. The average measured CH₄ emission rate is higher in this subset since canisters were only acquired in the more robust observations (with relatively stable and strong offsite plumes). The VOC emission estimates are based on the summation of a 37-compound set (excludes CH₄ and ethane) that assumes a zero VOC background and assigns zero to below detection limit values (\approx 0.2 ppbC).

Table 2: Summary of canister subset CH₄ (above background) and VOC concentration and emission estimate data. BETX = benzene, ethylbenzene, toluene, and xylene isomers.

	Colorado (N = 52)					Texas (N = 59)					Wyoming (N = 75)				
	CH ₄ Conc. (ppb)	CH ₄ Emis. Est.. (g/s)	VOC Emis. Est.. (g/s)	BTEX Emis. Est.. (g/s)	Benzene Conc. (ppb)	CH ₄ Conc. (ppb)	CH ₄ Emis. Est.. (g/s)	VOC Emis. Est.. (g/s)	BTEX Emis. Est.. (g/s)	Benzene Conc. (ppb)	CH ₄ Conc. (ppb)	CH ₄ Emis. Est.. (g/s)	VOC Emis. Est.. (g/s)	BTEX Emis. Est.. (g/s)	Benzene Conc. (ppb)
Mean	3491	0.84	0.81	0.02	8.50	2812	1.33	0.14	0.00	0.85	1717	2.03	0.83	0.10	4.62
Median	2843	0.21	0.16	0.00	1.83	1859	0.43	0.04	0.00	0.19	863.9	0.79	0.30	0.01	0.86
Stddev.	3121	2.52	2.27	0.07	19.4	4042	3.09	0.31	0.01	2.39	2386	3.08	1.49	0.26	10.7
Min.	183.9	0.02	0.00	0.00	0.00	171.8	0.03	0.00	0.00	0.00	101.2	0.05	0.00	0.00	0.00
Max.	16150	14.2	14.5	0.45	120	27820	20.6	1.46	0.05	16.0	12220	19.0	8.89	1.27	60.8

Individual compound VOC emission estimates are calculated by multiplying the VOC to CH₄ concentration and molecular weight ratios by the CH₄ emission estimate.^{10,11} VOC and benzene emissions and ground level concentrations are higher in CO and WY compared to the observed areas in TX primarily due to the wet gas nature of the production and higher density of condensate tank observations in the former.⁶⁻⁹ Offsite benzene concentration data is elevated in CO in part due to atmospheric conditions. The 37-compound VOC list utilized in this analysis is a subset of the ozone PAMS precursor list¹² that was selected based on the above detection limit occurrence frequency and relevance to oil and gas sources. The compound subset is shown in Figure 5 which displays the VOC to CH₄ concentration ratio used in the emission estimate for 60 canister acquisitions for each study (results with CH₄ levels <100 ppb were excluded from the Table 1 Summary). The wet gas (higher VOC to CH₄ ratio) vs. dry gas (lower VOC to CH₄ ratio) is visually evident in the comparison of the REQ CO and REQ WY data to the REQ TX data.

Figure 5a: Visual summary of VOC to CH₄ ratio for 60 canisters acquired in REQ CO studies (m-Diethylbenzene not shown). Each vertical column is an individual canister result. Compounds are in horizontal rows with the color bar indicating ratio.

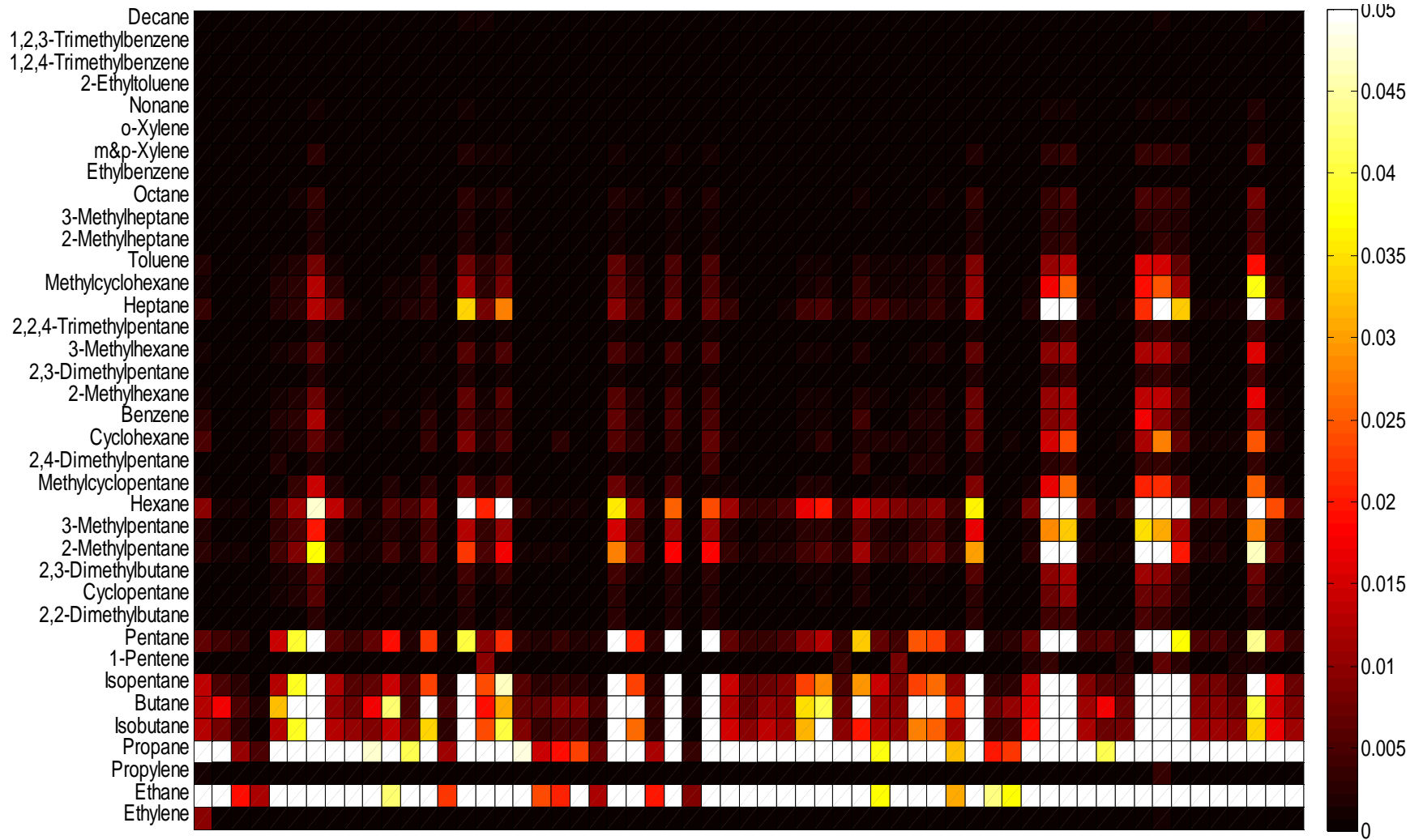


Figure 5b: Visual summary of VOC to CH₄ ratio for 60 canisters acquired in REQ TX studies (m-Diethylbenzene not shown). Each vertical column is an individual canister result. Compounds are in horizontal rows with the color bar indicating ratio.

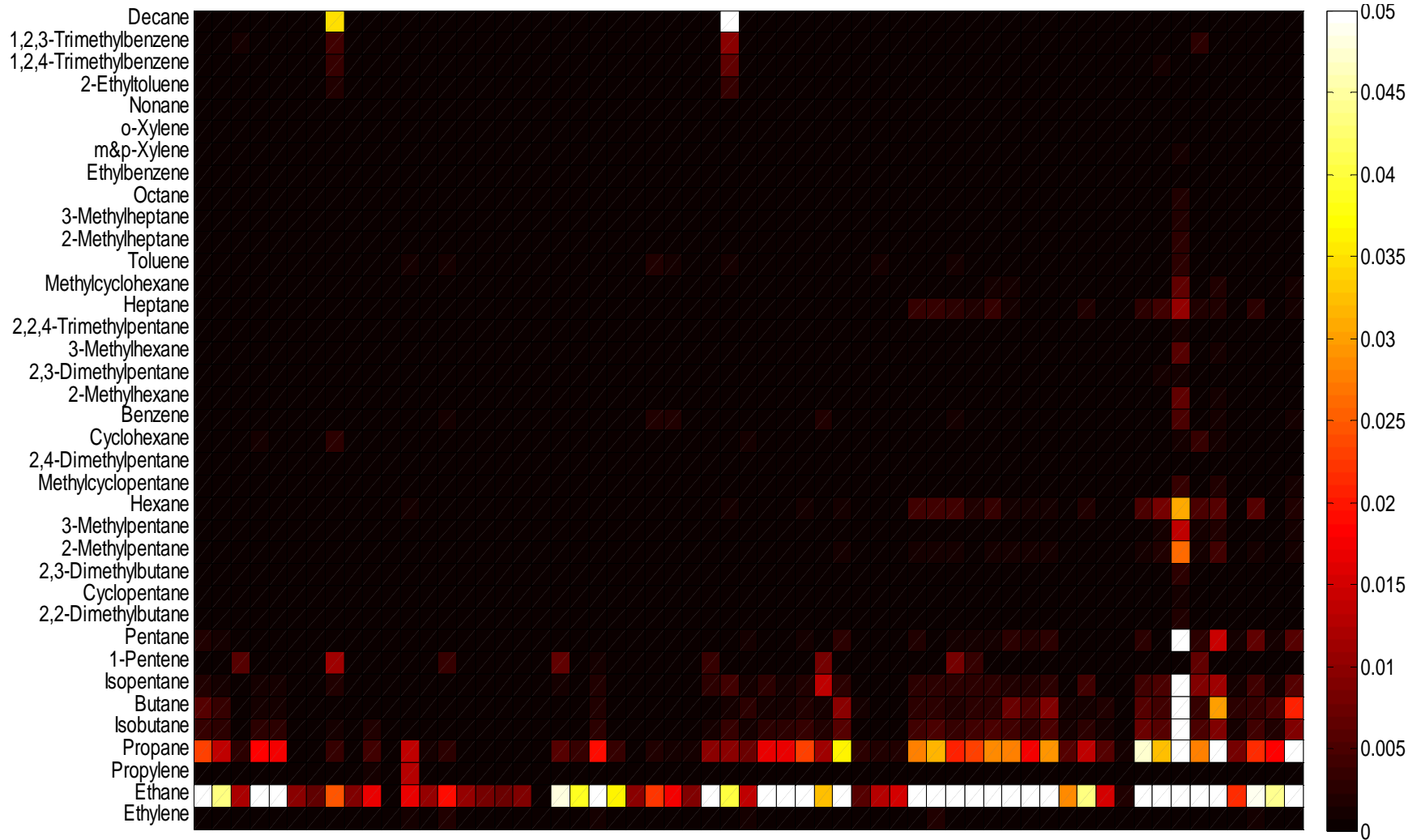


Figure 5c: Visual summary of VOC to CH₄ ratio for 60 canisters acquired in REQ WY studies (m-Diethylbenzene not shown). Each vertical column is an individual canister result. Compounds are in horizontal rows with the color bar indicating ratio.

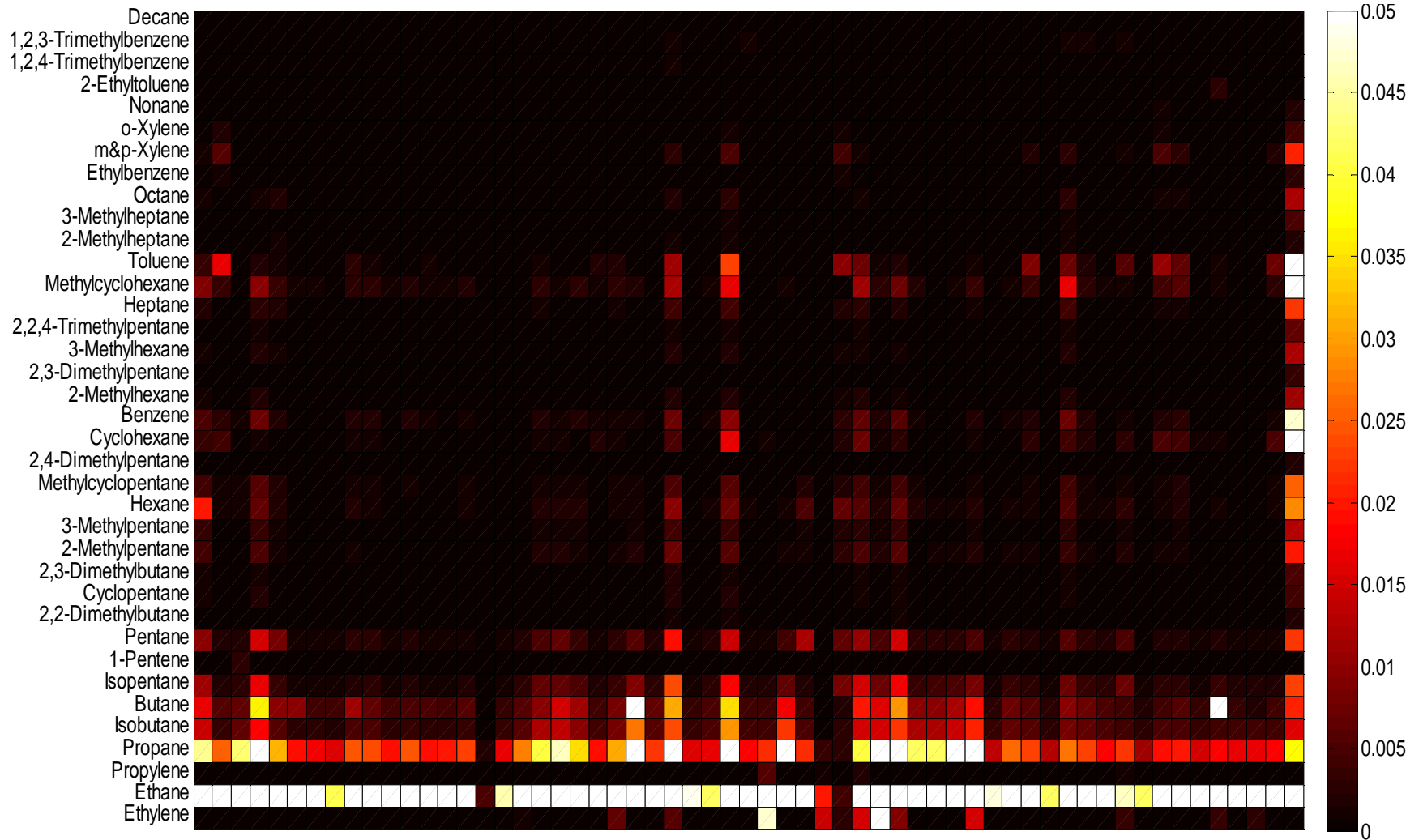
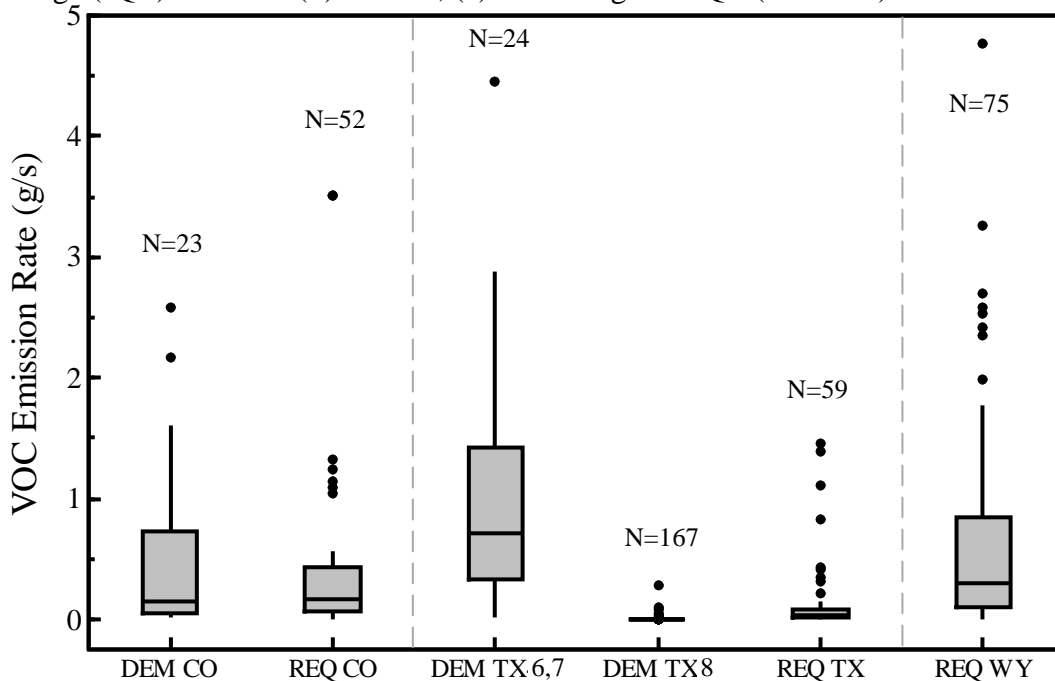


Figure 6 compares VOC emission data from several studies including results from two condensate and oil tank emissions projects ^{6,7} (DFW TX area, DEM6,7). The following values (in g/s) are off scale: REQ CO (14.5, 7.3), DEM TX6,7 (6.4) and REQ WY (8.9,7.2). DEM CO and DEM TX8 were performed by the same contractor and approach (except canister lab analysis) with the former study focusing more on condensate tank emissions.

Similarities in the DEM and REQ results are noted with large differences in DEM TX6,7 and DEM TX8 due to the focus of the latter study on dry gas sites and the former exclusively on tank battery emissions in wet gas and oil areas (illustrates range of emission potential). The REQ TX results contain a mixture of both cases with substantially more coverage in the wet gas areas in contrast to DEM TX8. With its broader mix of data, the REQ TX results confirm significantly lower overall VOC, and HAP emissions in these regions of TX in comparison to the CO and WY results and serves to illustrate the major differences in emission profiles in different geographical areas. Even though overall VOC emissions appear lower in REQ TX, there are sites with significant VOC and HAP emissions (at least in snapshot measure) that require consideration. Note that results in other areas of TX will likely differ.

As in Figure 4, the REQ studies show a considerable number emissions exceeding 1.5*IQR. In some cases, these emissions are believed to be of relatively short duration occurring as flash emissions from condensate tanks. In other cases, the emissions may be related to maintenance issues previously mentioned and could be more sustained as a result. Additional analysis of repeat measurements is underway to better understand the temporal variability of emissions. Infrared camera images will be used to illustrate these points in the presentation.

Figure 6: Comparison of VOC emission data from several studies, (□) interquartile range (IQR) box with (—) median, (•) exceeding 1.5*IQR (whiskers).



SUMMARY AND NEXT STEPS

Environmentally responsible development of oil and gas can be facilitated by advancement of emission measurement tools. This paper describes EPA's GMAP REQ mobile off-site measurement technique and its use in 2010 and 2011 oil and gas production pad emission survey studies in areas around Greeley, CO, Fort Worth, TX, and Pinedale, WY. Preliminary summary data are presented here with additional analysis available for presentation.

The near-field GMAP REQ approach can complement evolving on-site measurement approaches for upstream oil and gas applications. The strengths of the approach lie in its ability to survey larger geographic areas and to identify and quickly assess emissions in a range of scenarios. The weakness of the approach is the reliance on acceptable wind conditions for plume transport and the presence of downwind road access. The method is best applied in open flat areas with few obstructions and may not be usable in areas with high topographic relief or forests without close, line of site access to the sources under observation.

Continued analysis of this preliminary data set is underway, especially with regard to assessments of data quality filters and investigation of high outlier values. Work will continue to understand both the PSG and bLS emissions estimate approaches to help characterize uncertainty. This will include computational fluid dynamic simulations of wake flow around typically observed sources. Additional analysis will investigate the impact of assuming a zero VOC background in the emission calculation which currently leads to a positive bias. We will also investigate the use of CRDS-determined CH₄ concentration measured at the time of canister draw for the VOC calculation. This CRDS measure may be more accurate than the canister-determined value and will therefore improve the overall calculation. Additional work will focus on continued method protocol development and expansion of the approach to other concentration measurement instruments and potentially to applications such as large facility fence-line monitoring.

ACKNOWLEDGEMENTS

The authors would like to thank Michael Miller with EPA Region 6 for his efforts on project. We also thank Bill Mitchell, with EPA ORD NRMRL and Connie Oldham with EPA OAQPS for ongoing support on this project. Particular thanks to Frank Grainger and Parik Deshmukh with Aracdis for their field efforts. Thanks to many individuals at Enthalpy for analytical support for this project. Special thanks Chris Rella, Eric Crosson and many others at Picarro Inc. for ongoing support for GMAP development efforts. Thanks also to Loretta Lehrman, Marta Fuoco, Chad McEvoy, Jesse McGrath, in EPA Region 5 for ongoing collaboration in this area.

Disclaimer: This article has been reviewed by the Office of Research & Development, U.S. Environmental Protection Agency, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the agency nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

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KEYWORDS

oil and gas production, fugitive emission, VOC emission, HAP emissions, GHG emissions, mobile measurement, remote sensing, methane, inspection, mitigation, toxic air pollution