RESEARCH ARTICLE

Methane emissions from contrasting production regions within Alberta, Canada: Implications under incoming federal methane regulations

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Aggressive reductions of oil and gas sector methane, a potent greenhouse gas, have been proposed in Canada. Few large-scale measurement studies have been conducted to confirm a baseline. This study used a vehicle-based gas monitoring system to measure fugitive and vented gas emissions across Lloydminster (heavy oil), Peace River (heavy oil/bitumen), and Medicine Hat (conventional gas) developments in Alberta, Canada. Four gases (CO_2, CH_4, H_2S, C_2H_6), and isotopic $\delta^{13}C_{CH_4}$ were recorded in real-time at 1 Hz over a six-week field campaign. We sampled 1,299 well pads, containing 2,670 unique wells and facilities, in triplicate. Geochemical emission signatures of fossil fuel-sourced plumes were identified and attributed to nearby, upwind oil and gas well pads, and a point-source gaussian plume dispersion model was used to quantify emissions rates. Our analysis focused exclusively on well pads where emissions were detected >50% of the time when sampled downwind. Emission occurrences and rates were highest in Lloydminster, where 40.8% of sampled well pads were estimated to be emitting methane-rich gas above our minimum detection limits (m = 9.73 m^3d^{-1}). Of the well pads we found to be persistently emitting in Lloydminster, an estimated 40.2% (95% CI: 32.2%–49.4%) emitted above the venting threshold in which emissions mitigation under federal regulations would be required. As a result of measured emissions being larger than those reported in government inventories, this study suggests government estimates of infrastructure affected by incoming regulations may be conservative. Comparing emission intensities with available Canadian-based research suggests good general agreement between studies, regardless of the measurement methodology used for detection and quantification. This study also demonstrates the effectiveness in applying a gaussian dispersion model to continuous mobile-sourced emissions data as a first-order leak detection and repair screening methodology for meeting regulatory compliance.

Keywords: Methane; Fugitive and vented emission; Monitoring; Oil and gas; Vehicle-based; Regulation

Introduction

Methane is a short-lived greenhouse gas with a radiative heating potential 28–34 times that of carbon dioxide over a 100-year timespan, and is the main constituent of energy sector emissions (Rella et al., 2015; Gasser et al., 2017). In energy developments, methane is emitted during flaring, venting (reported and unreported), fugitive leakage, combustion, storage and handling losses, and accidental releases (Canadian Association of Petroleum Producers, 2004).

The Canadian government aims to reduce oil and gas methane emissions 40–45% from 2012 levels by 2025 (Government of Canada, 2018), with the province of Alberta setting a parallel target (Government of Alberta, 2016). Although proposed venting allowances are different amongst federal and provincial jurisdictions, both regulatory approaches will impose new emission caps to achieve their reduction goals. To-date, industry has not been required to routinely measure and record emissions, thus the assumed policy baseline is predicated on emission factor estimates (US EPA, 2013; Barkley et al., 2017) from 2011, extrapolated to present (Environment and Climate Change Canada, 2015). U.S. studies have shown that inventory estimates based on emission factors tend to show downward bias (Miller et al., 2013; Allen, 2014; Brandt et al., 2014), and that emissions from U.S. developments are higher than previous Environmental Protection Agency (EPA) estimates (Katzenstein et al., 2003; Karion et al., 2013; Miller et al., 2013; Pétron et al., 2014; Peischl et al., 2016). Most recently, Alvarez et al. (2018) estimates that measured emissions from the U.S. oil and natural gas supply chain are ~60% higher than current EPA inventory estimates. In Canada, measurements are sparse, but recent

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studies have documented higher methane emissions than those which are reported, or presently reflected in inventories (GreenPath Energy Ltd., 2016; Atherton et al., 2017; Johnson et al., 2017; Zavala-Araiza et al., 2018). A recent ground-based study recorded total measured methane emissions that were 15 times higher than total reported emissions in Red Deer, Alberta (Zavala-Araiza et al., 2018), and an aircraft mass balance study suggests that emissions within the Alberta Lloydminster region are 3.6 times greater than inventory estimates (Johnson et al., 2017). If Alberta's oil and gas sector releases 25–50% more methane than currently reported, as suggested in the aforementioned study (Johnson et al., 2017), total wasted methane represents annual losses of ~$213 million CAD in natural gas sales for industry, and ~$17 million CAD in royalties for government (Pembina Institute, 2018).

In any cap-based regulatory system, the mitigation economics depend on the proportion of infrastructure emitting above the established threshold, and the costs for fixing the affected infrastructure. Aircraft studies lack the fine-scale resolution to quantify emissions from specific classes of infrastructure (Allen, 2014), but vehicle-based emission surveys (Phillips et al., 2013; Brantley et al., 2014; Eapi et al., 2014) are well suited for this purpose and can cover more ground than Optical Gas Imaging (OGI)-based campaigns. Relative to the U.S., few ground-based studies have been conducted in Canada, and detailed measurements are urgently needed to guide industry and inform policy development.

This study describes vehicle-based surveys across three upstream energy developments in Alberta; Canada's largest oil and gas producing and exporting province (Alberta Energy, 2015). Our aims were to (i) broadly describe methane mole fractions and drivers of variation across several developments; (ii) quantify vented and fugitive emissions frequency and severity from well pads with varied infrastructure; (iii) determine the proportion of sites that exceed the emissions threshold (combined volume of hydrocarbon gas that is vented, destroyed, or delivered) of 40,000 m^3 $year^{-1}$ (~110 m^3 day^{-1}), in which venting must be limited to 15,000 m^3 $year^{-1}$ (~41 m^3 day^{-1}) under Canadian federal methane regulations (Government of Canada, 2018), and (iv) test the effectiveness in applying a gaussian dispersion model to continuous, mobile-sourced emissions data for leak detection and repair (LDAR) screening applications.

Methodology

a) Survey locations

Peace River (heavy oil/bitumen), Lloydminster (heavy oil), and Medicine Hat (conventional gas) were chosen as field sites for this project due to their contrasting operation practices, production types, and the extensive oil and gas development within each region (**Figure 1**). These areas are representative of several hydrocarbon production types in Canada.

Cold Heavy Oil Production with Sand (CHOPS) is prevalent in the Lloydminster Region, and to a lesser extent in Peace River, where thermal recovery is also common. CHOPS wells produce water, sand, oil, and associated gas, which get separated at the surface via battery production facilities. Excess associated gas, deemed uneconomic to capture or re-inject, is typically flared or vented at these batteries (Johnson and Coderre, 2012; Alberta Energy

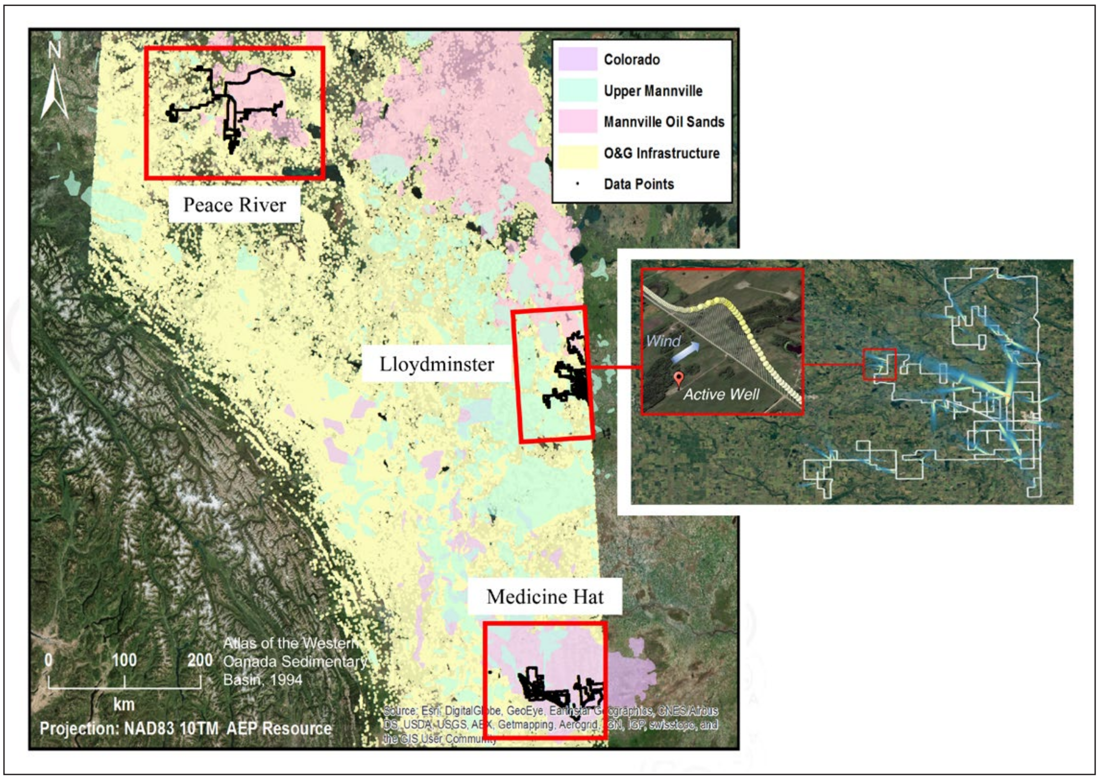

Figure 1: Survey locations in Alberta, Canada. Major geological formations are shown. Survey routes are depicted in black. DOI: https://doi.org/10.1525/elementa.341.f1

Regulator, 2017a). As a result, CHOPS sites commonly emit large and irregular amounts of methane.

Air quality has historically been a concern in Peace River, where the oil sands bitumen deposits have higher levels of sulphur and aromatic compounds than other areas of the province, prompting frequent public odour-related complaints. Following a 2014 report, 'Recommendations on Odours and Emissions in the Peace River Area (Alberta Energy Regulator, 2014), the Alberta Energy Regulator (AER) finalized Directive 084 in 2017, which requires heavy oil and bitumen operations in the area to flare, incinerate, or conserve all casing and tank-top gas, effectively eliminating venting in the area (Alberta Energy Regulator, 2017b).

Alberta's first discovery of natural gas occurred in the Medicine Hat region during the early 1880's. The largest natural gas pool in the province, the Medicine Hat sandstone produces sweet gas and a minor amount of oil (Canadian Society of Petroleum Geologists, 1994). Although it accounts for only 7% of provincial natural gas production, approximately half of Alberta's operating gas wells are located in the Medicine Hat play (GreenPath Energy Ltd., 2016).

b) Field measurements

A series of 45 truck-based air composition surveys were conducted in autumn 2016. These surveys sampled downwind of 2,670 individual active or suspended wells and facilities grouped on 1,299 unique well pads across the three developments. Each survey followed one of five region-specific pre-planned routes on public roads (SM – S1). Individual routes were repeated three times.

Gas mole fractions were measured in real-time while driving using Picarro G2210-*i* CRDS (CO_2, CH_4, C_2H_6, and $\delta^{13}C_{CH_4}$, CH_4 measurement error of <0.1ppb over 5-minute average) and Teledyne T101 (H_2S, precision within 0.5%) ppb-level analyzers recording at 1 Hz. Gases were transported to the analyzers through an inlet at the front of the vehicle (1 m height) using a 7 L m^{-1} pump through 6 mm inner-diameter tubing. The gas measurement data were merged with GPS and meteorological parameters recorded by a Garmin 16x Vehicle Rooftop GPS (position accuracy < 3 m), RM Young 2-D sonic anemometer (windspeed accuracy ± 2 ms^{-1}, direction accuracy ± 2°), and Campbell Scientific 107B thermal gradient temperature sensors (error ≤ ±0.01°C), all logged at 1 Hz using a Campbell Scientific CR1000 datalogger. Vehicle anemometer measurements from each campaign were corrected to remove the vector of the vehicle's motion. Our anemometer data was not reliable in a total 8 surveys, and on these days, hourly wind data collected from the nearest Environment Canada weather station was used. Lags and temporal offsets between measurements were carefully accounted for. Subsequent statistical analyses, including plume detection and source attribution, were performed using R 3.2.5 statistical software (R Core Team, 2016).

c) Detection, attribution, and quantification

Analysis consisted of isolating plumes based on geochemistry, applying corrections to subtract the fluctuating background, and applying geospatial algorithms to attribute observed anomalies to probable sources – as described in Atherton et al. (2017). Only active and suspended infrastructure types were considered for this study. We identified wells by their Unique Well Identifiers (UWI), and facilities by their unique facility IDs. A facility in this paper encompasses all types of infrastructure in our database that has an associated facility ID. These are: batteries, meter stations, regulator stations, injection plants, gas gathering systems, pump stations, gas processing plants, compressor stations, satellites, tank farms, and central treating plants. We defined methane-rich areas as ones in which mole fractions of CH_4 were enriched above background levels – where ratios of super-ambient (excess gas, written here as '*e*') CO_2 and CH_4 (eCO_2:eCH_4) were highly depressed relative to the global atmospheric average of ~215 (Atherton et al., 2017). Specifically, we looked for ratios <120 to define methane-rich areas in Peace River, <100 in Medicine Hat, and <65 for Lloydminster, where solution gas venting is the highest in the province (Alberta Energy Regulator, 2017a). These ratio cut-offs were chosen from kernel density plots of measured excess mole fraction ratios, as explained further in SM – S2. Described in Hurry et al. (2016), the excess ratio technique is effective in partitioning sources in complex environments where multiple emission source types could exist. The ratio threshold rarely has a significant effect on the number of plumes detected and tends to have more effect on the apparent width of the plume (duration while surveying), which is not pertinent to our study. The ratio-based technique is also less sensitive to pooling of gases in valleys and other factors that contribute to natural methane enrichment in air, lessening our chances of false positives. We frequently reset the ambient background mole fractions using a Running Minimum Reset Interval (RMRI), a time interval over which the lowest observed mole fraction was used to represent the localized ambient background mole fraction.

Distinguishing individual pieces of emitting infrastructure on a multi-infrastructure well pad during mobile measurement introduces attribution uncertainty, so we chose to cluster infrastructure to the well site (pad) level within a 45 m radius. Each grouping represents an individual well pad containing anywhere from 1–19 wells and/or facilities. To locate the estimated emission source (well pad) for each plume, the well pad had to meet the following geochemical, geospatial, and emission persistence criteria: (a) be upwind of an uninterrupted series of 3 or more anomalous on-road datapoints, (b) be within a defined upwind distance of on-road datapoints, (c) meet the above criteria >50% of the times it is surveyed (i.e. 2/2, or 2/3 etc.), and (d) be the closest well pad to the assumed plume centerline (maximum plume eCH_4), if multiple well pads met the aforementioned criteria for a single plume. As the majority of well pads we passed in Lloydminster and Medicine Hat were within 200 m of the road, we only considered well pads that were within 200 m of our survey routes to be potential emission sources for plumes detected on those campaigns. In Peace River, we extended our attribution distance to 400 m because well pads tended to be farther away from the road. Furthermore, we only considered well pads sampled at least twice within the triplicated routes, which excluded all sites that were only surveyed once from analysis. Throughout the three developments we sampled downwind of well pads on 3,973 unique occasions, enumerating 3,574 plumes that met these criteria. Emissions frequencies presented in this study are conservative due to the fact our method does not consider episodic or short-lived plumes detected on 50% or less of our survey passes. Instead, the focus of this study is on well pads we found to be persistently emitting, in which we can have higher confidence due to repeat measurements.

For each emitting well pad, the maximum eCH_4 mole fraction measured during an observed plume was fed into a point-source Gaussian Dispersion Model (GDM) (De Visscher, 2013) to estimate source emission rate, corrected to a standard temperature and pressure. Input parameters to the GDM were: wind speed, distance, estimated source emission height, and estimates of Pasquill atmospheric stability, with downwind sigma values based on Turner (1994) emission rate estimates. Our GDM followed a 'screening' approach comparable to EPA's ISCST3 (US EPA, 2017). Outputs from ISCST3 are proven to be within 0–2% of EPA's AERMOD dispersion model (US EPA, 2003), which is the Alberta government's recommended dispersion model for refined assessment (Alberta Government, 2013). Using an empirical formula derived from laboratory experiments, raw gas mole fractions were corrected for factors that damp the instruments response function, such as averaging filters in the instrument software, pulse broadening in the tubing, or dilution that occurs within the gas analyzer measurement cavity when our vehicle may be transiting a plume for only a few seconds, resulting in a mole fraction peak that is smaller than the true mole fraction. We determined the magnitude of these effects by introducing gas of known composition at the inlet for increasing duration and measuring the corresponding depression in plume centreline response. Volumetric emission rates presented in this study were the average of 1–7 downwind plume transect passes, depending on the well pad. For each emitting well pad, we used the GDM to test whether the average volumetric emission rate for a well pad exceeded the 110 m^3 day^{-1} (40,000 m^3 $year^{-1}$) venting threshold to trigger mitigation under incoming federal regulations (Government of Canada, 2018). We also estimated an emissions rate Minimum Detection Limit (MDL) at each emitting location using the 5th percentile of excess methane mole fractions for attributed plumes within each campaign, and a source height of 1 m. Sensitivity testing described further in SM-S3.2, showed that emission release height was the largest source of GDM uncertainty, as many emitting well pads were comprised of an infrastructure distribution with varying possible emission source heights. As emissions data were recorded while in continuous movement, our method does not have the specificity to pin-point the exact emission sources on an emitting well pad. To address this, we calculated a weighted mean emission rate per well pad. A range of emission heights were incorporated in the GDM parameterization, dependent on the distribution of emissions source heights of individual infrastructure types on a well pad. We then averaged this population of estimates to arrive at one emission rate estimate per well pad. Well pad specific maximum and minimum emission rates were calculated assuming the emissions originated exclusively from the tallest sources present on a pad, or the shortest, respectively. For simplicity, this manuscript focuses on mean emission rates, but maximum and minimum emission rates for each well pad are presented in the supporting documentation S2.Volumetric_Data.xlsx. In a small number of cases we found ourselves proximal to well pads where tall pieces of infrastructure were situated such as storage tanks, and where our position fell within the theoretical GDM downwash zone, thus underneath a hypothetical plume emitted from that source, where there is a non-Gaussian wake effect. In these situations, we excluded the particular source from the calculated well pad emission rate average.

d) Uncertainty

Mobile-based campaigns are akin to remote sensing, and inherently incorporate uncertainty related to both the detection and attribution of plumes. Using control routes and the same processing techniques as this study, Atherton et al. (2017) estimated the rate of false positive plume detections for this type of northern Canadian landscape, with the conclusion that we can have >99% confidence in emission detection. In this study, we also used geochemical verification (target gas ratios and Keeling plot intercepts, (Keeling, 1958)) to demonstrate certainty in that we are detecting reservoir-sourced gases. In addition to geochemical analysis of data recorded by the truck-based analyzers, we conducted a follow-up field study in Lloydminster during autumn of 2017 in which 31 grab samples were collected in Flexfoil bags (SKC Ltd.) within plumes downwind of emitting well pads. These samples were later analyzed at Royal Holloway University of London for species mole fractions using Picarro 1301, LGR EGGA, and UMEA instruments, and for $\delta^{13}C_{CH_4}$ by a CF-GC-IRMS (Fisher et al., 2006). Source $\delta^{13}C_{CH_4}$ signatures were calculated using Keeling plots with background composition fixed by ambient air samples collected on the same days.

Our plume detection confidence is high, however back-trajectory analysis for attributing plumes to emission sources incorporates more uncertainty because a plume's origin may be difficult to estimate under certain circumstances. Uncertainty in plume attribution is introduced by a) 'shadow' plumes originating in the distance and not from the most proximal well pad which is what our current method assumes to be emitting, b) regions with high infrastructure density where well pads are closely spaced, and c) sources other than those we have considered, such as pipelines, etc. We estimated attribution confidence by considering the assumed emitting well pad relative to all potential emitting sites for that plume (any site upwind and within the considered distance for each campaign). Attribution confidence was 74.7%, 92.5%, and 78.5% for Lloydminster, Peace River, and Medicine Hat respectively. High-density regions such as Lloydminster and Medicine Hat led to lower confidence in plume attribution, but overall rates of attribution suggest that the sources were clearly defined on most occasions. In certain cases, it is possible that actual emissions were not detected, most likely because we were upwind, or the emitting source was too high and not detected based on our proximity, or that it was emitting below our MDL. As a result, our emission persistency estimates are likely conservative to some degree.

There are several sources of emissions rate uncertainty that should be considered in this study including methodological uncertainty, GDM field data parameterization uncertainty, and other considerations such as the impact of obstructions on our modeled emissions rates. These are discussed in detail in SM-S3. Previous ground based mobile dispersion studies have recorded uncertainty estimates ranging from 50–350% (Caulton et al., 2017). To quantify methodological uncertainty in this study, we conducted a series of controlled release experiments at the Carbon Management Canada Research Institutes Field Research Station near Brooks, AB, described further in SM-S3.2. Mean measured emissions rates for populations of triplicate measurements were found to have an upwards bias of 30%, in contrast to median measured emission rates which displayed a downward bias of 18%. The mean measured rates were influenced by a small number of high emitting outliers. True emissions rates are likely to fall within the mean and median estimates, and as a result, we have presented both within the SM and Dataset S2. Future LDAR screening campaigns should consider these uncertainties when flagging a well pad with a potential emissions exceedance. Yet despite these uncertainties, we can be confident in our ability to reliably discriminate expected, or below regulatory emissions, from regulatory exceedances that are orders of magnitude larger.

In this study, emission rate uncertainty is primarily a function of emission height uncertainty. At many sites, a multiplicity of well and facility point sources may have contributed to the plumes we detected on-road, which is why we calculated and averaged multiple emission rate values for each emitting well pad, based on the assumed emission heights from all wells and facilities on-site. The extreme maximum and minimum values for each site represent a worst-case range of emission rate uncertainty specific to that well pad. It should be noted that vehicle-based data collection is somewhat biased toward emission sources close to the ground and may not always fully capture emissions from taller infrastructure, such as tanks and flares. Our average detection distance for the three campaigns ranged from 137–220 meters, and at these distances, it is possible to measure emissions from taller (>1 m) sources. Because we cannot be sure of the exact emitting source on a well pad, we have given equal weight to all possible sources. For improved height parameterization accuracy, additional data such as optical gas imagery would be needed to precisely determine emission point source heights.

Results and Discussion

We sampled a total 1,431 wells and 1,239 facilities operated by 59 unique companies of varying sizes. Each site was sampled downwind at least twice on survey routes that we replicated three times on different days, and at different hours. Survey route statistics by campaign are shown in **Table 1**.

a) Emission geochemistry

Average ambient methane values for Lloydminster, Peace River, and Medicine Hat were 2.41 ppm (n = 186,121), 1.97 ppm (n = 148,391), and 2.03 ppm (n = 147,450) respectively, with all regions more enriched than the current global mean of ~1.85 ppm (National Ocean and Atmospheric Administration, 2018). For context, a similar mobile surveying study conducted by Atherton et al. (2017) in the British Columbia Montney recorded a mean methane value of 1.90 ppm (s = 0.084, n = 444,585), which was very similar to global background values and indicative of lower infrastructure density and/or smaller emissions. In dense developments such as the Barnett Shale, recorded background methane values are as high as 11.9 ppm, (s = 63.58) (Rich et al., 2014). Summary ambient gas statistics are shown in SM – S4.

Mole fraction duration analyses illustrate how raw methane mole fractions were sustained over different time intervals (**Figure 2**). Individual surveys from each campaign were combined and running averages of the methane time series were computed for 1-min., 15-min., and 60-min. intervals. The highest 60-minute averaged value reflects the most severe regional scale anomaly observed, which was 5.40 ppm, 2.24 ppm, and 2.45 ppm in Lloydminster, Peace River, and Medicine Hat respectively. The 'all data' column is the arithmetic average methane mole fraction over all 15 surveys combined. A large 1-minute running average methane value, seen left of the bar chart in **Figure 2** suggests that disproportionately large, or many emission sources may exist locally. The horizontal red line represents the global atmospheric methane background mole fraction of 1.85 ppm (National Ocean and

Table 1: Summary route statistics. DOI: https://doi.org/10.1525/elementa.341.t1

	Lloydminster	Peace River	Medicine Hat
Total km surveyed	2,684	2,881	2,784
Total surveys	15	15	15
Geolocated datapoints collected	2,593,304	2,064,258	2,051,518
Sampled well pads	434	131	734
Total wells on sampled pads	474	219	738
Total facilities on sampled pads	627	246	366

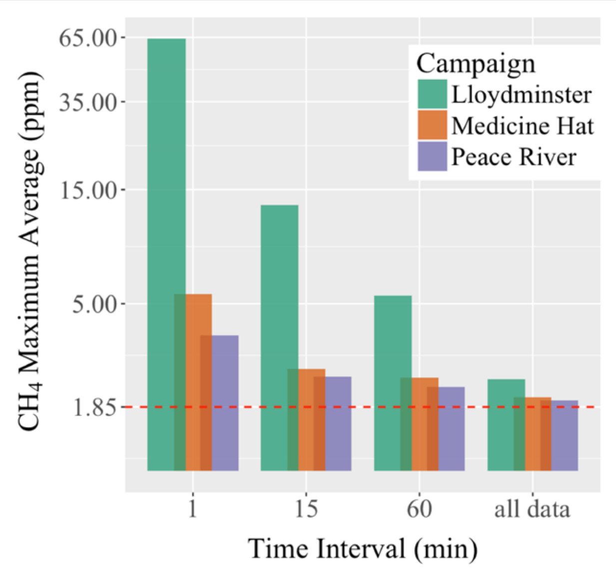

Figure 2: Methane plume mole fraction vs. duration. Individual surveys from each campaign were combined and running averages of the methane time series were computed for 1-min., 15-min., and 60-min. intervals. The maximum average mole fractions observed over these periods is observed on the logarithmic y-axis. DOI: https://doi.org/10.1525/elementa.341.f2

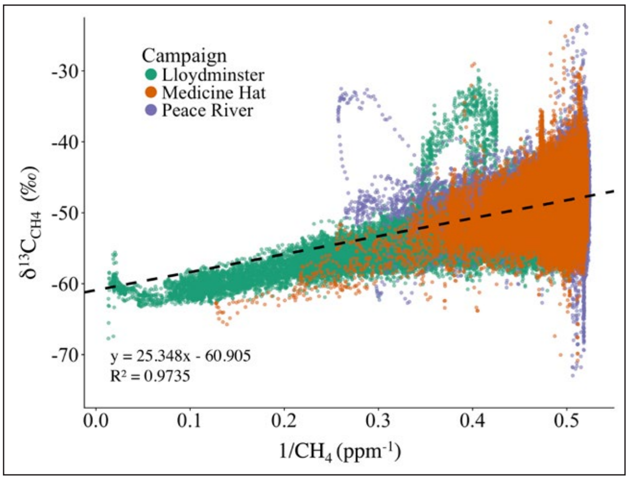

Figure 3: Keeling plot of $\delta^{13}C_{CH_4}$ versus inverse methane from all surveys (n = 45). Results from the follow-up isotopic study conducted in Lloydminster are represented by the trend line. All recorded datapoints are presented, no outliers have been removed. DOI: https://doi.org/10.1525/elementa.341.f3

Atmospheric Administration, 2018). We traverse energy developments at speeds of 60 km/hr to 80 km/hr, so the aforementioned moving averages also reflect mole fractions over space. A 1-minute moving average might reflect a spatial domain of some hectares, thus a local-scale, whereas a 1-hour moving average could reflect average mole fractions across many square kilometers, covering a regional-scale. Lloydminster is the anomaly amongst the three campaigns, having by far the highest methane mole fraction over all timescales.

Keeling plots of raw atmospheric gas mole fractions acquired directly by the Picarro analyzer in the truck (**Figure 3**) suggest that $\delta^{13}C_{CH_4}$ signatures of hydrocarbon point sources are approximately –63.5‰, –63.6‰, and –47.9‰ for Lloydminster, Medicine Hat, and Peace River respectively (indicated by the y-intercept in **Figure 3**). Lloydminster isotopic values fall within published values for the Mannville Group strata underlying Lloydminster ($\delta^{13}C_{CH_4}$ = –70‰ to –60‰), which are equivalent to values observed in the overlying Cretaceous Colorado Group associated with both Lloydminster and Medicine Hat (Rowe and Muehlenbachs, 1999). All three sample regions are characterized by immature, biodegraded reservoirs (Deroo and Powell, 1978; Canadian Society of Petroleum Geologists, 1994), and we expected the Peace River keeling intercept to be more depleted in $\delta^{13}C_{CH_4}$ as a result. However, it falls within the range (–52‰ to –47‰) of $\delta^{13}C_{CH_4}$ compositions from degraded Peace River oil reservoir gases measured by Jones et al. (2008).

From the follow-up grab sample campaigns in Lloydminster, plumes downwind of 31 emitting well pads showed an average $\delta^{13}C_{CH_4}$ of –60.9‰ (s = 1.25), and 0.84% C_2H_6 (s = 0.55) which matches well with our mobile-based calculations (trend line shown in black, **Figure 3**). Both the keeling plot and grab sample results show that measurements recorded within plumes from upstream oil and gas infrastructure were representative of known produced fluid ratios, which show a combined biogenic ($\delta^{13}C_{CH_4}$) and thermogenic (C_2H_6) fingerprint.

Compositional analysis demonstrates that one can recover produced fluid ratios through air sampling of fugitive and vented emissions near oil and gas infrastructure. It also highlights the fact that $\delta^{13}C_{CH_4}$ is an imperfect tracer for many oil and gas production environments, as processes such as biodegradation via methanogenesis often obscure distinctive isotopic signals of origin, particularly common in shallower formations with low thermal maturity (Jones et al., 2008). Past literature has shown $\delta^{13}C_{CH_4}$ fossil fuel signatures for ground-sourced methane in oil and gas developments that are enriched relative to the atmosphere (~–45‰ to –41‰), with a time averaged, globally weighted mean fossil-fuel $\delta^{13}C_{CH_4}$ of –44.0 ‰ (Schwietzke et al., 2016). It is apparent that several Albertan developments depart strongly from this mean. A $\delta^{13}C_{CH_4}$ depleted source component is common in several prolific Canadian petroleum reservoirs (Lopez et al., 2017).

b) Emission detection and attribution – patterns and comparisons across developments

In total, 177 well pads within the Lloydminster heavy oil region, 37 within the Peace River heavy oil/bitumen region, and 95 around the Medicine Hat shallow gas field were flagged as emitting. Frequencies of emissions were 40.8%, 28.2%, and 12.9% in Lloydminster, Peace River, and Medicine Hat, respectively. **Table 2** presents a breakdown of well and facility classes most commonly found on sampled and emitting well pads. For simplicity, suspended and active wells of all categories (gas, oil, etc.) have been combined. It should be noted that in some cases, emissions from well pads with suspended infrastructure may

Table 2: Breakdown of most commonly observed infrastructure (classes that were sampled >10 times total) on sampled and emitting well pads, ordered by occurrence. DOI: https://doi.org/10.1525/elementa.341.t2

	Infrastructure Class	**Present at Sampled Sites (*n*)**	**Present at Emitting Sites (*n*)**
Lloydminster	Battery	585	272
	Suspended Well	278	112
	Active Well	193	99
	Total Well Pads	**434**	**177**
Peace River	Battery	190	68
	Active Well	174	60
	Suspended Well	37	11
	Meter and/or Regulator Station	23	4
	Satellite	19	5
	Total Well Pads	**131**	**37**
Medicine Hat	Active Well	629	69
	Battery	289	130
	Suspended Well	58	11
	Commingled Well	51	5
	Compressor Station	26	13
	Meter and or Regulator Station	25	15
	Total Well Pads	**734**	**95**

originate from gas migration, old associated pipelines, or other infrastructure not considered here.

Well pads with persistent emissions, and thus those flagged as an emitting group on every survey pass, were most likely to contain batteries onsite in all developments. Additionally, multi-infrastructure sites were prone to larger emissions in all cases, when compared to single infrastructure well pads. We saw episodic emissions in all campaigns, but emissions in Lloydminster and Peace River displayed the highest degree of spatiotemporal variability, suggesting a need for monitoring technologies that can capture and quantify long-term variability within a range of emission intensities. Using the Conventional Volumetric Information report publicly available by Petrinex (2018), we performed a regression analysis to compare reported oil (Lloydminster and Peace River) and gas (Medicine Hat) daily production volumes for the month of our study to our measured emissions rates on a per-site basis. There were no statistically significant relationships between measured emissions and production for any of the three developments.

Lloydminster is an anomaly amongst Canadian developments. This is reflected in provincial flaring and venting inventories (Alberta Energy Regulator, 2017a), a 2017 airborne study (Johnson et al., 2017), and additionally in our measurements, through elevated emission frequencies and regional background CH_4. The mean emission intensity for the development was also high at 249 m^3 day^{-1} (95% CI: 173–325 m^3 day^{-1}) per emitting well pad. Considering not all well pads in the development are emitting, our average emission rate per well pad is 102 m^3 day^{-1}, once we multiply by our emission frequency of 40.8%. This value is ~50% lower than the emission rate of 195 m^3 day^{-1} per well pad that Johnson et al. (2017) estimated for the Lloydminster area. To make this comparison, we recalculated the total number of well pads in the Johnson et al. (2017) study area using the centroid provided, our infrastructural databases, and the same method of identifying well pads as described earlier. We then divided the study's oil and gas sector methane emission rate for the region by the number of well pads (n = 4381). This result is in-line with our expectation that a vehicle-based study would lead to lower emission intensity estimates because vehicle-based data collection is somewhat biased toward emission sources close to the ground and may not always fully capture emissions from taller infrastructure, pipelines and other leaks, or service events. Additionally, our emissions frequencies are conservative because well pads at which we detected emissions from 50% or less of the times they were surveyed from have been omitted from analysis. Episodic sources exist yet were excluded in our analysis as they are not the focus of our study.

We saw appreciable emission rate variability in Lloydminster, with s = 512, range = 0.840–4,850 m^3 day^{-1}, and a 75th percentile of 247 m^3 day^{-1}, indicating that a small number of high emitting sources are skewing the mean intensity values. In Lloydminster, plumes were detected on-road at an average distance of 137 m from source at an average MDL of 9.73 m^3 day^{-1}. On emitting pads, 42.7% of the infrastructural population that could have been contributing to emissions were assumed to originate from ground level sources, and 51.2% of the sources were from higher emission sources (i.e. tanks). Overall, the majority of emitting well pads in Lloydminster included tall emission sources, most often tanks.

In oil-producing regions such as Lloydminster, the impact of tank vents on overall emissions inventories could be significant. In this study, 500 active and suspended wells and facilities on 177 unique well pads potentially contributed to the measured emissions, with batteries and active wells being the most prevalent infrastructure types on-site. A study by Lyon et al. (2016) used aerial surveys across seven U.S. oil-producing basins to conclude that oil storage tanks were responsible for >90% of CH_4 emissions. Unreported venting from sources such as pneumatic instrument vent gas and tanks are a major contributor to the discrepancy between government and recent measurement-based inventories in this region (Johnson et al., 2017), and should be included in future inventories. Incoming federal legislation aims to reduce venting from CHOPS sites, but the stochastic nature of these emissions poses significant methane regulation compliance challenges for operators in this area.

In Peace River, active wells made up the majority of sampled wells in the region (174), which contributed to the 28.2% of well pads flagged as emitting. The calculated emission frequency was lower in comparison to that of Lloydminster, the other oil development sampled, and emission severity was intermediate, with a mean emission rate of 158 m^3 day^{-1} (95% CI: 97.6–217 m^3 day^{-1}, s = 186, range = 12.8–891 m^3 day^{-1}) and a 75th percentile of 176 m^3 day^{-1} per emitting well pad. Relative to Lloydminster, Peace River had fewer extreme emitters, as seen by the emissions distribution in **Figure 4**. Emission rate estimates from repeat downwind passes showed appreciable variability. In SM-S3.2 *a) Methodological Uncertainty*, we describe two approaches, one which considers the mean of repeat downwind measurements at a site, the other which considers the median. Had we used the more conservative of the two (higher probability of underestimation), inferred development-average rate estimates for emitting sites would have been ~60 m^3 day^{-1} lower, which lies at the lower edge of confidence intervals presented above. For other developments our approach to variability mattered less because differences in mean and median rates over multiple passes were not significant. In Peace River, we suspect that operational variability and topography both contributed to increased measurement variability.

Due to accessibility constraints, Peace River campaigns sampled >300 fewer sites than Lloydminster and Medicine Hat. In total, 37 plumes originating from emitting well pads in Peace River were enumerated from an average detection distance of 220 m, at an average MDL of 12.4 m^3 day^{-1}. Since 2014, operators in the affected regions of the Peace River area have implemented infrastructure improvements in response to AER regulatory requirements including tank top vapour recovery systems, flares for system upset and tank top volumes, additional compression to gather casing gas for injection into the gathering system, and capacity expansion for existing gas gathering systems. Beginning in 2015, the AER has reported substantial reductions in gas venting from oil and bitumen batteries, as documented on the Peace River Performance Dashboard of the AER website (Alberta Energy Regulator, 2018). Thus, regulatory action taken by

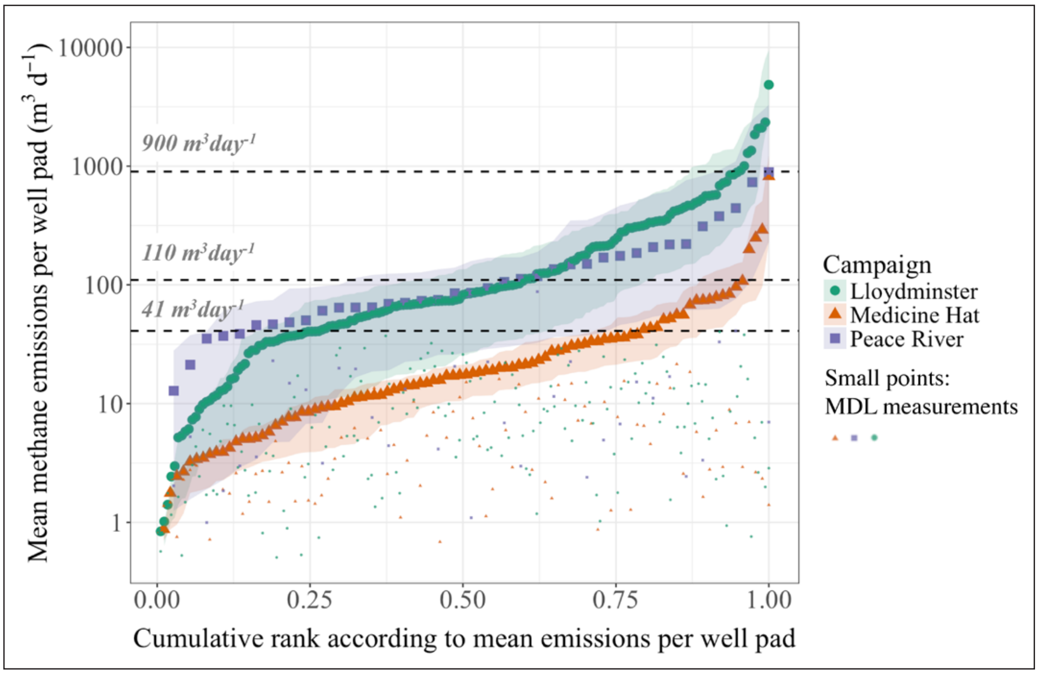

Figure 4: Mean emission rate per emitting well pad, colored by development. Regulatory thresholds of interest are the volumes in which affected infrastructure need to comply under existing and future federal venting regulations, and are labelled by dashed lines at 900 m^3 day^{-1}, 110 m^3 day^{-1}, and 41 m^3 day^{-1}. The small points are individual MDL measurements, coloured by campaign. The shadow for each campaign represents the range of possible emissions, based on maximum and minimum heights. Please note the logarithmic scale on the y-axis. DOI: https://doi.org/10.1525/elementa.341.f4

AER in Peace River over the past four years is presumed to be partially responsible for the relatively low incidence of hydrocarbon plumes within the region, as Bulletin 2014–17's recommendations focus on the reduction of venting and flaring in the area (Alberta Energy Regulator, 2014). AER's directive 084 released in early 2017 will have operators conserve at least 95 per cent of all solution gas (Alberta Energy Regulator, 2017b), and we expect emissions in this region to continually decline as new emission management practices are implemented. Therefore, we can infer from our results that increased historic scrutiny and more stringent regulation may have already contributed to the low $e\mathrm{CH}_4$ incidence in Peace River, demonstrating the effectiveness (and future potential) of regulation in improving environmental performance. Despite positive efforts in emission management by industry, Peace River also recorded the highest H_2S of all campaigns, as expected due to the sulfurous nature of the underlying deposits. Alberta's ambient air quality objectives for H_2S state that concentrations must not exceed a one-hour average of 10 ppbv, or 24-hour average of 3 ppbv (Alberta Government, 2017). Although our recorded values surpass such limits, we cannot classify this as an accurate exceedance due to the 'point in time' nature of mobile surveying, as opposed to stationary collection. However, our results do indicate a possibility of exceedances in the area, and further stationary monitoring is recommended to ensure compliance with ambient air quality guidelines.

Active wells were the predominant class of infrastructure surveyed in Medicine Hat, with 85.2% of all wells considered sampled classified as active. Favorable wind conditions and a dense development allowed us to survey 734 well pads 2 to 9 times each over the course of 15 surveys. Attributed plumes in this region had an average MDL of 5.50 m^3 day^{-1}. Infrastructure groupings containing only facilities were nearly three times more likely to emit than well-exclusive sites. Detected from an average distance of 138 m, 93 plumes that we attributed to sites upwind emitted methane at an average rate of 40.6 m^3 day^{-1} (95% CI: 21.8–59.8 m^3 day^{-1}, s = 93.8, range = 0.87–820 m^3 day^{-1}), with a 75th percentile of 35.8 m^3 day^{-1}. In Medicine Hat and adjacent regions, the majority of facilities do not use pneumatic pumps or controls, and wells generally have a pipe-in/pipe-out configuration with no surface facilities such as separators. As a result, there are limited emissions from pneumatic devices (GreenPath Energy Ltd., 2016), which in-part explains why Medicine Hat had the lowest plume incidence and magnitude amongst the three developments (**Figure 4**).

The mean emission intensity for each development fell near the range observed from seven previous independent ground-based studies at energy production sites across North America, where mean emissions intensity per-site determined from a central emission factor drawn from a probability distribution function specific to each region ranged from 42–258 m^3 day^{-1} (Zavala-Araiza et al., 2018). It should be noted that Lloydminster and Medicine Hat are on the high and low range of the distribution, respectively. To account for emissions that fall below our MDL, we have re-calculated development-wide means from a fitted lognormal distribution, following a similar procedure mentioned above by Zavala-Araiza et al. (2018) This process, explained further in SM-S3.3, enables us to more readily compare emissions across developments, as we apply the same lognormal fit assumptions across all campaigns. Due to the lack of an unbiased dataset upon which to compare our GDM estimates, the fitted distribution has not been corrected for a high-emitter bias. The derived development-wide emission rate means are summarized in SM-Table S1.

While OGI and mobile-based measurement have different sensitivities and operate on varied scales, we can draw qualitative comparisons with the 2016 Greenpath Alberta Fugitive and Vented Emissions Inventory Study, which occurred in nearby regions within three months of our mobile measurement campaign. Here, OGI detection was used for 676 inspections at 395 distinct facility locations across six geographical areas (GreenPath Energy Ltd., 2016). In the OGI study, an average of eight leaks or vents were visible via the thermal imaging cameras for every ten facilities inspected. In Bonnyville AB, a CHOPS heavy oil development north of Lloydminster, 225 leaks or vents were detected over 102 locations, 99% of which were from venting related sources. An emissions incidence in Bonnyville of 2.2, described as emissions detected/sites surveyed, is contrasted to the mere 0.11 observed in Medicine Hat, where 7 emissions were detected over 63 locations. Similar to our study, the CHOPS development has an emissions incidence several factors higher than the conventional gas development in Medicine Hat. Because the average OGI inspection takes ~2.7 hours per well-pad (ICF International, 2015), and an on-road survey can sweep several hundred pieces of infrastructure within a day (at MDL's as low as ~0.30 m^3 day^{-1}, depending on source proximity, emission strength, height, etc.), industry could experience significant time and cost savings if truck-based screening is adopted as a first-order monitoring technique, triggering OGI inspections only at identified sites. As the move from emission estimates (factors) to emission measurements becomes enforced, a multi-scale, triage approach will be fundamental to efficient monitoring on an operational scale.

Currently, the AER enforces solution gas flaring/venting conservation requirements on sites (defined as a single-surface lease where gas is flared or vented) that emit over 900 m^3 day^{-1} (Alberta Energy Regulator, 2016). Commencing in 2023, federal regulations will limit venting to ~41 m^3 day^{-1} (15,000 m^3 $year^{-1}$) on qualifying facilities that vent over 40,000 m^3 $year^{-1}$ (~110 m^3 day^{-1}) (Government of Canada, 2018). While this is not presently a legislated threshold, we can still consider the possible implications for mitigation. As regulations tighten, thousands of infrastructural retrofits will be required to meet conservation requirements. Of emitting well pads, 40.2% (95% CI: 32.2–49.9%), 43.2% (24.3–54.1%), and 4.3% (3.23–10.75%) of sites in Lloydminster, Peace River, and Medicine Hat respectively exceeded the 110 m^3 day^{-1} threshold to trigger emissions reduction compliance. Nine detected plumes in Lloydminster, or 5.2% of the total plumes, exceeded the current AER Directive 60 threshold for per-site emissions. We did not detect emission rate exceedances of 900 m^3 day^{-1} in Medicine Hat or

Peace River. All campaigns were characterized by a heavy tail emissions distribution, in which 74.8%, 57.6%, and 72.4% of cumulative emissions in Lloydminster, Peace River, and Medicine Hat respectively originated from 20% of emitting sites.

Considering both the emission frequencies and the volumetric distributions for each campaign, we extrapolated our results to estimate the approximate proportion of well pads that would be affected under the venting mitigation threshold of ~110 m^3 day^{-1} throughout a ~25,000 km^2 region within each development. Using these assumptions, we estimate 2,668 of 16,267; or 16.4% sites within the Lloydminster region, 418 of 3,433; or 12.2% sites within the Peace River region, and 283 of 51,016; or 0.55% of sites within the Medicine Hat region would be emitting over 110 m^3 day^{-1} according to our mobile surveying results. The estimated proportion of infrastructure that would require mitigation based on the 110 m^3 day^{-1} threshold is low in Medicine Hat, yet the infrastructure count is high, especially relative to the low rates of hydrocarbon production. Comparatively, a considerably larger proportion of sites in Lloydminster could be affected, yet the development is smaller and more productive per well. The Canadian Government predicts a total 7,590 sites across the country will be affected by the incoming venting regulations, which considers all existing and new facilities between 2018–2035 (Government of Canada, 2018). Applying our measured emissions distribution to the extrapolated region, our analysis above estimates approximately ~3,300 sites total could require mitigation in the Lloydminster, Peace River, and Medicine Hat regions alone. We do acknowledge that not all of the facilities in the above extrapolation region would meet the potential to emit criteria outlined in the Federal regulations which is a condition for venting restrictions to apply, but based on our results, it is likely that government estimates of affected infrastructure are conservative as a result of measured emissions being larger than those reported in government inventories. Quantifying the extent to which estimates are conservative would require emissions data recorded in additional developments across Canada and is beyond the scope of this paper. However, Canadian industry and policymakers are still assessing the impact of the incoming regulations, and in that context studies like this offer useful insight.

Conclusion

Patterns presented by different research groups working independently of one another show that Canadian-based studies thus far (emissions rates, frequencies, inventory validity by development style) are generally consistent regardless of methodology applied (GreenPath Energy Ltd., 2016; Atherton et al., 2017; Johnson et al., 2017; Zavala-Araiza et al., 2018), suggesting reasonable agreement between varying measurement approaches. In many developments, measured emissions are higher than emission factor-based inventory estimates, though discrepancies are development specific (Atherton et al., 2017; Johnson et al., 2017; Zavala-Araiza et al., 2018). Emissions varied largely by development, but were the most significant in terms of frequency and magnitude in the Lloydminster region. The relatively low incidence of hydrocarbon plumes detected in Peace River are presumed to be in part due to gas conservation regulations and resulting management practices implemented in recent years, thus demonstrating the effectiveness and potential of regulation in mitigating emissions.

The application of a gaussian dispersion model to continuous, mobile-sourced emissions measurement data was found to be an effective technique for flagging high emitting sites with potential regulatory exceedances, and for characterizing emissions occurrences, rates, and patterns. The large-scale efficiency benefits of the vehicle-based technique are in part offset by the increased uncertainty inherent with mobile techniques. Effective LDAR programs will involve additional technologies with greater specificity to refine emissions estimates and pin-point the exact emission on a well pad once a screening campaign is conducted. As the first large-scale mobile emissions study in Alberta, this work demonstrates the effectiveness of vehicle-based techniques as a first-order screening tool to meet regulatory leak detection and repair compliance. When properly combined with standard approaches such as OGI, the application of alternative LDAR technologies such as satellite, drone, mobile techniques, etc. pose promising time and economic advantages. This is important, as we suggest a greater number of sites then expected will be affected under the new regulations. Future ambient monitoring is needed to quantify the impact of the regulations on air quality once implemented, and this study has provided a development-specific baseline against which to compare.

Methane cannot be managed without the accurate collection of widespread, defensible measurements, and as additional data are collected in Alberta, a better statistical understanding of emission norms and variance will develop over time, leading to refined estimates of affected infrastructure and associated mitigation costs.

Data Accessibility Statement

Datasets produced in this work are available as online supplemental material accompanying this publication. Dataset S1 contains emission source heights used in the GDM. Dataset S2 contains a unique ID for each emitting well pad in which we calculated a volumetric emission rate, the mean emissions rate, the median emissions rate, MDL, standard deviation, standard error, and maximum and minimum volumetric emissions rates per individual well pads.

Supplemental files

The supplemental files for this article can be found as follows:

- **Dataset S1.** Infrastructure heights for GDM. DOI: https://doi.org/10.1525/elementa.341.s1
- **Dataset S2.** Volumetric emissions data. DOI: https://doi.org/10.1525/elementa.341.s2
- **Text S1–5.** Supplemental material. DOI: https://doi.org/10.1525/elementa.341.s3

Acknowledgements
The authors would like to thank Alex Marshall and Tara Hanlon for the help with data collection, and Altus Geomatics for the GIS database assistance.

Funding information
Funding for this work was provided by the Petroleum Technology Alliance of Canada (Alberta Upstream Petroleum Research Fund) and the Atlantic Canada Opportunities Agency (Atlantic Innovation Fund).

Competing interests
The authors declare no competing financial interests. The research presented in this manuscript was exclusively done under the contracts to the authors' academic affiliations. There is no overlap or relationship between the funding organizations of this study, and any work that authors EO, DR, and EA have been conducting with their private company, Arolytics Inc.

Author contributions
- Contributed to conception and design: EO, DR, EA
- Contributed to acquisition of data: EO, JB, DL, JJ
- Contributed to analysis and interpretation of data: EO, DR, EA, EB, CF, DL, JJ
- Drafted and/or revised the article: EO, DR, EA, EB, CF, JB, DL, JJ
- Approved the submitted version for publication: EO, DR, EA, EB, CF, JB, DL, JJ

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How to cite this article: O'Connell, E, Risk, D, Atherton, E, Bourlon, E, Fougère, C, Baillie, J, Lowry, D and Johnson, J. 2019. Methane emissions from contrasting production regions within Alberta, Canada: Implications under incoming federal methane regulations. *Elem Sci Anth*, 7: 3. DOI: https://doi.org/10.1525/elementa.341

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Knowledge Domain: Atmospheric Science

Part of an *Elementa* Forum: Oil and Natural Gas Development: Air Quality, Climate Science, and Policy

Submitted: 31 May 2018 **Accepted:** 18 November 2018 **Published:** 03 January 2019

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Science of the Anthropocene

Elem Sci Anth is a peer-reviewed open access journal published by University of California Press.

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