Onshore oil and gas: quantifying whole-site methane emissions and associated uncertainties

Chief Scientist's Group research report

November 2022

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Dr Robert Bradburne
Chief Scientist

Executive summary

Methane emissions from onshore oil and gas (OOG) sites and facilities contribute towards the UK inventory of methane emissions, and these could be significant if not well controlled. Understanding and quantifying whole-site emissions is important in terms of prioritising action to reduce methane emissions.

This report reviews methods for quantifying whole-site emissions from OOG sites regulated by the Environment Agency. It considers the wide range of approaches available to determine the combined plume of emissions from all sources within an OOG site, determines the methods best suited for different types of OOG sites, and presents information on the uncertainties associated with quantifying emissions for these selected methods.

A taxonomy of methods for detecting and quantifying whole-site methane emissions for OOG facilities has been determined. After assessing the available methods against important criteria, the following 5 methods were selected for more detailed consideration:

- plume-based flux recovery (US EPA Other Test Method 33a (OTM33a) - Geospatial Measurement of Air Pollution (GMAP))
- component-level measurements
- mass balance
- fenceline monitoring
- tracer method

The report introduces several concepts in uncertainties as part of the detailed assessment. Method uncertainties are outlined for the 5 selected methods, with evidence drawn from studies in the literature. In addition to method uncertainty, additional considerations relating to measurement campaign uncertainties are also presented in detail, including issues related to sampling, emission type uncertainty, changes over time (temporal variability), and method implementation uncertainty. Uncertainty caused by the representativeness of the sampling period is also discussed, in particular where the measurement period may, or may not, capture upset conditions which can bias annual emissions high or low, respectively.

The assessment of methods and related uncertainties have been applied to 4 different site types which represent OOG activities in England. A method selection table (see Table ES1) identifies the preferred methods for use at different site types. Indicative information on costs is also included to help determine the optimum approach to meet the purposes for which whole-site methane emissions quantification is being carried out.

Table ES1: Method selection table

Site type	Description of location	Potential emission sources	Preferred methods						Prohibitively uncertain methods
				1	2	3	4	5	
Small production site, complex topography	Wooded with complex aerodynamics	Relatively complex with some processing	**Method**	Tracer method	Mass balance	OTM33a	Component-level measurement		Fenceline measurement
			Method Uncertainty	2	2	1	1		
			Cost	£	££-£££	£	£		
Small production site, simple topography	Open setting	Wellheads, separator and condensate tanks only	**Method**	OTM33a	Component-level measurement	Tracer method	Fenceline measurement	Mass balance	
			Method Uncertainty	1	1	2	3	2	
			Cost	£	£	£	£	££-£££	
Large production site, complex topography	Wooded with complex aerodynamics	Multiple wellheads, on-site processing	**Method**	Tracer method	Mass balance	OTM33a			Fenceline and component-level measurement
			Method Uncertainty	2	2	1			
			Cost	££	££-£££	£			
Large processing sites	Very large and complex site. More open setting but aero-dynamically complex topography	Large number of individual sources	**Method**	Tracer method	OTM33a	Mass balance	Fenceline measurement		Component-level measurement
			Method Uncertainty	2	1	2	3		
			Cost	££	£	££	£-££		

Uncertainty key

Uncertainty range colour code	Uncertainty bounds
	<±20%
	±20% to ±50%
	>±50%

Uncertainty approach code	Approach used to determine uncertainty bounds
1	Controlled release with published data
2	Published, desk-based analysis
3	No data – Analysis through modelling

The uncertainty in an annual emission estimate depends on (a) the uncertainty in measurements and inverse modelling during a monitoring period/campaign, and (b) the uncertainty in how representative a monitoring period/campaign is of the overall annual pattern of emissions. Emissions from OOG operations can be both sporadic and large; consequently, methods that approximately quantify sporadic large emissions may provide a more representative estimate of total site emissions than methods that are precise but fail to capture sporadic large emissions. Therefore, approximate monitoring over 100% of the year may generate less uncertainty in the annual emissions estimate than very accurate monitoring for shorter periods of time. It follows that having site activity data during the monitoring is essential for establishing the representativeness of monitoring campaign data.

Main recommendations for quantifying whole-site methane emissions include the following:

- Ensure that controlled releases, in as realistic an environment as possible, are used to characterise method detection limits and quantification accuracy.
- Assess total uncertainty using empirical predictive power calculations that account for the non-normal distribution of facility emissions and, where applicable, skew in the uncertainty of the method.
- Standardise application of methods by developing and using method standards, implemented by sufficiently experienced personnel.
- Consider combining lower-cost higher-uncertainty methods with higher-cost lower-uncertainty methods to capture large emitting events, while also understanding the contribution from all other sources at a facility.
- Evaluate new and emerging methods.

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1. Introduction

Methane is an important greenhouse gas with a high global warming potential. Consequently, emissions of methane need to be substantially reduced to address the climate emergency and enable the UK to meet its net zero carbon emission commitments. Emissions from onshore oil and gas (OOG) sites and facilities contribute towards the UK inventory of methane emissions, and these could be significant if not well controlled.

Understanding and quantifying the contribution from these sites is important in terms of prioritising action to reduce methane emissions. Furthermore, since emissions from different individual pieces of equipment within a site can vary substantially on a temporal basis, whole-site emissions (often otherwise referred to as ‘full facility emissions’ or ‘facility-wide emissions’) need to be understood.

The term ‘whole-site emissions’ refers to all emissions from a specific geographic area, for example, from all point, area and line sources within the perimeter fence of a site. Following the definition in EN 17628 (CEN, 2022), there are 4 types of emissions:

- accounted channelled emissions (from stack emission points)
- unaccounted channelled emissions (for example, from vents)
- fugitive emissions (leaks, for example, from valves, seals)
- area emissions (for example, from water treatment basins)

Emissions may arise from intended engineered sources (for example, via stacks) and unintended sources (for example, leaks, fugitive discharges, process upsets), as well as occasional intended discharges during non-routine operations (for example, flares, maintenance activities). Whole-site emissions can be summarised over a formal accounting period, for example, annual emissions.

This report provides a review of suitable methods for quantifying whole-site emissions from OOG sites for which the Environment Agency has a regulatory remit (Environment Agency 2020). These sites include wells, central gathering stations, gas processing sites and compressor stations (Finlayson and others, 2021) and they constitute the ‘upstream’ and ‘midstream’ sectors of the oil and gas industry. They are sites where oil and gas are extracted from geological reservoirs and/or contain the infrastructure for initially gathering and processing it. Refineries, that is, the ‘downstream’ sector, are substantially larger and more complex facilities with more potentially very different emissions profiles and subject to more specific regulatory requirements. These were not considered in this study.

More specifically this report involves:

- reviewing the wide range of approaches available to quantify the combined plume of emissions from all sources within an OOG site
- selecting methods best suited for OOG sites
- understanding the uncertainties associated with these selected methods

Understanding the total uncertainty of quantification methods will help determine whether temporal changes at a site (for example, the effectiveness of control measures) are real and meaningful, as well as evaluating the emissions performance (and therefore importance) of different sites in comparison to each other. Uncertainty analysis also helps to target regulatory resources at those sources with the greatest potential emissions. In other words, it can support efficient and proportionate regulation.

This report builds on a previous Environment Agency scoping study of quantifying OOG whole-site methane emissions (Finlayson and others, 2021) which included a limited consideration of uncertainty. Here, currently available methods of quantifying OOG methane emissions and their component uncertainties are assessed to derive the combined, or total, uncertainty for whole-site methane emission rates for a particular type of facility and quantification method. A more detailed understanding of uncertainty allows the effectiveness of different quantification methods to be compared and a selection approach to be developed that can be used to identify the most suitable quantification method in different circumstances.

A ranking of the most suitable methods for different types of OOG facilities, along with the expected accuracy (the total method uncertainty) and costs of a particular method for a particular facility type, has been developed. This allows the most effective method for quantifying emissions from an OOG facility to be selected, which, as discussed in Finlayson and others (2021), may be a scoping, monitoring or research study. Furthermore, the required period over which emissions are being quantified (for example, ‘snapshot’ versus annual) or population size (single facility versus multiple facilities) may require sampling uncertainty and temporal variability to be considered.

1.1. Report structure

This report is structured as follows:

Section 2 lists the main characteristics of OOG facilities that will determine whether a particular quantification method should be considered. Available methods that could potentially be used for quantifying OOG whole-site methane emissions are then reviewed in section 3. The multiple criteria used to select 5 methods that are most suitable to quantify whole-site methane emissions from English OOG facilities are discussed in section 4.

The method uncertainties of the 5 selected methods for detailed consideration are discussed in section 5, while a general discussion of other sources of uncertainty and how they contribute to total uncertainty for these methods is provided in section 6. These underlying analyses and reviews are used to compile the method selection table. How this can be used for different types of facilities, while satisfying the particular quantification requirements, is then described in section 7.

Conclusions and recommendations, including suggested requirements for future standards for quantifying methane emissions from OOG facilities, are given in section 8, along with

comments on the transferability of the methods considered here to other sources and pollutants.

Details on relevant reference studies for each method, including where they were applied and the approaches to uncertainty, are given in the Appendix. A glossary of main terms is also provided.

2. Onshore oil and gas sites in England

There are 4 main types of OOG sites within England:

- **Small production sites located in aerodynamically complex topography areas:** Site size of up to 200 m x 200 m. As well as wellheads, separators and condensate tanks may include some processing equipment. Surrounded by trees up to 30 m tall.
- **Small production sites located in aerodynamically simple topography areas:** Smaller site size (typically 80 m x 40 m) and smaller configuration (typically wellheads, separators and condensate tanks) compared to small production sites in wooded areas. Surrounded by open land.
- **Large production sites located in aerodynamically complex topography areas:** Site size of 500 m x 300 m containing multiple wellheads and extensive on-site processing equipment. Surrounded by trees up to 30 m tall.
- **Large processing sites located in aerodynamically simple topography areas:** Site size of up to 1 km x 1 km with a collection of multiple individual sources of different heights. Includes gas pipeline terminals and large compressor stations. These sites are usually surrounded by open land.

In addition to facility size, the type of processing on a facility has an impact on emission sources, and on suitable emissions measurement methods. For small facilities, if there is liquid separation (for example, well field separators) or treating equipment (for example, dehydrators) on a facility (for example, a well pad), process equipment failures are likely to cause the largest emissions, and typically constitute the majority of emissions. When these types of equipment are not present at a facility (for example, at metering and regulation stations, distribution metering stations), then processes on the facility are less complex, and emissions tend to be caused by leaks in pressurised equipment or control equipment failures (for example, a valve open when it should be closed). Additionally, studies have found that the process equipment failures tend to produce larger emissions and are often more difficult to measure (for example, compare the well pad study by Allen and others (2013) to the Lamb and others (2015) study of distribution system emissions, both of which used similar measurement methods). The nature of emissions (magnitude and temporal variability) will depend on the number, size and type of sources at each type of site.

The physical environment in and around a facility also impacts the applicability of methods that rely on dispersion estimates or stationary sensors. Whether a facility is in a wooded setting or not has significant implications for the complexity of the aerodynamics of plume releases from a site, and whether these can be easily measured. Additionally, the effective height of the emission source, gas composition and micrometeorology are important for quantifying emissions. Methane enhancements caused by near-ground-level emission sources are more likely to be detected by sensors on the ground than emissions from sources higher up, as the plumes from elevated sources may ‘overfly’ ground-level monitors. Therefore, relatively inexpensive ground-level monitors may be suited to sites

with lower emission sources, while sites with a range of evaluated sources may require an approach that can measure at a range of heights.

The Environment Agency 2020 Pollution Inventory (PI) was used to identify typical methane emissions from regulated oil and gas facilities within England. Excluding refineries, there are 39 OOG facilities in the PI data. This is not an exhaustive list of all the OOG facilities in England due to reporting thresholds for the size of the activity and for the emissions released annually. Of these 39 sites, 18 sites reported that annual methane emissions were below the 10,000 kg/yr reporting threshold and so we assume annual emissions for these sites were 5,000 kg as an average value between zero and the reporting threshold. These 18 sites have all been identified as small well sites or crude oil terminals. As such, the emissions profile for the small sites is likely to be biased. Oil and Gas Authority (OGA) 2020 production data has been used to identify the typical production of oil and gas from onshore oil and gas facilities. The OGA production data includes 72 facilities, however 20 reported no production and, as such, were potentially not operational in 2020. It is important to note that processing sites are not included within the OGA production data. Table 2.1 presents the average reported annual emissions plus oil and gas production averaged by site category.

Table 2.1: Average annual methane emissions, oil production and gas production for 2020 from OOG sites in England

Site type	Average methane emissions (kg)	Average oil produced (t)	Average gas produced (t)
Small (with both complex and simple aerodynamic topography)	32,153	3,531	1,971
Large production sites	46,220	600,306	0
Large processing sites	379,783	n/a	n/a

3. Review of available methane quantification methods

This section considers classifications of methods that can detect an emissions plume from a facility and provides a brief summary of all existing methods that could potentially be used for quantifying OOG whole-site methane emissions. These methods primarily use downwind measurements of the gas concentration combined with modelling to estimate emissions. In general, modelling approaches either: (i) simulate the 3-dimensional dispersion when an emitted gas is entrained into the ambient airflow, or (ii) scale from known on-site emission rates. In addition, we consider component-level measurement which involves detecting and measuring individual component emissions at a facility and uses these measurements to infer the total mass of gas emitted from the site.

While there are overlaps with leak detection and repair (LDAR) programmes and requirements, particularly with respect to detecting leaks, the focus here is on the direct measurement of all emissions from all leaks and any other sources of emissions at an individual facility, rather than using standard ‘look-up’ tables of average emission rates from leaking components or other types of emission factors.

Available methods have been grouped as follows:

- Survey methods with tracer gases
    - whole-site tracer flux
- Survey methods without tracer gases
    - plume-based flux recovery
    - mass balance
    - solar occultation flux (SOF)
    - plume imaging and quantification
- Continuous monitoring methods
    - fenceline monitors (point and line sensors)
    - eddy covariance
- Component-level measurements
    - leak detection and high flow sampling

Most of the methods described observe the increase in methane emitted from a site when compared to any background emissions in the ambient upwind airstream. In part, the accuracy of the emission estimate depends on the accuracy of background measurements and an ideal background concentration would be as low and homogeneous as possible when compared to the site emission, so the site-to-background noise ratio is as high as possible.

Note that this study did not consider methods which are primarily intended to quantify an individual, typically large, source (or small cluster of sources) within a facility by identifying a distinct emissions plume and then estimating emissions represented in that plume. In

general, those types of method identify large emitters but do not estimate full facility emissions.

A more detailed listing of the main studies using each method, where they were applied and the approaches to uncertainty, is given in Table A1 in the Appendix. The approaches used to derive the uncertainty estimate have been used to determine the level of confidence in the uncertainty bounds. The quantification methods with uncertainty bounds generated by controlled release, with matching measurement data published, are of the highest standard. Published uncertainty analyses that were generated by desk-based analysis gives some confidence to the uncertainty bounds, while methods without any published analysis give little indication that the uncertainty bounds are realistic.

3.1. Approaches for detecting the emissions plume from a facility

All methods in this study (except component-level measurements) measure emissions by identifying volumetric mixing ratios (also termed ‘concentrations’) of emitted gases that are elevated above the background mixing ratio of the gas being measured (in this case, methane). Elevated mixing ratios, typically termed ‘enhancements’, are the result of the transport and dispersion of emissions from the target facility. Since facilities are often in regions with other sources, one important assumption is that sources upwind of the facility are sufficiently distant such that emissions from those sources have either (a) blended into the background or (b) can be identified, measured and subtracted from total emissions seen downwind of the target facility.

Figures 3.1 to 3.5 provide an illustration of the primary methods for detecting emission plumes that are included in this study. Figure 3.1 illustrates the general layout of sources and plumes for the subsequent Figures 3.2 to 3.5. Equipment within the loop road centred in the figure represents the ‘target facility’; equipment to the left is upwind of the facility. The goal of all methods is to measure sources from the target facility without including plumes from nearby sources.

Figure 3.1: Example facility showing idealised emissions from different sources at a facility plus sources upstream of the facility

Figures 3.2 to 3.5 show different sensing approaches used in estimating facility emissions. These approaches involve either continuous monitors (permanently installed systems making regular measurements over an extended period of time) and/or survey methods (intensive measurements made during short campaigns on a single or periodic basis).

Figure 3.2: Continuous monitors: Using point sensors to measure concentrations in or near the emissions plume from a facility

The point sensors (also known as fenceline monitors) are shown as black poles. The sensor returns a time series of concentrations from each sensor location. Sample frequency varies from ~1 Hz to averages over several minutes. Wind direction data is typically used to position sensors downwind of equipment for predominant wind directions.

Figure 3.3: Continuous monitors or survey methods: Using line (laser) sensors (yellow lines) to measure concentrations in or near the emissions plume from a facility

Sensors may be fixed in one line-of-sight or be gimballed to target multiple retroreflectors. The laser sensor returns a time series of path-integrated concentration from the laser source round trip to the retroreflector. Sample frequency depends on whether the system is gimbaled, and the number of retroreflectors sampled.

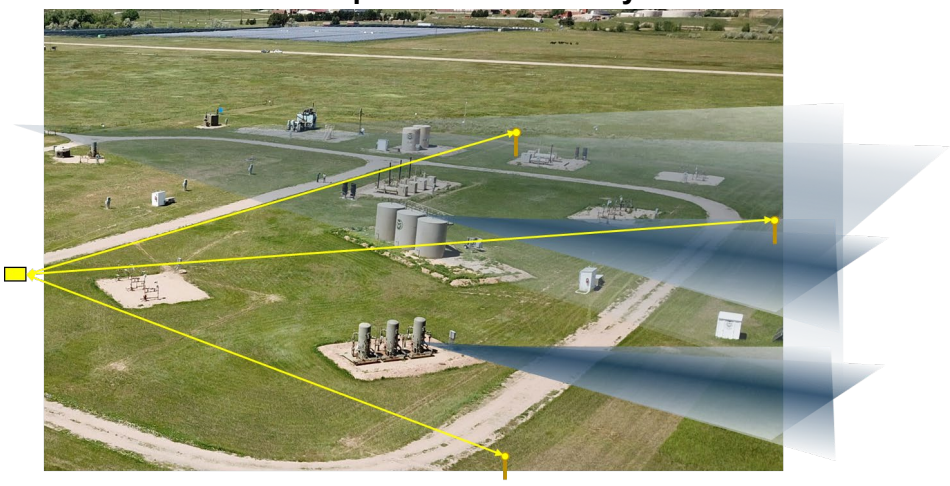


Figure 3.4: Survey methods: Using a concentration survey of the emissions plume from a facility

A high-sensitivity gas analyser is combined with a means of moving it along a path through the plume (red line). While the figure shows ground-level transport of the sensor, aircraft or drones can be used to transport the sensor around the site, above ground level. The sensor returns a geo-located time series of concentrations. Sample frequency is typically fast – 0.5 Hz or faster.

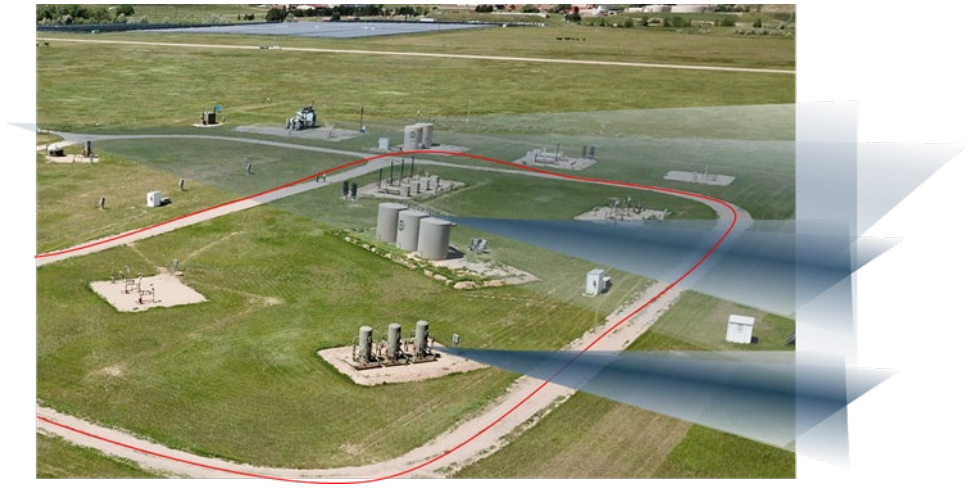
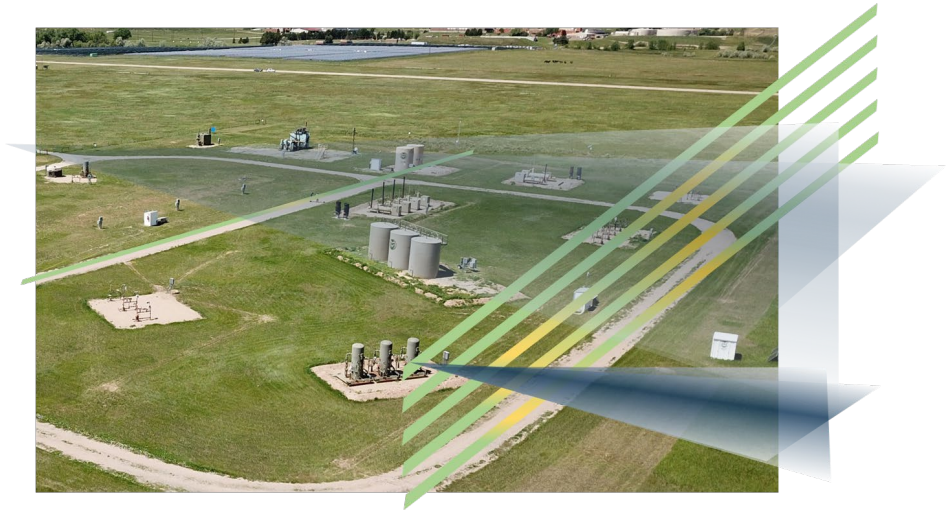


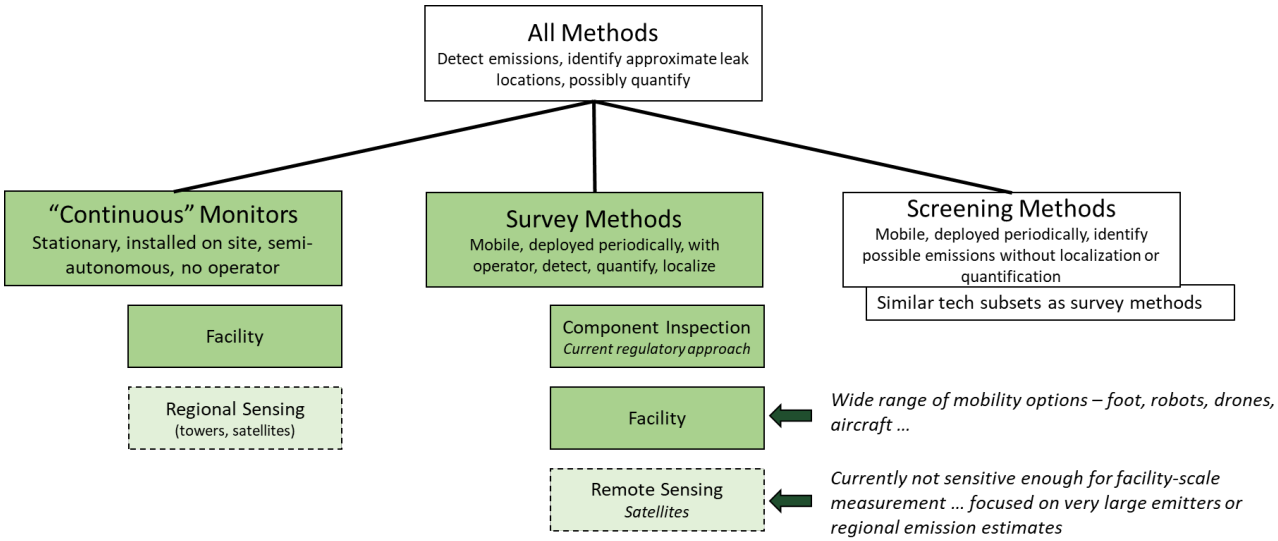
Figure 3.5: Survey methods: Using a flux plane method to survey the emissions plume from a facility

A trace gas analyser carried on a drone or aircraft flies through the plume at multiple levels (represented by light brown lines). Sensing equipment and sample frequency is similar to Figure 3.4. Flight lines should be long enough to pass outside the plume on each transect, and flight height should be sufficient to collect at least one transect with no enhancement.



A taxonomy of methods for detecting and quantifying whole-site methane emissions for OOG facilities is presented in Figure 3.6. Both continuous monitors and survey methods are suitable for quantifying whole-site emissions, and details of specific implementations are discussed in the following subsections.

Figure 3.6: A taxonomy of methane detection and quantification methods



3.2. Survey methods with tracer gases

3.2.1. Whole-site tracer gas methods

This method involves releasing one or two 'tracer gases' at known rate, most usually nitrous oxide or acetylene[1], close to points where emissions are known or expected. Typically, the emission will be identified using an optical gas imaging (OGI) camera and the source bookended by the 2 tracer gases, and the flow rates of the tracer gases are optimised for downwind detection. Concentrations of the tracer gas and target gas are then measured downwind while moving perpendicular to the plume. The release rate of the target gas is inferred from the known release rate of the tracer gas, as the ratio of the gases downwind is equal to the ratio at source (Herndon and others, 2013, Roscioli and others, 2015). Ideally, the types of gas used as tracers are distinct from the gases emitted from the source being quantified, so that tracer and source gases can be readily differentiated. Also, dual tracers can be placed at either side of the source (that is, not completely co-located) and the shapes of the tracer and source plumes can be used to increase confidence that the targeted source is being measured. This method requires mobile access perpendicular to the wind direction between 200 m and 1500 m downwind from the site such that the sensor can be moved across the plume, and measurement equipment capable of measuring at least 2 gas species. Currently, required instrumentation is too heavy for drones and requires road access downwind. However, future instrumentation development could reduce weight sufficiently for deployment on drones, reducing the need for road access downwind of the facility.

[1] Past practice also used sulphur hexafluoride (SF_6) as a tracer, but the measurement community is moving away from this type of tracer due to its long atmospheric life and climate impact.

3.3. Survey methods without tracer gases

3.3.1. Plume-based flux recovery

Rather than use tracer gas concentration measurements to indicate plume dispersion, it is possible to model the dispersion, and combine this information with the downwind concentration enhancement measurements of the emitted gas to give emission estimates. Here, we define the enhancement as the measured downwind concentration minus the measured upwind concentration.

For this method data is gathered in 2 steps. Practitioners drive through/around a facility using a high-speed, high accuracy, trace gas analyser to identify enhancements in the concentration of the desired pollutant. Once a plume is identified, the practitioner places the analyser downwind of the emission source – as centred as possible in the plume – and records wind and mixing ratio data for approximately 20 minutes.

Gaussian or Lagrangian approaches to modelling plume dispersion are both options. Using a Gaussian plume model typically requires good characterisation of micro-meteorology to obtain reliable estimates of plume dispersion. This is particularly challenging in areas with obstructions to the air flow, and a number of assumptions may need to be built into the modelling, for example, surface roughness and stability of air.

A backward Lagrangian stochastic model, for example, WindTrax (Crenna 2012), can also be used to characterise plume dispersion, and estimate emissions from downwind concentration measurements. This particular model is not constrained by the shape or size of the source, and it can model emissions from point sources, line sources or area sources. However, it relies on an idealised calculation of atmospheric dispersion and can result in inaccurate emissions under light wind conditions and both extremely stable and unstable stratification (Gao and others, 2008).

Either of these modelling options for assessing plume dispersion can be combined with a variety of different approaches for measuring downwind concentrations. In each case, the effects of methane emissions further upwind must be considered, and the enhancement caused by the source should be large compared to the background. Ideally, fieldwork should be planned during dispersion conditions that result in only small and homogeneous background methane concentrations due to upwind emissions.

3.3.2. Mass balance methods

The objective of a mass balance method is to spatially characterise the incoming and outgoing concentration of the target gas. Spatially resolved concentrations can then be combined with wind data to compute the transport of the target gas into and out of the facility. Concentrations are measured using trace gas analysers mounted on aircraft or drones. Two methods are commonly used:

- An aircraft/drone flies a plane upwind and downwind of the facility, building up a concentration map between the ground and the aircraft, from as close to ground level as possible to above the plume height. This flight pattern is often termed the

'flux plane' method. Differential Absorption LiDAR (DIAL) can also be a flux-plane method if it is used to 'sweep' through successive vertical sections across the wind, for example, upwind and downwind of a source.

or

- The aircraft/drone flies in a spiral pattern around a facility to measure concentrations both upwind and downwind, from as close to ground level to above plume height. This flight pattern is often termed 'spiral flight'.

With either flight plan, mass balance is then used to determine the target gas flux, using the concentration of the emitted gas and the normal of the wind across the flight path. Only aggregate air movement and meteorology is required.

For manned aircraft approaches, there is appreciable risk that some portion of the emissions may pass below the lowest aircraft flight line; typically, concentrations measured in lower flight lines are extrapolated to ground level using some assumptions.

3.3.3. Solar occultation flux (SOF)

Absorption spectra of a plume is measured using the sun as the infrared source. As for tracer flux or mass balance methods, mobile access is required to transect the plume downwind of the facility while instruments follow the direction of the sun. Wind measurements must also be made, which along with the measured concentration, are used to calculate the emission rate. Current solar occultation instruments have difficulty resolving enhancements of main greenhouse gases, notably methane, and work better with volatile organic compounds (VOCs) that have lower atmospheric concentrations.

3.3.4. Plume imaging and quantification methods

Thermal infrared imaging spectrometers are used to map target gas plumes. The imaging cameras are filtered to specific wavelengths applicable to the target gas. Several plume-based modelling methods are then used to quantify the emission rate (Thorpe and others, 2016). Imaging spectrometers can be fixed to drones and aircraft, and even satellite data can be used in this way, although for the latter resolution can be an issue. There has also been some development of handheld methods which work with data from optical gas imaging cameras to estimate emissions rate.

If one emitter dominates the emissions at a facility, estimates of the plume emission rate may be reflective of whole-site emissions, but generally these methods do not produce whole-site emissions estimates, and are not included in the method uncertainty discussion in section 5.

3.4. Continuous/autonomous monitoring methods

3.4.1. Fenceline fixed-position monitoring methods

This method is relatively new, and as such, there are much less data available. However, this is considered a rapidly developing, highly promising method due to low cost and long-duration observation periods. Sensors are fixed, for example, to poles or similar structures

at the fenceline of the facility, and are paired with analytics that convert sensor readings into emissions detections. Concentrations are measured more or less continuously, albeit with substantial uncertainty. These sensors are positioned so emissions pass through or around the sensor and the concentration is determined at a point or along a line. Emissions are then calculated using an assumption of plume dispersion (as for plume-based flux recovery) but with different assumptions. Since the sensors cannot move, algorithms must assume the location of the plume, which increases uncertainty, but are also able to calculate a longer time series of emissions, which generally reduces uncertainty. A principal challenge for fenceline methods is the trade-off between the cost, sensitivity, and the number of sensors.

At the time of writing, fenceline continuous monitors are under rapid development and performance testing is underway[2].

3.4.2. Eddy covariance methods

Eddy covariance is a micrometeorology technique for calculating surface emissions. Concentrations and meteorology, giving the vertical flux of the target gas, are measured at a single point downwind of the source (Coates and others, 2017). Dispersion modelling, for example, Gaussian plume or backward Lagrangian stochastic, is then used to determine the release rate. Location of sources and a background flux measurement are required for this technique, and the monitors used for concentration measurements require a relatively high response rate (~3 Hz). Therefore, the approach is more commonly used in research studies rather than permanent monitoring.

3.5. Component source quantification methods

Rather than determining whole-site emissions by identifying and measuring the aggregated plume of emissions from all sources, emissions can be quantified at the component level and then aggregated to derive the whole-site emissions. This involves identifying and measuring emissions from all equipment types, typically in close proximity to each individual source. Component source quantification methods are unlikely to be suitable for calculating whole-site emissions from larger sites. However, these methods may be the most accurate and cost effective for small sites with no liquid separation or gas treatment on site. In these cases, component-level sources will primarily consist of leaks from flanges, valves and connectors.

3.5.1. Leak detection methods for component sensors

Traditionally, leak detection was done using US EPA Method 21 (US EPA 1990, 2010) and numerous variants of this, such as EN 15446 (CEN 2008), which uses a portable probe to directly inspect each component. This is a sensitive and relatively precise method but is resource intensive (Zimmerle and others, 2020b). More recently, optical gas imaging

[2] For example: https://energy.colostate.edu/metec/aded/

(OGI) has become widely used for leak detection from oil and gas facilities. OGI uses imagers filtered to mid-infrared wavelengths to image gas plumes and to target leaks (Zimmerle and others, 2020b). OGI has also been used to additionally quantify emissions from component sources using a response factor (Zeng and others, 2017). However, quantifying emissions using OGI is still very much in the development phase, because the difference between the temperature of the leak seen and of the background, which is the crucial factor for visualisation, also has a large influence on measurement uncertainty as it affects how many pixels the OGI camera can see.

3.5.2. High flow sampling

High flow samplers for measuring methane have been on the market since 2001. While initially developed for natural gas streams where methane was the predominant component, it is increasingly used on other methane containing gas streams. High flow samplers operate by drawing in a high volumetric rate of air, entraining the target emission (leak) and then measuring both the emissions concentration and total flow rate of the air and entrained target gas. Background measurement is also made and subtracted from the sample measurements. While the method can be used for any gas species, currently available production instruments measure only hydrocarbons, and are generally not methane-specific.

Since high flow instruments have upper limits on the size of emissions that can be measured, most research teams using these instruments also have available backup methods to capture and measure emissions such as static/dynamic flux chambers.

4. Selecting methane quantification methods for detailed consideration

The full set of available methane quantification methods (Section 3) were reviewed to select methods that are most suitable for the types of OOG sites in England.

Two main criteria were used to select the most suitable methods:

1. They either:

    a. have been tested substantially by use in scientific studies (for example, tracer flux, plume-based flux recovery, component-level measurements) and may also have been approved by a major regulator (for example, plume-based flux recovery, component-level measurements)
    b. are rapidly gaining traction for use in whole-site measurements (for example, flux plane, fenceline monitors)

2. They can quantify whole-site methane emissions

The latter criterion implicitly excludes aircraft imaging methods, which can only measure individual plume(s) from a facility, and satellites, which have too low resolution and too high a lower detection limit (LDL) to resolve emissions from individual facilities. This criterion also eliminates methods which are just screening methods (for example, OGI) that only detect the presence of emissions, but cannot reliably quantify emissions.

Two additional criteria were considered:

- Whether the method has been proven for OOG operations, which excludes eddy covariance.
- Whether the method has been implemented widely, that is, by more than one vendor or academic group, and so would be suitable for widescale deployment. This criterion excludes Bayesian convergence surveys and solar occultation flux (SOF).

Qualitative evaluation of each of the methods described in section 3 against the main and additional criteria is shown in Table 4.1.

Table 4.1: Qualitative evaluation against main and additional criteria to select methods for detailed consideration

		Whole-site tracer flux	Plume-based flux recovery	Flux plane methods	Plume imaging & quantification	Solar occultation flux	Bayesian convergence survey	Fenceline monitors (point & line sensors)	Eddy covariance	Backward Lagrangian stochastic	Component-level measurements
Main criteria	Tested substantially in a number of studies?										
	Approved by a major regulator?										
	Suitable for whole-site emissions?										
	Proven for OOG operations?										
	Implemented by more than one vendor or academic group?										
Additional criteria	Controlled release validation?										
	Speed of method?										
	Ease of use?										
	Suitable for small facilities?										
	Suitable for larger facilities?										
	Access restrictions?										
	Suitable for a complex fetch?										
	Suitable for sites surrounded by woodland?										
	Suitable for continuous measurement?										
	Summary - Recommended for detailed study?	**Yes**	**Yes**	**Yes**	No	No	No	**Yes**	No	No	**Yes**

Key: **Green – Highly/very suitable,** **Amber – Intermediate,** **Red – Poor/not suitable**

The 5 classes of methods that then best meet the selection criteria are:

- plume-based flux recovery (US EPA Other Test Method 33a (OTM33a), see Thoma and Squire, 2014)
- component-level measurements
- mass balance
- fenceline monitoring
- tracer method

While not all of these methods would apply to every type of OOG facility, the overall range of methods ensures that at least one method can be expected to be effective for each type of OOG facility. Furthermore, these methods are a mix of continuous monitoring and survey (campaign) methods, therefore providing the capability to make more accurate measurements over short intervals or to understand variations over long periods, depending on a study's particular needs and goals.

In practice, methods could be combined, where lower-cost higher-uncertainty methods (for example, OTM33a) are used frequently and calibrated against higher-cost lower-uncertainty methods (for example, tracer methods) less frequently. This would supply a greater amount of data that could be used to infer any temporal variability in emissions, while helping to reduce costs of measurement. This does not remove the relatively high uncertainty in the OTM33a measurements but gives more confidence in the method's ability to quantify emissions. Additionally, lower-cost, higher-uncertainty methods could be used for an initial screening, with escalation to higher-cost, lower-uncertainty methods if warranted by the initial results, including both the absolute level and uncertainty of those results.

For each of the 5 selected classes of methods, a detailed assessment of the component and total uncertainties for the most common implementation is made in section 5.

5. Uncertainty of selected methods

Uncertainties specific to each of the selected methods are discussed in this section, while other sources of uncertainty are discussed in section 6. A value for the total method uncertainty for each method, applicable in different circumstances, is then derived. To help compare methods (see section 7), the method uncertainties have been grouped as those with an uncertainty less than ±20%; between ±20% and ±50%; or greater than ±50%. Additionally, the approach used to derive the method uncertainty estimate is included. These approaches are:

1. Those generated by analysing published controlled release data, specifically experiments where the method was used to quantify known releases multiple times, at multiple emission rates, in realistic, or at least pseudo-realistic, environmental conditions. This is the ideal approach that gives high confidence in the method uncertainty range.
2. The uncertainty analysis was generated by desk-based analysis and has been published – this approach gives limited confidence in the method uncertainty range.
3. No data on uncertainty bounds were available and uncertainty analysis has been carried out through modelling – the method uncertainty range is consequently highly uncertain.

5.1. Plume-based flux recovery (OTM33a)

5.1.1. Description

Methods used to quantify above ground, point-source emissions typical of emissions from OOG facilities using plume-based flux recovery have been assembled into the US EPA 'Other Test Method 33a' (OTM33a), which is a defined US EPA method for the geospatial measurement of air pollution (GMAP; see https://www.epa.gov/emc/emc-other-test-methods). Thoma and Squire (2014) describe the method including suitable instrumentation, required quality control procedures, appropriate environmental conditions, and suitable source types. In short, the method is implemented in 2 steps:

- Practitioners drive through/around a facility using a high-speed, high accuracy, trace gas analyser to identify elevated mixing ratios ('plumes') of the desired pollutant, commonly called 'enhancements'. These enhancements correspond to increases above the concentrations in background air that is upwind of the plume source.
- Once a plume is identified, the practitioner places the analyser downwind of the emission source – as centred as possible in the plume – and records wind and mixing ratio data for approximately 20 minutes. The method requires high speed recordings of wind speed, wind direction and pollutant mixing ratio; 1 Hz instrumentation is commonly used. The distance from analyser to source is also required; teams often use optical gas imaging or other techniques to locate the plume. The resulting data is binned by wind direction and fitted to a Gaussian (normal) distribution. A Gaussian plume assumption is then used to estimate the

emission rate. As a point source, the emission is described in 3-dimensional space and assumptions may be made about the source's location and height.

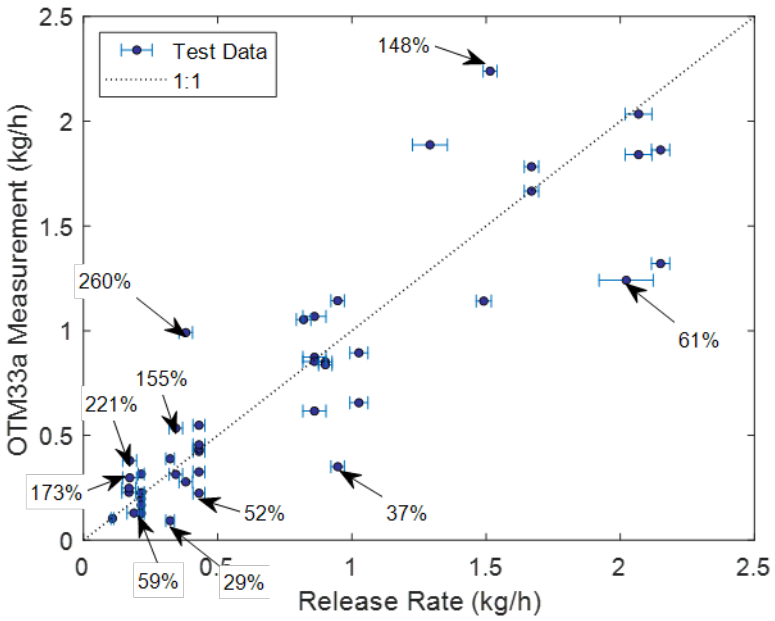
The Gaussian plume assumption limits the method to emitters that can be treated as point sources, either a single dominant emitter or by sampling sufficiently downwind that a group of spatially proximate sources appear to be a single source.

A variant of the method simultaneously samples both the pollutant and a tracer gas, released at a known rate near the emissions source. The same calculation method is applied to both. If the release rate of the tracer gas can be properly recovered during the analysis, confidence increases that the pollutant is also properly recovered. The extension using tracers is not analysed in the following subsections.

5.1.2. Uncertainty calculation

The OTM33a method was tested at the Colorado State University by personnel from the University of Wyoming using controlled releases of methane or natural gas (Edie and others, 2020). Releases consisted of 21 tests released from a gas bottle or small manifold in an open area, and 24 releases from equipment at the Methane Emissions Technology Evaluation Center (METEC; Bell and others, 2020). Testing was performed over a short period, with limited variation in weather conditions. The team used a high-speed methane analyser (see Edie and others, 2020 for details) and a multi-axis anemometer, mounted on a mobile laboratory. Controlled test data are summarised in Figure 5.1.

Figure 5.1: Test data for OTM33a controlled testing, from Edie and others (2020)
Results with the 10 largest relative errors are identified on the plot. The unity line is also shown. A least squared fit to the data, not shown, results in y=0.996x, with R^2=0.766.

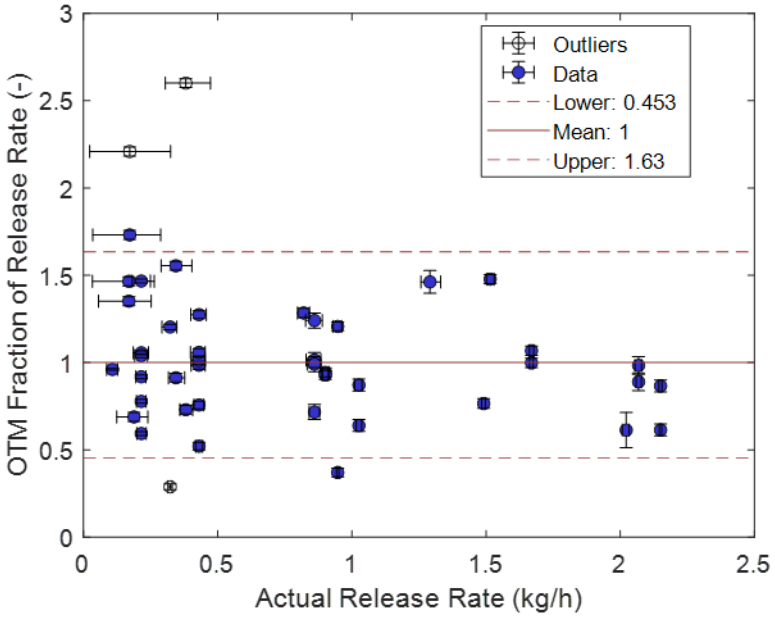


Uncertainty is characterised by estimating the probability distribution, *f*, that relates controlled release rate to measurements in:

$$M_i = f A_i$$

where M_i is the OTM33a measurement result for release i, and A_i is the actual release rate. For application purposes, it is desirable for f to be a relative uncertainty that is valid across the entire operational range of method. As can be seen from Figure 5.1, more points with large relative error occur at lower emission rates. If the method is used across a wide range of emission rates, it is useful to trim these outliers to avoid overstating the uncertainty of the method for the larger emitters that typically contribute a larger fraction of emissions. Outlier trimming is shown in Figure 5.2. Measurements were trimmed if relative error was >200% or <50% (2:1 range) and the controlled rate was less than 0.5 kg/h.

Figure 5.2: Relative correction factor for OTM33a, indicating which controlled releases were trimmed as outliers
After trimming, the 95% empirical confidence interval is shown for the remaining data.



5.1.3. Total method uncertainty

After trimming and using bootstrap methods to account for release rate uncertainty, the method shows no bias ($\bar{f} = 1$) and a somewhat skewed confidence interval: $f = 1.0\ [+61\%/-49\%]$. If outliers are not removed, the method is biased slightly above unity, and confidence limits are wider: $f^* = 1.05\ [+119\%/-65\%]$ (95% confidence intervals).

5.2. Component-level measurement

5.2.1. Introduction

For small facilities with a low probability of large emitters, direct detection and measurement of leaks from components is often an effective method of quantifying emissions. Typical facility types include end-point distribution equipment (residential and commercial meter sets), metering and regulating stations, wellheads with limited processing equipment on site, pig launchers and receivers (equipment associated with pipeline maintenance) that are not on other facilities, and similar locations containing dispersed equipment.

Direct component quantification requires a detection method and a quantification method (see Zimmerle and others, 2020a for an example implementation). The method uncertainty is a combination of both.

5.2.2. Detection

Common leak detection methods include US EPA Method 21 (M21; US EPA 1990, 2010) and European Standard EN 15446 (CEN 2008), as well as OGI, which is also a US EPA method (US EPA 2015) and for which development of a new European Standard is underway. In both cases, numerous variants are practised in the field. M21 and EN 15446 use a gas concentration sensor to detect emissions, typically by placing the sensor close to the targeted component. A leak is detected if the concentration exceeds a set threshold. OGI uses a gas imaging video camera that makes leak plumes visible. Surveyors look for visible emissions to detect leaks. While M21 and EN 15446 require close contact with each component, OGI can be used at a distance; see US EPA (2015) for method specifics.

In both cases, a surveyor moves through the entire facility and examines each component that has a potential to emit. For smaller facilities, surveying can commonly be completed from ground level. For larger facilities, surveyors may use an elevating work platform to access components. In the USA, upstream and midstream operations are more likely to use OGI, while refineries, gas plants and distribution systems use variants of M21.

5.2.3. Quantification

Detected emitters are quantified using a second method. While other methods exist (US EPA 1996), most commonly accepted methods capture emissions and measure the flow rate of the captured emissions. One of the most effective methods of measuring small emitters is high flow sampling (HFS). HFS quantifies emissions by (1) capturing emissions in a higher flow rate of ambient air, (2) measuring the mixing ratio of the desired pollutant and total mass flow rate, and (3) computing the emission rate from the ratio of mixing ratio to mass flow rate.

This analysis focuses on OGI leak detection followed by HFS measurement, which is suitable for a wide range of small facilities. Substantial controlled testing of OGI and the predominant HFS instrument (Bacharach Hi Flow®) helps in this analysis. Note, however, that the Bacharach instrument is obsolete at the time of writing, and several new products are under development by Sensors Inc.[3], AddGlobe[4], and Hetek[5] which have been tested by the METEC at the Colorado State University (Zimmerle and others, 2022).

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[3] Sensors Europe GmbH, Feldheider Str. 60, 40699 Erkrath, Germany, https://sensors-inc.com/

[4] AddGlobe, LLC, 1650 Arabian Drive, Loxahatchee, FL 33470, USA, https://addglobe.com/

[5] Hetek Solutions, Inc., 2085 Piper Lane, London, ON N5V 3S5, Canada, https://www.hetek.com/

The Bacharach instrument has exhibited a number of accuracy issues related to gas streams with high hydrocarbon content (Howard and others, 2015, Zimmerle and others, 2020a) or when concentrated gas streams are quickly introduced to the instrument (Connolly and others, 2019). As a result, measurement uncertainty is a function of how the instrument is used and can vary widely. The supplementary information in Zimmerle and others (2020a) provides an example of the required analysis.

OGI emissions detection performance is best represented by controlled testing of the entire survey process, where a trained surveyor locates leaks in realistic conditions in a single-blind trial. Such a trial was performed at the Colorado State University and is presented in Zimmerle and others (2020b). Other studies have characterised cooled cameras (Ravikumar and others, 2018, Zeng and others, 2018, Connolly and others, 2019) and uncooled cameras (Ravikumar and others, 2016, Zeng and others, 2017). The Zimmerle and others (2020b) study found that surveyor experience had a definitive impact on detection efficacy. For this analysis, only the results from the most experienced subgroup of surveyors were used – those that had surveyed 700 or more sites before the efficacy study. For these surveyors, the probability of detection, p, is a function of the emission rate, r, in kg/h of methane:

$$p = \begin{cases} 1.58r^{0.28} & r < 0.19\frac{kg}{h} \\ 1 & r > 0.19\frac{kg}{h} \end{cases}$$

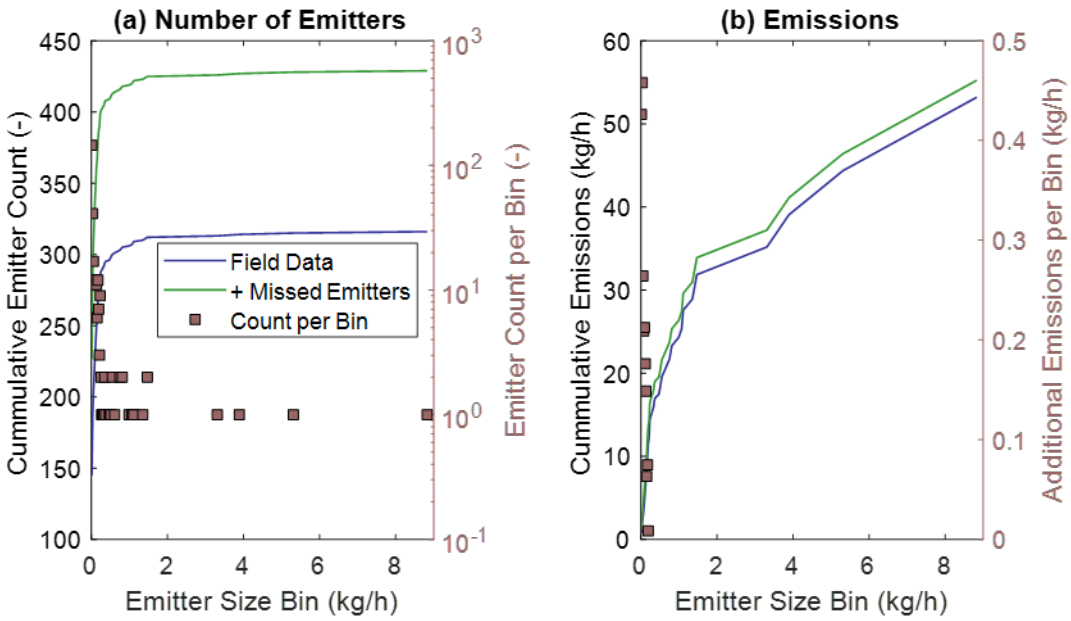
The HFS was characterised by a 90% lower detection limit (LDL) from Zimmerle and others (2020a) of 0.2 standard cubic feet per hour (whole gas) or approximately 0.0038 kg/h methane.

For this type of method, uncertainty is due to 2 factors: (a) what fraction of emitters were detected, and (b) once detected, what uncertainty is in measured emissions. Factor (b) can be further subdivided into 2 sources of uncertainty: (b1) estimated emissions missed when a detect was reported as zero by the instrument – that is, under the LDL of the instrument, and (b2) uncertainty in the instrument measurement itself.

Uncertainty is therefore a function of the emissions profile, since small emitters are both more likely to be undetected (factor (a)) and also reported as a zero emission rate (factor b1). These 2 are combined with the measurement uncertainty of the instrument (factor b2) which can be added after estimating the other 2. Importantly, ‘this makes the uncertainty of the method a function of the distribution of emission size’, as well as other factors which may impede detection and/or measurement, and results in an uncertainty with a low bias.

This uncertainty analysis uses data from the Bell and others (2017) study, which occurred on the type of facility where the combination of OGI and HFS is well suited, using only detected and measured emissions. OGI detection performance was estimated by applying the detection probability to all emitters and computing emissions from emitters that were not detected by OGI. For each detected emitter, the HFS LDL was applied to estimate the faction of non-zero emissions that would be reported as zero emissions by the HFS instrument. Results are shown in Figure 5.3.

Figure 5.3: Results of a combined OGI/HFS quantification method applied to emissions found during the Bell and others (2017) study
The left panel illustrates the number of emitters that were not detected during the field campaign. The right panel illustrates the emissions that would be missed by detection or measurement failure.



In the Bell and others (2017) study, 316 emitters were detected, resulting in a total emission rate of 53 kg/h. Applying the OGI probability of detection, an additional 113 emitters were possibly missed during the Bell and others (2017) field campaign. (Note that this analysis neglects repeat visits to some facilities during the Bell and others (2017) study, which would likely have decreased the number of non-detected emitters.) The additional emitters would have totalled approximately 2 kg/h or a bias of +0%/-3.8% of emissions.

During the Bell and others (2017) study, the Bacharach HFS instrument reported zero emissions after an OGI detection 53 times, resulting in an estimated unmeasured emission rate of 0.1 kg/h, or a bias of -0.19% of emissions. Combining these 2 results, estimated method uncertainty for OGI/HFS survey at small facilities is +0%/ to 4% of emissions, exclusive of the HFS instrument uncertainty.

Due to issues with the Bacharach instrument, no instrument uncertainty is included here, and it is recommended that the uncertainty analysis be updated when new instruments are available and characterised. For example, if the instrument has a 2-sigma uncertainty of +/-10% and an identical LDL as the Bacharach instrument, overall method uncertainty when applied to facilities similar to Bell and others (2017) would be approximately +6%/-14%.

5.2.4. Total method uncertainty

Based on measurements made during a controlled release and published in Bell and others (2017), the total method uncertainty for the component-level detection and measurement method for small facilities is estimated at +0% and -4% (excluding HFS instrument uncertainty and based on highly experienced surveyors). It is expected that as

the site size and complexity increases, the underestimation of emissions will increase due to missing sources.

5.3. Mass balance

5.3.1. Description

Trace gas analysers mounted on drones or aircraft are used to measure gas concentrations upwind and downwind of a facility as they circle it at different heights. The emission rate of the facility is then calculated from these measurements and the horizontal windspeed across the flight path. See Conley and others (2017) and Allen and others (2019) for examples of implementation of this method. DIAL can also be used to ‘sweep’ through successive vertical sections upwind and downwind of a facility.

5.3.2. Uncertainty in concentration measurements

The mass balance approach uses high precision methane analysers mounted on a mobile platform, such as an airplane or drone and whole-site quantification is achievable using the mass balance approach (Conley and others, 2017, Vaughn and others, 2017, Schwietzke and others, 2019). As an example, the ABB microportable greenhouse gas analyzer is a laser absorption spectrometer that measures methane mole fractions in air every second, with a precision of <2 ppb (1σ at 1 Hz) over an operating range of 0.1 to 100 ppm (Paul and others, 2001). This corresponds to an uncertainty in measurement of ±0.1% while measuring ambient background methane concentrations in air. The horizontal windspeed measurement is also required and the uncertainty is estimated at ±0.2 m/s (Conley and others, 2014).

5.3.3. Modelling uncertainty

Controlled releases of hydrocarbons (methane and ethane) were used to test the airplane flux balance method in Colorado in 2014, and in Arkansas in 2015 and 2017 (Conley and others, 2017, Schwietzke and others, 2019). Emissions were calculated to within +30% and -17% of the known emission rate between 5.5 and 91 kg/h (Figure 5.4). Controlled releases of ethane of 1.2 kg/h were also tested, but calculated emissions were underestimated by a factor of 2 (Figure 5.5). Data on repeatability are limited, but from the 2 repeat measurements made by Conley and others (2017), the calculated emission are within 5% of the average.

Figure 5.4: Known and calculated hydrocarbon emissions used to test the airplane flux balance method in Colorado and Arkansas in 2014 and 2015

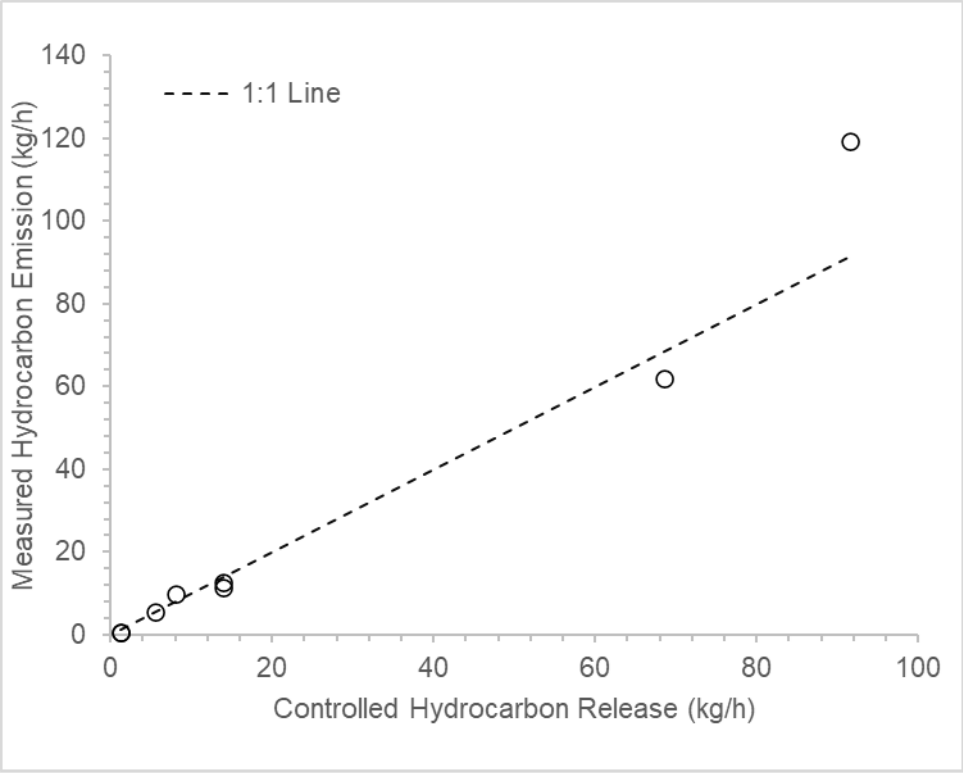
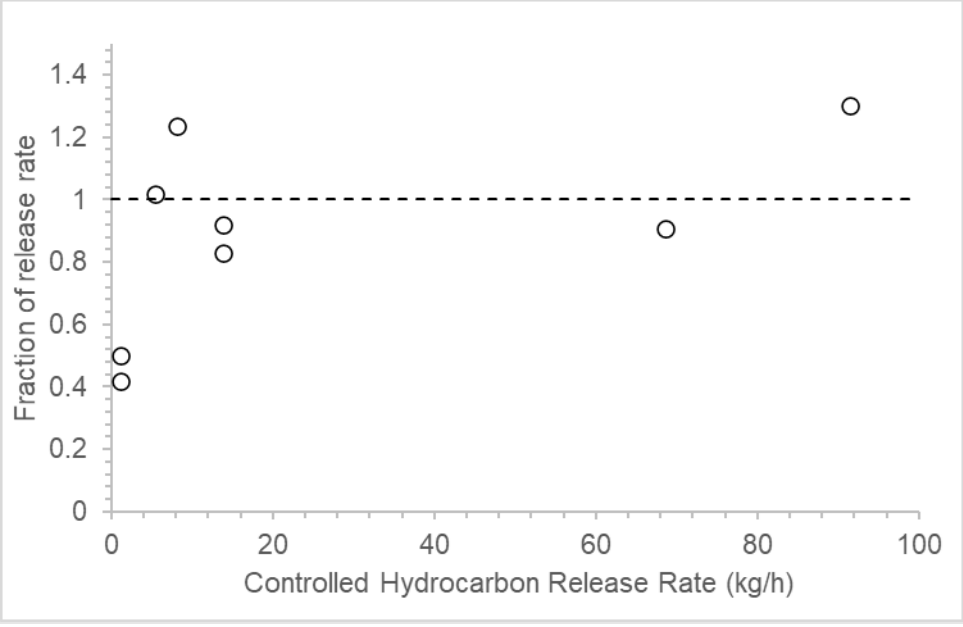


Figure 5.5: Controlled hydrocarbon release rate plotted against the fraction of release rate as calculated by the airplane flux method



5.3.4. Total method uncertainty

Based on measurements made during a controlled release and published in Conley and others (2017) and Schwietzke and others (2019), the total method uncertainty for the mass balance method is estimated at +30% and -17%. There is insufficient data to understand by how much this varies between the size of the facility. An additional caution is that the

data sets available in the above studies are limited, lowering confidence in uncertainty calculated from controlled testing.

5.4. Fenceline monitoring

5.4.1. Description

Fenceline monitoring uses multiple instruments fixed at the boundary of OOG operations to continuously measure gas concentrations. If the measured gas concentrations downwind of a source are sufficiently higher than the background concentration for a set pre-defined period, an emission is detected, and the operator is alerted. For example, SoCalGas uses 8 line-integrated infrared sensors at its Aliso Canyon natural gas facility in California. SoCalGas defines normal background at between 1.9 and 2 ppm and will look for leaks if sensors detect CH_4 mixing ratios averaging 8 ppm or higher for more than 20 minutes (SoCalGas, 2020). Currently, there are no published studies investigating the use of fenceline systems to monitor and quantify methane emissions from OOG production well pads. However, a major study into the quantification efficacy of these systems is currently underway at the Colorado State University[6], and results should be published by late 2022.

5.4.2. Uncertainty in concentration measurements

Low-cost sensors, typically including metal oxide methane sensors, catalytic oxidation sensors or compact sensors, are generally used for the fenceline monitoring approach. These low power (~0.5 W), low-cost (~£10 to £50) sensors are designed to measure methane mixing ratios between 1 and 100 ppm and after calibration can be used to measure concentration between 1 and 20 ppm (Riddick and others, 2020). Sensors tend to be unreliable in low humidity (<30%) conditions and sensor response is sensitive to both temperature and humidity.

Eugster and Kling (2012) reported that methane mixing ratios calculated from the output of a TGS2600 at Toolik Lake, Alaska, USA were in good agreement (R^2 = 0.85) with mixing ratios between 1.85 and 2 ppm, as measured by a Los Gatos Research FMA 100 methane analyser (Eugster and Kling, 2012). Riddick and others (2020) reported a TGS2600 could be used to reliably measure mixing ratios between 1.85 and 5.85 ppm that agree to a high-precision instrument output to ±13%, with the majority of the uncertainty caused by the propagation of temperature uncertainty in the emission estimate (Riddick and others, 2020).

Commercial solution developers have also adopted more expensive, accurate and stable sensors. Examples include sensors from Lunar Outpost Environmental (https://outpostenvironmental.com/) or Sensirion (https://sensirion.com/). These sensors

[6] https://energy.colostate.edu/metec/aded/

typically provide better stability and higher sensitivity, but their higher cost may also lead to fewer sensors being placed at the facility.

5.4.3. Meteorology uncertainty

Measurement of meteorology is vital to both concentration measurements (temperature and relative humidity) and modelling (wind speed, wind direction and atmospheric stability). Simple meteorological stations should be capable of measuring, to a reasonable accuracy, wind speed ±0.5 m/s; air temperature ±0.5°C; relative humidity ±0.5%; and wind direction ±0.5°. These individual uncertainties result in different combined uncertainties in the calculated emission depending on the modelling approach.

A measure of atmospheric stability can be generated by a range of approaches. In increasing complexity, methods include: (1) calculating a Pasquill-Gifford stability class (PGSC) using a lookup table based on cloud cover and wind speed (propagates an uncertainty of ±1 PGSC class); and (2) calculating a Monin-Obukhov length (L) using a sonic anemometer (uncertainty of ±1%, depending on the specifications of the sonic anemometer).

The roughness length (z_0, m) of the fetch can be estimated by measuring the height of the aerodynamic obstructions between source and detector. Approximations suggest z_0 is equal to one-tenth of the obstruction height with associated uncertainty, based on the uncertainty in measuring device, of ±10% (Seinfeld and Pandis, 2016). Roughness length can also be derived from sonic anemometer measurements with a lower associated uncertainty (uncertainty of ±1%, depending on the specifications of the sonic anemometer).

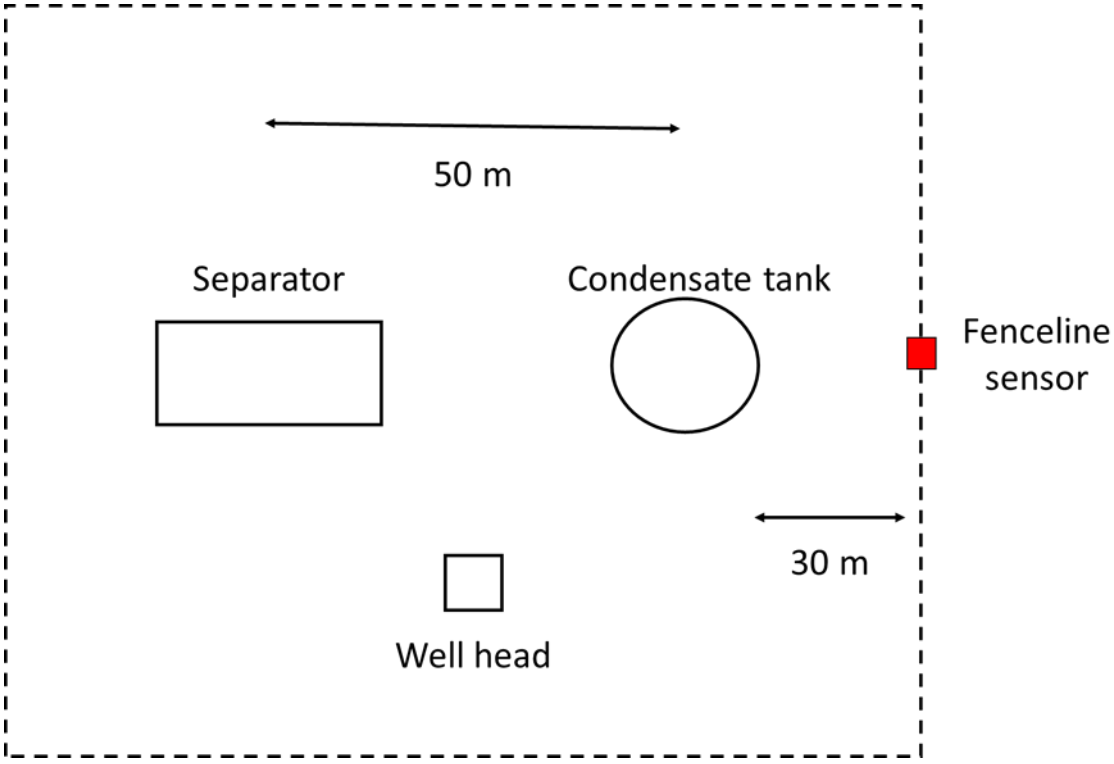
5.4.4. Uncertainty in approach

Two typical modelling approaches can be used for fenceline monitoring: (1) the Gaussian plume (Hunt 1982, Hunt and others, 1988, Seinfeld and Pandis 2016, CERC 2017); and (2) the backward Lagrangian stochastic (bLs) method (Flesch and others, 1995, 2004). The major shortcoming of both approaches is that the number of sources that can be measured depends on the number of detectors and the number of emission sources is less than or equal to the number of detectors. Input to both the modelling approaches includes measured methane concentrations, distance to the source, wind speed, wind direction and atmospheric stability. In ideal measurement conditions, where the location of the emission source is known and highest precision micrometeorological measurements are available, the Gaussian plume and bLs approaches can estimate methane emissions of 200 g/h to ±33% and ±11%, respectively (Riddick and others, in review).

Peer-reviewed uncertainty data on fenceline monitors are limited. One recent study reported average calculated emissions to within ±4% of the known emission rate (Riddick and others, 2022). However, the emission estimates were based on prior knowledge of the emission location and filtering of data, that is, limitations of the sensors and limitations of the model to parameterise lateral dispersion at close range (<100 m), which meant that emission estimates were only be calculated for 1% of the time. To estimate the uncertainty in emission, we can consider a simple well pad, comprising one wellhead, one separator

and one condensate tank surrounded by a fenceline 30 m away (Figure 5.6) and simulate the emissions in a bLs model. This configuration represents the smallest OOG sites in England without any aerodynamic obstructions.

Figure 5.6: Simple well pad site with a fenceline 30 m from all equipment



At wind speeds of 4 m/s, the wind direction of 270° to the north and in a neutral atmosphere (PGSC D), a methane emission from the tank of 47 g/h and a methane emission from the separator of 172 g/h will result in the same concentration, 166 µg/m^3, being observed by the fenceline sensor. The concentration of 166 µg/m^3 was chosen for this experiment as it is the 50th percentile of measured well pad leaks (Allen and others, 2013, 2015, Bell and others, 2017). Here, we estimate the uncertainty caused by horizontal distance between leak and detector at ±57%. Leak location in the vertical will also result in emission uncertainty. A methane emission of 142 g/h on top of the condensate tank at 3 m would result in a concentration of 166 µg/m^3 and uncertainty of ±50%. Low-cost sensors have an uncertainty of ±13%, propagating an uncertainty in emission of ±17%. High precision micrometeorological measurements propagate an uncertainty in emission of ±1%.

Taking the root of the sum of the individual uncertainties squared, results in an overall uncertainty for a small production site (one wellhead, one separator and one tank) of ±78%. The largest source of uncertainty is associated with the uncertainty in the location (horizontal and vertical) of the leak. This is the simplest and smallest of arrangements seen at English OOG sites. Larger sites will have a larger emission uncertainty as the lateral distance will be larger. This is highlighted if we imagine the wellhead to be leaking: in the scenario of a wind direction of 270°, the fenceline sensor will not detect the emission and the wellhead leak will remain undiagnosed. The distribution of the fenceline sensors

and prevailing wind directions will determine which leaks will be ‘seen’ by the fenceline sensor.

5.4.5. Representativeness of measuring period

As it is a continuous, autonomous measurement system, a fenceline monitoring system can be used to observe temporal variability in the site emissions on any scale. This gives fenceline monitoring an advantage over all other systems, despite an emission uncertainty of each measurement in the simplest, best-case scenario of ±78%. It is reasonable to assume that the representativeness of the emission estimates will average out over a longer period, that is, for an annual emission rate. However, it will be difficult to distinguish a large, upset condition, such as an open pressure release valve or tank hatch, from the noise in the system. For example, in a recent study (Zimmerle and others, 2020a), a thief hatch large emitter exhibited mean emission rates of 45 kg/h compared to typical component emissions of 0.2 kg/h, therefore, the upset emissions from the thief hatch would not be discernible against the smaller but continuous component emissions when considered over a longer timeframe. Temporal uncertainty for all methods is discussed further in Section 6.4.

5.4.6. Completeness of Site Description

As shown above, a complete description of the site is required to be able to quantify emissions using a fenceline system. Emission uncertainties will be significantly larger for larger sites with more equipment. At present, no studies or data have been published showing how fenceline systems quantify emissions from multiple point sources even though there are several private companies that manufacture, install and run fenceline systems for leak detection and potential quantification purposes at active well pads in the US, Canada and Europe. Companies such as, Project Canary (www.projectcanary.com), ChampionX (www.championx.com), Qube Technologies (www.qubeiot.com) and Fluxsense (www.fluxsense.com) all use low-cost sensors to measure concentrations on the fenceline of oil and gas facilities, but it is not currently clear how they convert the measured concentrations to emissions or how well their emission estimate is constrained.

5.4.7. Total method uncertainty

Based on analysis carried out using an atmospheric dispersion model, the total site uncertainty for fenceline systems monitoring the simplest OOG production site in the best-case atmospheric conditions is estimated at ±78%. This uncertainty is likely to be the smallest realistically and will be much larger for larger and more complex facilities.

5.5. Tracer method

5.5.1. Description

Tracer gases are emitted at known rate next to the site of interest. The tracer and pollutant concentrations are then measured downwind while moving through the plume. The pollutant emission rate is then inferred from the known release rate of the tracer gases. See Lamb and others (1995) for an example implementation of this method. The timing of

the release of tracer gases, that is, the measurement period, can be tailored to meet the needs of the study.

5.5.2. Concentration measurement uncertainty

Subramanian and others (2015) used a dual-laser Aerodyne Quantum Cascade Tunable Infrared Laser Differential Absorption Spectrometer (QC-TILDAS) instrument to measure methane, ethane, nitrous oxide, and acetylene at 1-Hz and a Picarro cavity ringdown spectroscopy (CRDS) instrument for methane at 3 to 5 Hz (Subramanian and others, 2015). Minimum detection limits were: CH_4 5 ppb, C_2H_2 0.5 ppb, N_2O 0.2 ppb, C_2H_6 0.3 ppb.

5.5.3. Meteorology uncertainty

Meteorological data are not required for quantifying emissions.

5.5.4. Uncertainty in approach

The uncertainty in measuring point source emissions is simply a function of the gas concentration measurement uncertainty and the release rate uncertainty. The results from controlled tests and from replicate measurements indicate that the total source emissions can be obtained with an accuracy of ±15% (Lamb and others, 1995). Lamb and others (1995) stated that this uncertainty was derived using both controlled releases and a desktop study. The data for the desktop study were presented in full in the publication, the data for the controlled release were not.

Allen and others (2018) and Fredenslund and others (2018) present multiple downwind mass balance measurements made downwind of a 10.9 kg/h methane release and a 5.3 kg/h methane release. The data presented shows the mean emissions bias from 10+ individual measurements, therefore, it is impossible to present uncertainty from a single measurement. The mean emission bias from these experiments ranged from +40% to -15%. These studies also report that in using statistical techniques the calculated uncertainty could be reduced further to ±20%. Another tracer method with a controlled methane release of 4.7 kg/h (Mønster and others, 2014) reliably overestimated emissions by a factor of 2. Similar to the Frendenslund and others, (2018) study, the uncertainty was presented as an average of between 10 and 13 transects and individual measurement uncertainties were not presented.

5.5.5. Completeness of site description

Irrespective of the size and complexity of the site, the tracer method has been used by many studies to quantify site-wide emissions from oil and gas operations in the US (Marchese and others, 2015, Vaughn and others, 2018, 2017, Schwietzke and others, 2019). This method is simpler than other downwind methods as the atmospheric dispersion is implicitly accounted for in the movement of the tracer gas and does not need to be explicitly defined.

5.5.6. Total method uncertainty

Based on a desk-based study and published in Lamb and others, (1995), the total method uncertainty for the tracer method is estimated at ±15%. There is insufficient data to understand by how much this uncertainty could vary between the size of the facility.

6. Other sources of uncertainty

While measurement methods are often the focus of uncertainty calculations for measurement campaigns, in practical field situations, other sources of uncertainty may be as large, or larger, than the measurement method uncertainty. Additionally, variability in emissions rates over either a number of facilities, or over time at one facility, will affect the net uncertainty of measurements in practical field applications. This section provides a general overview of the sources of uncertainty other than the measurement method uncertainty presented in section 5.

Section 5 considered the uncertainty of a single measurement made by each method. These measurements may vary in the duration of data required. For example, a single tracer flux measurement may take hours to complete (multiple transects), an OTM33a measurement may take 20 to 30 minutes after the source is identified, or a fenceline monitor may produce estimated emissions every 15 to 240 minutes. This section provides an overview of how the uncertainty in the method interacts with the deployment of the method in the field.

For this analysis, we consider 4 sources of uncertainty other than the measurement method itself:

- **Sampling** – Emission rates from OOG facilities are often highly skewed, leading to uncertainty driven by sample size. For methods with asymmetric confidence intervals, which is often the case when dispersion assumptions are required, the asymmetric uncertainty interacts with asymmetric (skewed) emissions distribution to produce larger uncertainties than would be the case with symmetric method uncertainties and normally distributed emissions distributions.
- **Emission type uncertainty** – While most field campaigns focus on unplanned or unexpected emissions (often termed ‘fugitive’ emissions), many OOG operations have emissions of identical chemical species due to routine operations, including vented and combusted emissions. When the primary interest is in ‘total emissions’, regardless of cause, this uncertainty need not be considered. When the primary interest is to identify unexpected/unpermitted/abnormal emissions, then it is necessary to separate emissions caused by planned venting, or incomplete combustion, from fugitive emissions.
- **Temporal variability** – OOG emissions may vary substantially over time, particularly due to maintenance operations or failure conditions at facilities, such as process upsets, failed equipment or human error. If no knowledge of site operations is available to the measurement campaign, these variations can cause significant uncertainties, particularly when extrapolating to annual emission rates. Additionally, most facility-scale quantification methods are largely incapable of measuring short, large emission sources, as the sampling speed or fundamental averaging time of the method cannot capture these emission transients accurately. This includes most continuous monitors, which typically have long averaging times, tracer flux or OTM33a downwind measurements, and commonly used on-site measurement methods. Therefore, a method which can accurately characterise near-constant,

long-duration emissions may not characterise short-duration emissions, and for many OOG facility types, short-duration emissions may account for large fractions of annual emissions from a facility (for examples see Allen and others, 2013, Zimmerle and others, 2015, 2020a).

- **Method implementation uncertainty** – Most methods undergo controlled testing performed by expert practitioners in near-ideal conditions. Field conditions often deviate from these conditions and can lead to uncertainty. For example, Mønster and others (2014) tested single-tracer flux methods, using several calculation techniques, but report data only for a single measurement with 10 to 13 transects downwind of the emission source. This number of transects is exceptionally rare in field conditions, and as a result, reported uncertainty is understated.

In most cases, practitioners are interested in representative estimates of emissions from some population of facilities over some duration. Since uncertainty in such an estimate depends on 'what' is being measured, and 'how' and 'for how long' it will be measured, there are 3 cases to be considered when estimating uncertainty:

Case 1: **A single measurement at a single facility**: In this case, the method uncertainty in section 5 provides an estimate of uncertainty for the method being used. The result is a single measurement of a single facility. Additionally, this case must also consider potential increase in uncertainty if there are implementation issues which deviate substantially from the assumptions used to compute the method uncertainty.

For example, uncertainty computed in section 5.4 for fenceline monitors assumes a simple facility configuration and a limited range of dispersion parameters. If a deployment deviates from these assumptions, uncertainty may increase. (Practical testing currently underway at the Colorado State University shows that these increases may be substantial, but the results have not been published at the time of writing.)

Case 2: **Repeat measurements at a single facility**: Repeated measurements may be completed by multiple deployments of a survey method, such as tracer flux or OTM33a, or deployment of a continuous monitoring method, such as a fenceline monitor. In this case, there is no sampling uncertainty, as one facility has been selected for measurement, and results should be characteristic of that facility. Uncertainty analysis should focus on uncertainty impacts due to temporal variability and implementation uncertainty. Uncertainty due to temporal variability may often be controlled by tracking facility operations during the measurement period, and treating different facility operational modes as separate sub-groups of measurements.

Case 3: **Single, or small number of, measurements at a selected set of facilities:** This type of measurement is often termed a 'field campaign', and is typically carried out over one or more short-duration periods. This case is applicable to survey methods, such as on-site component-level measurement, tracer flux, flux plane or

mass balance. While short-duration measurement is possible for fenceline monitors, most often these continuous monitors are used for extended periods at selected facilities (See case 2).

In this case, all the uncertainties described must be considered. Do the sampled facilities represent the full population of facilities? When measured, will the facility be in a representative state, or will emissions be non-characteristic? Can the method be properly used in the permitted, environmental and access conditions at the sample facilities?

Additional notes:

- If characterisation of a single facility (case 1 or 2) is meant as a surrogate for emissions at a larger population of facilities, then the measurement effort is case 3, with a sample size of one.
- Knowledge of the activities and processes taking place at a site during the measurement period is needed. Such 'metadata' or contextual information will be necessary to interpret estimates of whole-site emissions, especially when measurements are being made for regulatory purposes.
- As noted earlier, in all cases emission type uncertainty may or may not be a consideration, depending on the goals of the measurement campaign. If the goal is to characterise total emissions, regardless of source, any of the whole-site methods in section 5 estimate all emissions of the target analyte. However, if the goal is to identify unexpected, unpermitted or otherwise exceptional emissions 'in the presence of expected emissions', then the source and cause of emissions must be considered. See section 6.3.
- For the methods recommended here, analytic assumptions, including dispersion, efficacy of OGI leak detection are substantially larger than sensor and instrument uncertainties. Where relevant, instrument uncertainties are included in the uncertainties presented in section 5 (for example, on-site measurements). However, practitioners should be aware that using low-cost sensors or instruments could increase method uncertainties beyond those in section 5. This potential source of uncertainty is not analysed in this section.

Given the asymmetric uncertainties of most of the measurement methods, and the skewed distributions of emissions at most OOG facilities, the recommended method for estimating the total uncertainty is:

Step 1: Select the desired campaign plan from the 3 cases described.

Step 2: For case 3, determine the population of facilities to be characterised, and select a sampling strategy within that population.

Step 3: For cases 2 and 3, develop a prior estimate of variability between facilities (case 3) and variability over time (cases 2 and 3). Since uncertainty is easily normalised, the mean of the estimated prior probability distribution is less important

than properly representing the expected relative skew in emissions – both between facilities and over time.

Step 4: For each measurement method being considered, simulate the measurement effort, computing a net uncertainty. While various statistical methods may be used, Monte Carlo simulation is often readily used for this purpose.

Step 5: Considering the results from step 4, modify method selection and sampling strategy to minimise uncertainty given practical constraints (for example, budget, access, travel times).

The following sections illustrate the impact – quantitatively if possible, qualitatively if not – of each of the uncertainties indicated above, and represent subsets of the simulation proposed for step 4. To better illustrate the results, examples of facility emissions are introduced in section 6.1 (an example of step 3) and are reused in other sections.

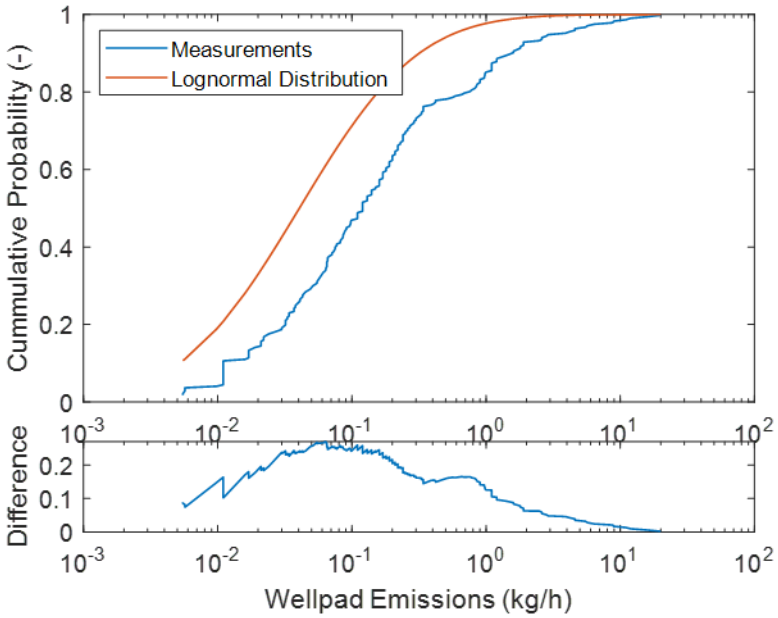
6.1. Example data

For the examples of different types of uncertainty in sections 6.2 to 6.5, we use data sets drawn from 2 studies, as well as a completely fabricated population of facilities with normally distributed emissions.

Study 1: The first study (Bell and others, 2017) measured fugitive emissions and simulated non-fugitive emissions from 262 well pads in the Fayetteville Shale production basin in Arkansas, USA in 2016. The Fayetteville is a dry gas basin that produces negligible oil. Three facilities that were measured during liquids unloading are not included in this analysis, leaving 259 facilities with a mean emission of 0.738 [0.0056 to 6.46] kg/h (empirical 95% confidence interval used throughout here).

Facility emissions from study 1 are shown in Figure 6.1, compared with a lognormal distribution with the same mean and standard deviation. As is typical of OOG emissions (and air emissions in general), emission rates are substantially more skewed than a lognormal distribution. For these data, 5% of facilities are responsible for 59% of emissions (emissions >3.5 kg/h), and the lowest emitting 50% of facilities are responsible for 3.1% of emissions.

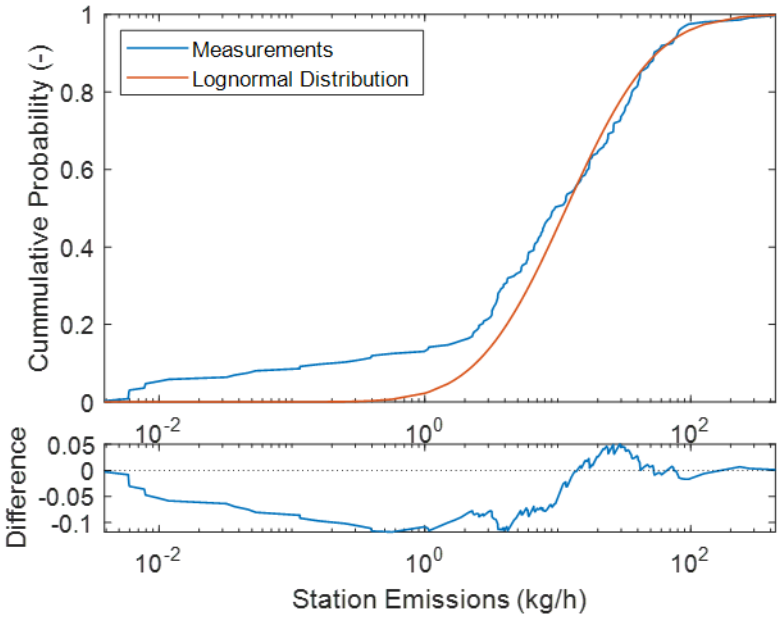
Figure 6.1: Per-facility emissions rates for the Bell and others (2017) study
Top panel shows a cumulative distribution function (CDF) plot of the facility emissions, compared to a lognormal distribution with the same mean and standard deviation. Bottom plot shows the difference (residual) in cumulative probability between the data and lognormal approximations.



Study 2: A second data set is taken from a 2018 national study of US gathering compressor stations (DataTable.xlsx, Sheet D23 Station Emissions in Zimmerle and others, 2020a). The data set from that study summarises emissions from 180 gathering compressor stations, including both measured fugitive and engine exhaust emissions, supplemented with simulated emissions for other active venting sources. Note that the reciprocating engines used on these sites have non-trivial methane emissions in their exhaust, increasing emission rates for facilities where engines are running.

Facility emissions from study 2 are illustrated in Figure 6.2, compared to a lognormal distribution with the same mean and standard deviation. Per facility mean emissions are 24.5 [0.00592 to 96.3] kg/h. Data are less skewed than the Bell and others (2017) study, and largely follow the lognormal distribution for facilities with emission rates above 3 to 4 kg/h. For these measurements, the highest emitting 5% of facilities emit 35% of emissions (emission rates >80 kg/h), while the lowest emitting 50% of facilities emit 7.5% of emissions (emission rates <9.5 kg/h).

Figure 6.2: Per-facility emissions rates for the Bell and others (2017) study
Top panel shows a CDF plot of the facility emissions, compared to a lognormal distribution with the same mean and standard deviation. Bottom plot shows the difference (residual) in cumulative probability between the data and lognormal approximations.



Normal control: In addition to the study data from the Bell and others (2017) and Zimmerle and others (2020a), normally distributed emissions are used here as an illustrative control. This will be termed the 'normally distributed control' (NDC) population in subsequent simulations. In practice, normally distributed emissions are vanishingly rare in OOG facility populations. For this purpose, we use a distribution with mean of 1 kg/h, standard deviation of 0.25 kg/h with lower tail truncated at 0, and pull 1,000 random samples from this distribution to simulate a population of facilities.

While essentially never seen in real OOG emission studies, comparing normally distributed emissions to the other 2, non-normally distributed emissions in studies 1 and 2 illustrates how non-normality impacts uncertainty calculations, and why careful selection of a study simulation method is required.

6.2. Sampling uncertainty

Considering case 3, field campaigns intend to produce a central estimate of emissions – typically mean or median – from a population of facilities, and assess the uncertainty in those emissions, and, if possible, variability between facilities or over time. In practice, field campaigns are always limited in the number of facilities that can be sampled, and since emissions are highly skewed, this increases uncertainty that the field campaign sample will reproduce mean emissions from the population of facilities. This section illustrates how sampling a population with skewed emissions impacts method uncertainty.

Method: To illustrate this uncertainty, we use Monte Carlo methods to simulate 1,000 field campaigns. As a metric, we compute the error in the campaign's mean emissions relative to the population mean. To separate method uncertainty from other causes, in each

campaign N facilities are ‘measured’ with zero error. Campaign sizes range from 10 to 100 facilities (5 to 55% of the population represented by the example Bell and others (2017) study).

Note that both studies represent one sample of a larger population and therefore the distribution of emissions are only an approximation of the true population, that is, the US gathering stations (Zimmerle and others, 2020a) or Fayetteville well pads (Bell and others, 2017).

Results: For the NDC population the mean emission rate computed from each campaign reduces (as expected) as the campaign size increases from 10 to 100 facilities. In no case does the error in any simulated campaign exceed ±30%. Since many statistical approaches assume normality, this provides a contrasting example: uncertainty in mean emission estimate reduces at approximately $\frac{1}{\sqrt{N}}$, where N is the number of facilities in the field campaign.

For the other studies, we compare them to this metric: What is the probability that a simulated field campaign will estimate mean facility emissions to within ±30% of the population mean? When the emissions distribution is highly skewed, sampling a disproportionate number of high emitting sites skews the mean upward. Conversely, since many facilities in the populations have near-zero emissions, disproportionately sampling these facilities skews mean emissions below the population mean.

Figure 6.3 shows results for the Zimmerle and others (2020a) study, which has a somewhat lognormal emissions distribution. The bottom panel of the figure illustrates the probability that any one field campaign will have an error in mean estimated emissions in excess of 30%. For example, a field campaign measuring 50 facilities has a 1 in 4 (25%) chance of an error greater than 30%. Skew and resulting error are more severe for the Bell and others (2017) study (Figure 6.4); the same field campaign sampling 50 facilities has a 1 in 2 chance of mean emissions being within ±30% of the population mean.

Figure 6.3: Simulation of field campaigns for the Zimmerle and others (2020a) data set
Top panel illustrates the result of 1,000 simulated field campaigns, with mean, inner quartile, whiskers at 1.5x the quartile, and outliers. The bottom panel illustrates the probability that the mean emissions measured in any one field campaign estimate a mean that deviates more than 30% from the population mean. Campaign sizes range from 5.6% to 56% of the population.

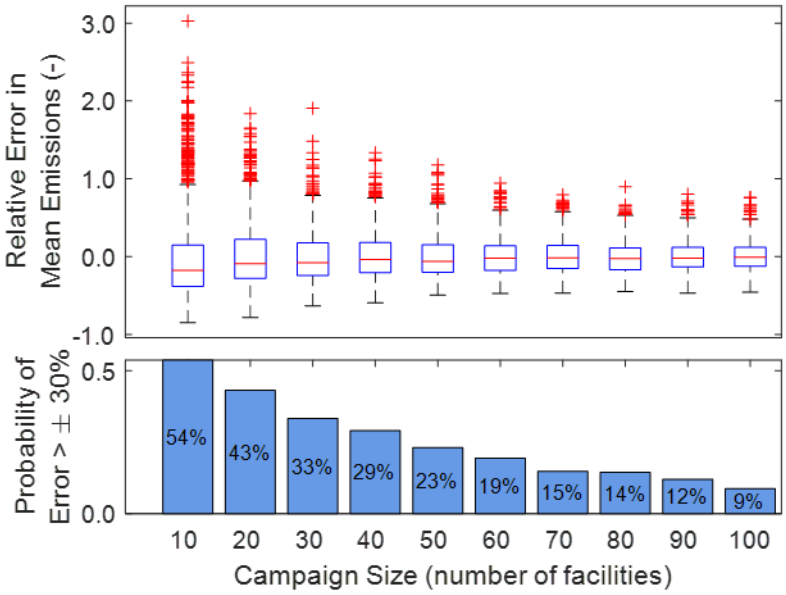
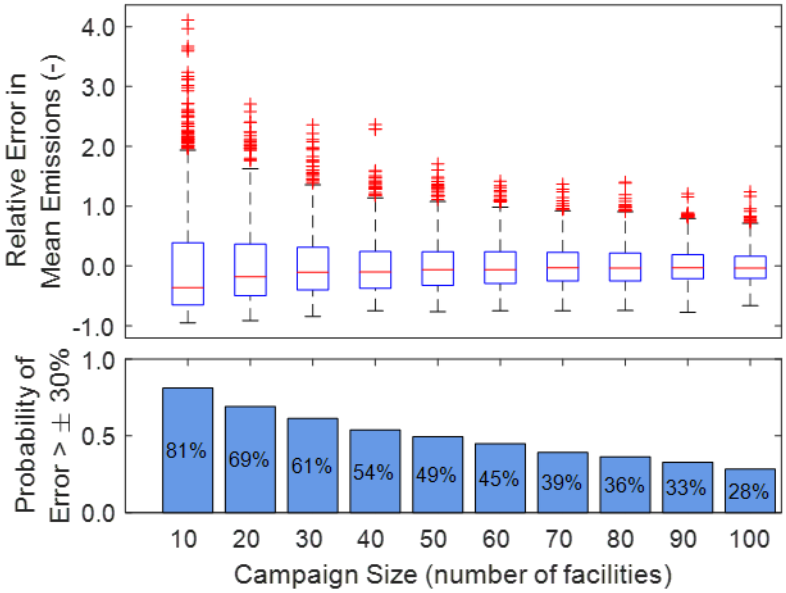


Figure 6.4: Simulation of field campaigns for the Bell and others (2017) data set
Top panel illustrates the result of 1,000 simulated field campaigns, with mean, inner quartile, whiskers at 1.5x the quartile, and outliers. The bottom panel illustrates the probability that the mean emissions measured in any one field campaign estimate a mean that deviates more the 30% from the population mean. Campaign sizes range from 3.9% to 39% of the population.



Conclusions: In addition to measurement method uncertainty, field observations consistently indicate that OOG facility emissions are highly skewed. As a result, sampling

uncertainty may create uncertainties in mean emissions on par with, if not larger than, the uncertainty in the methods used for measurement. These simulations illustrate an important point regarding the reduction of uncertainty from any measurement campaign: for typical emissions profiles, larger sample sizes lead to substantially lower uncertainties.

The simulation above assumed a 'perfect measurement method': for every simulated facility measurement, 'exactly' the true emissions were reported. The result is non-trivial uncertainty, caused only by skew in the emissions from the target population of facilities, and the resulting uncertainty is similar to the uncertainty in the method. Therefore, a method with higher uncertainty but lower cost, enabling larger sample sizes, repeat sampling, or more frequent sampling may produce lower uncertainty than an expensive, highly accurate method that can only be used for small sample sizes.

Recommendations: When selecting measurement methods, simulate the measurement campaign as described in the 5-step process to estimate total uncertainty, combining both measurement method uncertainty and sample skew.

6.3. Emission type uncertainty

The source of emissions affects all measurement cases (cases 1 to 3), if differentiating between source categories is a goal of the study. Emissions from OOG operations are generally categorised into 3 types of emissions:

- **Vented emissions** (may be referred to as 'unaccounted channelled emissions') – uncombusted gas intentionally released during operations or maintenance at the facility. Typical sources include equipment blowdowns (depressurising equipment), routine leakage through shaft seals or rod packing on compressors, gas-powered pneumatic controllers, and similar sources.
- **Combustion emissions** (may be referred to as 'accounted channelled emissions') – gas species emitted in the combustion exhaust of prime movers, heaters, flares, and other combustion sources. Of particular interest is 'combustion slip' – fuel gas species which are not combusted and are emitted in the combustion exhaust. For example, a flare with '98% destruction efficiency' would 'slip' 2% of gas sent to the flare.
- **Fugitive emissions** – uncombusted gas unintentionally released during operations. These emissions include equipment leaks as well as excess ('upset' or unintentional) emissions from vented or combusted sources.

Note that there are substantial differences in the categorisation of sources. For example, a regulatory jurisdiction may not categorise venting or combustion emissions in excess of design specifications as fugitive emissions.

If the objective of the field campaign is to characterise total emissions, then the source category is of secondary interest: it affects determination of effective mitigation methods, but not total emissions. However, the source of emissions may affect method uncertainty. For example, combustion emissions may be transported and dispersed differently than vented or fugitive emissions.

However, if the objective of the field campaign is to identify fugitive emissions and stimulate corrective action to reduce them, the field campaign will need to distinguish fugitive emissions from expected vented and combusted emissions. The remainder of this section is focused exclusively on this case.

For the example data sets used here, the origin of emissions was categorised (with some exceptions) by on-site observers who recorded active operations at the time of measurement. In the Bell and others (2017) study, 80% of emissions were from non-fugitive sources, and on 76% of sites non-fugitive emissions exceed 50% of facility emissions. A significant fraction of the non-fugitive emissions originates with gas-driven pneumatic controllers. For the Zimmerle and others (2020a) study, 48% of emissions were non-fugitive sources, and non-fugitive emissions exceeded half of facility emissions at 44% of facilities. A primary non-fugitive source on these stations is the exhaust from reciprocating engines (37% of emissions).

Method: The critical issues when detecting fugitive emissions against a ‘background’ of non-fugitive emissions is whether the measurement method is capable of distinguishing the excess fugitive emission rate as an enhancement over the non-fugitive emission rate. To assess the impact of measurement uncertainty on emissions classification, we consider a hypothetical whole-site measurement method that has uncertainty of ±50% across its entire operational range. This arbitrary number range is selected to be roughly similar to the best downwind methods (see section 5).

Given the method uncertainty, we consider 2 stages to detect fugitive emissions. First, the facility emissions must exceed ‘expected emissions’ by the method uncertainty. For this demonstration we use the population mean emissions as the expected emissions, and a facility is ‘detected’ as having unusually higher emissions (fugitive emissions) if measured rates exceed the mean by the measurement uncertainty. Second, to estimate fugitive emissions, the measurement campaign must also separate fugitive and non-fugitive emissions. Assuming there is reasonably complete knowledge of vented and combusted emissions on the facility, we assume fugitive emissions can be identified if total emissions exceed expected emissions from non-fugitive sources by the measurement uncertainty – that is, total emissions are 50% more than expected non-fugitive sources.

The method above classifies each facility measurement as one of 3 types:

- Not detected – emissions were below the threshold where the facility is suspected of having excess fugitives.
- Detected/not quantifiable – emissions were above the detection threshold, but fugitives were within the confidence interval of the method.
- Detected/quantifiable – emissions were above the detection threshold and the total exceeded expected non-fugitives by uncertainty of the method.

The latter category is a reasonable test for finding ‘large emitter’ facilities, which is often the goal of measurement efforts. Also, note that this analysis sidesteps many confounding issues and it is therefore a best-case estimate of uncertainty caused by emission type uncertainty. Two primary issues are (a) temporal variability, which makes the ‘expected

emissions' estimate uncertain, and (b) asymmetric method uncertainty, which is typically skewed high, in the same direction as the quantity being detected (fugitive emissions).

Results: Figure 6.5 shows analysis results for the Zimmerle and others (2020a) study. All facilities in this study had some fugitive emissions. The top panel shows non-fugitive emissions, of all types, relative to total facility emissions. The lower panel classifies sites into the 3 detection types. Green points representing detections with quantifiable fugitives are clustered at the higher facility emission rates (due to detection threshold) and above non-quantifiable sites (due to quantification requirements). Results for the Bell and others (2017) study are shown in Figure 6.6; 98% of facilities in this study had fugitive emissions. Totals are summarised in Table 6.1.

Figure 6.5: Detection simulation for Zimmerle and others (2020a) study data
Upper panel illustrates non-fugitive emissions, per facility, relative to total emissions for the facility. Lower panel shows detection simulation results; see text.

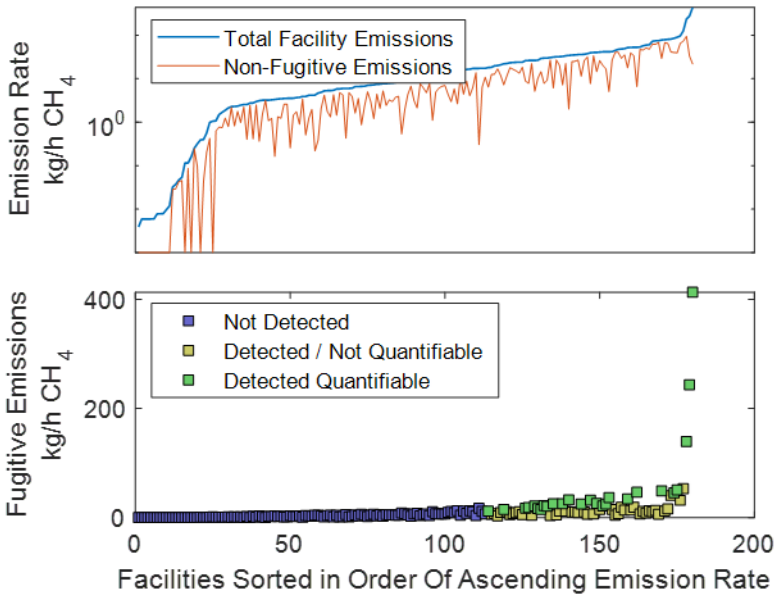


Figure 6.6: Detection simulation for Bell and others (2017) study data
Upper panel illustrates non-fugitive emissions, per facility, relative to total emissions for the facility. Lower panel show detection simulation results; see text.

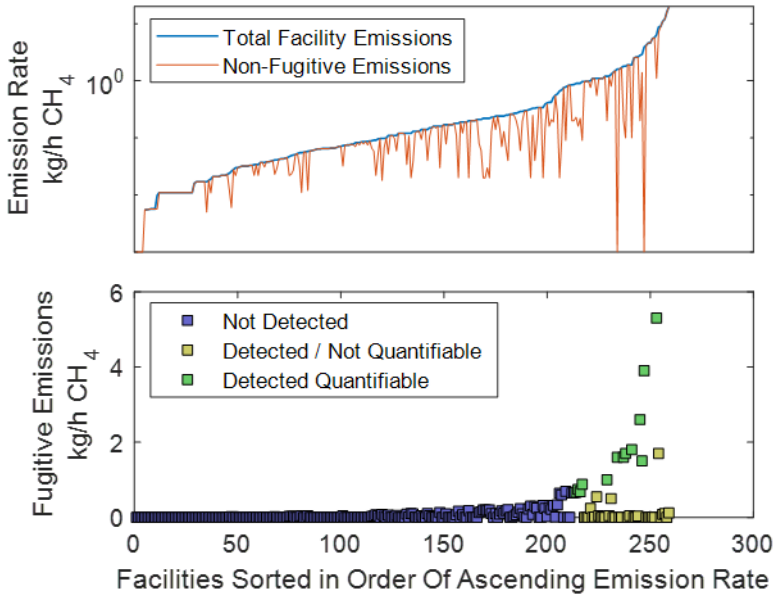


Table 6.1: Summary of probabilities for emissions type uncertainty

Metric	Zimmerle and others (2020a) study		Bell and others (2017) study	
	Fraction of sites	Fraction of emissions	Fraction of sites	Fraction of emissions
Not detected	63%	16%	82%	28%
Detected/not quantifiable	23%	21%	13%	9.2%
Detected/quantifiable	14%	62%	5%	63%

Conclusions: For the example data, which are widely representative of emissions distributions seen in the field, a method with a ±50% uncertainty detects two-thirds of emissions, despite 'not identifying' most facilities as having fugitive emissions. Since emissions mitigation is largely driven by the time between detection and repair of fugitive emissions, these data indicate that more frequent surveys, even if using methods with higher uncertainty, may result in better emissions detection and mitigation than a more certain method used less often.

Recommendations: This section is focused exclusively on where the measurement effort is to detect and quantify fugitive emissions. The simple simulations performed here indicate that resulting detection and quantification is non-trivial, highly dependent on the

emissions distribution, and also dependent on the mix of fugitive and non-fugitive emissions in the population of facilities. This leads to 2 recommendations:

First, for this measurement effort goal, step 3 of the uncertainty analysis (estimating expected emissions profile) should also subdivide emissions into the components of interest.

Second, the combination of (a) frequent surveys using more uncertain methods that can measure many facilities, coupled with (b) less frequent surveys using less uncertain methods may provide an optimal balance in fugitive emissions detection. In effect, this hybrid approach 'detects large emitters quickly and detects all emitters eventually'.

6.4. Temporal uncertainty

All of the methods discussed in section 5 provide a snapshot of emissions that generally result in one emission estimate from data collected over the duration (typically a few minutes or longer) it takes to make the measurements. However, that duration may not be representative of the average annual emission estimate as the measurements will either take place during 'normal' operations where the estimate will be biased-low or during large, upset emission events, which will be much higher than the average annual emission.

When simulating temporal uncertainty, it is useful to consider 2 types of measurement effort: field campaigns that measure a set of facilities one or more times, and a continuously operating measurement method (commonly called a 'continuous monitor') operating at one facility for an extended period of time.

Field campaigns: For a field campaign in cases 2 and 3, facilities may be measured multiple times, therefore making temporal variability a factor in total uncertainty. For case 3, most measurement efforts studying a population of facilities assume that temporal variation can be approximated by sampling variation. In other words, if a (typically rare) event occurs at some frequency in the sample, approximately that number of events will be occurring at all times, albeit at random locations in the population. Therefore, the results of studies such as the examples used here (Bell and others, 2017, Zimmerle and others, 2020a) can be interpreted as either:

- the long-term variation in emissions between facilities, as seen during any one survey

or

- temporal variation in facility emissions within a population of facilities, in effect assuming that any measurement made at one facility could just as readily have happened at any other facility, had measurements been made at a different time

Since most published data comes from this type of field campaign, little data exists to separate sampling uncertainty and temporal variation in emissions. Therefore, without any

other information on temporal variability, simulations like those in section 6.2 can be used to estimate the impact of temporal variation on uncertainty for survey methods.

However, if temporal data is available, it can be inserted into the prior probability distribution from step 3 to provide insight into uncertainty due to temporal variation independently of facility-to-facility variability (as covered in section 6.2). As an example, several recent campaign approaches, for example, aerial surveys in US production basins (Carbon Mapper[7], Cusworth and others, 2021) have attempted to better separate sampling and temporal variation by clustering wide-area surveys into multiple, closely-spaced surveys that better estimate persistence of large emission sources.

While not directly related to method uncertainty, temporal variation in emissions does impact the probability of detection. Assuming (a) an emission is large enough to be detected, and (b) persists for sufficient time, a survey method will detect the emitter, on average, in half of the survey cycle.

This type of data is specific to regional conditions, individual OOG operator practices, or other factors, such as the age of the facility or type of equipment. It is therefore not generally available for a first measurement effort for a population of facilities. However, if the first measurement effort is properly constructed, results can provide an improved prior probability distribution for planning subsequent studies.

Continuous monitoring: Unlike a field campaign, a fenceline continuous monitoring (CM) system installed at an OOG facility operates a large fraction of the time, making it possible to detect a new emission source soon after it starts. Since the quantification accuracy of fenceline CM systems is highly uncertain, these systems often use high thresholds to avoid false positive detections, which are costly to operators. These systems also require time to identify elevated gas concentrations relative to background variations (or to detect visible emissions for camera systems), and typically wait for some time before issuing an ‘emissions detected’ alert. As a result, these semi-autonomous systems often have relatively low probability of detection in any given time period. But since they are continuously functioning, they have many sequential opportunities to attempt a detection.

Prior OOG measurements indicate a large fraction of emissions is due to a small number of large emitters. Therefore, the uncertainty of a continuously operating measurement system is highly affected by the speed at which the system can detect these emitters.

Method: For illustrative purposes, varying emitter size was not modelled. Instead, we considered an emitter that is above the lower detection limit of the CM system – that is, one that is detectable by the system, and persists until the system detects it. Obviously,

[7] https://carbonmapper.org/data/

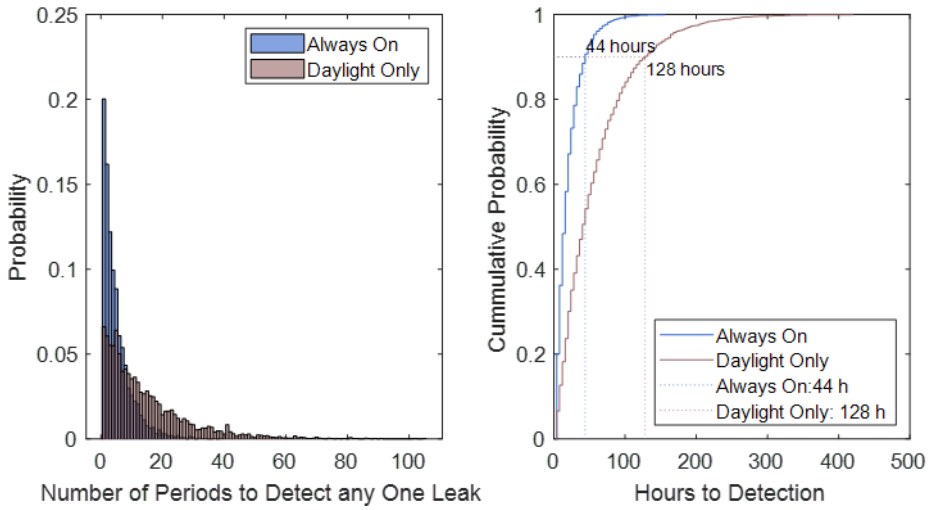
more sensitive CM systems will detect more emissions (and likely create more false alarms) than CM systems with less sensitivity.

Two fenceline CM systems were considered:

- A concentration sensor that operates continuously.
- A visible light sensor (for example, camera) which operates 8 hours per day, when sufficient ambient daylight is available (as with concentration sensors, camera CM systems are being developed for quantification as well as detection).

Both systems have lock-outs for extreme weather, which is not simulated here, but can be simulated using tools such as FEAST (Kemp and others, 2015). We assume the system takes 4 hours to collect and analyse data before making a detection[8], and each 4-hour period has a 20% probability of detecting the emitter, an intentionally conservative estimate. The detection process was simulated using 5,000 Monte Carlo iterations at a 4-hour time step. Simulation results are shown in Figure 6.7.

Figure 6.7: Continuous monitor detection simulation for a system with a 20% probability of detecting an emitter in a 4-hour period
Right panel illustrates a histogram of the number of periods before the emitter is detected. Left panel illustrates the time required to achieve a 90% probability of detection. The 'daylight only' system operates 8 hours/day. The 'always on' system operates 24 hours per day.



Results: While the method's probability of detection (a component of method uncertainty) is low (20%), continuous operation leads to a 90% probability of detection in 44 hours (11

[8] Solution developers testing with the ADED project at the Colorado State University (https://energy.colostate.edu/metec/aded/) provide time-to-detection in the range of 1 to 4 hours. The ADED project varies the duration of emitters from 1 to 8 hours.

periods) for the always-on system, and 128 hours for the system operating only daylight hours.

Note that this, as with the other simulations in sections 6.2-6.4, illustrates only one aspect of the recommended measurement effort simulation (step 4, above): that is, is the CM method aware that an emission is occurring? In a full simulation, each detection would be followed by an uncertain estimate of emission size.

Conclusions: A monthly survey method with a nearly 100% probability of detection has a time to detection of half of the survey period, or approximately 360 hours; quarterly surveys exceed 1000 hours. Therefore, a continuous monitor with a low probability of detection is likely to find an emitter faster than a survey method (assuming in both cases that the emitter is not found first via other means). Since total emissions are the product of time emitting multiplied by emission rate, and 50% or more of total emissions are due to large emitters, time-to-detection is a critical element to accurately quantifying emissions.

Recommendations: In the common situation where total emissions are dominated by large emitters, quantification may be improved by identifying large emitters as quickly as possible, estimating the duration of those emissions, and measuring their size – either with the CM system or a more accurate method. To improve emissions quantification, practitioners should consider disaggregating detection and duration estimation from quantification. One method could be used to detect larger emitters quickly and to estimate duration – functions which are currently supported by many fenceline CM systems. Quantification of those emitters could then be completed using a different method to improve quantification accuracy.

6.5. Method implementation uncertainty

Uncertainty for selected methods is discussed in section 5. The uncertainty analyses for these methods assumes:

- the method is implemented by well-trained personnel. Several of the methods are scientific in nature, and typically implemented by trained scientists; others require less expert knowledge
- quality control checks are properly implemented, and data failing quality control is discarded. Tracer flux, OTM33a, and OGI and Method 21 surveys have defined quality control actions which practitioners must use. Other methods are less rigorously defined, requiring user judgement, and therefore more expertise and caution
  Note that this is often a harsh requirement: downwind facility methods (tracer flux, OTM33a, mass balance) all require reasonably strong and steady winds. Therefore, it is not unusual for an entire day of measurement attempts to result in no successful measurements with these methods.
- the method is applied to the right type of facility and emitter, in suitable environmental conditions

There is no simple means to include implementation uncertainty in the measurement effort analysis (steps 1-5) recommended here. However, this uncertainty can be better

estimated, understood and reduced by considering how the method uncertainty was calculated in the first place, as discussed in section 5.

In the ideal case, method uncertainty is the result of controlled release testing in realistic conditions, over the range of emission sizes expected in use. For example, estimation of the OTM33a method uncertainty (section 5.1) used a relatively small, but non-trivial, set of controlled releases in an open field and in realistic equipment conditions. Analysis was done using a well-defined procedure. Therefore, uncertainty in (a) open conditions, and (b) similar equipment complexity, is likely well represented by the method uncertainty in section 5.1, provided proper procedures and quality control are applied.

The same method applied to different equipment types (for example, compressor stations), to emissions from an area source (for example, underground leaks) or in complex dispersion conditions (for example, wooded areas) is likely to have a different method uncertainty.

When method uncertainty is largely the result of theoretical studies or laboratory testing, results from prior test programmes indicate the uncertainty is almost always underestimated, often substantially (see Zimmerle and others, 2020b for an example).

Method implementation uncertainty can be substantially affected by the expertise of personnel carrying out or supervising the measurements. The tracer method must be implemented by trained scientists, whereas the OTM33a and mass balance methods can be implemented by technicians supervised by a trained scientist. Several academic groups in the UK are actively engaged in applying the methods described here and they could provide training and support to the measurement and operator communities. Tracking the qualifications and experience of personnel and ensuring adherence to method protocols and any available standards can help to reduce method implementation uncertainty.

Recommendations: Where possible, methods should be characterised using well-designed controlled release studies, in conditions reflective of where the methods will be used. Additionally, method protocols should be well documented, used in the controlled release studies, and required for field campaigns. Methods should be implemented by sufficiently trained and experienced personnel.

6.6. Assessing total uncertainty

To assess total uncertainty for any measurement effort, we recommend simulating the measurement effort as outlined at the start of this chapter. Where possible, empirical uncertainty (actual controlled testing results) distributions should be used for methods. These should be combined with a robust prior estimate of emissions that includes the size of the emissions, skew in the emissions distribution, and temporal variation if used with long-duration measurements. If desired, this estimate should also divide emissions between those of interest (for example, fugitive emissions) and the background of expected emissions.

Summarising other conclusions from this section:

1) Temporal or facility-to-facility variability and skew in emissions will often contribute uncertainty ‘in the same order of magnitude’ as the method uncertainty.

2) Uncertainty from (1) is convolved with method uncertainty, and the results are typically unpredictable without simulation. This is particularly true if the emissions distribution is highly skewed (typically the case), and the method uncertainty is asymmetric (also typically the case).

3) To measure emissions over an extended period (case 2) emissions must be both ‘detectable’ and ‘quantifiable’. Additionally, estimating ‘total’ emissions also requires the ‘duration’ of temporally varying emissions to be understood. As a result, methods which make many measurement attempts – even if highly uncertain – may produce lower total uncertainty than a more certain method used less often. Again, simulation is typically required to compute these uncertainties.

4) Currently, methods which make many measurements quickly and inexpensively have higher uncertainties than methods which have lower uncertainty but take more time and cost more. This dichotomy is likely to persist for the foreseeable future. In these conditions, a hybrid of 2 methods, one intended to identify and quantify large emitters quickly, together with a second method that is used less frequently but detects most emitters and quantifies with less uncertainty is likely to minimise total uncertainty for a given measurement investment. Any hybrid approach can be (at least approximately) optimised by simulating the measurement approach with the method recommended here.

5) Any measurement effort should clearly define the quantity being measured. As noted in measurement case 1, any of the methods described in section 5 can produce a whole-site measurement, with uncertainty that can be estimated. If a spot-check of emissions is desired, any one measurement in case 1 is a complete result. However, if the goal of measurement is to characterise a population of facilities (for example, a ‘field campaign’) or to calculate total emissions from one or more facilities over an extended period (for example, annual emissions), then it is necessary to simulate, at a minimum, both sampling and temporal uncertainties, as these will typically be as large, if not larger, than the uncertainty of the measurement method.

7. Selecting quantification method for different site types

This section discusses the selection of quantification methods by site type. Additionally, a qualitative cost analysis of the methods is presented. The purpose for which methane emissions quantification is being carried out (such as scoping assessment, routine monitoring or research study; see Finlayson and others, 2021) drives the required uncertainty of the results. This criterion, as well as the number of separate measurements required to characterise temporal variability at the same facility or variations between facilities (see section 6), will dictate the selection of the most cost-effective method.

7.1. Selection table

A selection table can be used to assess the suitability of the identified methods for each site type. As previously discussed in section 2, there are 4 main types of sites within England:

- **small production** sites located in aerodynamically **complex topography**
- **small production** sites located in aerodynamically **simple topography**
- **large production** sites located in aerodynamically **complex topography**
- **large processing** sites located in aerodynamically **simple topography**

Each type of site presents its own challenges for methane measurement and therefore there is no one method most suitable for all. Table 7.1 presents the method selection table developed as part of this work. It includes information on the most suitable methods, and their associated uncertainty and costs. Table 7.1 also identifies any method that should not be used for each type of site as the uncertainty associated with the emission estimate is prohibitively high, as a result of aerodynamically complex topography (trees, slopes, valleys and buildings) or the methods not being able to measure all emission sources.

Table 7.1 presents 2 types of information on the method uncertainty:

- the uncertainty bounds, which are represented by green/amber/red colour coding
- the approach used to determine the uncertainty bounds which is represented by the number 1-3, explained in the key below the table

Indicative field time, field cost, equipment costs, analysis costs and total costs are provided in section 7.2. Additional detail on the advantages and disadvantages of each method and the reasons for the ranking provided are discussed in sections 7.3 to 7.6.

Table 7.1: Method selection table

Site type	Description of location	Potential emission sources	Preferred methods						Prohibitively uncertain methods
				1	2	3	4	5	
Small production site, complex topography	Wooded with complex aerodynamics	Relatively complex with some processing	**Method**	Tracer method	Mass balance	OTM33a	Component-level measurement		Fenceline measurement
			Method Uncertainty	2	2	1	1		
			Cost	£	££-£££	£	£		
Small production site, simple topography	Open setting	Wellheads, separator and condensate tanks only	**Method**	OTM33a	Component-level measurement	Tracer method	Fenceline measurement	Mass balance	
			Method Uncertainty	1	1	2	3	2	
			Cost	£	£	£	£	££-£££	
Large production site, complex topography	Wooded with complex aerodynamics	Multiple wellheads, on-site processing	**Method**	Tracer method	Mass balance	OTM33a			Fenceline and component-level measurement
			Method Uncertainty	2	2	1			
			Cost	££	££-£££	£			
Large processing sites	Very large and complex site. More open setting but aero-dynamically complex topography	Large number of individual sources	**Method**	Tracer method	OTM33a	Mass balance	Fenceline measurement		Component-level measurement
			Method Uncertainty	2	1	2	3		
			Cost	££	£	££	£-££		

Uncertainty key

Uncertainty range colour code	Uncertainty bounds
	<±20%
	±20% to ±50%
	>±50%

Uncertainty approach code	Approach used to determine uncertainty bounds
1	Controlled release with published data
2	Published, desk-based analysis
3	No data – Analysis through modelling

7.2. Costs

In addition to the uncertainty, the cost of methods needs to be accounted for when selecting the most suitable method for quantifying whole-site emissions. While a quantitative cost analysis has not been carried out, the relative costs of each method have been assessed qualitatively.

The factors taken into account when assessing the costs of each method are:

- time and staff requirements for measurement in the field
- time and cost of getting to/from the site
- equipment costs
- analysis time

In addition to the time required for measurement, there is significant variation in the number of staff and their skillsets that are required for field measurement. Staff requirements for analysis do not vary as much between methods for analysis. It is estimated that the analysis time will be roughly one day of analysis per day in the field for an experienced team and will involve one analyst and a scientist for supervision. Table 7.2 summarises the field time and staff required for each method by site size. Field time covers time setting up and collecting data; time for administrative matters such as safety briefings is not included here. As flux plane methods can be carried out using drones or aircraft and the associated costs vary between the two, these have been analysed separately. The resulting total costs from the measurement, equipment and analysis are then summarised in Table 7.3.

Table 7.2: Indicative field time and costs

Site size	Method	Estimated required field time (hours)	Field staff	Field cost
Small	OTM33a	2	Two senior technicians and a fraction of office support scientist	££
Small	Tracer method	5.2	One scientist, one support staff	££
Small	Mass balance, aircraft	4	One technician, one pilot and a fraction of office support scientist	£££
Small	Mass balance, drone	1.67	One technician, one pilot and a fraction of office support scientist	£££
Small	Component-level measurement	2	Two technicians	£
Small	Fenceline	2	Two technicians and a fraction of office support scientist	£
Large	Tracer method	27.2	One scientist and one support staff	£££
Large	Component-level measurement	16	Two technicians	££
Large	Mass balance, aircraft	5	One technician, one pilot and a fraction of office support scientist	£££
Large	Mass balance, drone	3	One technician, one pilot and a fraction of office support scientist	£££
Large	OTM33a	3	Two senior technicians and a fraction of office support scientist	££
Large	Fenceline	4	Two technicians and a fraction of office support scientist	£

Table 7.3: Indicative costs

Site size	Method	Field cost	Equipment cost	Analysis cost	Indicative total cost
Small	OTM33a	££	£	£	£
Small	Tracer method	££	£	£	£
Small	Mass balance, aircraft	£££	££	£	££
Small	Mass balance, drone	£££	£££	£	£££
Small	Component-level measurement	£	££	£	£
Small	Fenceline	£	£	£	£
Large	Tracer method	£££	£	££	££
Large	Component-level measurement	££	££	£	££
Large	Mass balance, aircraft	£££	££	£	££
Large	Mass balance, drone	£££	£££	£	£££
Large	OTM33a	££	£	£	£
Large	Fenceline	£-££	£-££	£	£-££

7.3. Small facilities (aerodynamically complex topography)

As shown in the method selection table, the preferred methods for small facilities located in wooded areas are 1) tracer method, 2) mass balance, 3) OTM33a and 4) component-level measurement. Fenceline measurement is considered prohibitively uncertain.

The advantages and disadvantages of these methods for these sites are summarised in Table 7.4. The limiting factor for many of the methods is the complex aerodynamic environment created by the surrounding trees. These limit the air flow and make plume modelling, due to the assumptions about meteorology required, and measurement of micrometeorology difficult.

Table 7.4: Reasons for preferred methods for small facilities in aerodynamically complex topography

Method	Advantages	Disadvantages
Tracer method	Allows emissions in a complex aerodynamic environment to be quantified as no micrometeorology measurement or plume modelling is required.	Access to emissions downwind from the site is required, which is not always possible for these smaller sites.
Mass balance		A reasonably flat fetch is required and the surrounding trees mean that the gas may not be entrained in the air; outflow may not be representative of gas emission rate. Additionally, measurement needs to be done between 30 m and 50 m from the source, which may still be within the treeline.
OTM33a	This method is potentially suitable if measurement is done very far downwind, 20 times the height of the surrounding trees. Emissions would therefore need to be large enough. Measurement is quick compared to other methods and only requires an analyser capable of measuring one gas.	Plume modelling is required and therefore good measurement of micrometeorology and assumptions on these parameters is also required. This will result in high uncertainty for these sites in the complex aerodynamic environment.
Component-level measurement	If the number of equipment is low, it is unlikely that an emission source is missed.	Conversely, if there is a higher number of equipment, the site is too complex for this method and a source is likely to be missed. Additionally, the method inherently has a low bias for emissions.

Fenceline measurement	If left deployed, a timeseries of emissions can be collected, which reduces the uncertainty.	Similar disadvantages to plume-based flux recovery due to the complex aerodynamic environment. The vegetation could additionally mean that the gas is not entrained in the air and the outflow therefore not representative of the gas emission rate. Additionally, the monitors are stationary and therefore will generate an emission estimate only when the wind is in the correct direction.

7.4. Small facilities (aerodynamically simple topography)

The preferred methods for small facilities located in areas without surrounding vegetation are 1) OTM33a, 2) component-level measurement, 3) tracer method, 4) fenceline measurement, 5) mass balance. No methods were identified as prohibitively uncertain due to the relatively open setting of these facilities and their small size. Table 7.5 presents the advantages and disadvantages of the methods for these types of small sites.

Table 7.5: Advantages and disadvantages of methods for small sites in aerodynamically simple topography

Method	Advantages	Disadvantages
Tracer method	Highly accurate method if downwind road access is available. Method can generally separate on-site emissions from those of nearby sources.	Requires an analyser capable of measuring 2 gases, and measurement takes longer than other methods, making this an expensive and over-complicated method if a simpler method could be used.
Mass balance	It is possible to get within 30 to 50 m of the source at these sites.	There is high uncertainty in measurements at smaller sites due to the low emissions.
OTM33a	Measurement is quick compared to other methods and only requires an analyser capable of measuring one gas. The open setting allows for easy measurement of the micrometeorological parameters required reducing uncertainty.	Method is inherently highly uncertain.
Component-level measurement	If the number of equipment is low, it is unlikely that an emissions source is missed.	Conversely, if there is a higher number of equipment, the site is too complex for this method and a source is likely to be missed; therefore, the method inherently biases emissions low. More time and expertise are required compared to other methods, such as plume-based flux recovery.
Fenceline measurement	If left deployed, a timeseries of emissions can be collected, reducing uncertainty.	The monitors are stationary and therefore will only generate an emission estimate when the wind is in the correct direction. Higher uncertainty due to the assumptions required in micrometeorology as these parameters are unlikely to be measured constantly.

7.5. Large production facilities (aerodynamically complex topography)

As previously discussed, large OOG production facilities in England are located in aerodynamically complex topography (wooded) areas. The preferred methods for these facilities are 1) tracer method, 2) mass balance, and 3) OTM33a. Fenceline and component-level measurement were identified as prohibitively uncertain (fenceline) or costly (component-level measurement). Table 7.6 presents the advantages and disadvantages of the methods for these large production sites. As with smaller facilities located in wooded areas, the limiting factor for many of the methods is the complex aerodynamic environment created by the surrounding trees, requiring assumptions about meteorology and making measurement of the micrometeorology difficult.

Table 7.6: Advantages and disadvantages of methods for large production facilities in aerodynamically complex topography

Method	Advantages	Disadvantages
Tracer method	Allows emissions in a complex aerodynamic environment to be quantified as no micrometeorology measurement or plume modelling is required.	Downwind road (or marine) access required; 'downwind' depends on current meteorological conditions, making planning difficult.
Mass balance		A reasonably flat fetch is required and the surrounding trees mean that the gas may not be entrained in the air; outflow may not be representative of gas emission rate. Additionally, measurement needs to be done between 30 m and 50 m from the source, which may still be within the treeline.
OTM33a	This method is potentially suitable if measurement is done very far downwind, 20 times the height of the surrounding trees. Emissions would therefore	Plume modelling is required and therefore good measurement of micrometeorology and assumptions on these parameters is also required.

	need to be large enough. Measurement is quick compared to other methods and only requires an analyser capable of measuring one gas.	This will result in high uncertainty for these sites in the complex aerodynamic environment.
Component-level measurement	If the number of equipment is low, it is unlikely that an emissions source is missed.	Conversely, if there is a higher number of equipment, the site is too complex for this method and a source is likely to be missed.
Fenceline measurement	If left deployed, a timeseries of emissions can be collected. Monitors can be instrumented to detect large emitters quickly, allowing fast response to abnormal emissions.	Similar disadvantages to plume-based flux recovery due to the complex aerodynamic environment. The vegetation could additionally mean that the gas is not entrained in the air and the outflow therefore not representative of the gas emission rate. Additionally, the monitors are stationary and therefore will only generate an emission estimate when the wind is in the correct direction.

7.6. Large processing facilities

Large OOG processing facilities within the UK are usually in aerodynamically simple topography. The preferred methods for large processing facilities are 1) mass balance, 2) OTM33a, 3) tracer method, 4) fenceline measurements. Component-level measurement was identified as prohibitively costly and may be challenged by the variety of emissions sources. Table 7.7 presents the advantages and disadvantages of the methods for these large production sites. The limiting factor for many of the methods is the size and complexity of the site, which can create a complex aerodynamic environment.

Table 7.7: Advantages and disadvantages of methods for large processing facilities

Method	Advantages	Disadvantages
Tracer method	Allows emissions in a complex aerodynamic environment to be quantified as no micrometeorology measurement or plume modelling is required.	Emission estimates are more uncertain when unable to co-locate tracer gas with the emission source. Downwind road access required.
Mass balance	A drone mounted trace gas analyser can capture the impact of multiple individual sources. It is possible to get within 30 to 50 m of the source at these sites. Emissions likely to be large enough to minimise the uncertainty.	
OTM33a	Measurement is quick compared to other methods and only requires an analyser capable of measuring one gas. The open setting allows for easy measurement of the micrometeorological parameters required, reducing uncertainty.	Measurement would need to be done far enough downwind so that the plume is that of a point source. Measurements may therefore be confounded by unknown sources between facility and detector.
Component-level measurement	.	The high number of equipment makes these sites too complex for this method and a source is likely to be missed. Additionally, the method inherently has a low bias for emissions.
Fenceline measurement	If left deployed, a timeseries of emissions can be collected. Monitors can be instrumented to detect large emitters quickly, allowing fast response to abnormal emissions.	For large facilities, fenceline methods are unlikely to produce high quality measurement of total facility emissions but will be capable of quickly detecting upset conditions that lead to large emitters.

8. Conclusions and recommendations

8.1. Uncertainty determination

8.1.1. Method uncertainty

For this study, we consider evaluation with controlled releases as the best available test for assessing method accuracy. Unfortunately, a review of a wide selection of methods indicates that little data has been released, particularly in peer-reviewed sources, to characterise method accuracy. Many papers mention ‘controlled testing’, but release no data for the controlled releases, while other papers rely on prior publication of similar methods as a justification for both use and uncertainty estimates.

For each method in this study the most commonly implemented approach was considered when assessing uncertainty. Practitioners currently make wide-ranging modifications to a particular method, often to speed up measurement times or to tailor measurement methods to a particular site type. While these modifications may provide a degree of optimisation, the resulting variations may decrease the comparability between studies carried out by different vendors and the resulting measurements.

8.1.2. Sampling uncertainty

Total uncertainty for a population-wide field campaign (or an equivalent series of measurements from a group of facilities) depends not only on the measurement method uncertainty, but also on the variability, skew and prevalence of outliers in the measured facilities' emissions (see section 6). Therefore, field campaigns and reporting programmes should be assessed for total uncertainty using empirical predictive power calculations that account for the non-normal distribution of facility emissions, and where applicable, skew in the uncertainty of the method. Empirical predictive-power calculations are recommended, as emissions distributions from OOG facilities are often highly skewed. When, as is often the case, budget limits the size of field campaigns or the completeness of reported data, simulated field campaign results support pre-campaign uncertainty assessment to determine if the campaign will meet programme objectives.

8.1.3. Measurement purpose

The best methods for identifying large emitters, which typically account for the majority of emissions at most OOG facilities, may not produce whole-site quantification of emissions. Often these methods are substantially faster than methods producing whole-site quantification. Therefore, the choice of method depends on the objective of the measurement campaign. If the objective is to identify and repair the majority of emissions as quickly as possible, frequently using methods that identify large emitters may be preferable to less frequently using methods that producing whole-site emission rates. Conversely, if the objective is to characterise all emissions from a group of facilities (or one facility over a period of time), selected methods must produce whole-site quantification estimates; large emitter detectors will inherently fail to capture some emissions.

8.1.4. Method development and testing

Methods presented here use well-known, open-source approaches for computing emissions from sensor outputs. Tracer method, OTM33a and component-level measurements use well-understood algorithms for computing emission rates. For other methods, however, there are both variations in open-source algorithms and numerous proprietary algorithms. Many solution developers are concentrating on fenceline monitoring (or similar continuous monitoring systems) using a wide variety of algorithmic approaches. These include a range of plume models; statistical or machine learning approaches that assume little-to-no physical model; and computational fluid dynamics algorithms that model each facility in substantial detail – an approach aided by laser detection and ranging (LIDAR) structure mapping. These algorithms are developing rapidly and may produce superior results – or surprising failures – relative to the open-source algorithms described here. Most of these developments are also proprietary and difficult to assess without controlled testing. Each implementation also has a range of settings which, when adjusted, can cause a 2:1 or higher change in both accuracy and detection limits.

8.2. Recommendations

8.2.1. Review or engage in controlled testing

Controlled testing for the selected methods should be reviewed or carried out to better understand the method uncertainty of each particular method. Well-designed single-blind trials using controlled releases, in as realistic an environment as possible, are strongly recommended to characterise method detection limits and quantification accuracy, as a function of environment variables (wind speed, solar irradiation) and of emissions rate and type. A project funded by the US Department of Energy at the Colorado State University has developed protocols for this type of testing[9]. Controlled testing should also be combined with field verification to assure that controlled test results reflect field performance. The downside of controlled testing is that the results are expensive, limited in realism relative to field conditions, and are difficult to make single-blind, particularly for aircraft-based methods.

8.2.2. Combine methods

Paired-method studies have been highly informative on the performance of methods in field conditions (Zimmerle and others, 2015, Bell and others, 2017, Vaughn and others, 2017). Combining component-level measurements with multiple downwind whole-site methods also supports analysis of how emissions of different types affect accuracy of the whole-site measurements. If implemented as part of a measurement study, paired methods – even if implemented on a subset of facilities – provide excellent quality control on measurement results. However, paired-method studies are expensive, and small differences in the timing and duration of measurements cause uncertainties that are difficult to control.

[9] https://energy.colostate.edu/metec/aded/

Methods could be combined, where lower-cost, higher-uncertainty methods (OTM33a) are used frequently and calibrated against higher-cost, lower-uncertainty methods (tracer) less frequently. This would supply a greater amount of data that could be used to infer any temporal variability in emissions, while helping to reduce costs of measurement. Similarly, fenceline measurement, which can be used to detect larger emitters quickly and to estimate their duration, could be periodically calibrated with tracer measurements. This does not remove the relatively high uncertainty in the OTM33a or fenceline measurements, but gives more confidence in the methods' ability to quantify emissions and to be used for initial screening purposes followed by use of higher-cost, lower uncertainty methods if necessary.

8.2.3. Standardise application of methods

The field performance of any method is highly dependent on quality control and staff expertise, before, during and after the measurement campaign. Main elements and the application of the selected methods should be standardised to improve comparability and to ensure a minimum quality standard for selected aspects of the relevant method. Method protocols should be well-documented, used in the controlled release studies, and required for field campaigns.

The European Committee for Standardization (CEN) is currently developing a new standard 'Fugitive and diffuse emissions of common concern to industry sectors – Standard method to determine diffuse emissions of methane into the atmosphere'. This new standard is expected to include a toolkit of methods similar to those in the recently finalised EN 17628 standard (CEN 2022) which is focused on refinery VOC emissions. Learnings from this study can feed into the development of the new standard, which will ultimately be a useful resource for practitioners and for ensuring consistent and thorough documentation of whole-site emissions measurements.

Method implementation uncertainty can be substantially affected by the expertise of personnel carrying out or supervising the whole-site measurements. Several academic groups in the UK are actively engaged in applying the methods described here to measure whole-site methane emissions from oil and gas facilities as well as sources in other sectors, and they could provide training and support to the measurement and operator communities. Nevertheless, availability of skills may be a limiting factor on the application and uptake of these methods.

8.2.4. Evaluate new and emerging methods

For new and developing methods, the following should be considered:

1) Systems provided by individual vendors may need to be tested separately to understand uncertainty.
2) Testing programmes should include contingencies for periodic retesting.
3) Settings used in testing and in deployment should be well documented and audited.
4) Applicability of methods may change rapidly.

As an example of testing programmes, see the ‘Advancing Development of Emissions Detection’ (ADED) project (https://energy.colostate.edu/metec/aded/), and specifically the testing protocols developed by that project.

8.3. Transferability to other sources and pollutants

8.3.1. Quantifying methane from other sources

The methods outlined in section 5 can be used to quantify emissions from point sources in other sectors provided a sufficiently strong signal above background for the plume of emissions from the facility can be detected. For example, these methods have been applied to biogas plants and wastewater treatment plants (Bakkaloglu and others, 2021, Knudsen and De Rossi 2022). Furthermore, in some cases, the methods are also transferable to quantifying emissions from area sources with diffuse emissions. It will be important to consider the instrument’s sensitivity (signal-to-noise ratio), the method’s suitability for the area footprint, and the method’s sensitivity to complex topography if the source is aerodynamically complex.

The mass balance method has been shown to successfully estimate methane emissions from landfill sites (Abichou and others, 2010, Goldsmith and others, 2012, Fredenslund and others, 2018, Allen and others, 2018, 2019). Both the plume-based flux recovery (OTM33a) and tracer gas methods are also capable of measuring emissions from area sources, provided that the sensors are located at a distance from which the area can be treated as a point source. Plume-based flux recovery methods have been used to quantify landfill emissions (Hensen and Scharff, 2001, Foster-Wittig and others, 2015, Riddick and others, 2017, Terent’eva and others, 2017) and agricultural emissions (Flesch and others, 2005, 2013, Bjorneberg and others, 2009, Ro and others, 2012). Tracer methods have been used to quantify emissions from landfills (Börjesson and others, 2009, Scheutz and others, 2011, Mønster and others, 2015), wastewater treatment plants (Delre and others, 2017, Samuelsson and others, 2018), biogas plants (Scheutz and Fredenslund, 2019) and livestock production facilities in North America (Arndt and others, 2018, Daube and others, 2019). As well as downwind road access at a sufficient distance, a strong signal and the co-location of the tracer gases are required in order for the tracer gas method to be successfully applied to larger area sources.

The plume-based flux recovery, tracer and mass balance methods have all been applied to other sectors. Many of the learnings from this report on method uncertainty (section 5) and on other sources of uncertainty (section 6) will be relevant when quantifying methane emissions from both point and area sources in other sectors.

8.3.2. Quantifying other substances

The methods discussed in this report can also be used for quantifying emissions of other light gas species, for example, nitrous oxide or ammonia, provided the instrument used is capable of detecting the gas species, that the expected concentrations are within the detection threshold and that a difference from background mixing ratios can be discerned. Within the dispersion models there is no differentiation for gas speciation, therefore, the use of one method can be ported between gases.

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List of abbreviations

ADED	Advancing Development of Emissions Detection
BLs	Backward Lagrangian stochastic
CDF	Cumulative distribution function
CM	Continuous monitoring
CRDS	Cavity ringdown spectroscopy
DIAL	Differential absorption LiDAR
EPA	Environmental Protection Agency
GMAP	Geospatial measurement of air pollution
HFS	High flow sampling
LDAR	Leak detection and repair
LDL	Lower detection limit
LiDAR	Light or laser detection and ranging
M21	(US EPA) Method 21
METEC	Methane Emissions Technology Evaluation Center
NDC	Normally distributed control
OGI	Optical gas imaging
OGA	Oil and Gas Authority
OOG	Onshore oil and gas
OTM33a	(US EPA) Other Test Method 33a
PGSC	Pasquill-Gifford stability class
PI	Pollution Inventory
QC-TILDAS	Quantum cascade tunable infrared laser differential absorption spectroscopy
SOF	Solar occultation flux
US EPA	United States Environmental Protection Agency
VOC	Volatile organic compound

Glossary

Area source – Source of emissions released to the atmosphere from an extended area, for example, the surface of a wastewater treatment pond.

Backward Lagrangian stochastic model – Calculation of an ensemble of fluid element (particle) trajectories that are distinguished by each passing through an observation point.

Component-level measurement - Identifying and measuring emissions from all equipment types, typically in close proximity to each individual source at a facility.

Condensate tank – Tank used to store light liquid hydrocarbons.

Continuous monitoring – Permanently installed systems making regular measurements over an extended period of time.

Emission type uncertainty – Ability to distinguish between planned or expected emissions and unexpected/unpermitted/abnormal emissions.

Fenceline measurement – Use of sensors fixed to poles or similar structures at the fenceline of the facility paired with analytics to convert sensor readings into emissions detections.

Flux plane – An aircraft or drone flies a plane upwind and downwind of the facility, building up a concentration map between the ground and the aircraft.

Fugitive emissions – Fugitive emissions are specifically defined within EN 15446 as an ‘emission to the atmosphere caused by loss of tightness of an item which is designed to be tight’. These can therefore be considered leaks and are often the subject of LDAR programmes which aim to identify and fix the leaks.

Gaussian plume model – A formula that describes concentrations in 3-dimensions generated by a point source under steady meteorological and emission conditions.

Mass balance method – Spatial characterisation of the incoming and outgoing concentration of the target gas.

Method implementation uncertainty – Deviation from implementation of a method by expert practitioners in near-ideal conditions.

Method uncertainty – The expected accuracy of a measurement made with a particular method.

OTM33a (Other Test Method 33a) – A US EPA method that formalises plume-based flux recovery.

Plume-based flux recovery – Use of instrumented, ground-based vehicles to identify and then measure concentration enhancements in a plume produced from a point source, and to then model the dispersion to derive an emission rate.

Point source – A specific localised source of emissions, such as a vent stack. In practical terms, a point source is one giving rise to a narrow plume of emissions.

Pressure relief valve – Relief valve that ensures equipment is not subjected to pressure higher than its design pressure.

Processing site – Location which processes gas produced at other locations to recover natural gas liquids (condensate, natural gasoline and liquefied petroleum gas) and to remove impurities such as water, carbon dioxide, VOCs and other heavy hydrocarbons and condensates.

Production site – Location with at least one wellhead. Some initial processing may take place where gas, oil and produced water are separated. Some liquids may be stored on site prior to collection by road tanker.

Sampling uncertainty – Potential variation in the composition of a sample.

Survey method – Intensive measurements taken over a short period of time on a single or periodic basis.

Temporal variability – Change in activity and emissions within the period being studied.

Separator – Pressure vessel used for separating a well stream into gaseous and liquid components.

Total uncertainty – Aggregation of uncertainty from all sources, including method uncertainty, sampling uncertainty, temporal variations and method implementation uncertainty.

Well pad – Cleared area encompassing a wellhead and associated drilling and production equipment.

Wellhead – Component at the surface of an oil or gas well that provides the structural and pressure-containing interface for the drilling and production equipment.

Tracer method – Tracer gases are emitted at known rate next to the site of interest. The tracer and pollutant concentrations are then measured downwind to infer the pollutant emission rate from the known release rate of the tracer gases.

Whole-site emissions – The combined plume of emissions from all sources at a site. Also referred to as ‘full facility’ or ‘facility-wide emissions’.

Appendix: Main characteristics of methane emissions quantification methods

Table A1: Main studies for each method, where each method has been used and the approach to uncertainty

Reference	Method class	General description	How/where used	Uncertainty based on controlled release or desk-based study?	Description of controlled release
Albertson and others (2016)	Bayesian convergence survey	Describes a multilevel approach to detect leaks and quantify release rates.	Well pads	Controlled release	Limited controlled release, one release rate measured three times.
Alden and others (2019)	Line sensor	Open-path dual frequency comb laser spectrometer is used to monitor methane concentrations and is combined with met data in an inversion to characterise emissions from a site.	METEC	Controlled release	Single blind leak detection tests performed at the METEC facility.
Allen and others (2013)	Whole-site tracer flux	Measurement of methane at 150 US natural gas sites was carried out and compared to EPA	Onshore natural gas sites	Desk	-

		estimates from the national inventory.			
Allen and others (2019)	Flux plane	Unmanned aerial system (UAS) used to make proxy measurements of CO_2 concentration and wind data to infer methane flux.	Landfill sites	Desk	-
Bell and others (2017)	Whole-site tracer flux; Plume-based flux recovery	Compares OTM33a with the dual tracer flux method.	Gas production facilities	Desk	-
Bell and others (2020)	Various	A test of 12 different next generation emission measurement technologies, carried out under controlled conditions at METEC.	METEC	Controlled release	Tests involved both single and multiple emission sources. Single emission sources were operated at a continuous rate, multiple emission sources operated both continuously and intermittently.
Brantley and others (2014)	Plume-based flux recovery	OTM33a was used to quantify short-term emissions.	OOG production pads	Controlled release	A large number of single point releases were measured in obstruction free areas.

Caulton and others (2019)	Plume-based flux recovery	Mobile lab used to measure emissions at unconventional natural gas well pads.	OOG well pads, large emission sources	Controlled release	Limited controlled releases.
Coates and others (2017)	Eddy covariance; Backward Lagrangian stochastic	Combination of 2 methods used to measure continuous emission source.	Cattle paddock	Controlled release	CH_4 released at 8 points in test field and measured over 15-minute periods.
Conley and others (2017)	Flux plane method; Point sensor or line sensor	Presents an airborne method for sampling sources of trace gas and investigates the relationship between the time spent sampling and the accuracy of the result.	OOG facilities	Controlled release	Single point sources are measured at 2 sites.
Connolly and others (2019)	Component methods - high flow sampling	Investigated the operation of Bacharach Hi Flow® sampler by splitting into 3 modes: catalytic oxidation, thermal conductivity, and transition region. Operational guidelines developed as part of this work.	Lab testing	N/A	CH_4 gas in varying concentrations delivered direct to equipment.
Edie and others (2020)	Plume-based flux recovery	Studies the accuracy of OTM33a and explores several approaches to assessing accuracy.	Oil and gas production facilities	Controlled release	Test releases for both single and multiple points, with different source heights and release rates.

Feitz and others (2018)	Various	Compares 8 quantification methods for both CH_4 and CO_2. All estimates of methane emissions were found to be within 20% of the actual release rate, with the tracer ratio method providing the closest estimate. Focus is on instruments.	Open field	Controlled release	Single point source, 3 release rates.
Foster-Wittig and others (2015)	Plume-based flux recovery	Describes 2 Gaussian plume inverse approaches for measuring ground-level continuous emissions.	Ground level point sources	Controlled release	Controlled release data gathered from 14 studies.
Gao and others (2009)	Backward Lagrangian stochastic methods	Measures continuous emissions from agricultural facilities over a period of several days.	Agricultural facilities	Controlled release	CH_4 was released at a continuous rate over 5 days from a 2 m x 2 m grid at the centre of the enclosure.
Humphries and others (2012)	Bayesian convergence survey	Method is tested using controlled releases of N_2O and CO_2.	Point source in open field	Controlled release	Single point on field site (2 sources; one known and one unknown).
Lamb and others (1995)	Whole-site tracer flux	Estimates emission rate by comparing downwind plume mixing ratio of methane to mixing	Whole site (OOG), urban areas	Controlled release	Small scale, controlled releases. Measurements initially

		ratio of a tracer gas released at a known rate.			made at a fixed position 50 m from the release point and then later from traversing through plume 100 m downwind of source.
Mauder and others (2013)	Eddy covariance	Data gathered at several hundred sites is used to characterise ecosystem exchanges of trace gases, water and energy.	Ecosystems, long-term measurements	Desk	-
Mønster and others (2019)	Various	Reviews various methods for whole-site methane quantification. Methods measuring downwind to the landfill such as tracer gas dispersion and DIAL were thought to be best, while aerial measurement methods showed promise.	Landfill sites	Desk	-
O'Connell and others (2019)	Plume-based flux recovery	Applies a Gaussian dispersion model to measure continuous emissions data at well pads located in Alberta, Canada. Suggests the method could be combined with standard approaches such as OGI and LiDAR.	Well pads	Controlled release	Release rates lower than those observed in the field study and raw results not presented.

Peterson and others (2017)	Point sensors	Metal oxide semiconductors were used to measure ozone and NO_2 concentrations. Methane was not the focus of the study but was detected.	Urban environments, industrial safety systems	Desk	-
Rella and others (2015)	Flux plane	Ground based method used at OOG facilities.	Well pads	Desk	-
Riddick and others (2017)	Backward Lagrangian stochastic method	Compares different approaches to estimating emissions at near source, middle distance and the landscape scale.	Landfill	Desk	-
Riddick and others (2019)	Plume-based flux recovery	Fishing boats used to collect methane concentration data downwind and upwind of oil and gas platforms in UK waters.	Offshore oil and gas platforms (UK)	Desk	-
Ro and others (2011)	Plume-based flux recovery; Backward Lagrangian stochastic (bLs) method	Vertical radial plume mapping (VRPM) and the bLS methods are compared.	Open field	Controlled release	Single and multiple emission sources used.
Roscioli and others (2015)	Whole-site tracer flux	Dual tracer gases used to quantify emissions at a range of facilities. On-site infrared imaging and	Gathering and processing facilities	Desk	-

		equipment surveys also carried out.			
Sherwin and others (2021)	Whole-site tracer flux; Plume imaging and quantification: airborne	The quantification abilities of an airplane based hyperspectral methane detection imaging system is tested for 'super-emitting' sources.	Large emission sources	Controlled release	Large-volume single-blind controlled releases with some negative controls.
Subramanian and others (2015)	Whole-site tracer flux	Equipment and site-level methane emissions from compressor stations measured with downwind tracer flux techniques and compared to direct measurements of fugitive and vented sources. The US EPA emission factors were shown to be comparable to the average methane emissions of the non-super emitters.	Compressor stations (gas transmission and storage)	Desk	-
Thorpe and others (2016)	Plume imaging and quantification: airborne	An airborne visible/IR imaging spectrometer, equipment not originally designed for methane detection, was used for high resolution mapping of methane seeps at various OOG facilities.	Oil field testing centre	Controlled release	Controlled releases carried out over 6 days at 3 sites.

Tratt and others (2014)	Plume imaging and quantification: airborne	Emissions are quantified using a hyperspectral camera mounted on an aircraft and plume constituents are identified.	Various	Controlled release	Single emission source located above ground level.
Yacovitch and others (2015)	Whole-site tracer flux	Dual tracer gases are used to measure emissions and constrain source locations.	OOG facilities	Desk	-
Yacovitch and others (2020)	Plume-based flux recovery	Gaussian inversion methods used to estimate methane emission rates, with shipboard measurements taken during periods of favourable weather conditions.	Offshore oil and gas platforms	Desk	-
Zeng and others (2017)	Leak detection methods for component sources	Response factors were derived for a number of hydrocarbons (methane included) based on detection of propene.	None	Desk	-
Zeng and others (2019)	Leak detection methods for component sources	Investigates the detection limits of OGI.	OOG, LDAR	N/A	-
Zimmerle and others (2020b)	Leak detection methods for component sources	Paper focuses on the human factors affecting OGI use for leak detection. The experiment was	METEC	Controlled release	Single and multiple emission sources used, with realistic

		designed to study leak detection performance (surveyors all brought their own OGI cameras) and is intentionally biased towards low emission rates.			variations in weather conditions.

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