Quantification of landfill methane emissions – a method comparison study

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This report presents the results of monitoring of methane emissions from three landfill sites located in Sweden using; the tracer gas dispersion method, UAV-based and ground-based plume measurements and surface flux chambers provided by the Technical University of Denmark (DTU), FORCE Technology, ReSource, Lund University, Sweco Environment, respectively. The results from these monitoring techniques are compared. The monitoring campaign took place during the last week of October 2019. This study is financed by the participants in this project, including the Swedish Waste Management Association (Avfall Sverige).

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Executive Summary

This study, commissioned by Technical University of Denmark (DTU), Lund University, Sweco Environment, FORCE Technology, and ReSource, has been designed to provide detailed methane emission data using tracer gas dispersion technique, drone-based plume measurement, ground-based plume measurement and surface flux chambers at three landfill sites (one closed and two operational) located in Sweden. This report presents the results of the measurement campaigns, which were carried out across the last week in October 2019 and the first in November 2019. The four measurement techniques are compared in this report.

The objectives of the study are detailed in the Swedish Waste Management Association (Avfall Sverige) report (in Swedish) “Ytemissioner av deponigas - en studie” and briefly described here. The overall aim is to recommend a method for standardising measurements of fugitive methane emissions from landfills in Sweden. Hence, the aim of this study is to compare different measurement methods in terms of methane flux quantification, duration of measurement, uncertainties, suitability for different kinds of landfills etcetera.

Four different techniques for methane emission measurements were applied simultaneously in order to compare the performance of the methods. The first method was the tracer gas dispersion technique (TDM), which uses a tracer gas release at one or several points on the landfill. The concentrations of the dispersed tracer gas as well as methane are measured downwind the landfill by driving a vehicle with a sensitive methane/tracer gas analyser across the plume. Using the concentration ratio of the two gases and the known tracer gas release rate, the total landfill methane emission rate can be determined. The second method, drone-based plume quantification, uses a sensitive methane analyser mounted on an unmanned aerial vehicle (UAV, e.g. drone). By measuring the wind speed and direction while flying transects at increasing heights downwind of the landfill, a “curtain” cross-section of the plume can be reconstructed and the flux through the cross-section estimated representing the total emission from the landfill. The third method, ground-based plume measurement, also known as OTM 33A, is a method consisting of an initial screening of the site by car with a methane analyser to locate emitting sources or hotspots. When these have been identified, the methane emission from each of them are quantified by installing a tower with a sensitive methane analyser, anemometer and weather station. This method provides an estimation of methane emissions from identified sources at a site and cannot be used to quantify the whole landfill methane emission. The fourth method in the comparison study is the surface flux chamber method where a box with valves mounted on the bottom are placed upside down at several spots at the landfill surface. The methane concentration is measured in the box over time, which makes it possible to calculate a flux from the area covered by the box. A large number of measurements is needed in order to scale up the emissions measured in specific locations. A protocal is available describing the number of measurements needed.

Moreover, a controlled release test was performed where a known flow of methane was released in three release points and the combined emission was measured simultaneously applying the different methods. It was not possible to use the surface flux chambers in this test. The error when comparing the actual methane release rate with the quantified emission rate was determined: the TDM had an error of +6.8 %, the UAV-based plume method an error of -19 % and the OTM 33A method an error of -12.5 % compared to the actual release flow.

At Slite, the smallest of the three landfills, emissions for day 1 range between 1.4 and 4.7 kg CH_4 h^{-1} among the three methods (TDM, UAV and flux chambers). For day 2, the results ranges between 1.6 and 5.8 kg CH4 h^{-1} (only TDM and UAV). During both days, the UAV method monitored higher emission rates than the TDM method (with an approx.. factor of 3.5). The flux chambers monitoring resulted in similar results (4.3 kg CH_4 h^{-1}) to the UAV method. The screenings performed by both the TDM-team and the UAV indicated that emissions were mainly originating from the southeastern part of the landfill. The two emission hotspots found by the TDM team made up 0.75 kg CH_4 h^{-1} corresponding to about 40% of the total methane emissions from the site, which based on the walk-over screenings seem reasonable as the majority of the site did not emit much methane.

At Filborna, the largest and most complex of the three landfills, the TDM and the UAV method obtained very similar emission rates of 139.5 and 130.7 kg CH_4 h^{-1}. The highest emission rate of 310 kg CH_4 h^{-1} was obtained by the surface flux chamber method. As both TDM and UAV are performed

at a distance from the landfill, methane emissions from on-site sources or sources located right next to the landfill will be included in the quantified emission rate. The flux chambers, however, only monitor emissions from the landfill surface, which contradicts the high emission rate in comparsion to the emission rates obtained by the two remote sensing methods.

At Vankiva landfill, methane emissions ranged between 1.1 kg CH_4 h^{-1} for the surface flux chambers, 14.5 kg CH_4 h^{-1} for the TDM and 52.0 kg CH_4 h^{-1} for the UAV method. The surface flux measurement were not performed according to the standard method, hence the results are not entirely accurate and should not be seen as an estimation of the methane flux.

In general, the emission rates obtained by the UAV method compared reasonable well to the TDM method, especially at Filborna. At the two other sites, the UAV method resulted in a higher emission rate than the TDM method by a factor of 3.5 in both monitoring campaigns. The OTM 33A method is developed for quantification of point sources and cannot be used at larger areas and for monitoring more diffuse emission sources. For this reason, this method could not be compared to the others. However, when applied at a point source or a single source where emissions are localised and blend into one plume within a short distance, the OTM 33A method seems to result in reasonably comparable emission rates.

This study suggests that the most accurate, whole-site methane emission quantifications are performed using methods measuring plumes downwind of the landfill, such as tracer gas dispersion or UAV plume measurements. This study also for the first time clearly demonstrates that the UAV method can give similar results to the currently-accepted tracer dispersion method, based on the controlled release trial and results from actual landfills: Hence, the UAV method offers an alternative or supplementary method of measurement. Obtaining proper wind information (wind speed profiles) when applying the UAV method is important as the emission is a direct result of the the applied wind speed through the vertical plume plane. Further validation and documentation of the method compared to controlled releases and known sources is however required.

Ground-based method OTM 33A is not suitable for quantification of emissions from all landfills. It is suitable as an additional method for investigation methane leaks from infrastructure at the landfill or localised area sources e.g. smaller lagoons, or sludge storages. Combined with whole-site methane emission quantifications, on-site emission sources can be determined.

This study demonstrates that heterogeneous emissions from landfills, both spatial and temporal, result in larger variability when quantifing total methane emissions using the surface flux chamber method. However, the method can, when combined with walk-over surveys, provide information about leak patterns and point sources at the landfill, which the landfill operator can use as an action plan for reducing the surface emissions from the site. Walk-over surveys and flux chamber measurements can also be undertaken by trained site personnel and although more time consuming than the other methods, requires less expensive and readily available equipment.

This study also demonstrated that UAV can be used to identify areas with elevated methane concentrations. Due to the larger measuring distance from the source, specific emission hotspots and locations can be difficult to pinpoint. For this, a walkover methane screening survey combined with visual surface inspection is more appropriate.

1 INTRODUCTION AND OBJECTIVE

1.1 INTRODUCTION

Since the start of the Industrial Revolution in the mid-eighteenth century, human activities have substantially contributed to the increased concentrations of carbon dioxide (CO_2), methane (CH_4) and other greenhouse gases (GHGs) in the atmosphere. CO_2 and CH_4 are primary anthropogenic drivers of climate change and their impact is expected to grow unless their emissions can be dramatically reduced. To limit the increase in global average temperatures to less than 2 °C above pre-industrial levels, the 21st session of the Conference of the Parties (COP21) of the United Nations Framework Convention on Climate Change (UNFCCC) agreed to implement an ambitious effort to reduce GHG emissions. While the atmospheric CH_4 concentration is still much smaller than that of CO_2, the observed CH_4 increase will still have a substantial climatic impact due to inherent characteristics of the CH_4 molecule, which has 28 to 36 times (weight based) stronger global warming potential then CO_2, based on a 100 years scale (IPCC, 2013). Mitigating climate change is therefore a key scientific and societal challenge.

Methane can enter the atmosphere from several sources such as fossil fuel industry, waste management, livestock farming, and agriculture. There is a growing interest in the waste management industry, intended here as a constellation of both municipalities and private operators, to adopt precise methods to detect and quantify fugitive emissions, particularly from landfills. The reason is twofold: methane leakages from waste facilities may represent a loss of marketable product and an infringement of regulatory requirements in the environmental permitting work.

In Europe there are a significant number of directives regulating and impacting on landfill practices which, in turn, affect the generation and control of landfill gas. Key EU directives, which were relevant for this study are the following:

- Landfill Directive, 1999/31/EC;
- Waste Framework Directive, 2008/98/EC;
- Industrial Emission Directive, 2010/75/EU;
- Directive on Integrated Pollution Prevention and Control, 96/61/EB;
- Directive on the Promotion of electricity produced from renewable energy sources in the internal electricity market 2001/77/EC;
- Directive on the Harmonisation of the laws of the Member States relating to equipment and protective systems intended for use in potentially explosive atmospheres, 2014/34/EU.

According to the Swedish law, *Förordning (2001:512) om deponering av avfall,* the landfill operator must collect landfill gas at landfills where organic waste is or has been disposed. Landfill gas should be monitored during the active phase of the landfill. According to the Swedish Energy Agency (2016, 2018), there were 51 landfills in Sweden where landfill gas was extracted in 2017. The total amount of gas sent to flare or gas plant was 145 GWh in 2017. Due to decreasing amounts of organic waste being landfilled, the gas extraction has decreased between 2005 and 2017 by about 68 %. Swedish Energy Agency states that the decrease can also be due to lack of landfill owners reporting their gas extraction data.

Currently, methane emissions from Swedish landfills are often not monitored due to lack of standardised techniques for measuring and controlling landfill gas emissions. Prior to construction of landfill capping, landfill gas emissions are sometimes monitored using surface flux chambers, or the tracer gas dispersion method, but in most cases, emissions are not quantified. For the statutory environmental reporting in Sweden, the gas emissions and production are commonly reported as the amount of gas flared or sent to gas plant, .

1.2 OBJECTIVE

The temporal and spatial variability of surface emissions from landfills results in challenging conditions for accurate measurement of the emissions. Not only are the emissions a result of internal parameters of the landfill, such as waste type, depth, leachate, landfill shape and cover, but also the meterological and atmospheric conditions have a significant impact on the emission variation.

This study, commissioned by Technical University of Denmark (DTU), Lund University, Sweco Environment, FORCE Technology, and ReSource, has been designed to provide detailed methane emission data using tracer gas dispersion technique, UAV-based plume measurement, ground-based plume measurement and surface flux chambers at three landfill sites (one closed and two operational) located in Sweden. This report presents the results of the monitoring campaigns, which were carried out across the last week of October and the first of November 2019. The four monitoring techniques are compared in this report.

The objectives of the study are detailed in the Avfall Sverige report “Ytemissioner av deponigas – en studie”, published in 2020 in Swedish, and briefly described here. The overall aim is to recommend a method for standardising monitoring of fugitive methane emissions from landfills in Sweden. Hence, the aim of this study is to compare different monitoring techniques in terms of methane flux quantification, duration of monitoring, uncertainties, suitability for different kinds of landfills and practicability.

2 SITE DESCRIPTION

This section provides information about the three landfills where methane emission measurements where performed. Table 2.1 provides an overview of main characteristics of the investigated landfills.

2.1 SLITE, GOTLAND

Slite landfill is situated in the Northern part of Gotland. The total landfill area is about 10 ha and consists of two parts; one older clay-covered part (~7 ha) with no basal liner and one more recent part (~3 ha) temporary covered with gravel/sandy soil and with a basal liner in place. The landfill has been active since the 1960's and in 1996 the old part was closed. Landfilling is still taking place at an average yearly rate of 4000 tonnes waste in the new part of the landfill. At the old part of the landfill there are two gas collection wells whereas four gas collection wells are installed in the new part. Gas is continuously extracted and treated in a flare. The combined average gas collection rate is about 20 Nm^3 LFG h^{-1} with a methane concentration oscillating between 25 and 34 %vol. The landfill has a leachate collection and treatment system (located next to the landfill) consisting of an equalisation lagoon, two sandfilters, a sequencing batch reactor, and a wetland. The treated leachate is routed to the local municipal wastewater treatment plant for final treatment.

Slite is classified as a non-hazardous landfill and can accept a maximum of 8000 tonnes of waste per annum according to its environmental permit. The total residual disposal capacity, as of 2019, is 14 068 m^3. The total fill volume including restoration layer on the active part is about 248,000 m^3. Waste types deposited at Slite includes household waste, commercial and industrial waste, ash, waste sludge and garden waste. The total amount of landfilled waste at the old part is approximately 640,000 tonnes while 83,000 tonnes of waste has been disposed of in the new part. The filling height of the active part of the landfill is about 46 m. The maximum height allowed by permit is 51.5 m.

New cells are under construction as of 2019.

Figure 2.1 Overview of Slite landfill site. The flare at the top of the landfill is shown by an red arrow while gas collection wells are marked with red dots. The composting pile and garden waste are shown in the figure. Aerial photo from ReSource/Region Gotland.

Figure 2.2 The areas delimited by yellow and red lines represent active and closed parts, respectively. In the upper right corner, there is a recycling station. On the right side of the landfill, the leachate treatment is located. South of the landfill, there area composting areas. Aerial photo from ReSource/Region Gotland.

2.2 FILBORNA, HELSINGBORG

Filborna landfill is located in Helsingborg and covers about 32 ha. The landfill has been in operation since 1951. Parts of the landfill are temporarily capped while some parts are permanently capped (Figure 2.3). The landfill is still in operation, though organic waste is no longer deposited. The landfill is a “land raise” with a filling height of about 40-45 m. The capping consists of clay with a thickness of 0.3 m. Waste types landfilled at Filborna includes household waste, commercial and industrial waste, ash, waste sludge and garden waste. The total amount of landfilled waste at the site is approximately 6,300,000 tonnes. The landfill is unlined. Collected leachate is treated on site in aerated lagoons.

The site has an active landfill gas collection system, which has been in operation since 1985 and consists of over 100 wells, though the system is largely nonfunctional and only a small number of gas wells remain efficient. The gas wells are both horizontal and vertical. The landfill gas extraction system is coupled to a gas boiler, which delivers heat to the district heating system. The gas collection is operated continuously, and the gas collection rate is about 250 Nm^3 LFG h^{-1} with a methane concentration of 39 %vol. The gas collection system is due to be upgraded during the winter 2019-2020. This site has several potential methane emission sources; an anaerobic digester, food waste storage, sludge storage lagoons, composting area, etc. (Figure 2.3).

Figure 2.3. Overview of Filborna landfill, Helsingborg. Potential methane emitting areas at Filborna landfill and the area around are described. The landfill is marked with an orange line and the capped areas are coloured green. Photo from NSR AB.

2.3 VANKIVA, HÄSSLEHOLM

Approximately 1 million tonnes of waste has been landfilled at Vankiva landfill located in Hässleholm. The landfill was operated between 1979 and 2008. The waste fractions received at the landfill includes residual household waste, wastewater treatment sludge, industrial waste including construction and demolition waste, ash, among others. The landfill is unlined.

Closure and capping works with a low-permeability HDPE membrane started 2008 and is ongoing, see Figure 2.4. The landfill covers about 10 ha (base area) and the maximum filling height is 105 m. Approximately 2/3 of the landfill area is capped with a HDPE-membrane. The HDPE membrane is covered with 1.2 m of cover soils. Uncapped areas of the landfill are covered with a layer of fines (residue from fragmentation of waste), which constitutes a temporary cap.

A leachate collection drain surrounds the landfill. There is an active landfill gas extraction system with 32 vertical wells on site, which is coupled to a generator for electricity generation. The gas collection is operated continuously, and the average gas collection rate is about 100 Nm^3 LFG h^{-1} with a methane concentration of 36 %vol.

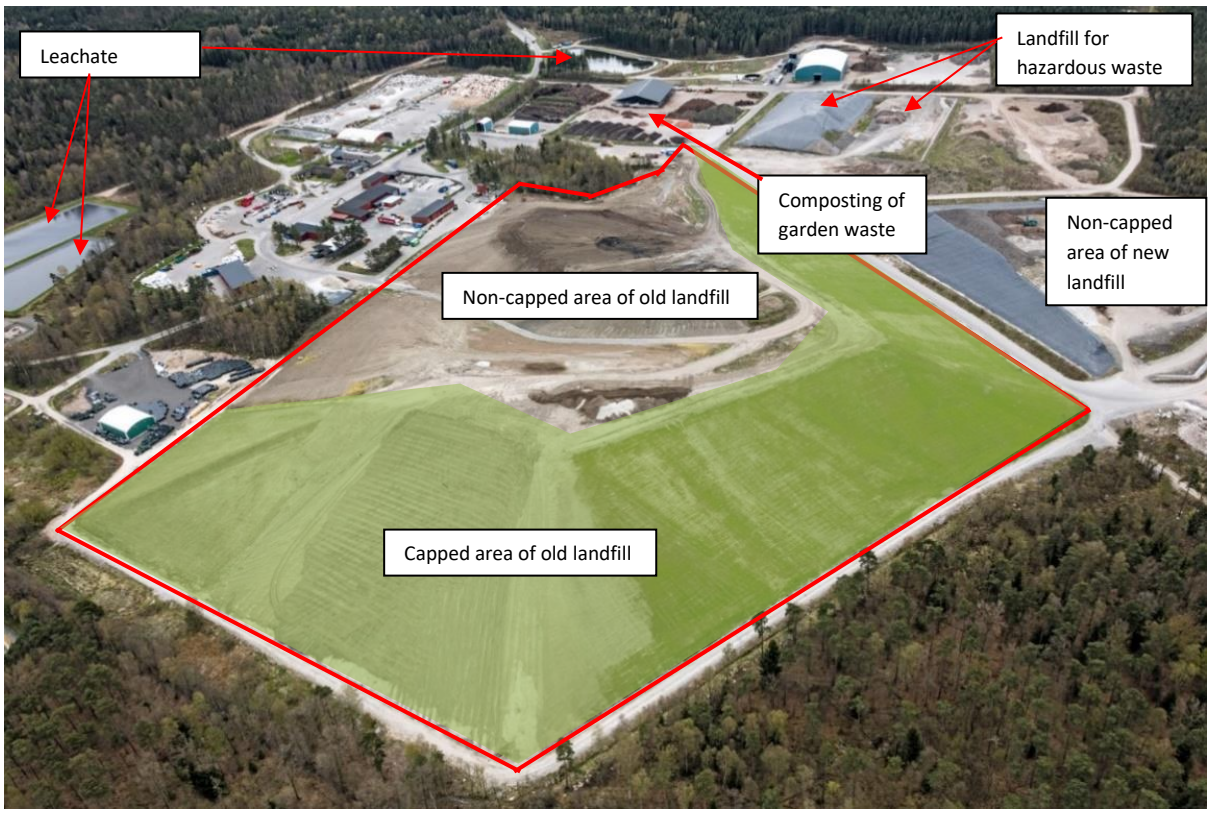


Figure 2.4. Overview of Vankiva landfill. The landfill to the right in the image is a new, active landfill, not included in the campaign. Photo from Hässleholm Miljö AB.

Table 2.1. Overview of landfill characteristics.

Landfill	Disposal period	Amount of waste disposed until the end of 2018 (tonnes)	Waste categories received	Area of landfill (ha)	Maximum filling height (m)	Cover soil (type and height)	Gas collection (yes/no) Collection rate (Nm^3 LFG h^{-1}) [a]
Slite Gotland	Old part: 1965-1999	640,000	Household waste, sewage sludge, industrial waste including construction and demolition waste, ash, etc.	Capped: 6.2	18	Clay 0.3 m	Yes, 20
	New part: 2000-ongoing	83,000	Industrial waste including construction and demolition waste,	Non-capped: 1.8	32	Temporary capped (gravel/sand)	
Filborna Helsingborg	1951-ongoing	6,300,000	Household waste, sewage sludge, industrial waste including construction and demolition waste, ash, etc.	Capped: 13 Non-capped: 17.7	45	Clay, 0.3 m and 1.5 m soil	Yes, 250
Vankiva Hässleholm	1979-2008	1,000,000	Household waste, sewage sludge, industrial waste including construction and demolition waste, ash	Capped: 8.4 Non-capped: 4.3	35	HDPE and 1.2 m soil	Yes, 100

[a] The reported gas collection rate is the annual average.

3 DESCRIPTION OF MEASUREMENT METHODS

3.1 THE TRACER GAS DISPERSION METHOD

Method principle
Total landfill methane emissions were quantified using a mobile tracer dispersion method (TDM) that combines a controlled release of tracer gas from the landfill with concentration measurements downwind of the landfill, by using a mobile high-resolution analytical instrument (Galle et al., 2001; Scheutz et al., 2011). The method has been used successfully in the last few decades, and with new developments in analytical technology it has become a powerful tool for quantifying methane emissions from landfills (Mønster et al., 2014; 2015; Rees-White et al., 2019). The tracer dispersion method in general is based on the assumption that a tracer gas released at the landfill, will disperse in the atmosphere in the same way as methane emitted from the landfill. Assuming a defined wind direction, well mixed air above the landfill for the methane and tracer gas to be fully mixed, and a constant tracer gas release, the methane emission rate can be calculated as a function of the ratio of the integrated cross-plume concentration of the emitted methane and the integrated cross-plume concentration of the released tracer gas, as follows:

$$E_{gas} = Q_{tracer} \cdot \frac{\int_{Plume\ end\ 1}^{Plume\ end\ 2} C_{gas}\, dx}{\int_{Plume\ end\ 1}^{Plume\ end\ 2} C_{tracer}\, dx} \cdot \frac{MW_{gas}}{MW_{tracer}} \quad \text{(Eq. 3.1)}$$

Where E_{gas} is the methane emission rate (kg/h), Q_{tracer} is the release rate of the tracer gas (kg/h), C_{gas} and C_{tracer} denote cross-plume concentrations (ppbv) above the background concentration, MW denotes molecular weights and x corresponds to distance across the plume. The principle is shown in Figure 3.1.

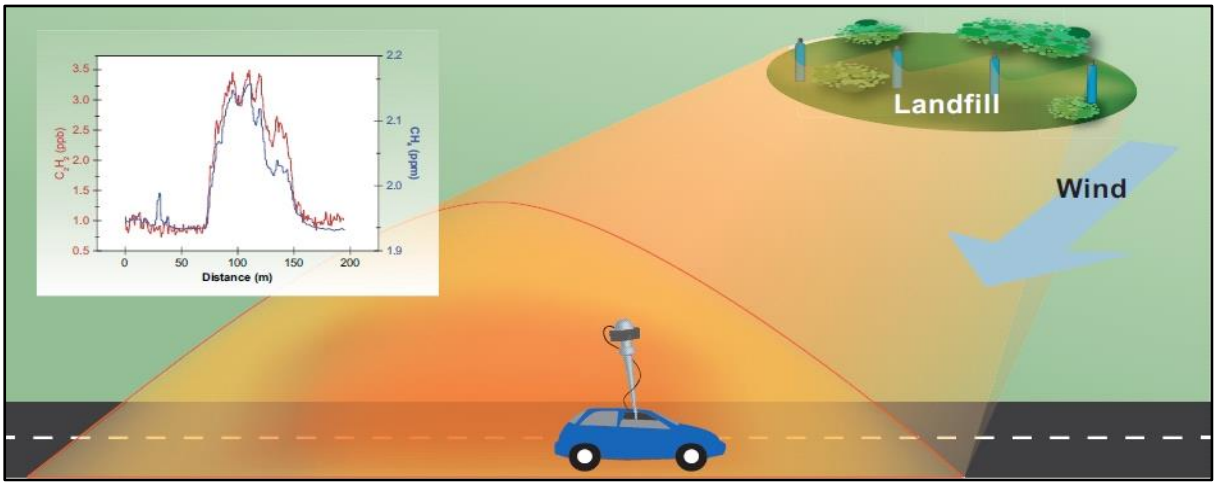


Figure 3.1. The principle of the dynamic plume method for quantifying greenhouse gas emissions from fugitive sources.

The optimal distance for measuring a site's total emissions depends on the size of the emission area, the topography of the site, weather conditions such as wind speed and solar radiation (Mønster et al., 2014), and the sensitivity of the analytical platform. It is important that the tracer gas simulates the methane emission from the landfill and the tracer gas release should correlate with the methane emission area at the landfill. If the landfill covers a large area several tracer gas release points might be needed and placed at the main emission areas at the site. In order to obtain the best simulation of the methane emission, it is recommended to do a qualitative assessment of the emission pattern by conducting methane screening measurements on-site and around the landfill (when possible) and on all possible roads nearby.

Quantifications are made by performing multiple traverses across the plume and then calculating the

methane/tracer ratio (Eq. 3.1) for each traverse. In this way, a change in dilution due to a change in wind speed, or turbulence changing vertical mixing, would be the same for both gasses at each individual plume measurement. At each traverse it should be ensured that the whole plume is measured before turning the vehicle to measure the plume again. This enabled the establishment of a baseline of background concentrations to be subtracted from the measurements, in order to obtain the landfill's contribution to the plume. Several factors will affect the ideal measuring distance between the landfill and measurement location. Long measuring distances assure mixing of methane and tracer gases. However, long measuring distances also increase the dilution of the gases, and at a certain distance the uncertainty in the measured concentration will increase. The ideal measuring distance is thus the distance where the tracer gas plume and the methane plume match each other and the signal of both is sufficient to be distinguished from the background. A guideline for best practice measurement performance using the tracer gas dispersion method is presented in Scheutz & Kjeldsen, 2019.

Instrumentation
At Slite landfill, measurements were performed with a cavity ring-down spectroscopy (CRDS), methane/acetylene analyser (G2203, Picarro, Inc., Santa Clara, CA), where acetylene was used as tracer gas. Atmospheric air was sampled from the roof of a vehicle and brought to the analyser via an external pump, thus enabling a fast response time when a plume was entered while driving. The atmospheric concentrations of methane, acetylene and water were measured with a frequency of 2 Hz and logged together with the atmospheric conditions and GPS position. The precision of methane and acetylene concentrations were 0.48 ppb and 0.40 ppb, respectively, making it possible to detect small changes in atmospheric concentrations while driving and monitoring the concentration on the screen attached to the analyser. At Filborna and Vankiva landfills and at site where the controlled release test was carried out, two cavity ring-down spectroscopy (CRDS) analysers were used. The first was a carbon dioxide/methane analyser (G1301, Picarro, Inc., Santa Clara, CA), while the second was an acetylene/nitrous oxide analyser (S/N JADS2001, Picarro, Inc., Santa Clara, CA) (Mønster et al., 2014; Yoshida et al., 2014). The atmospheric concentrations of methane, acetylene and water were measured with a frequency of 0.3 Hz. A GPS (model R330 GNSS Receiver and A101 Smart Antenna, Hemiphere, Canada) was attached to the roof of the vehicle in order to log the position of the vehicle measured the location within 20 cm precision.

Calibration procedure
The CRDS instruments were calibrated by the manufacturer. The methane measurements were calibrated using four gravimetrically prepared mixtures of methane with concentration ranging from 1.7 to 2.3 ppm. Ultrapure nitrogen was used as a zero calibration. The acetylene measurement was not calibrated directly with a standard. Instead, the spectroscopy calibration constant that was measured on another acetylene instrument that uses the same spectral line (Model G1203, SN: DFADS002, Picarro, Inc., Santa Clara) was applied to our measurements. That other instrument was calibrated against a 103 ppb acetylene in air mixture (Linde Premium Products (formerly SpectraGases), Stewartsville, NJ) that was guaranteed with an accuracy of 10%. A series of laboratory tests were then performed to establish the basic performance of the analyser, consisting of continuous measurements on prepared gas mixture (Mønster et al., 2014). The CRDS instrument is occasionally tuned by adjusting the focus of the lasers wavelengths to reduce the signal to noise ratio.

Tracer gas release
At Filborna and Vankiva landfills and at the site where the controlled release test was carried out gas bottles of 21 L filled with 3.92 kg acetylene (purity >99.5%) were used. The acetylene flow was controlled with a two-stage regulator, with a safety valve for back flush, connected to a 150 mm flow meter (Sho-rate, Brooks) calibrated for measuring between 0 and 25 L per minute and with an uncertainty of 5% on the maximum value. The acetylene was dissolved in acetone in a ratio 25:1 allowing a stable and safe release, but also limiting the release rate to approximately 1 kg per hour. At Slite landfill, the controlled tracer gas release was performed using slightly smaller gas bottles (1.3 kg acetylene) with a build-in pressure regulator. An additional pressure regulator, a back flush safety valve and a flowmeter was added to the tracer gas bottle. The flowmeters were calibrated by a certified laboratory (FORCE Technology, gas calibration) to measure acetylene flow between 4 and 27.5 L per minute with an uncertainty of 2.5% on the maximum value. The tracer bottles are placed in areas identified from initial screening measurements at the landfill to have the highest methane emissions, in order to simulate the emission in the best possible way. Between two and three tracer

gas bottles were used at each site. The bottles were weighed before and after the release and the weight loss checked against the flow meters.

Measurement uncertainty

The overall uncertainty of TDM measurements is a combination of several uncertainties, some of which are instrument specific and some of which vary according to specific measurement conditions (Fredenslund et al., 2019). Factors which may contribute to the overall uncertainty include: analytical uncertainties; uncertainties in the tracer gas release rate; data processing, and tracer gas placement and source simulation (Fredenslund et al., 2019). In addition to the listed factors, the variability of an emission quantification should be added. Fredenslund et al. (2019) presents a systematic assessment of the potential uncertainty of the tracer gas dispersion method and discuss how the uncertainty of each factor can be assessed. Several controlled release tests have been carried out to assess the accuracy of the tracer gas dispersion method (Mønster el at. 2014; Delre et al., 2018; Fredenslund et al., 2019). Following best practice when performing measurements, the overall error of a TDM measurement is less than 20% (Mønster el at. 2014; Delre et al., 2018; Fredenslund et al., 2019). However, if the source is not correctly simulated by the tracer gas release, larger errors can be seen. In this study, the total uncertainty on a quantification was estimated by following the method in Fredenslund et al (2019), which provides the uncertainty of the measurement method and by adding the variability of a measured emission rate. The variability of a measured emission rate is calculated as the standard error of the mean based on the number of performed plume traverses.

Weather measurements

During each measurement campaign, meteorological data were recorded at an on-site weather station (Kestrel 5500 Weather Meter, Kestrel Instruments, USA). The weather station provided 2 min averages of continually monitored meteorological data including temperature, wind speed, wind direction, and atmospheric pressure. The weather data are not used in the calculation of emission rates but mainly serves a purpose during the measurements e.g. establishment of wind direction and wind speed. The weather information can be used to interpret methane concentration recordings and locate potential methane sources.

Description of data processing

It was sought to obtain the highest possible number of successful plume traverses at each measurement day at all three landfills measured. Initially, a visual screening of each traverse was made to check for anomalous data and excess noise, which may be caused by instrument error, changing weather patterns and from interfering methane or acetylene sources. Plume traverses were excluded if they were performed outside of the tracer gas release period, defined as the period from start of release (plus the travel time for the tracer gas to reach the measurement point) until shut off of tracer gas. Incomplete plume traverses, which could occur if there were no drivable roads to complete a full plume traverses were also excluded. Typically, plume traverses clearly affected by other sources of methane emissions would be excluded as would plumes where the measuring vehicle was stationary for any period of time, for example due to traffic queues or traffic lights. The plumes passing the visual screening were integrated individually to find the methane/tracer gas ratio for each plume traverse, as this has been found to be the most accurate method to obtain the methane/tracer gas ratio (Mønster et al., 2014; Delre et al., 2018). This was also done in cases where the methane and tracer gas plumes did not completely overlap. The ratio of the areas can be used, as measurements were done far from the landfill and both gasses (methane and acetylene) undergo the same atmospheric dispersion. Figure 3.2 shows an example of recorded methane and acetylene concentrations in a plume traverse measured downwind Slite landfill. The emission rate from the individual measurement campaigns was calculated by taking the average of all the emission rates calculated from the accepted plumes rations. The variability of the averaged emission rate was then calculated as the standard error of the mean.

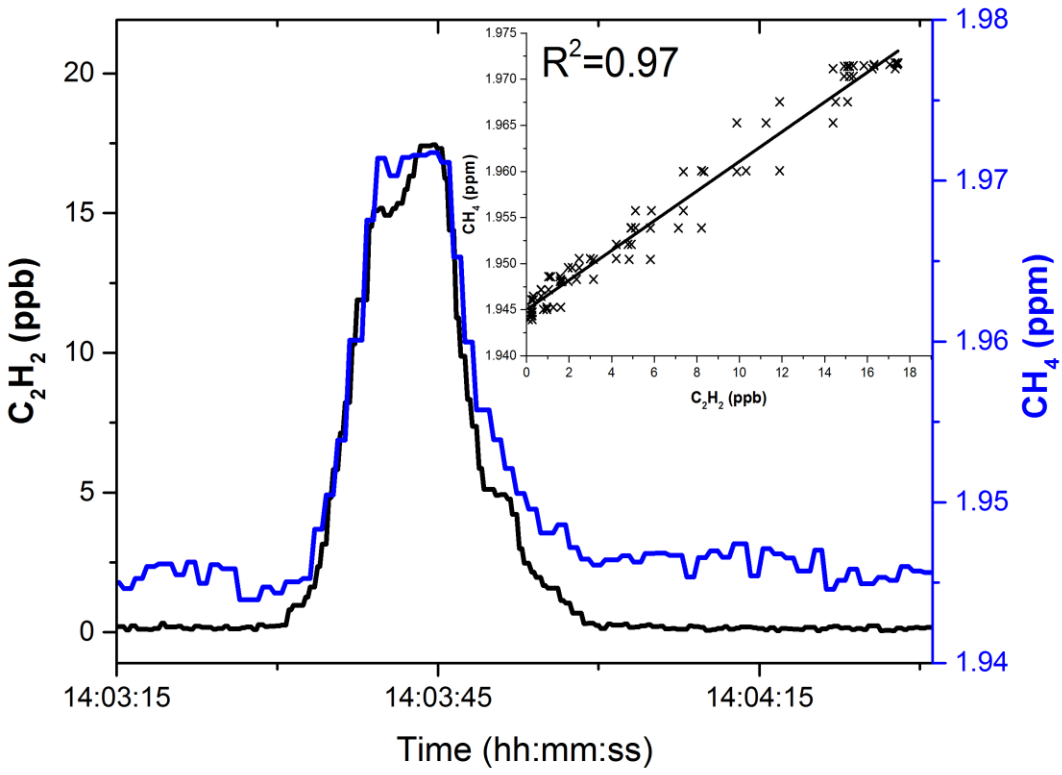


Figure 3.2. Integration of methane and acetylene plume concentrations recorded in a traverse measured downwind Slite landfill.

3.2 UAV-BASED PLUME MEASUREMENTS

Method principle

UAV-based plume measurements are conceptually simple, but the method is technically tricky to accomplish precisely. A UAV equipped with a sensitive methane gas sensor is flown at 8-10 m s^{-1} downwind of a landfill at increasing heights, usually between 10 m and 120 m above ground. It is typically possible to complete transects at six to eleven altitudes within the ten-minute flight time of the current UAV/sensor combination for a landfill of medium size. These transects are combined together using a geostatistical spatial averaging technique known as kriging. This technique uses the acquired data to build up a 2D cross-section of the plume, as in Figure 3.3.

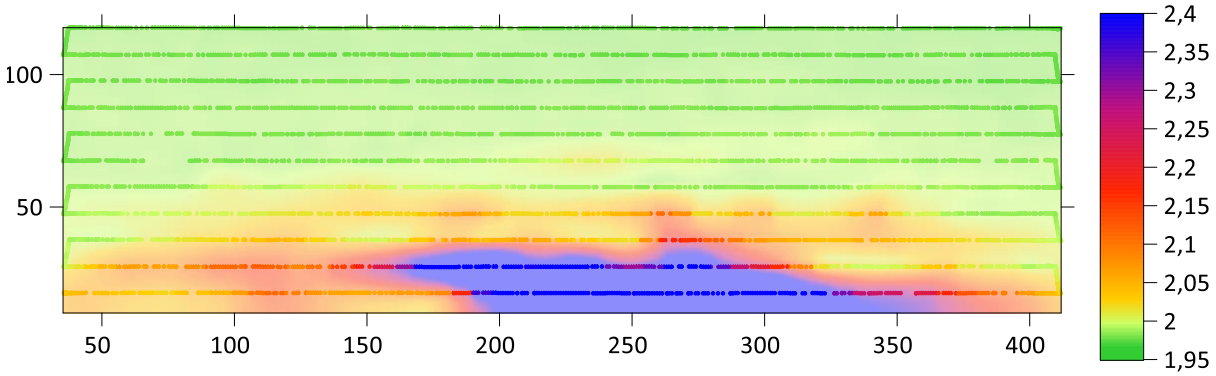


Figure 3.3. Example of methane plume cross-section. Methane concentrations are in ppm; axes are in meters.

Atmospheric background methane concentrations must then be subtracted from the plume. Background measurements are most efficiently taken by flying upwind, and/or above or to the side of

the target plume, if there are no nearby emitting sources. Background values are then subtracted from the reconstructed plume. As this method only relies on the precise repeteability of measurements and not the actual accuracy of an instrument compared to a reference sample, long-term drift present in all instrumental measurements is not a component of the error. Background values may also be taken by flying directly upwind of the transect on the other side of the emitting source, thus measuring an upwind plane of inbound atmospheric methane to subtract from the downwind plane. However, this reduces the number of measurements that can be taken downwind and would only be used where there was significant background variation in methane.

Plume measurements in this case were taken between 10 m and 500 m downwind of plumes for all of the landfill tests (the transect varies in distance from the closest emitting point of a large emission source, often considerably) and between 100 and 900 m for the release tests. Having a high degree of mixing by going about 500 m downwind is ideal, but not possible in some cases due to local topography, complications posed by tree cover and the size of the emitting plume.

Each concentration, measured in ppm and with background averages subtracted, is converted into a point-flow of methane. First, the units are changed to kg·m^{-3} using the following formula:

$$C_{methane} = ppmv \cdot \frac{V_{m.air.standard}}{M_{methane}} \cdot \frac{T_S}{T_M} \cdot \frac{P_M}{P_S} \quad \text{(Eq. 3.2)}$$

Where
$C_{methane}$ = concentration of methane (kg·m^{-3})
ppmv = parts per million, by volume basis
T_M = Median measured temperature (K)
P_M = Median measured pressure (mB)
T_S = Standard temperature (273.15K)
P_S = Standard pressure (1013mB)
$V_{m.air.normal}$ = Molar volume of air at Standard conditions (22.4 liters)
$M_{methane}$ = Molecular weight of methane (16.0425g/mol)

In order to convert this data to flux data (i.e., to take the value of kg/m^3 and convert it to kg/h), a measure of wind speed must be incorporated. This is complicated by the fact that the wind field varies across the height of the plume and in some cases along transects. In this study, ground-based measurements were used to estimate an average logarithmic wind profile to account for height effect. The velocity at the height z above ground, v_z was calculated using the following equation:

$$v_z \approx v_{ref} \cdot \frac{\ln\left(\frac{z}{z_0}\right)}{\ln\left(\frac{z_{ref}}{z_0}\right)} \quad \text{(Eq. 3.3)}$$

Where
v_{ref} = velocity measured at the ground-based anemometer (m s^{-1})
z = altitude above ground (m)
z_0 = surface roughness length (m)
z_{ref} = the measurement height of the ground-based anemometer (m)

Roughness coefficients are usually based on some estimate of the local topography and a table of values (Table 3.1), although, by using vertical profile measurements on the drone, it was possible to constrain this number more precisely to local conditions by comparing predicted values of V_z with measured values and adjusting the figure to better fit the model (Figure 3.4).

Table 3.1. Table of standard values for surface roughness length (z_0) after Burton, Sharpe, Jenkins, and Bossanyi (2001). Estimates are complicated by features such as shelter belts.

Terrain	z_0 (m)
Mud Flats, Ice	$1\text{-}3 \times 10^{-5}$
Calm Sea	$2\text{-}3 \times 10^{-4}$
Sand	2×10^{-4} to 10^{-3}
Mown Grass	10^{-3} to 0.01
Low Grass	0.01 to 0.04
Fallow Field	0.02 to 0.03
High Grass	0.04 to 0.1
Forest and Woodland	0.1 to 1
Built-up Area, Suburb	1 to 2
City	1 to 4

The value of wind speed for each individual concentration measurement was derived from Eq. 3.3 and then concentration was divided by this value to give an calculated point flux in kg m^{-2} s^{-1}. It is anticipated that this may be done entirely on the drone-mounted anemometer in future, removing the estimated component giving better precision particularly for lateral variation in wind speed. As two different anemometers were used, two different values for z_{ref} were used; due to the logarithmic nature of the function, values near the ground create greater imprecision in the final results.

Finally, the point fluxes were kriged together to make a curtain view of the gas plume. This kriged surface was then integrated to find the total flux volume.

Instrumentation and calibration
See measurement uncertainty for details about precision.

Methane sensor: Los Gatos MGGA Microportable (ABB, Switzerland). Calibrated by the manufacturer; relative measurements claimed to be unaffected by absolute drift. Time setting was calibrated daily by spiking of gas at a fixed time, and it was adjusted in processing phase to account for this.

Anemometer, Barometer, Thermometer: Trisonica Mini (Anemoment, USA). Calibrated prior to start of the measurement campaign for zero value, temperature and humidity. Otherwise calibrated by the manufacturer.

UAV: Matrice 600 Pro (DJI, China): Calibrated by the manufacturer. Ground-level calibrated during each flight.

Measurement uncertainty
There are quite a few sources of potential uncertainty following application of this method. As it is a new method with this being one of its first trials, its ongoing development relies on understanding, quantifying and mitigating the largest of these. Repeated measurements and calibration with other methods offers the best possibility to reduce most of the uncertainty. The following is a list of uncertainties and assumptions.

Instrument uncertainty:

1. Methane sensor. This is low, especially given the elimination of instrument drift. Los Gatos lists its measurement uncertainty at 0,004 ppm 1 σ for 1 Hz measurements. There may be some drift as a result of changes in internal pressure as a result of the increase in wind speed with height, but this is unquantified. Methane inlet source was 60 cm above rotor plane.
2. GPS. This is complicated to define as it affects many of the components of the processing but should have negligible effects on the data at the scale of the plume (metres and above). This uncertainty can however be reduced by acquiring topographic models of the area investigated prior to measurements.
3. Anemometer. This is 0.1 m s^{-1} and 1° for the UAV-mounted anemometer.
4. Thermometer and barometer. These were measured using the on-board anemometer, with temperature extrapolated from humidity and the speed of sound. Thermometer uncertainty is 0.1 °C. Barometer uncertainty is 10 mB.

Analytical uncertainty:

1. Background emissions. The background inbound emissions for the different sites were measured during downwind measurements by averaging of areas outside of the plume. However, these ranged in the order of 1.955 to 1.983 ppm; at low emissions levels, any error in this measurement can have a large impact.
2. Wind speed. Wind speed varies and rises with height, implying higher flux for a given concentration of gas with increasing distance from the ground. Therefore, an anemometer was mounted on the UAV and flown.This anemometer was used in combination with ground-based measurements (where available) to fit a log curve of wind speed with altitude, which was used to offset flux estimates. Mounting anemometers on copter-type UAVs is still experimental and returns mixed results. Data from the UAV-mounted anemometer did not return precise enough results to be used on its own in some circumstances, hence the use of a fitted curve.
   Wind direction. Wind direction varies during the course of measurements. Transects are plotted at roughly 90 degrees to the wind, and fluxes are corrected for average wind direction using a cosine transformation. The error associated with this grows as the angle grows, but is typically <5%.

Systematic uncertainty:

1. Uncertainty of height above ground. Given that concentrations are typically highest in the lowest 50 m of the measurements and tend to peak near ground level, this error may be significant. Currently, ground height is estimated using NASA SRTM or similar data which has a typical relative error of 6 m (Elkrachy, 2018) and combined with the barometric altimeter on the UAV, which is accurate to 0.5 m. The size of this error is less certain where trees were flown over, as an assumed canopy height was used as a baseline. In future, better digital elevation models and/or an on-board rangefinder will be used to reduce this error.
2. UAV influence on wind speed. Side-by-side anemometer measurements show that UAV wind speed approximates that of ground measurements while stationary. However, this has only been verified at ground level and issues remain in extracting wind speed while in motion. The anemometer was mounted on a 60 cm pole, and measured 0.5 m s^{-1} of downdraft.

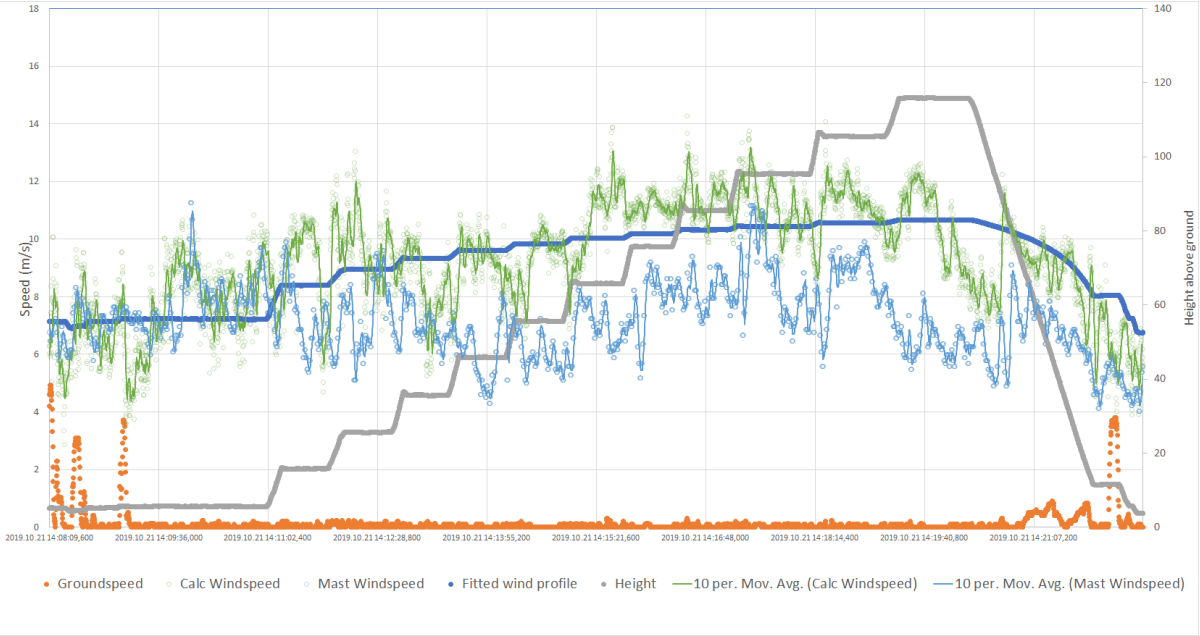


Figure 3.4. Wind speed measurements from UAV (green) and Icelandic Met Office mast (blue), along with groundspeed (orange) and height above ground (grey). A fitted logarithmic profile based on mast height, estimated roughness and drone height is in blue.

3. Effect of fitting curve. There will be an unknown amount of simplification in fitting an atmospheric height profile to windspeeds based on ground-level data; this was minimised by using drone measurements to estimate the coefficient of roughness.
4. Lack of measurement points near ground level. The UAV is limited in how close to the ground it can fly. As the gas plume is highly concentrated near ground level, there may be some underestimation of high-volume flows along the ground, but there are additional issues with

estimating near-ground wind flow and turbulence. This uncertainty is currently not quantifiable.

5. Tree cover. Currently, gas is assumed to mostly escape across the top of forest cover. On several of the sites, extensive forest surrounded landfills. In this study, it is assumed that no wind (and thus no gas) flows underneath canopy. This will underestimate emissions, depending on the density and depth of the stand. However, trees typically stop the vast majority of wind flow. This uncertainty is currently unquantifiable.
6. Turbulence. Transects were chosen in part to avoid this, by e.g. not flying immediately downwind of obstructions.
7. Vertical mixing. Vertical mixing and movement of the plume is assumed to be net zero or at least less than the distance between vertical transects (at least 10 m) during the time of measurement.

Some uncertainty was not quantifiable. Uncertainty estimates were derived using the following formula (Allen et al. 2018):

$$\sigma_T = F \sum_i^N \sqrt{\frac{\sigma_f^2}{\bar{f}} + \frac{\sigma_\varepsilon^2}{\bar{X}} + \frac{\sigma_{ws}^2}{\overline{ws}} + \sigma_{wd}^2 + \frac{\sigma_B^2}{\bar{B}}} \qquad \text{(Eq. 3.4)}$$

Where
σ_T = Total flux uncertainty (1σ), summed over N grid cells for entire flux plane (kg h^{-1})
F = Total flux (kg h^{-1})
σ_f = Standard deviation of measured mass concentrations in each cell, i (kg m^{-2} h^{-1})
f = mean point flux (kg m^{-2} h^{-1})
σ_ε= Instrument measurement precision (ppm)
X = mean measured concentration (ppm)
σ_{ws} = Standard deviation of measured windspeed across measurement period (m/s)
ws = mean windspeed (m s^{-1})
σ_{wd} = standard deviation of the cosine of wind angle variability perpendicular to flux plane (unitless)

Health, safety and risk assessment
UAVs pilots must follow airspace limits and gain certification before flying, as well as comply with local bylaws. This was done prior to the mission. In all cases, a flight plan was made so that flying over people and buildings was avoided, as was flying over active landfill flares. A lower boundary was set at 10 m to prevent collision with the ground. PPE was worn.

3.3 GROUND-BASED PLUME MEASUREMENT METHOD (OTM 33A)

Method principle
Other test method 33A (OTM 33A) is one of the measurement techniques described in details by United States Environmental Protection Agency. This method, fully described in U.S. EPA, 2014, consists of two parts; screening and quantification. The aim of the screening part is to localise emission sources. This can be obtained by driving near to potential sources with an online analyser and a GPS tracking system. The screening part should be performed relatively close to the source, because the quantification part is limited by the distance, up to 200 m from the source. Site access is thus required to perform proper screening. Information collected from outside of a landfill, could be hard to use for the quantification part.

After the location of a source its emission is quantified. The measurement point, which is where the analytical equipment is positioned, should be at a distance of 20 m to 200 m directly downwind of the source. At the chosen measurement point, a mast with a 3D sonic anemometer, a methane gas analyzer and a weather station has to be set up. The measurement period should be around 20 minutes with 2 Hz data acquisition frequency of both the 3D sonic anemometer and the methane gas analyzer. In case of a slower instrumentation recording frequency, a longer measurement period should be applied. Data collected during the measurement period includes wind speed and a wind direction in 3 axes, a methane concentration, a temperature and a relative humidity. Additionally, the distance from the emission source to the measurement point has to be recorded. If possible, a laser range finder should be used. If not available, the distance can be determined based on GPS position recordings.

OTM 33A calculates a methane emission rate based on the Gaussian dispersion equation as shown by Equation 3.5. Written scripts for solving Eq. 3.5 are available and can be downloaded in Matlab and in R (https://www.epa.gov/sites/production/files/2017-07/p14_otm_33a_ver._1.3.zip)

$$Q = 2\pi\sigma_y\sigma_z\bar{U}C_{peak} \qquad \text{(Eq. 3.5)}$$

Where Q is the methane emission rate (g s^{-1}), σ_y (m) and σ_z (m) are the vertical and horizontal dispersion coefficients, respectively, $\bar{U}$ is the wind speed (m s^{-1}) and C_{peak} is the maximum concentration of the methane gas (g m^{-3}) above background concentration at a given point in the downwind plume. The first part of equation 3.5, the vertical and horizontal dispersion coefficients, is related to the stability of the atmosphere. Based on recorded 3D wind speed data, the local stability class is calculated. In Table 3.2 parameters necessary to perform the calculation are presented.

Table 3.2. Parameters used to calculate the local atmospheric stability class.

Atmospheric Stability Indicator (ASI)	Definition	Standard deviation of wind direction	Turbulent intensity	Average local class (PGI)
1	Extremely unstable	27.5º < StdWD	0.205 < turbuint	pgi < 1.5
2	Moderately unstable	23.5 < StdWD ≤ 27.5º	0.180 < turbuint ≤ 0.205	1.5 ≤ pgi < 2.5
3	Slightly unstable	19.5 < StdWD ≤ 23.5º	0.155 < turbuint ≤ 0.180	2.5 ≤ pgi < 3.5
4	Neutral	15.5 < StdWD ≤ 19.5º	0.130 < turbuint ≤ 0.155	3.5 ≤ pgi < 4.5
5	Slightly stable	11.5 < StdWD ≤ 15.5º	0.105 < turbuint ≤ 0.130	4.5 ≤ pgi < 5.5
6	Moderately stable	7.5 < StdWD ≤ 11.5º	0.008 < turbuint ≤ 0.105	5.5 ≤ pgi < 6.5
7	Extremely stable	StdWD ≤ 7.5º	turbuint ≤ 0.080	6.5 ≤ pgi

The Atmospheric Stability Indicator (ASI) is calculated for two indicators; standard deviation of wind direction and turbulent intensity. Turbulent intensity is defined as standard deviation of horizontal wind speed divided by mean wind speed. Then ASI is calculated for wind direction (StdWD) and turbulent intesity. The two ASI values are averaged and average local class (PGI) is determined. Combination of the PGI value and distance from the source gives parametrized plume dispersion in horizontal σ_y (m) and vertical σ_z (m) dimension.

Wind data are also used to calculate the mean wind speed during the measurement $\bar{U}$ (m s^{-1}). The second part of the equation is the methane enhancement C_{peak} (g m^{-3}), which is estimated through the following procedure: a background value is calculated as the lowest 5% of measured concentrations and averaged. This value is subtracted from all recorded methane concentrations because only the enhancement is needed. Then the recorded methane enhancement concentrations are binned into 10º wind directions bins. In each bin, the average value for the methane enhancement is calculated. A Gaussian curve is fitted for that prepared data and the C_{peak} is calculated from that.

This method was originally developed for measuring methane emissions from point sources/leakages in the oil and gas sector. Applying this method for a quantification of landfill emission is possible, but only under certain conditions and at some landfills. With this method it is possible to estimate methane emissions from landfill infrastructures, e.g. gas collection wells, gas venting wells, lagoons, on-site storages of organic residuals (e.g. sludge) and other emitting areas/installations, which could be treated as a point source when applying a proper measuring distance. A stable wind direction is preferred in order to distinguish properly measured source from other methane sources on a landfill. Quantification of whole site emissions using OTM 33A is possible only if the emissions from the landfill is confined and localized to a smaller area. The wind speed has to be high enough to blend small leaks into one plume. If the measured plume contains several peaks, the method is not suitable.

Instrumentation

Measurements were performed with off-axis integrated cavity output spectroscopy (OA-ICOS), methane analyzer (Los Gatos Research Microportable Greenhouse Gas Analyzer - LGR MGGA). For the screening part of the method, atmospheric air was sampled from the roof of the car directly to the analyzer by a built-in pump. Delay between sampling and reading was short, around 2 seconds, so additional pumping wasn't required. The atmospheric concentration of methane was measured with 1 Hz frequency. GPS position data were measured using a smartphone with GPSLogger for Android at 1 Hz. After measurements GPS positions and methane concentration data are combined. Clock differences between devices and delay in a methane reading are included into this combination. The

precision of the methane concentration was 4 ppb at 1 second. Monitoring changes in methane concentration during measurements was possible on a tablet connected to the analyzer by Wi-Fi. The methane analyzer was powered from the car lighter socket via a 12V to 230V converter. A more detailed description of the LGR MGGA methane analyser is available in the manufacturer's datasheet (LGR-ICOS, 2019).

For the quantification phase of the method, wind components were measured by a 3D sonic anemometer (METEK GmbH Germany uSonic-3 Omni) at a frequency of 1 Hz. The methane concentration was measured by the LGR MGGA analyzer. Ambient air was sampled from below the 3D sonic anemometer. In that setup delay was around 2 seconds, so additional pumping wasn't required. Temperature and relative humidity were measured by a probe CS215 (Campbell Scientific) housed in a solar radiation shield with 1-minute interval. During the quantification phase of the measurements, data were collected on CR1000 Series Datalogger (Campbell Scientific). The sonic anemometer and the data logger were powered from 12 V motorcycle batteries. The methane analyzer was powered from built-in batteries.

Calibration procedure
LGR MGGA was calibrated before campaign in laboratory conditions, at Lund University. Three bottled gases with a known concentration (1.996, 2.012, 4.937) were measured. Based on three points, a calibration curve was performed.

Measurement uncertainty
Measurements uncertainty for the screening phase of the method had two sources. One of them is the precision and stability of the LGR MGGA readings. The standard deviation of methane concentration measurements was 4 ppb at 1 second. The LGR MGGA methane analyser doesn't have temperature and pressure stabilization. During screening measurements, changes in air temperature and and air humidity could influence the readings. No correction function has been developed yet. This uncertainty has influences on the size of a detected leak. A second source of uncertainty is GPS. The GPS from the smartphone could be unstable and imprecise. During cloudy days GPS signal could be lost or precision could only be around 100 m. This could affect the accuracy of the source location.

Measurements uncertainty for the quantification part of the method is correlated with the source strength. A lower detection limit for this method is 0.036 kg CH_4 h^{-1} (Brantley et al. 2014). Brantley et al. (2014) reported that quantification should be done when an average enhancement of the concentration inside plume is higher than 0.1 ppm. The controlled release tests showed that the uncertainty is +/- 30% reported by Brentley et al. (2014) and +/- 56% in 95% confidential level reported by Robertson et al. (2017). Further studies presented by Edie et al. (2019) shows that part of the uncertainty is dependant on the size of the emission. For lower emission, lower than 0.5 kg CH_4 h^{-1} higher uncertainty is more appropriate.

Health, safety and risk assessment
The tower for the quantification part of the method has to be placed where interference with daily site operations is minimised. Inappropriate placement of the tower could cause damage to operators or/and equipment.

Description of data processing
Data were analysed by using a script in R, downloaded from EPA website. Collected data have to be prepared according to a template, presented in the EPA script. Using this script, the results consist of plots binned by wind direction, methane enhancement (Figure 3.5) and fit of Gaussian curve (Figure 3.6). As well as tables with estimated emissions and associated uncertainties.

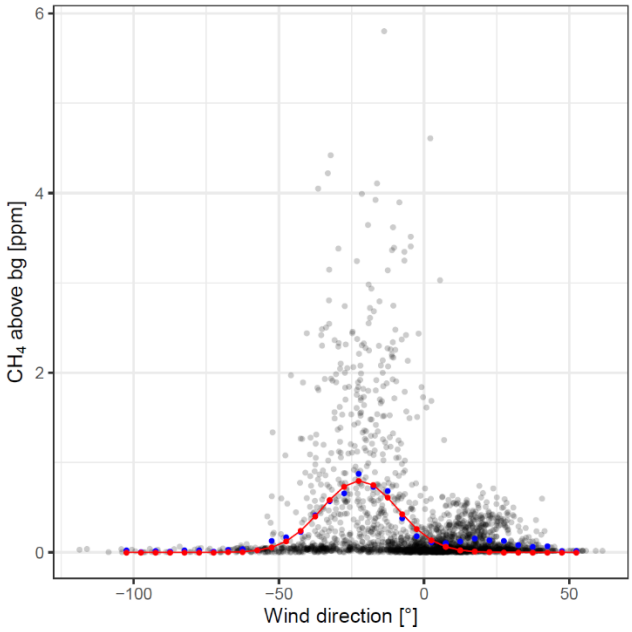


Figure 3.5. Example of methane enhancement binned by wind direction, where grey dots are methane concentrations above background, blue dots are average methane concentrations above background and the red line is the fitted Gaussian curve.

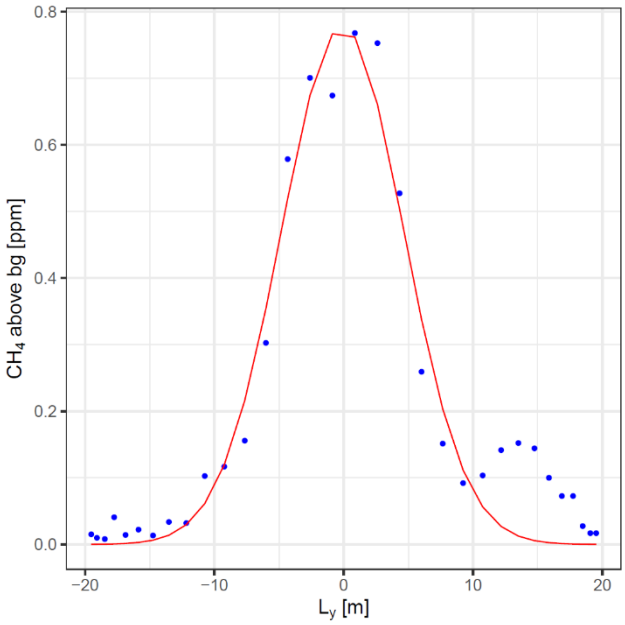


Figure 3.6. Close up of a Gaussian curve fitted average methane concentrations above background.

3.4 SURFACE FLUX CHAMBERS

Method principle

Total landfill methane emissions were quantified using surface flux chambers, which is a well-documented method recommended by the UK Environment Agency in LFTGN07 (Environment Agency, 2010). The flux chamber method has been successfully used for a long period of time due to its relatively simple measuring principle and inexpensive equipment. The method is based on the assumption that a series of time-based measurements of methane fluxes from representative locations can be converted into a total flux for the entire investigated landfill area. According to Environment Agency (2010) the flux chamber survey should follow an initial stage where the main characteristics of the landfill cap are recorded and mapped in a desk study, and a walkover survey is

conducted to identify areas and hotspots where methane concentrations measured close to the landfill surface are high.

A sensitive, hand-held gas detector is used in the walkover survey to record methane concentrations in the air close to the surface of the cap and thus detect areas with significant methane leakage. The walkover survey should be systematic and give a semi-quantitative ranking of the methane emissions from various features (Environment Agency, 2010). According to Environment Agency (2010), the flux chamber survey should be performed if ambient methane concentrations larger than 100 ppm are recorded during the walkover survey. In addition, the flux chamber method is, according to Environment Agency (2010), only valid if the emission rate of methane through the surface is between 5×10^{-5} mg m^{-2} s^{-1}, and 5 mg m^{-2} s^{-1}.

During the monitoring period, a flux chamber is sealed against the ground using low permeability material (e.g. clay, clayey soil) to provide a gas tight seal. Consequently, flux chamber monitoring is best suited for relatively smooth landfill caps, where a good seal can be prepared. The methane concentration within the chamber is measured at short time intervals (30 seconds) over a period of up to an hour. Thereafter the flux chamber is moved to the next location. The monitoring locations should be regularly spaced across the site. If the site varies in e.g. level of capping, the monitoring can be divided in various zones and features of the area. The number of flux measurements that preferably should be carried out to provide a statistically validated result is calculated according to Equation 3.6.

$$Number\ of\ boxes = 6 + 0{,}15\sqrt{x} \quad where\ x\ is\ the\ area\ in\ m^2 \quad (Eq.3.6)$$

Surface methane fluxes are measured at a number of representative sampling locations within each landfill zone and feature. An average flux of methane from the zone or feature is calculated from the emission rates measured by individual flux chamber measurements. Where a discrete feature within the zone is identified, this is not included in the main flux chamber array of the zone and the emissions from this feature are calculated separately. The total emission rate of methane from a zone is calculated by multiplying the average emission rate by the area of the zone. The overall emission rate of methane from the site is the sum of average emissions from the surface of each zone and feature (Environment Agency, 2010).

Instrumentation
In this study measurements were performed with flux chambers with a footprint of 0.49×0.64 m (=0.31 m^2) (Figure 3.7). The height of the chamber was 0.21 m resulting in a chamber volume of 0.065 m^3. At Slite landfill, different chambers were used: two surface flux chambers with a volume of 0.065 m^3, an area of 0.314 m^2 and a height 0.21 m and one chamber with a volume of 0.007 m^3, an area of 0.043 m^2 and a height 0.16 m. Each flux chamber had two ports fitted to the top of the chamber; an inlet port used for pressure equilibration and an outlet port for sampling. A gas detector, Sewerin Extec HS680, using a gas-sensitive semiconductor in the measurement range 0 – 1 %vol. and an infrared sensor for detecting methane up to 100 %vol. were used for detecting methane. A stopwatch and map of the site and GPS device were used for time logging and mapping of measurement location.

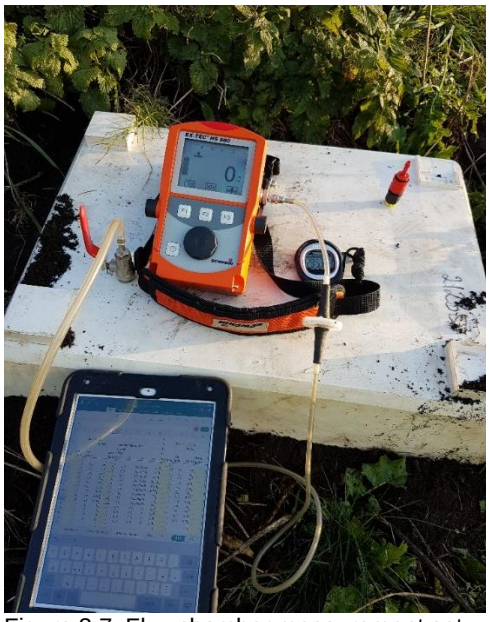


Figure 3.7. Flux chamber measurement set-up. Photo: Sweco Envionment.

Measurement of weather conditions
Data quality can be directly influenced by the meteorological conditions prevailing before and during the monitoring period. In particular, surface emission rates can be affected by factors such as rain and changes in barometric pressure. If surface soils and clay capping are saturated with water, they will be less permeable to gas and this will greatly reduce the surface flux (Environment Agency, 2010). Survey work and flux measurements should therefore be avoided following periods of unusually heavy rainfall. Surveys and flux measurements must never be conducted on areas where there is standing water (Environment Agency, 2010).

Calibration procedure
The instrument used in the study was calibrated by the supplier in June 2019 (calibration valid for 12 months).

Measurement uncertainty
Potential sources of error or uncertainty coupled to this monitoring method can be due to following factors:

- Instrument errors
- Uncertainties for the instrument in different intervals (measuring error of ±3% for the infrared sensor (0-100 %vol.) and ±30% for the gas-sensitive semiconductor (0-1 %vol.))
- Heterogeneity of the landfill capping
- Lower number of monitoring locations in our surveys than recommended in LFTGN07 (Environment Agency, 2010)
- Air intrusion from the base during monitoring
- Temperature variation in individual monitoring points
- Uncertainty due to the timing of the survey or failure to place the individual box appropriately
- Factors likely to make the monitoring invalid such as interference from other sources of methane or unrepresentative sampling locations in the zones.

Health, safety and risk assessment
A risk assessment was created for all of the site operators a few weeks before the measurements were due to start. The largest risks were slips and trips when walking on the slopes of the sites carrying the equipment. These risks were discussed and plans to avoid them were set. No health and safety issues arose during the measurements at the landfill sites.

Description of data processing
The methane flux (Q) from each measurement point is calculated by multiplying the rate of change of recorded methane concentrations (dc/dt) by the internal volume of the flux chamber (V) and dividing this value by the area (A) of the chamber connected to the ground (Equation 3.7).

The gradient of a graph of the methane concentrations (mg m^{-3}) versus time (seconds) is referred to as dc/dt and can be calculated using Equation 3.8 where n is the number of measurements and c is the concentration of methane. The methane concentration versus time is plotted on a graph and applying a 'best-fit slope' as the rate of change in methane. In some cases, individual data points in initial or late readings might deviate from a linear increase. Initial deviations can be due to for instance wind effects in the period before the chamber is fully sealed or the effect of the landfill gas collection system or the gas sampling. Decreasing or stabilizing values in later readings can be due to factors such as methane saturation in the flux chamber, back diffusion into the ground, or the effects of methane oxidation. In order to achieve a linear increase of methane concentration, individual data points can be removed from flow calculations (Environment Agency, 2010).

In order to evaluate the correlation between the measured data and the 'best-fit slope' the correlation coefficient R^2 can be calculated using Equation 3.9. If the correlation coefficient is below 0.8 the measurement is considered as not quantifiable according to Environment Agency (2010).

$$Q = \frac{V}{A} * \frac{dc}{dt} \qquad (Eq. 3.7)$$

$$\frac{dc}{dt} = \frac{n * \sum(t * c) - \sum t * \sum c}{n * \sum t^2 - (\sum t)^2} \qquad (Eq. 3.8)$$

$$R^2 = \left(\frac{n * \sum(t * c) - \sum t * \sum c}{\sqrt{(n * \sum t^2 - (\sum t)^2) * (n * \sum c^2 - (\sum c)^2)}} \right)^2 \qquad (Eq.\ 3.9)$$

Uncertainty analysis
In general, the results from the surface flux chamber method will be overestimated if the chambers are placed on top of hotspots with high emission rates, which are not treated as discrete features; the flux from the hotspots are thus assumed to be valid for a larger area (large radii of influence) than what is the actual situation. If chambers are placed randomly, the emission might very likely be underestimated as the chance of randomly hitting an emission hotspot is small. In both situations, the method will be more vulnerable to an over or underestimation the fewer chamber measurements are carried out. According to Appendix D in the LFTGN07 standard method, a kind of uncertainty analysis can be carried out where three different estimates of methane emission are calculated:

- The crude average of all measured emission rates within the zone multiplied with the total area of the zone (*this is refered to as the max, crude average*)
- Assuming a 20 m^2 area of influence around each sample point (*this is refered to as the mean, with 20 m radii of influence*). The total methane emission for the zone is thus the sum of the emission rates multiplied with an area of 20 m^2. In this case the emission from the remaining area (total area minus the number of flux measurements times 20) is assumed to be equal to the limit of quantification for the method.
- Assuming the area of a flux chamber itself placed on each of the surface points within the zone; that is, no zone of influence around each sampling point. (*this is refered to as the min, with no radii of influence*). In this case the emission from the remaining area is assumed to be equal to the limit of quantification for the method.

4 DESCRIPTION OF THE LANDFILL MONITORING CAMPAIGNS

The measurement campaigns were performed across the last week in October 2019 and the first in November 2019. During the measurements, the gas collection system was in operation at all three landfills. The collection rate of landfill gas is presented in Table 4.1.

Table 4.1. Landfill gas collected at the landfills during the measurements. At Filborna landfill, the 30th of October the gas collection rate was not recorded (marked unknown).

Date	Landfill	Flare/engine on or off	Engine (E) or flare (F)	Amount of methane flared/to engine (kg h^{-1})	Methane concentration (%vol)
Oct 28th	Slite	On	F	7.4	34.9
Oct 29th		On		7.3	33.3
Oct 30th	Filborna	On	E	81	38.5
Oct 31st		On		81	38.5
Nov 1st	Vankiva	On	E	48.6	36

4.1 TRACER GAS DISPERSION MEASUREMENTS

Table 4.2 provides an overview of the measurement campaigns performed at the three landfills using the tracer gas dispersion method. In total, four campaigns were performed; two campaigns at Slite landfill, and one at Filborna and Vankiva landfill, respectively. Each campaign consisted of an initial screening of methane in the area surrounding the landfill in order to assure that there were no other methane sources, which could interfere with the measurements of the landfill methane emission. To identify methane landfill emission hotspots and locate tracer gas release points, an on-site screening of the landfill was performed. Finally, a quantification of the methane emission was carried out consisting in one to three hours of acetylene release and downwind plume monitoring. Downwind measurements were carried out on suitable roads near the three landfills, and distances varied between the landfills, depending on the wind direction, the degree of dispersion and the accessibility of roads and possible interference with other methane sources in the area. Quantifications at the three landfills were made within a distance of 350 to 2200 m. Each plume traverse took between 1 and 5 minutes to perform and between 7 and 16 plume traverses were performed. The traverse time depended on the width of the plume at the measurement distance (depending on dispersion and distance) and driving speed. The driving speed was typically 15-30 km h^{-1}, depending on road and traffic conditions.

At Slite landfill, in addition to quantifying the emission from the whole landfill, emissions from two emission hotspots were quantified. The exact emission locations were found using a flame ionisation detector (FID) as well as the human nose (landfill gas has a distinct smell that can be noticed several meters downwind from an emission hotspot). The two emission hotspots included a hole/fracture in the ground on the slope near the top of landfill (point source A) and a leachate well at the base of the southern slope of the landfill (point source B). Tracer gas was then released at the exact emission point and tracer gas and methane concentrations were measured 5-20 meters downwind from the emission/release point. The measurements were done as slow traverses or stationary measurements.

Table 4.2. Overview of the TDM measurements at the three landfills. Weather data is from on-site weather station.

Landfill	Date and time interval	Average temperature (°C)	Average pressure (hPa)	Average wind direction (degree)	Average wind speed (m s^{-1})	Tracer gas release (bottles/total release in kg h^{-1})	Plume traverses	Measurement distance (m)
Slite - whole site	Oct 28th 13:50 - 14:30	5.6	1008.0	318	3.5	2/1.265	16	350
	Oct 29th 10:10 - 10:40	5.7	1014.4	329	4.3	2/1.277	16	350
Slite - point sources	Oct 29th 11:00 -11:15	5.7	1014.4	329	4.3	1/0.44	11	5-10
	Oct 29th 13:40 - 14:00	5.7	1014.4	329	4.3	1/0.44	Stationary	15-20
Filborna	Oct 31th 17:30 - 18:10	8.0	1019.5	275	3.2	3/2.851	9	2200
Vankiva	Nov 1st 18:20 - 19:10	3.8	997.8	87	1.3	2/1.826	7	1300

4.2 UAV-BASED PLUME MEASUREMENTS

Table 4.3 provides an overview of the measurement campaigns performed at the three landfills using the UAV-based plume measurements. In total, four campaigns were performed; two campaigns at Slite landfill, and one at Filborna and Vankiva landfill, respectively. A methane quantification consisted of a number of plume transects performed 10-150 m downwind the landfill, measured to the closest point. During the flight the plume was traversed 1 to 5 times at different heights (10 to 120 m above the ground). At two of the sites (Slite and Filborna) horizontal mapping of the landfill was carried out with the aim to screen the landfill surface for significant emission hotspots.

Table 4.3. Overview of the UAV-based measurements at the three landfills. Weather data is from visualcrossing.com.

Landfill	Date	Flight trial	Target	Time (UTC)	Wind speed (m s^{-1}) and direction	Average temperature (°C)	Average pressure (hPa)	Profiles (compiled of 5-11 traverses)	Distance of closest point of transect to landfill (m)
Slite	Oct 28th	1	Plume transects	1200-1600	5.9; 314°	6.4	1007.7	5	100
	Oct 29th	2	Plume transects	1100-1400	4.2; 321°	5.4	1015.6	3	100
	Oct 29th	3	Horizontal mapping	1030-1100	3.1; 333°	6.4	1014.8	1	10-20 m above
Filborna	Oct 31st	4	Plume transects	1500-1700	2.6; 283°	8.2	1014.7	2	150
	Oct 31st	5	Horizontal mapping	1200-1230	4.0; 289°	8.9	1014.9	1	30 m above
Vankiva	Nov 1st	6	Plume transect	1700-1715	3.1;107°	4.3	995.9	1	10

4.3 GROUND BASED PLUME MEASUREMENT (OTM 33A)

Table 4.4 shows an overview of the OTM 33A measurement campaigns on three landfills. At Slite landfill, screening was performed at the beginning of each measurement day. On the first day, the emission from a venting leachate well was quantified. During the second day a leak was found on the edge of the landfill, which was then quantified. The location of leaks and measurement locations are presented in Figure 4.1. At Filborna landfill one screening and two quantifications were performed. Many leaks and on-site sources were found on the landfill. However, only two of them were quantified. First the emission from a contaminated soil/sludge storage was quantified. Secondly the emission from a biogas plant and pretreatment was quantified. Emission sources and measurement locations are presented in Figure 4.2.

Table 4.4. Overview of the OTM 33A at the three landfills.

Date	ID	Time (Local)	Target	Wind speed (m s^{-1}) and direction	Average temperature (°C)	Average pressure (hPa)	Measurement period (min)	Measurement distance (m)	Source type	Stability class from PGI
Oct 28th	SE_1_S_1	10:50-12:20	Area screening	3.81 NW	7.23	1006.7	-	on landfill	Whole landfill	-
Oct 28th	SE_1_Q_1	13:10-13:40	Plume quantification	3.73 NW	5.96	1007.7	30	20	Leachate well	6
Oct 29th	SE_1_S_2	9:30-10:00	Area screening	3.48 NW	5.38	1014.2	-	on landfill	Whole landfill	
Oct 29th	SE_1_Q_2	10:20-11:10	Plume quantification	3.54 NW	6.22	1014.8	50	200	Leak located at one side of landfill	6/5
Oct 31st	SE_2_S_1	10:40-12:00	Area screening	3.97 W-NW	8.57	1015.2	-	on landfill	Whole landfill	-

Oct 31st	SE_2_Q_1	13:50-14:30	Plume quantification	3.87	W-NW	8.7	1014.5	40	45	Contaminated soil/ sludge storage	6
Oct 31st	SE_2_Q_2	15:15-15:35	Plume quantification	3.5	W-NW	8.62	1014.2	20	220	Biogas plant and pretreatment	6
Nov 1st	SE_3_S_1	9:45-10:30	Area screening	-		-	-	-	on landfill	Whole landfill	-

Figure 4.1. Map of the Slite landfill: purple squares show the measured area, purple dots show where measurments were perfomed; text in boxes refers to ID in Table 4.4.

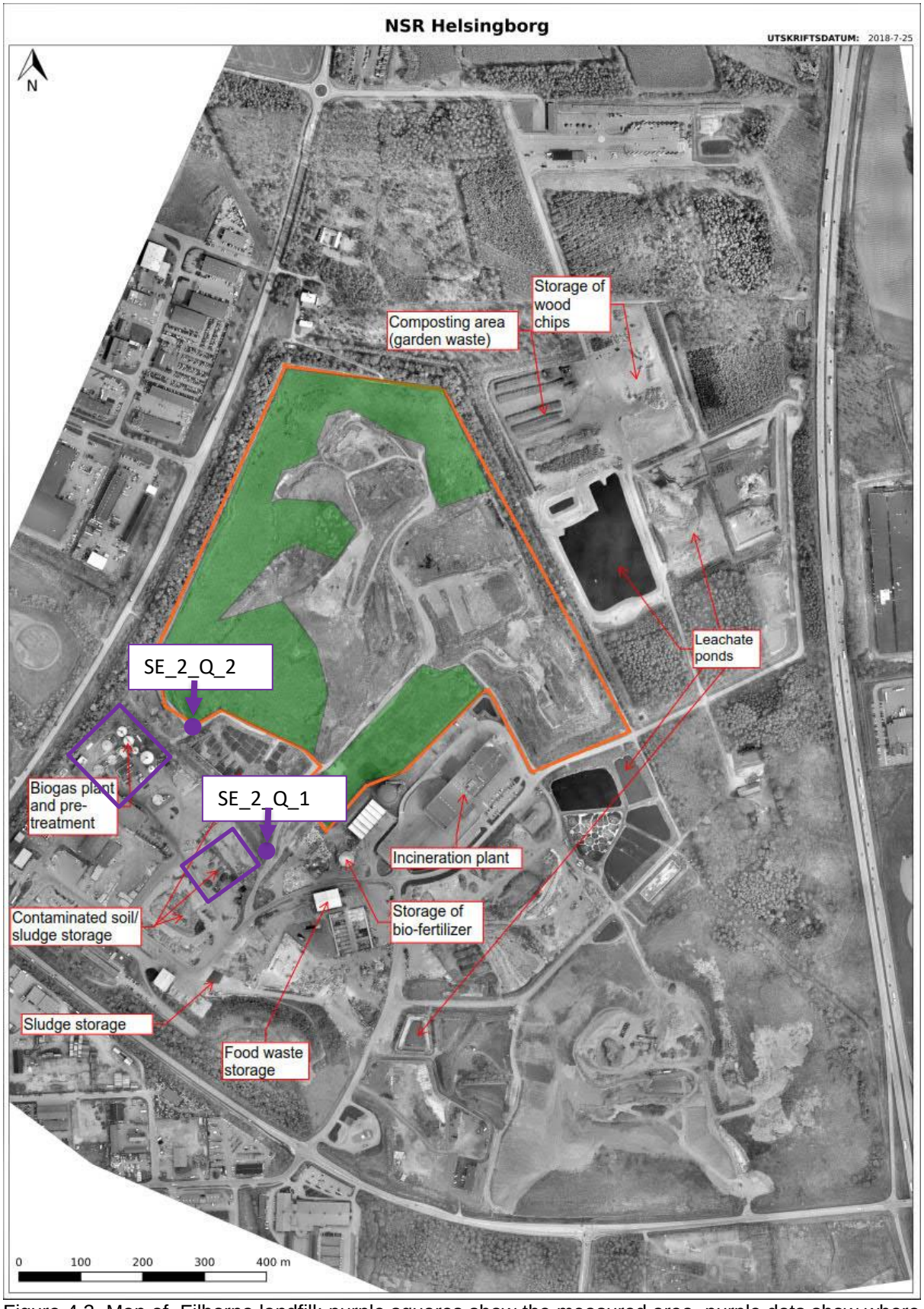


Figure 4.2. Map of Filborna landfill: purple squares show the measured area, purple dots show where measurments were perfomed; text in boxes refers to ID in Table 4.4.

4.4 SURFACE FLUX CHAMBERS

Walkover survey
Walkover surveys were carried out by Sweco at Slite and Filborna landfill in October 2018. Both surveys showed methane concentration at the landfills above 100 ppm. No walkover survey was performed at Vankiva landfill.

Designation of the landfill area into zones
Due to time limitations during the measurement campaigns, only a limited number of surface flux chamber measurements could be done at each site, see Table 4.5. The landfill area was split into zones (as described in the methodology section 3.4) based on the cover condition; capped and non-capped landfill areas. For Filborna landfill, the division into zones was made according to the designation of areas made by the landfill owner (based on capping, cells, etc.). However, this designation was only used when calculating emission rates for Filborna, not when calculating the number of chambers for each zone.

The flux chambers were placed on the surface semi-randomly: the approximate locations were randomly decided beforehand and the exact location decided on site. The exact location was decided so that the chamber was placed at a spot without vegetation and preferably sand, gravel or soil that easily let gas pass through the surface. The locations of the flux box are presented in section 6.5 Results: Figure 6.10, 6.11 and 6.12.

Table 4.5. Summary of number of flux chambers (where measurements were performed) and theoretical number of flux chambers according to the method in LFTGN07 by Environment Agency (2010). The number of flux chambers is based on the area of the zone according to section 3.4.

Landfill site	Zone	Number of surface flux chambers used in the campaign	Number of surface flux chambers according to LFTGN07	Area (m^2)
Slite	Capped area	15	18	6,200
	Non-capped area at the northwest slope	5	11	1,080
	Non capped area	15	27	19,920
Filborna	Capped area	2	60	129,570
	Non-capped area	28	69	177,450
Vankiva	Capped area	0	50	84,500
	Non-capped area	17	37	42,600

Atmospheric pressure
Atmospheric pressure data was downloaded from SMHI (Swedish Hydrological and Metrological Institute). For Slite, data was extracted from the weather station “Visby airport”, for Filborna the weather station “Helsingborg A” and for Vankiva the weather station “Hästveda Mo” when downloading data.

Assumptions
For each landfill a couple of assumptions have been made. These are described in the following.

Slite
It was not possible to place flux chambers on compacted gravel surfaces at the non-capped part of the landfill. A limited number of chambers were therefore placed at this part of the landfill. Furthermore, two different kinds of flux chambers were used.

Filborna
A surface flux chamber campaign was performed at Filborna landfill by Sweco in November 2018. In total, methane fluxes were measured in 93 locations. Due to time constraints, such a detailed survey was not possible to perform during the short campaign in 2019. Instead, the results from 2018 were partly used to estimate the emission in 2019. This was done by re-using some of the same locations (30 out of 93) as were monitored in 2018.

Capped areas were not prioritized; only two chambers were placed at capped areas. The emissions for capped areas were set to 5×10^{-5} mg m^{-2} s^{-1}, which corresponded to the average estimated flux in 2018 when 42 chamber measurements at capped areas was conducted and also corresponds to the limit of quantification. The measurements in 2018 showed no emissions above the limit of quantification at the capped areas. Furthermore, a few areas were not accessible for monitoring due to on-going site activities (construction of non-permeable surface, newly temporary capping of a landfill cell or storage of excavated soil).

According to Environment Agency (2010), data can be removed when calculating flux from each chamber in order to achieve a linear increase of methane concentration. Data removal has been done for the chambers: 7_2, 7_14 and 7_29. Two of the performed individual chamber measurements resulted in a flux larger than 5 mg m^{-2} s^{-1}, which is the higher limit of quantification for the method as described in section 3.4. There is no guidance in Environment Agency (2010) on how to deal with emission rates higher than 5 mg m^{-2} s^{-1}. For Filborna, calculations of the total emission has been done both with and without including the two measurements, assuming them to be 5 mg m^{-2} s^{-1} when included. Total flow calucations with and without these two measurements included are presented in Table 6.4.

During the 2019 monitoring campaign, measurements were only performed at or near one third of the locations from the 2018 campaign (30 out of 93) due to time constraints. To compensate for the low number of monitoring locations and attempt to avoid an under- or overestimation of the emission rate, the monitoring locations during the 2019 campaign were choosen such that the ratio between areas with a flux higher than 5×10^{-5} mg m^{-2} s^{-1} (red chambers) and areas with a methane emission lower than 5×10^{-5} mg m^{-2} s^{-1} (green chambers) during the 2018 campaign was to remain the same. (For further explanation of “green/orange/red chambers”, see chapter 6.5.) Due to construction work and soil stock piles on top of the landfill, all planned such monitoring locations were not accessible during the 2019 campaign. This resulted in the ratio between “green” and “red” chambers from 2018 differing, and a higher ratio of “red” chambers (points with known emissions) were monitored in 2019. This was subsequently compensated for in the emission rate calculations, (marked with “compensated for 2018 measurements”) by including a certain number of fictive measurements set to the limit of quantification in each zone, see Table 6.4.

An average and a standard deviation has been calculated including the four different ways of calculating the total emission (with/without high emission measurements and with/without compensation for 2018 measurements). The average of the four calculations is then used in the comparison of different methods in section 7.

Vankiva

Surface flux chambers were only placed at the non-capped part of the landfill. The flux from the capped area was assumed to be 5×10^{-5} mg m^{-2} s^{-1}, which is the limit of quantification for the method used.

IN accordance with the Environment Agency (2010) method, individual data points can be removed when calculating flux from each chamber in order to achieve a linear increase of methane concentration. Such data point removal was undertaken for chamber 16.

Weather conditions and gas extraction

The weather conditions during the measurements is presented in Table 4.4.

Table 4.4. Overview of weather conditions during the measurements

Landfill site	Date	Time interval	Atmospheric pressure (hPa)	Atmospheric pressure gradient 3/6/12 hours before (hPa)
Slite	Oct 28th	10:00-15:00	1012-1014	1010/1009/1007
	Oct 29th	8:30-14:00	1012.7-1014.9	1010.5/1009.1/1007.5
Filborna	Oct 30th	9:00-16:00	1031.0-1028.7	1030.4/1030.8/1030.3
	Oct 31st	10:00-15:00	1025-1023.8	1025/1025/1027
Vankiva	Nov 1st	12:00-17:00	1014-1007.8	1017.6/1019.1/1022.1

5 DESCRIPTION OF THE CONTROLLED RELEASE TEST

5.1 EXPERIMENTAL SETUP

With the objective to determine the accuracy of the quantification of three different methods (tracer gas dispersion, UAV-based plume measurement, and OTM 33A) a controlled methane release test campaign was carried out. The methane control release campaign took place on November 1st, 2019 in an old airfield close to Vankiva and measurements were performed between 13:00 to 16:00. The terrain where methane release took place was flat and no significant methane sources in the area were identified prior to the measurements.

Three bottles of methane were placed at the site. The exact configuration of the bottles can be seen in Figure 5.1. The methane release bottles were placed in a triangle with an approximate distance between the methane bottles of 40 m. Methane bottles (50 L) with purity of 99.95% were used for the controlled methane release, with a double pressure gauge valve and a 150 mm flow meter (Sho-rate from Brooks) calibrated for measuring between 0 and 50 L per minute and with an uncertainty of 5% on the maximum value. The combined uncertainty of the methane release was 8.7% of the total methane release rate. After the campaign flow rates were verified by performance of release test where methane gas bottles were weighed continuously and the weight loss and time recorded.

The total methane release rate was 2.70 ± 0.23 kg h^{-1} and occurred over a period of 2 ½ hours from 13:30 - 16:00 on November 1st, 2019.

Figure 5.1. The tracer gas configurations in methane controlled release campaign. Red pins mark the methane release. The location of the tracer gas releases are marked with white circles.

5.2 MEASUREMENTS PERFORMED

All three remote sensing teams quantified the methane emission from the controlled release.

Tracer gas was released at two locations as seen in Figure 5.1. The combined release rate was 1.36 kg h^{-1} equally distributed between the two gas bottles. Plume traverses were performed 175 m downwind the methane release area. Between 15:30 and 16:00, 7 plume transects were carried out. Initially, plumes were measured at a distance of 500 m downwind of the release area (7 incomplete

plume traverses). However at this distance the plume could not be completely traversed due to inaccessible road access.

UAV measurements were performed from 13:00 to 14:30. In total three flights were carried out each consisting of six to eleven traverses distributed from 3 m above ground and up to 120 m above ground. The flights were performed 100 m (once) or 900 m (twice) downwind of the release area. The first distance was limited by power lines and tree cover and was considered to be too close; the second distance was limited by tree cover and should ideally be closer for a source of this size. In addition, the plume was on the edge of the plane due to a lack of a data stream from the drone on the day. Thus, it may be only partly measured.

Two quantifications were carried out using OTM 33A. The first quantification was done at a distance of 22 m. Measurement was performed for 30 minutes. Averaged temperature was 7.57 ºC and average pressure was 1006.3 hPa. The wind speed and wind direction was 2.52 m s^{-1} and south-east, respectively. The calculated stability class from PGI was 1. The second quantification was done at a distance of 240 m. Between the measurement point and the release area there was a row with trees. The measurement was performed for 30 minutes. Averaged temperature was 7.11 ºC and average pressure was 1005.4 hPa. The wind speed and wind direction was 1.8 m s^{-1} and from south-east, respectively. The calculated stability class from PGI was 1.

6 RESULTS OF THE MEASUREMENT CAMPAIGNS

6.1 WEATHER CONDITIONS PRIOR AND DURING MEASUREMENTS

This section reports the weather conditions during the three measurement campaigns.

As previously mentioned, the weather conditions result in a variability in the emitted flux from a landfill. Notably, changes in atmospheric pressure drives a "breathing" phenomenon, where a high atmospheric pressure keeps the landfill gas in the ground (waste mass), and as the atmospheric pressure drops, landfill gas is release into the atmosphere. This phenomenon can be exacerbated by wind, reduced by saturated ground conditions etc. Furthermore, the measurement methods are all impacted by the weather conditions in different ways (as described elsewhere). The meteorological condtions prior to and during the measurement campaign were therefore carefully recorded. As a general comment, the weather conditions were favourable during the campaign and all measurements could be undertaken as planned.

Figure 6.1 shows the weather conditions in Slite for the two measurement days and the days prior to the campaign. During the two measurement days, temperature and wind speed were about 5.5 ^{o}C and 4 m s^{-1} respectively, whereas the absolute pressure increased from 1008 hPa to 1014 hPa between the first and the second campaign. However, the pressure trend prior to the measurements was similar both days and ranged from 0.5 hPa h^{-1} to 0.37 hPa h^{-1} 8 h before the measurement (Table 6.1).

In Filborna temperature and wind speed during the measurement period were 8 ^{o}C and 3.2 m s^{-1}, respectively (Figure 6.2). The absolute pressure on the measurement day was 1018 hPa. However, the measurements where performed duing a period with decreasing pressure. The pressure gradient was approximately -0.12 hPa h^{-1} 8 h before the measurement (Table 6.1).

On the day where measurements were performed at Vankiva landfill the temperature and wind speed were 6 ^{o}C and 4 m s^{-1}, respectively during the measurement period. The absolute pressure was 1002 hPa. Measurements were performed during a period with a pressure decrease and the pressure gradient was approximately -1.25 hPa h^{-1} 8 h prior to the measurement (Table 6.1).

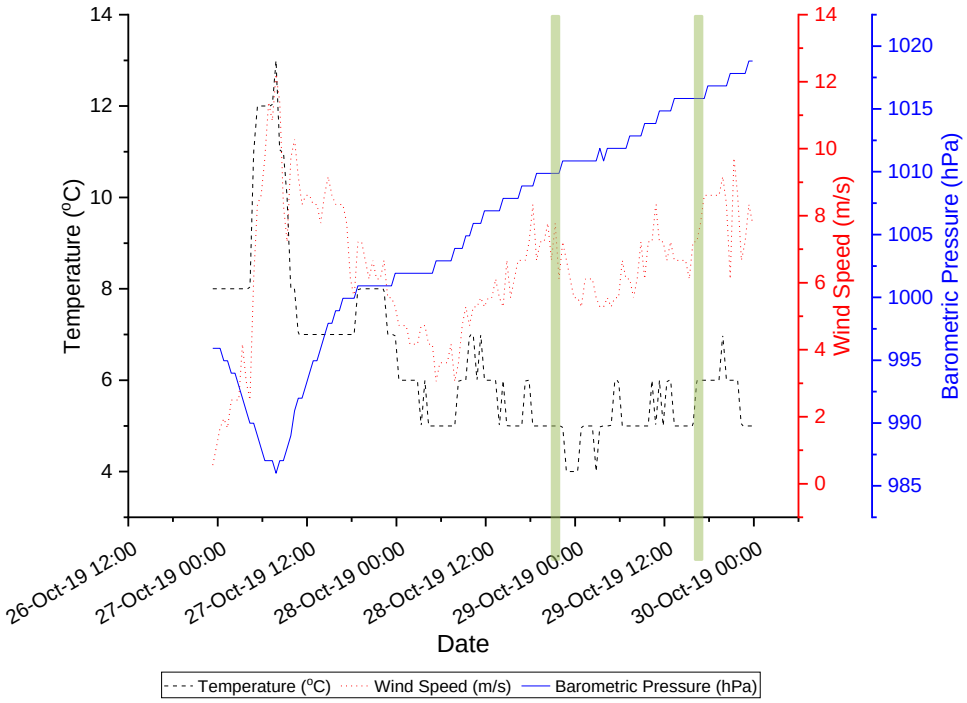


Figure 6.1. Weather data (Temperature, wind speed and barometric pressure) measured at Visby Airport Station 25 km from Slite landfill prior and during the measurement campaign. Light green shaded periods represent measurement periods of each TDM experiment. All data were retrieved by www.wunderground.com.

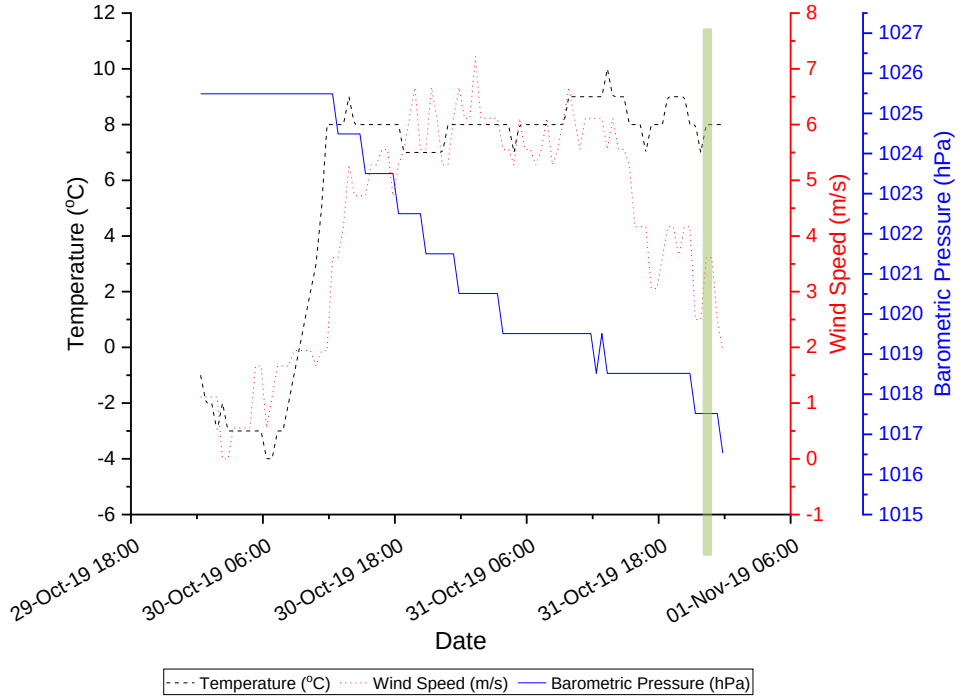


Figure 6.2. Weather data (temperature, wind speed and barometric pressure) measured at Angelholm-Helsinborg Airport Station 24 km from Filborna landfill prior and during the measurement campaign. Light green shaded period represents the measurement period. All data were retrieved by www.wunderground.com.

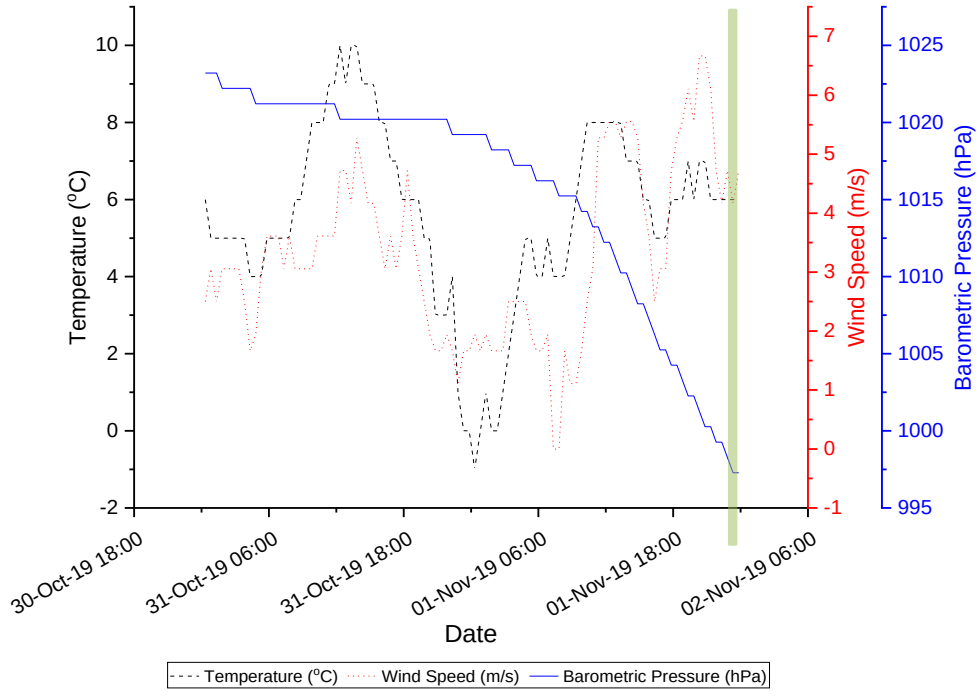


Figure 6.3. Weather data (temperature, wind speed and barometric pressure) measured at Kristianstad Airport Station 38 km from Vankiva landfill prior and during the measurement campaign. Light green shaded period represents the measurement period. All data were retrieved by www.wunderground.com.

6.2 TRACER GAS DISPERSION MEASUREMENTS

Initial area and on-site screening and tracer gas position
Initial screening of atmospheric methane concentrations in the area surrounding Slite landfill was performed during the first measurement day. No significant methane sources were found upwind of the landfill. Due to the limited time available for measurements only partial screening of the surrounding areas were performed at Filborna and Vankiva landfills. Nevertheless, from visual inspection of the areas around the landfills no significant methane sources upwind of the landfills, which potentially could interfer with the emission quantifications, were identified.

On-site methane screening was conducted at all three landfills by driving along all accessible roads. Figures 6.4 to 6.6 show the recorded atmospheric concentrations of methane above the background. The white targets mark the tracer gas placement locations. Tracer gas bottles were placed in specific locations to represent the most important emitting on-site areas/sources and at the same time to ensure coverage of the whole landfill area. The number of tracer gas bottles depended on the size and complexity of the site and the heterogeneity of emissions. For Slite the majority of the emissions exuded from the uncapped part of the landfill, and therefore two bottles of tracer gas were placed in this area (Figure 6.4). The area of Filborna landfill is about three times larger with several on-site emitting sources and souces right next to the landfill area, so three tracer gas bottles were used (Figure 6.5). Emissions from sources located right next to the landfill area (e.g. composting area, biogas plant, food waste storage, etc.) will be included in the measured emission. Finally, for Vankiva landfill two tracer bottles were used (Figure 6.6).

Quantification of the whole landfill site emission was performed two times for Slite and a single time for Filborna and Vankiva as shown in Table 6.1. At Slite landfill, the wind came from north-northwest on both measurement days. Due to relatively confined emission occurring primarily at the uncapped area of the landfill, measurements took place at a relative short distance from the landfill. During the campaign at Filborna, the wind direction was from the west and methane plumes were measured east-northeast of the landfill at a distance greater than 2000 m. Due to solar radiation atmospheric dilution was large, and so measurements were performed in the late afternoon and in the evening when the lower solar flux resulted in more stable atmospheric conditions and better noise-to-signal ratio. For Vankiva, the wind direction was from the east and the methane plumes were measured west of the landfill at a distance greater than 1000 m.

Figure 6.4. Relative atmospheric methane concentrations recorded when following the road around Slite landfill and on the road just downwind. Background methane concentration is subtracted and the maximum

concentration above the background level was found to be 102.7 ppb. Cyan targets mark the tracer gas placement used for whole site emission quantification.

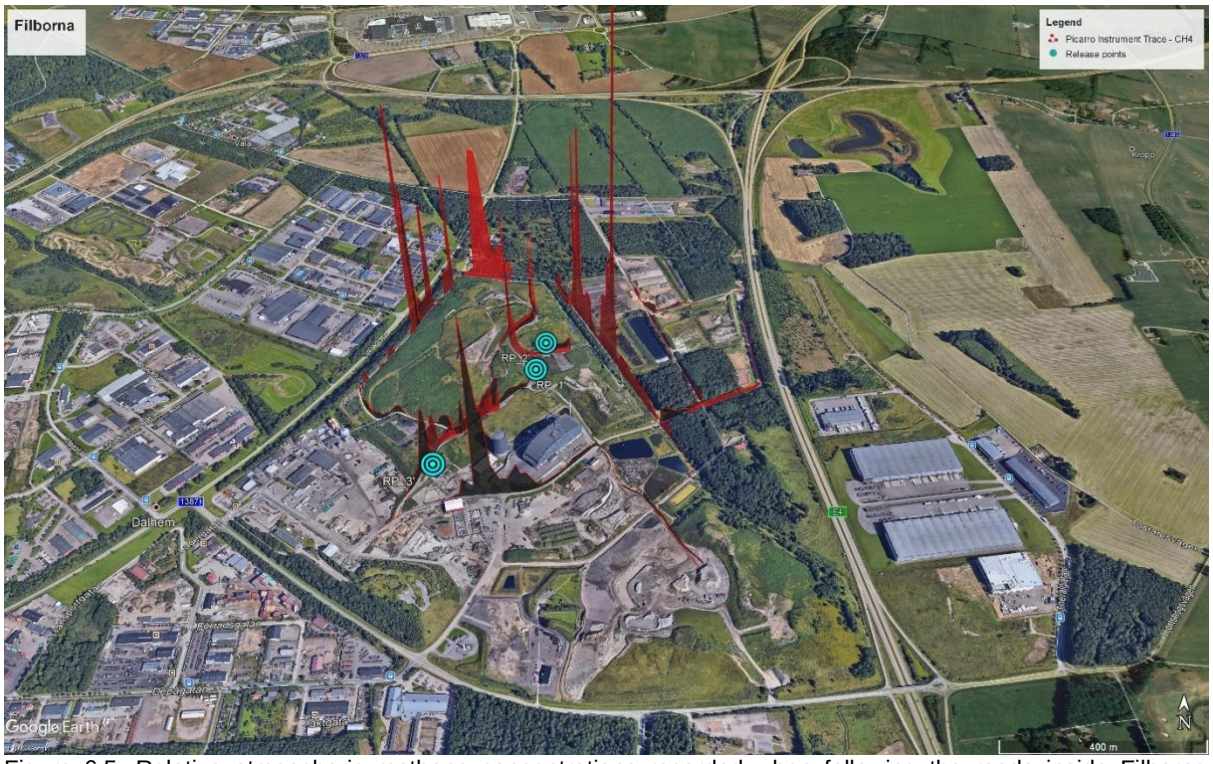


Figure 6.5. Relative atmospheric methane concentrations recorded when following the roads inside Filborna landfill. Background methane concentration is subtracted and the maximum concentration above the background level was found to be 19.91 ppm. Cyan targets mark the tracer gas placement used for whole site emission quantification.

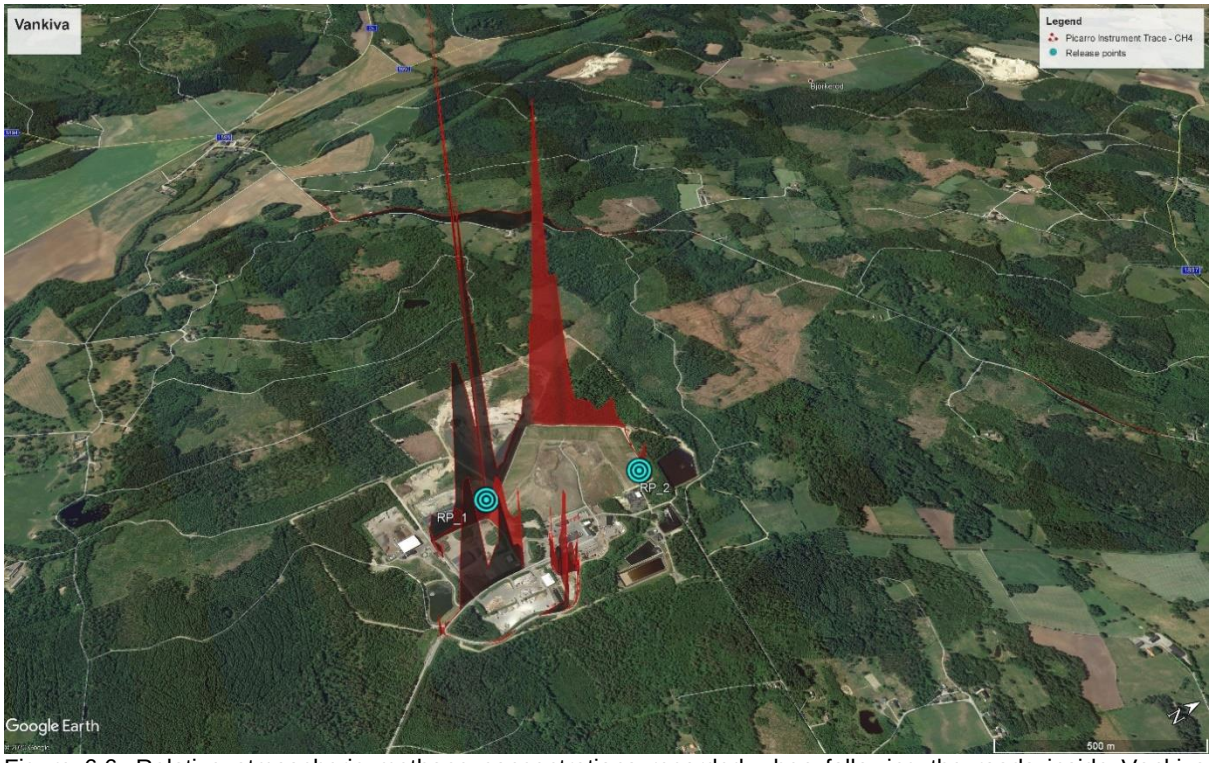


Figure 6.6. Relative atmospheric methane concentrations recorded when following the roads inside Vankiva landfill and on the road just downwind. Background concentration is subtracted and the maximum concentration above the background level was found to be 3.58 ppm. Cyan targets mark the tracer gas placement used for whole site emission quantification.

Whole site methane emissions quantification

Table 6.1 gives a summary of the methane emission rates measured on the three landfills. Also listed in the table is the atmospheric pressure during the measurements and the change in the pressure 8 hours prior to the measurement. The methane emission from Slite landfill was very low – around 1.4 to 1.6 kg h^{-1}. Figure 6.7 shows an example of a methane and acetylene plume measured downwind of the landfill. Measurements were performed during a period with a general increasing trend in the barometric pressure, which could have affected the emission rate resulting in a lower emission than under other pressure conditions. The two emission hotspots, which were quantified encountered a hole/fracture in the ground on the the slope near the top of landfill (point source A) a leachate well at the base of the southern slope of the landfill (point source B). The methane emission rates from these two hotspots were 0.41 and 0.34 kg h^{-1}, respectively. Combined, these two emission hotspots made up 46% of the total measured methane emission.

The highest methane emission rate of 139.5±9.4 kg h^{-1} was measured at Filborna landfill, which is also the largest of the three landfills. The emission from Vankiva was 14.5±2.8 kg h^{-1}. It is worth mentioning that all 3 sites had several activities besides landfilling such as composting, treatment of food waste, leachate treatment, sludge storage, which could also result in emissions of methane. The whole site methane emissions quantification encounters emissions from all on-site activities.

Table 6.1 Summary of methane emissions measured at the three landfills. The table also shows the atmospheric pressure behaviour over the period. Weather data prior to the measurement were collected from official nearby weather stations accessed through www.wunderground.com.

Landfill	Date	Time interval	Average pressure (hPa)	Atmospheric pressure gradient 8 hours before (hPa h^{-1})	Measured whole site methane emission (kg h^{-1}) and variability [a]	Total uncertainty of quantification (kg h^{-1}) [b]
Slite – Whole site	Oct 28th	13:50 - 14:30	1008.0	0.50	1.4 ± 0.05	0.2
	Oct 29th	10:10 - 10:40	1014.4	0.37	1.6 ± 0.03	0.2
Slite - Point sources	Oct 29th - A	11:00 - 11:15	1014.4	0.37	0.41 ± 0.08	0.08
	Oct 29th - B	13:40 - 14:00	1014.4	0.37	0.34 ± 0.07	0.07
Filborna	Oct 31th	17:30 - 18:10	1019.5	-0.12	139.5 ± 3.12	22.0
Vankiva	Nov 1st	18:20 - 19:10	997.8	-1.25	14.5 ± 1.05	2.4

[a] The variability of the averaged emission rate was calculated as the standard error of the mean (SEM). [b] The total uncertainty of the quantification was assessed by establishment of an error budget following the method in Fredenslund et al (2019) and adding the variability of the measured emission rate.

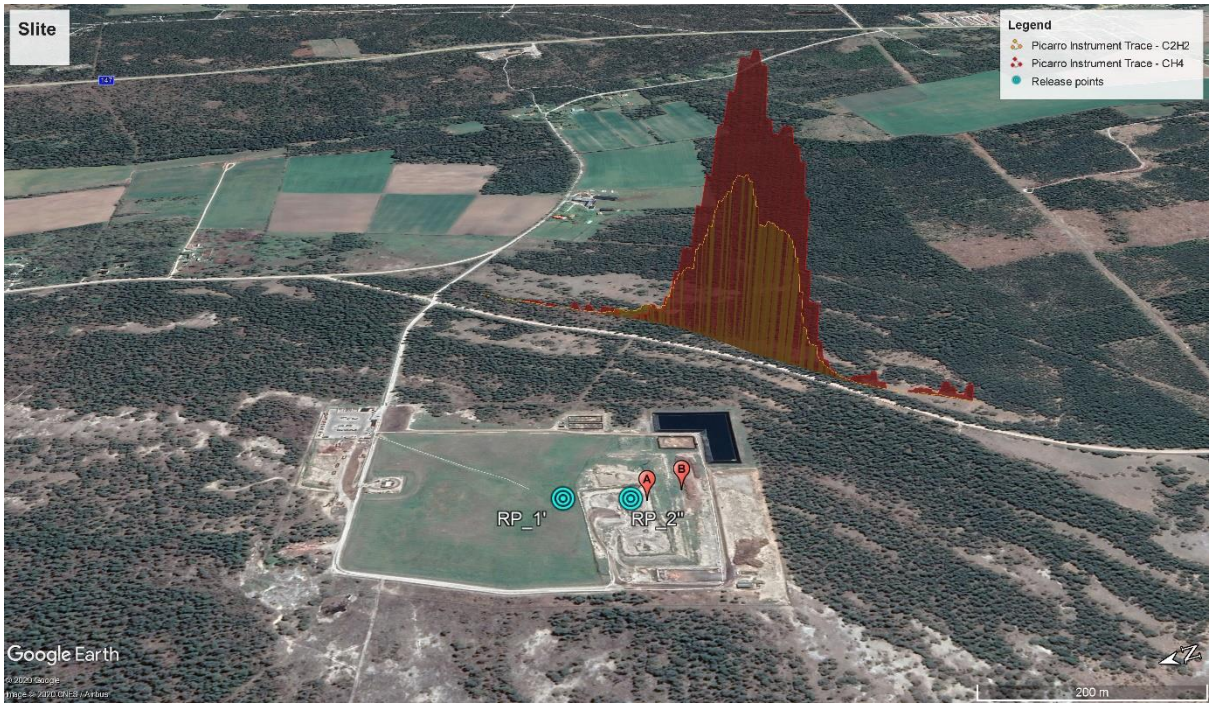


Figure 6.7 Geographical location of the plume in relation to Slite landfill on 29/10/2019 – 10:17. Relative methane

(red) and acetylene (yellow) concentrations above background. Maximum methane concentrations above background were 46 ppb and maximum acetylene concentrations above background were 21 ppb. The location of the two hotspots (A and B) are shown in red.

6.3 UAV-BASED MEASUREMENTS

Area screening

Screening flights were done at Slite and Filborna landfills. This involved flying the sensor at a roughly constant height in a pre-programmed grid pattern, and then Kriging the resulting data. This gives an indication of the location of methane production across the landfill, though this does not account for the horizontal offset caused by the time the plume takes to rise. Mean windspeed and direction are given as a visual aid to the direction and size of this offset. At Filborna, a precise digital elevation model was not available.Therefore the height was programmed to be 30 m. At Slite, an initial height of 10 m was chosen, but inaccuracies in the digital elevation model and high fencing meant that this was increased to 20 m for the southern part of the landfill. Two flights were done at high density on Slite and one flight was done at low density at Filborna.

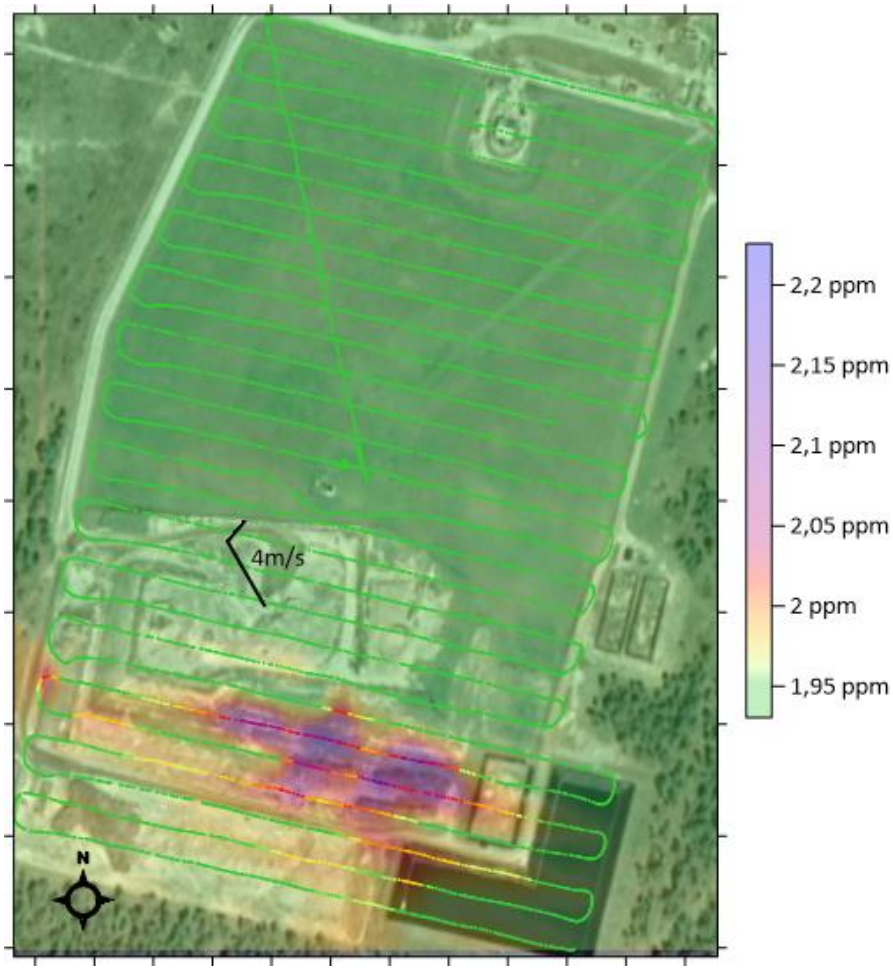


Figure 6.8 Horizontal methane plane at 10-20 m AGL, Slite

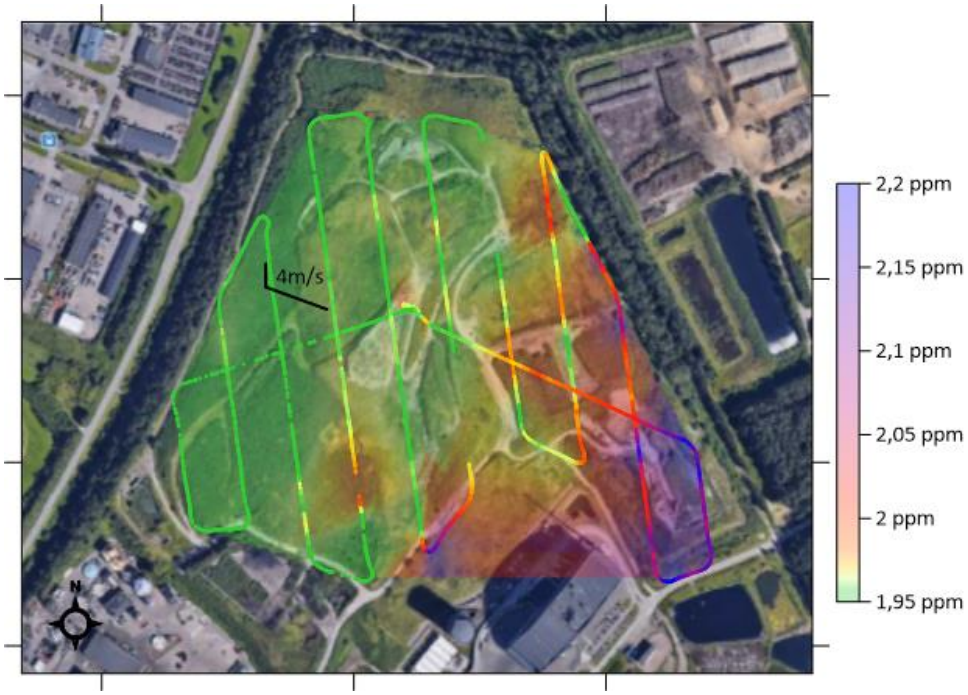


Figure 6.9 Horizontal methane plane at 25-35 m AGL, Filborna

Quantification
Four or five flights were undertaken on each measurement day. Mapping was conducted using two flights at Slite and one flight at Filborna; in addition, one flight at Filborna was aborted. This was due to differences between digital elevation models and the landfill surface resulting in excessively low altitude. Due to constraints of time, battery and daylight, only one transect was flown in Vankiva and the more precise sonic anemometer was not used; instead the Kestrel anemometer (with 2 minute averages) was used. In all cases, anemometers were placed as high as possible on the landfill; in Slite this was concordant with the lowest flight plane but was higher in Filborna and Vankiva. Some anemometer data for the mast-mounted sonic anemometer were lost or not collected at Slite; the Kestrel was used in these instances.

The following diagrams show example plume models for each site, with values in kg h^{-1} m^{-2} above background averages. Axes are in metres, with the X axis expressing distance from satellite elevation (the bottom of the transect is typically 15 m above this due to forest cover) and the Y axis expressing distance from a nominal start point of the transect. Gaps are due to occasional losses of drone telemetry due to interference rather than absence of measured values.

Methane emission rates are compiled in Table 6.2. As can be seen, the plume is relatively well-defined in Slite, coming from a small area; this corresponds with the observation that emissions arise from a small area. In future camaigns, the plume could be sampled more densely for better accuracy. In Vankiva and Filborna, the plume is much more extensive; in Filborna it extends up to the maximum height above ground of 120 m, which is an unexpected finding as methane usually disperses relatively horizontally. The results are an order of magnitude larger in Filborna and Vankiva; as there was not time to do many transects in these locations due to logistical issues, the range of possible values is larger.

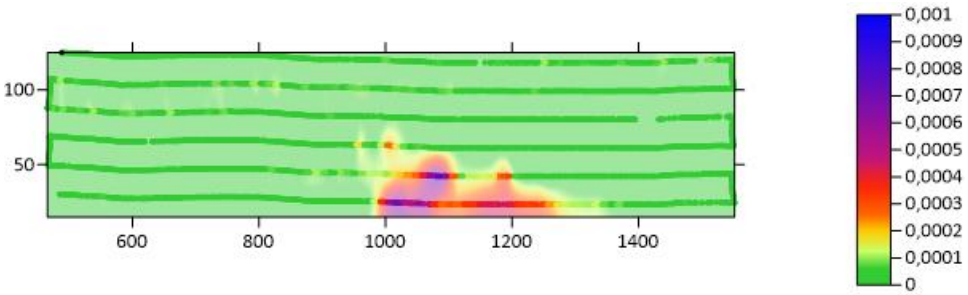


Figure 6.10 Methane plume Slite 29-1

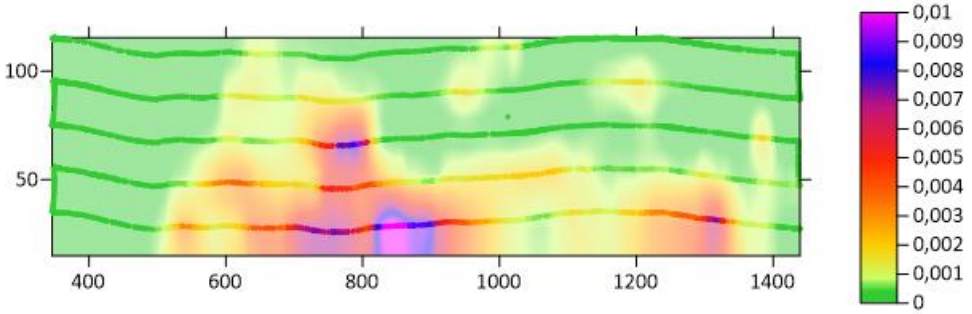


Figure 6.11 Methane plume Filborna 31-1

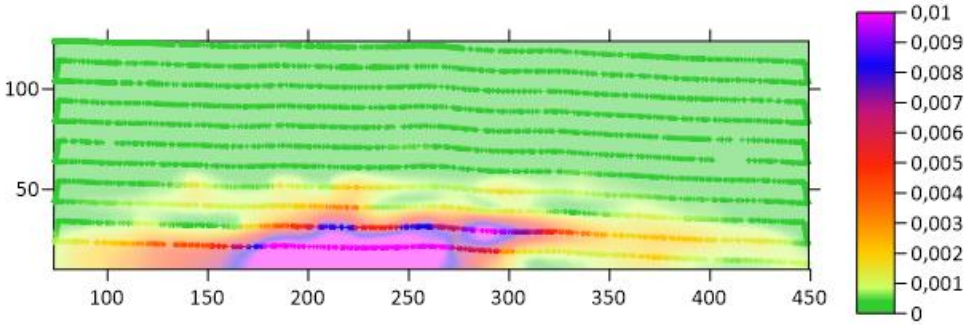


Figure 6.12 Methane Plume Vankiva 1-1

Table 6.2. Overview of the UAV-based measurements at the three landfills.

Landfill	Date	Flight trial	Time (UTC)	Wind speed (m s^{-1}) and direction	Distance of closest point of transect to landfill (m)	Transect	Methane emission rate (kg h^{-1})	Uncertainty (kg h^{-1})	Average methane emission rate (kg h^{-1}) and variability [a]
Slite	Oct 28th	1	1200-1600	5.9; 314°	100	Slite 28-1	3.66	13.65	4.7 ± 1.2
						Slite 28-2	7.23	12.19	
						Slite 28-3	5.54	13.47	
						Slite 28-4	6.46	13.00	
						Slite 28-5	0.39	5.08	
	Oct 29th	2	1100-1400	4.2; 321°	100	Slite 29-1	8.12	20.25	5.8 ± 1.2
						Slite 29-2	4.64	17.53	
						Slite 29-3	4.59	14.92	
Filborna	Oct 31st	4	1500-1700	2.6; 283°	150	Filborna 31-1	146.70	37.47	130.7 ± 22.6
						Filborna 31-2	114.73	22.80	
Vankiva	Nov 1st	6	1700-1715	3.1;107°	10	Vankiva 1-1	51.97	113.82	52.0

[a] The variability of the averaged emission rate was calculated as the standard error of the mean (SEM).

6.4 GROUND-BASED PLUME MEASUREMENTS (OMT33A)

Table 6.3 provides a summary of methane emission rates from landfills an d on-site emission sources quantified using OTM 33A. During the screening on the Slite landfill (site SE_1) emissions of methane were observed from the southwestern part of the landfill and from a leachate well. Both of these emission sources were quantified. First, on 28th October, the emission from the leachate well was quantified at 0.037±0.003 kg CH_4 h^{-1}. According to the Robertson et. al 2017 this emission rate is close to the detection limit and the measurement uncertainty could be higher than calculated from the data. On the next day, 29th October, the emission from a hotspot located at the southwestern part of the landfill was quantified to 0.356±0.050 kg CH_4 h^{-1} or 0.533±0.076 kg CH_4 h^{-1} depending on the choice of stability class. The calculated stability class was in between 5 and 6, so results are presented for both. The stability class has a big influence on the calculation of the methane emission rate.

Quantification of the total emission from the Filborna landfill (site SE_2) was not possible using OTM 33A. A defined methane plume from the landfill area was detected about 1 km downwind. This distance is too far and the plume was too disperse for the OTM 33A method to be applied. Instead two on-site methane emission sources were quantified. The first on-site source quantified was a contaminated soil/sludge storage consisting of a number of small sludge or soil basins. The combined area with the sludge basins was treated as a point source. At a distance of around 40 m, small methane plumes blended into one confined plume. The methane emission from this source was calculated to 0.264±0.044 kg CH_4 h^{-1}. The second on-site source was a biogas plant and pretreatment. At a distance of around 220 m, the biogas plant and pretreatment could be considered a point source. Emission from the biogas plant and pretreatment was calculated as 35.1±4.1 kg CH_4 h^{-1} suggesting that this source contributed significantly to the overall emission from the landfill area.

Table 6.3 Summary of methane emission rates from landfills and on-site emission sources quantified using OTM 33A.

Date	ID	Time (Local)	Source	Emission (kg CH_4 h^{-1})	Std of emission (kg CH_4 h^{-1})	Stability class from PGI
28.10.2019	SE_1_Q_1	13:10-13:40	Slite, Leachate well	0.037	0.003	6
29.10.2019	SE_1_Q_2	10:20-11:10	Slite, Emission hotspot at the southwestern part of the landfill	0.356	0.050	6
29.10.2019	SE_1_Q_2	10:20-11:10	Slite, Emission hotspot at the southwestern part of the landfill	0.533	0.076	5
31.10.2109	SE_2_Q_1	13:50-14:30	Filborna, contaminated soil/ sludge storage	0.264	0.033	6
31.10.2109	SE_2_Q_2	15:15-15:35	Filborna, biogas plant and pretreatment	35.1	4.1	6

6.5 SURFACE FLUX CHAMBER MEASUREMENTS

Quantification
In calculations of this study, three estimates (min, mean and max) as described in section 3.4 have been calculated. The mean and min values are likely to be an underestimation of the true emission: it is very likely that there are a number of hotspots that are not discovered during the measurements. Hence the max (crude average) is used when comparing the emission rates in this study.

Table 6.4 presents the results from the different ways of calculating the total methane emission for Filborna landfill. An average and standard deviation of the different calculation methods has been calculated. A similar calculation was not possible to perform for Slite and Vankiva landfill (only min, mean and max were calculated, see Table 6.6).

Table 6.4. Summary of methane emission calculations at Filborna landfill. The different ways of calculate are described in section 3.4 and 4.4.

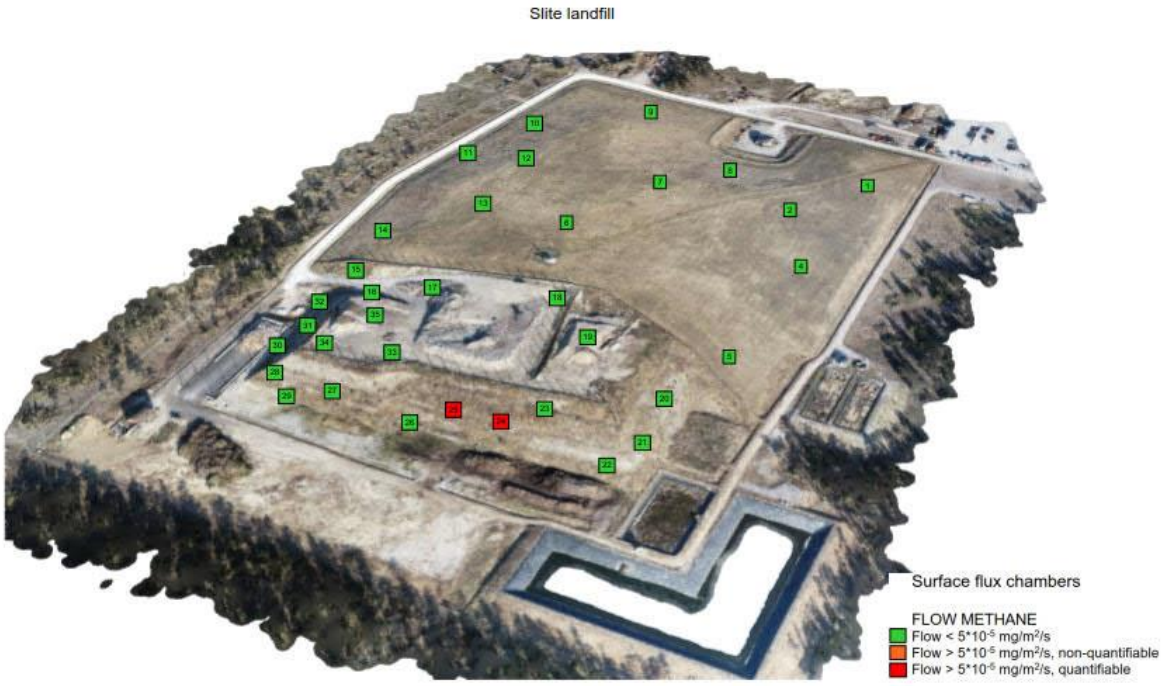
	Compensated for 2018 measurements	Flux measurements >5 mg m^{-2} s^{-1} included	Total landfill methane emission rate (kg h^{-1})		
			Min (no radii of influence)	Mean (20 m^2 radii of influence)	Max (crude average)
Calculation 1			0.066	0.77	242
Calculation 2		X	0.078	1.49	441
Calculation 3	x		0.066	0.71	213
Calculation 4	x	X	0.077	1.48	345
Average, calculation 1-4			**0.072**	**1.12**	**310**
Standard deviation, calculation 1-4			**0.0065**	**0.43**	**104**

Table 6.5 presents the results from the surface flux chamber. To illustrate the results, the calculated emission from each chamber measurement has been coloured according to Table 6.5. Chambers with a methane flux larger than 5×10^{-5} mg m^{-2} s^{-1} but not quantifiable (R^2 below 0.8), have in this survey been coloured orange and, according to the LFTGN07 standard method, assumed to be equal to the limit of quantification in the calculations of total emission from the landfills. Surface chambers where

no or low fluxes of methane was measured (< 5×10^{-5} mg m^{-2} s^{-1}) have been coloured green and chambers with a methane flux larger than 5×10^{-5} mg/m^2/s and R^2 above 0.8 have been coloured red. The colouring of the chambers is not a part of the LFTGN07 method but are for illustrative purposes. The results for each site are presented in Figure 6.10 (Slite), Figure 6.11 (Filborna) and Figure 6.12 (Vankiva).

Table 6.5. Summary of the results of the surface flux chamber method. *Measurements with a flow >5 mg m^{-2} s^{-1}**mg m^{-2} s^{-1}** is included.

Landfill site	**Total number of chamber measurements**	**Number of green chambers**	**Number of orange chambers**	**Number of red chambers**	**Number of measurements with flow >5 mg m^{-2} s^{-1}**	**Emission range (min-max) (mg m^{-2} s^{-1})**	**Average emission flux in red chambers (mg m^{-2} s^{-1})**	**Emission range (min-max) (mg m^{-2} s^{-1})**
Emission rate (mg m^{-2} s^{-1})		≤ 5×10^{-5}	>5×10^{-5}	> 5×10^{-5}	>5	> 5×10^{-5}	> 5×10^{-5}	> 5×10^{-5}
Quantifiable		No	No	Yes	No	Yes	Yes	Yes
Slite	35	33	0	2	0	0.81 - 4.7	2.8	0.81 - 4.7
Filborna	30	16	3	11*	2	3.4×10^{-4} – >5	1.1	3.4×10^{-4} – 7.1
Vankiva	18	13	2	3	0	2.6×10^{-4} – 0.11	0.029	2.6×10^{-4} – 0.11



Figur 6.10. Slite landfill, results from surface flux chamber measurements in October 2019. All chambers are marked in the figure and the rate of the flux is decribed by colour: chambers where no methane was measured (< 5×10^{-5} mg m^{-2} s^{-1}) have been coloured green and chambers with a methane flux larger than 5×10^{-5} mg m^{-2} s^{-1} and R^2 above 0.8 have been coloured red. Chambers with a methane flux larger than 5×10^{-5} mg m^{-2} s^{-1} but not quantifiable (R^2 below 0.8), have been coloured orange.

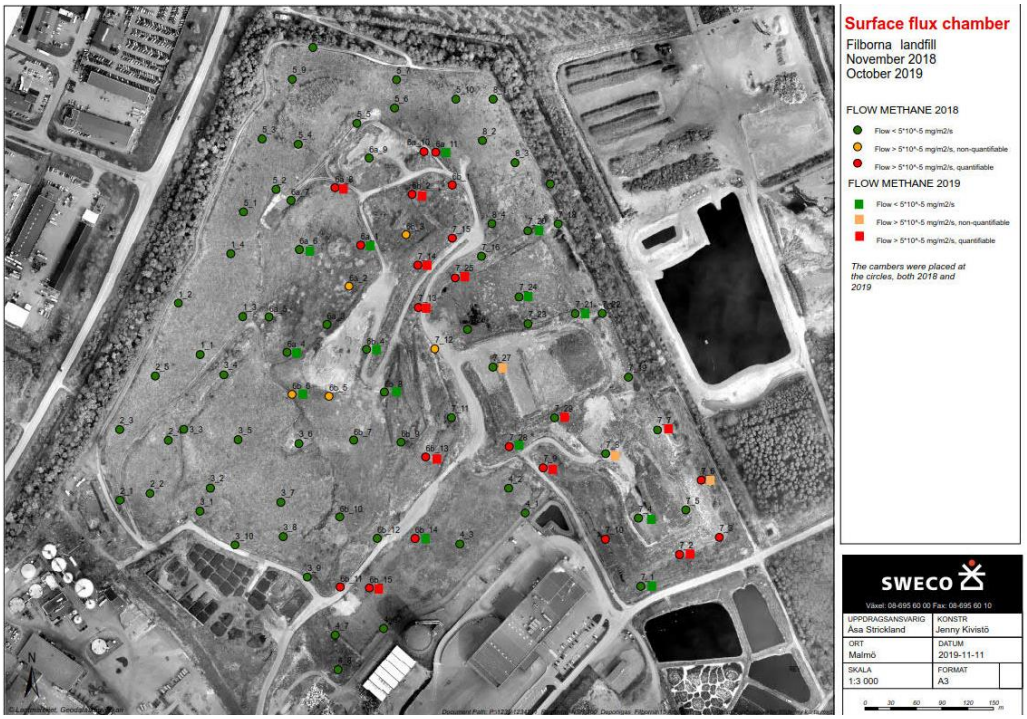


Figure 6.11. Filborna landfill, results from surface flux chamber measurements in November 2018 and October 2019. All chambers are marked in the figure and the rate of the flux is decribed by colour: chambers where no methane was measured (< 5×10^{-5} mg m^{-2} s^{-1}) have been coloured green and chambers with a methane flux larger than 5×10^{-5} mg m^{-2} s^{-1} and R^2 above 0.8 have been coloured red. Chambers with a methane flux larger than 5×10^{-5} mg m^{-2} s^{-1} but not quantifiable (R^2 below 0.8), have been coloured orange.

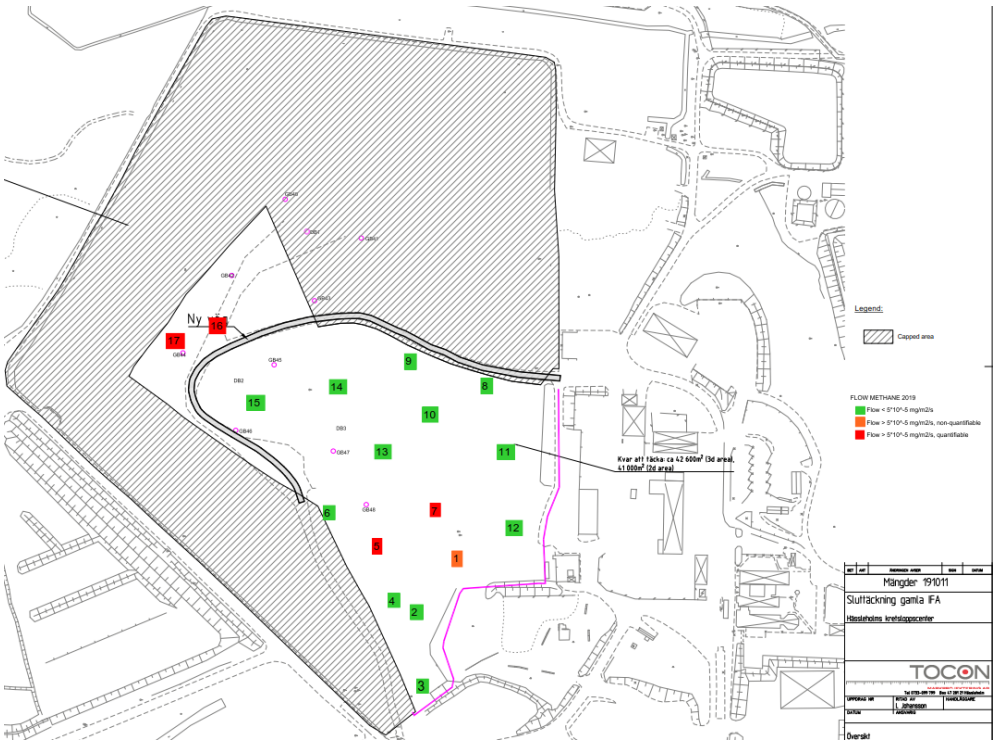


Figure 6.12. Vankiva landfill, surface flux chamber measurements. All chambers are marked in the figure and the rate of the flux is decribed by colour: chambers where no methane was measured (< 5×10-5 mg m^{-2} s^{-1}) have been coloured green and chambers with a methane flux larger than 5×10-5 mg m^{-2} s^{-1} and R2 above 0.8 have been coloured red. Chambers with a methane flux larger than 5×10-5 mg m^{-2} s^{-1} but not quantifiable (R2 below 0.8), have been coloured orange.

The methane emissions were calculated to 4.3 kg h^{-1} for Slite landfill, 310 kg h^{-1} for Filborna landfill and 1.1 kg h^{-1} for Vankiva landfill. This is based on the max value (crude average) for each landfill (Table 6.6). Standard diviation was not possible to calculate for Slite and Vankiva landfill.

Table 6.6 Summary of methane emissions measured at the three landfills Oct 28th-Nov 1st. For statistical data, see section 3.4, where min, mean and max are explained.

Date	Landfill site	Min emission (kg h^{-1})	Standard deviation	Mean emission (kg h^{-1})	Standard deviation	Max (crude average) emission (kg h^{-1})	Standard deviation
Oct 28th Oct 29th	Slite	0.31	-	0.41	-	4.3	-
Oct 30th Oct 31st	Filborna	0.072	0.0065	1.12	0.43	310	104
Nov 1st	Vankiva	0.023	-	0.031	-	1.1	-

Certainty analysis

For Slite landfill, the calculated maximum methane emission can be compared with the calculated flux from the measurements in 2018: 10.2 kg h^{-1}. The atmospheric pressure was first increasing and then decreasing during the measurement in 2018. In 2019 the atmospheric pressure was increasing, which could explain the lower methane emission rate in 2019.

The calculated maximum methane emission rate for Filborna landfill can be compared with the calculated methane surface emission in 2018: 400 kg h^{-1}. During the measurements in 2018 the weather was comparable to the weather during this campaign 2019, except for the atmospheric pressure, which was *increasing* during three out of the four days of measurements in 2018. The atmospheric pressure in 2019 was decreasing or stable. For Filborna, calculation of total emissions have been performed in four different ways, presented in Table 6.4. The values presented in Table 6.6 are the min, mean and max and the standard deviation of the four measurements. The standard deviation is calculated using the Microsoft excel formula STDEV.S, which calculates the standard devation based on a sample of a population.

6.6 RESULTS OF THE CONTROLLED RELEASE CAMPAIGN

Figure 6.13 shows the comparison of quantifications of the controlled methane release using three different measurement methods, Table 6.6 lists methane emission rates. Measured emission rates are compared to the controlled methane release of 2.70 kg h^{-1}.

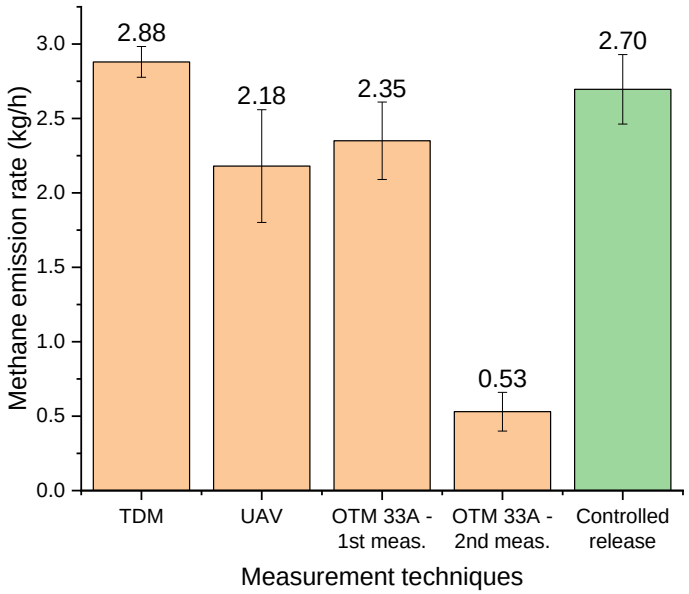


Figure 6.13. Comparison of quantifications of the controlled methane release using three different measurement methods; tracer gas dispersion measurements (TDM), UAV-based plume measurements (UAV) and Other test method (OTM 33A). The controlled release rate was 2.70 ± 0.23 kg h^{-1}.

Table 6.6. Methane emission rates measured in a controlled methane release campaign using three different ground-based remote sensing techniques (trace gas release, UAV and OTM 33A). All emissions are compared to the controlled release of 2.70 kg h^{-1}. The error is calculated as 100*(Measured rate - Controlled release rate)/Controlled release rate.

Method	**TDM**		**UAV**		**OTM 33A**	
Time interval	15:30 – 16:00		13:00 – 14:30		13:25-13:55 (close) 14:05-14:25 (far)	
	Average methane emission rate (kg h^{-1}) and variability [a]	**Total uncertainty (kg h^{-1})** [b]	**Average methane emission rate (kg h^{-1}) and variability** [a]	**Uncertainty (kg h^{-1})**	**Average methane emission rate (kg h^{-1}) and standard deviation**	**Uncertainty (kg h^{-1})**
			2.49	1.38	2.35[c] ± 0.26	
			2.63	5.70	0.53[d] ± 0.13	
			1.43	4.39		
Average	2.88 ± 0.10	± 0.44	2.18 ± 0.38			
Error (%)	6.8		-19.0		-12.5[c] / -80.3[d]	

[a] The variability of the averaged emission rate was calculated as the standard error of the mean (SEM). [b] The total uncertainty of the quantification was assessed by established of an error budget following the method in Fredenslund et al (2019) and adding the variability of the measured emission rate.

The measured methane emission rate using the tracer dispersion method was 2.88 kg h^{-1} with a variability on the measurement of ±0.10 kg h^{-1}. The emission was overestimated by 6.8% in comparison with the controlled methane release rate of 2.70 kg h^{-1}. The error of +6.8% is low and within the total uncertainty of the measurement method, which for this specific measurement campaing was assessted to be ± 15.2% (corresponding to ± 0.44 kg h^{-1}). The quantified emission of 2.88 kg h^{-1} ± 0.44 kg h^{-1} is comparable to the controlled methane release rate of methane (2.70 kg h^{-1} ± 0.23 when considering the uncertainty of the release rate. The measuring distance was slightly lower than recommended (4-5 times the width of the source), hence there could be a different evolution of methane and tracer gas plumes. Methane bottles were slightly closer to the measuring traverse than the two tracer gas bottle. This means the tracer gas travelled a slightly longer distance

than the methane gas and would therefore be slightly more diluted. This could cause an over-estiamtion of the methane emission.

In total, three UAV flights were carried during the controlled release test. The flights were performed 100 m (once) or 900 m (twice) downwind of the release area. The first two flights gave comparable emission rates of 2.49 kg h^{-1} and 2.63 kg h^{-1} whereas the last flight gave lower emission rate of 1.43 kg h^{-1}. All three flights gave emission rates that were lower than the controlled release rate of 2.70 kg h^{-1} (between -2 and 47%). The quantification was challenged by the local topographic conditions as well as the wind conditions. The area of the release was an open field surrounded by trees and pylons, and windspeed was measured using a 3D anemometer placed on site. The two possible downwind intervals for measurement with a consistent height were 100 m and 900 m; the signal from the latter distance was difficult to distinguish from background noise for an emission source of this relative small size. Due to uncertainty about the precise direction of the local wind, the two distant transects were off-centre, despite spanning 600 m. The nearest transect was done entirely manually due to the dangers of flying near pylons – all other transects were programmed prior to the flight. The second measurement covered most of the plume, but the off-centered plume measured during the third flight lead to a underestimation of the emission rate. On the contrary the first flight, which covered the whole plume, resulted in an emission rate similar to the controlled release rate. It is interesting again to note the elevation of the methane plume, parts of which rose to more than 120 m above ground level in under a km. Improvements in future measurement technique will prevent this kind of off-centering.

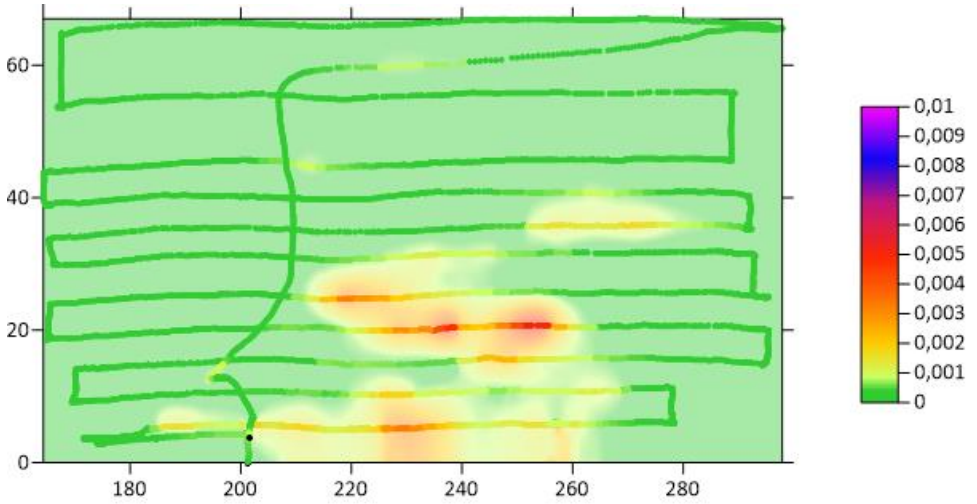


Figure 6.14. Tracer gas plume transect 1, conducted 100 m downwind

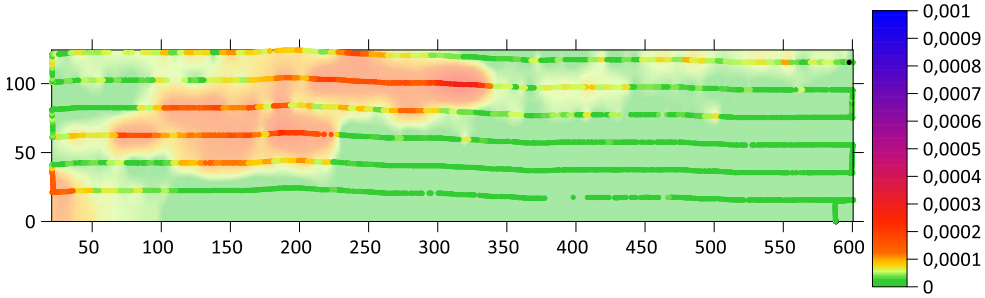


Figure 6.15. Tracer gas plume transect 3, conducted 900 m downwind

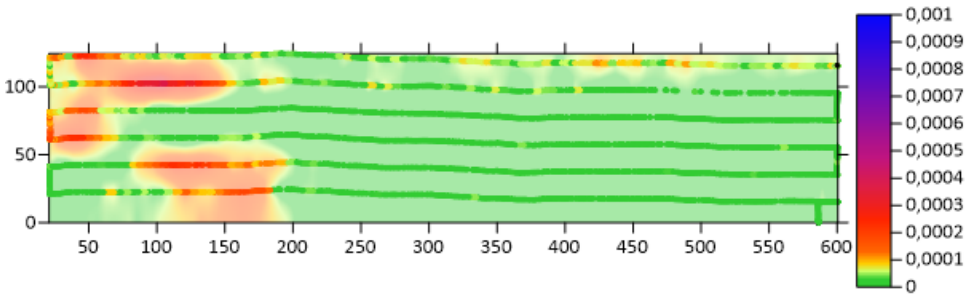


Figure 6.16. Tracer gas plume transect 3, conducted 900 m downwind

OTM 33A

During the controlled release test the atmosphere stability class was 1, extremely unstable. The wind speed during the first quantification was slightly higher than required whereas during the second quantification, it was slightly below requirements. Also the direction of the wind varied a lot. A location for quantification, in distance of 22 m, was chosen, where the plume was measured multiple times, despite the varying wind direction. For this quantification enough data were collected. A second quantification was performed at a distance of 240 m. Between the release location and the measurement location was a patch of forest. The measuring location was chosen, to check the influence of a barrier to the source estimation. In this case, varying wind direction was problematic and quantification was only possible for a very short time period, around 4 minutes. After that time, the wind changed direction and the plume wasn't measured at all. Nevertheless, a source strength was estimated.

The emission measured at a closer distance, was calculated to 2.35 ± 0.26 kg h^{-1}, whereas the emission measured at a greater distance, was 0.53 ± 0.13 kg h^{-1}. In both cases the quantified emission rates were lower than the controlled release rate (by 12.5 and 80.3%, respectively). The highest error obtained during the second measurement was most likely due to a combination of several factors. The wind speed during the second quantification was lower and the wind direction varied more. Because of this vaiability, the plume didn't reach the measurement points sufficient times, and the estimation was thus performed on smaller data set. Due to the short time interval where the plume was actually captured, the emission rate can not be considered valid. The estimation was performed only to show the limitation of this method and how large the error could become, when method is not applied properly.

In summary all three methods were able to quantify the controlled methane release. The TDM method showed an error of +6.8% whereas the error of the UAV and the the OTM 33A methods were -19% and in between -12.5 and -80.3%, respectively. The larger error for the UAV method obtained during the third flight could be explained by the the less good coverage of the whole plume and the low signal-to-noise caused by the larger measuring distance resulting in a more dilute plume. Disregardingthe third flight reduce the average error to -5%. For the OTM 33A method, the second quantification were not in accordance with the required standard conditions for measurements. Furthermore, during the second measurement the methane plume was most of the time not captured by the measurement point. Disregarding the second quatification the error is -12.5%.

6.7 COMPARISION OF MEASUREMENTS

An illustration comparing the results of the different methods used in the study is shown in Figure 6.17. The ground-based plume measurements (OTM 33A) is not included in the comparision figure since it was not possible to quantify the total emission from the landfills with that method, only specific "hotspots" were quantified.

Slite

The Slite emissions for day 1 range between 1.4 and 4.7 kg CH_4 h^{-1} among the three methods. For day 2, the results ranges between 1.6 and 5.8 kg CH_4 h^{-1} (only TDM and UAV). The surface flux chamber results is presented for day 1, however the measurements were performed on both day 1 and 2 and a total value was calculated. For both days, the UAV method showed higher emission rates than the TDM method (about a factor of 3.5). The screenings performed by both the TDM-team and the UAV indiated that emissions were mainly coming from the southeastern part of the landfill. The

two emission hotspots found by the TDM team made up 0.75 kg CH_4 h^{-1} corresponding to ~40% of the total methane emission from the site, which based on the screenings seems reasonable as the majority of the site did not seem to emit much methane. The OTM 33A team also identified two on-site emission sources/hotspots, which gave a combined emission rate of 0.39 til 0.57 kg CH_4 h^{-1}, comparable to the hotspots emissions quantified by the TDM team. Both teams quantified the methane emission from a leachate well. TDM got an emission rate of 0.34 kg CH_4 h^{-1} while OTM 33A got an emission rate of 0.037 kg CH_4 h^{-1}.

Filborna
The TDM and the UAV method gave very similar emission rates of 139.5 and 130.7 kg CH_4 h^{-1}. The highest emission rate of 310 kg CH_4 h^{-1} was obtained by the surface flux chamber method.

Screenings made by the TDM and the UAV shows several on-site areas with elevated methane concentrations (e.g. open waste cells at the northeastern part of the landfill) as well as off-site activities in the vincinity of the landfill with high methane concentrations (e.g. a food waste storage and a storage for bio-fertiliser). Due to the large measuring distance applied by the TDM method emissions from off-site sources located next to the landfill will be included in the measured emission rate of 139.5 kg CH_4 h^{-1}. The UAV flight were performed along a full downwind transect of the landfill and would thus include sources like the anaerobic digester, the sludge lagoons and the food pre-treatment area In comparsion, the flux chambers measured only emissions from the landfill surface which contradics the high emission rate in comparsion to the emission rates obtained by the two remote sensing methods.

Vankiva
The emisssions at Vankiva was quantified to be between 1.1 and 52 kg CH_4 h^{-1}. Contrary to the measurements at Filborna, the surface flux chamber method gave significantly lower emissions of methane than both the TDM and UAV method. The emission rate measured with TDM was 14.5 kg CH_4 h^{-1}, which was about a factor of 3.5 lower than the emission rate measured by the UAV method.

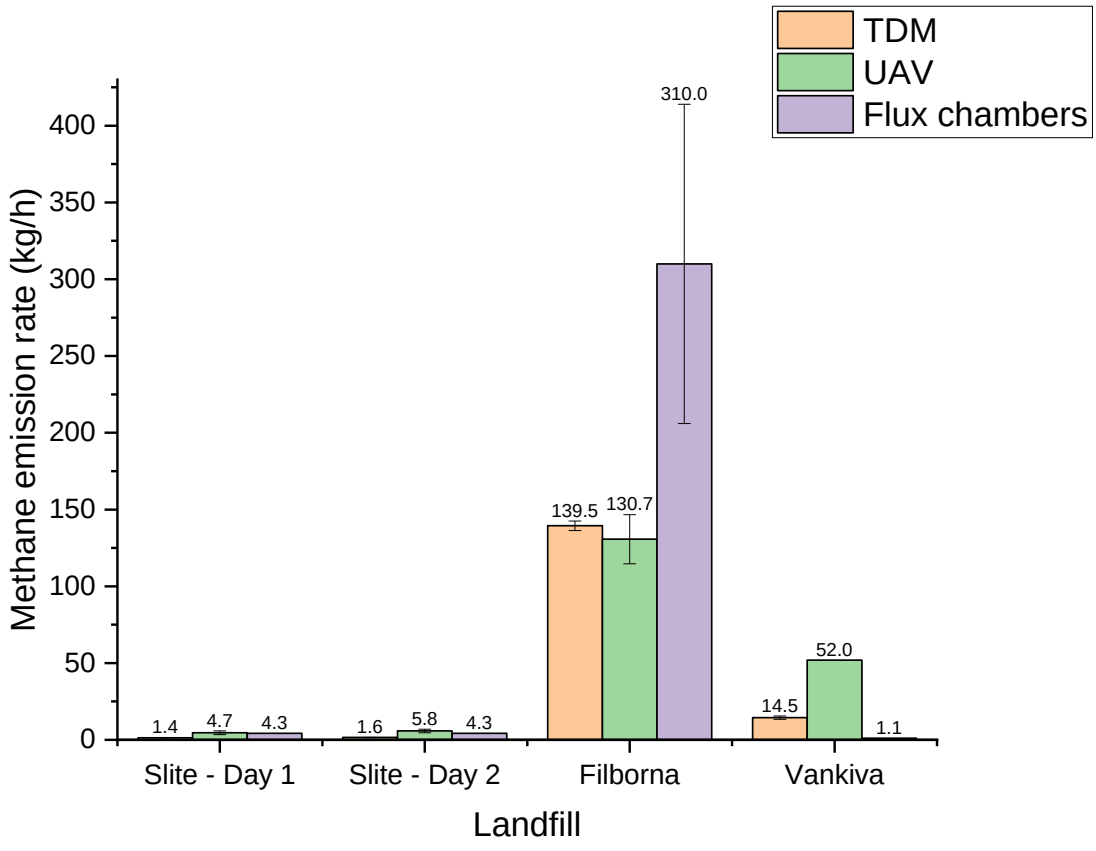


Figure 6.17. Comparison of whole landfill site methane emission rates using tracer gas dispersion measurements

(TDM), UAV-based plume measurements (UAV) and surface flux chambers. Error bars depict the variabilty of the measurements calculated as the standard error of the mean value. Please note the logaritmic Y-axis. The surface flux chambers results for Slite were performed over two days – the same emission rate is given for the two days. The surface flux results from Vankiva should not be seen as a good estimation according to the standard method due to a too low number of performed measurements.

Table 6.7 provides the methane emissions rates in carbon dioxide equivalents (CO_2-equivalents).

Table 6.7. The results presented as emissions of carbon dioxide equivalents for each site. When converting from CH_4-emissions to CO_2-equivalents, a global warming potential (GWP) of 25 was used for methane considering a 100-year horizon. In the lastest IPCC 5th assessment report (Myhre, 2013), the GWP of methane was set at 28 (without including climate change feedbacks), but since 25 is still the most commonly used GWP used for emission reporting this GWP was also used in this case.

Landfill site	**TDM (kg $CO_{2\ ekv}$ h^{-1})**	**UAV (kg $CO_{2\ ekv}h^{-1}$)**	**Surface flux chambers (kg $CO_{2\ eqv}$ h^{-1})**
Slite day 1	35	118	108
Slite day 2	40	145	108
Filborna	3488	3268	7750
Vankiva	363	1300	28

7 DISCUSSION

Comparison of measured methane emission rates
Efforts were made to undertake the measurements with the different techniques at the same time, to reduce the impact of the temporal variability of surface emissions. This is important as methane emissions can vary significantly even over relative short time intervals due to variations in barometric pressure. In terms of timing, the measurements campaigns were reasonably successful. In general, the TDM, UAV and OTM measurements were performed approximately at the same time. The performance of flux chamber measurements is more time consuming than the other method. At Slite, flux chamber measurements were carried out in parallel with the other methods, whereas at Filborna chamber measurements started one day earlier than the other methods and continued the second day when the other methods were also applied. At Vankiva landfill, the surface flux chamber method was performed earlier in the afternoon compared to the other methods, which should not have had a significant impact on the results. During the measurements, the landfill gas extraction systems at all three landfills were operational. Since the systems were extracting gas constantly and with the same rate during the whole measurement campaign, this should not impact the emission rates. It is also expected that methane emissions at landfills with gas extraction systems in operation are less influenced by changes in barometric pressure in comparison to landfills without gas extraction.

In general the emission rates obtained by the UAV method compared reasonable well with the TDM method, especially at Filborna where monitored emission rates were similar. At the two other sites the UAV method gave a higher emission rate than the TDM method by a factor of 3.5 in both campaigns. The OTM 33A method is delveloped for quantification of point sources and cannot be used at larger area and more diffuse emission sources. Therefore it was not possible to compare the performance of this method with the other methods. However, when applied at a point source or a simple source where emissions are localised and blend into one plume within a short distance, the OTM 33A method seems to provide reasonably comparable emission rates. In this study this was demonstrated in the controlled release test, where the OTM 33A method performed well when the required standard conditions given in the method guideline were fulfilled. Also the TDM and the UAV measured emission rates similar to the controlled release rate. In the landfill campaigns, the surface flux chamber method measured a higher (a factor of ~2.2 and 2.4 at Filborna) and a much lower (a factor of ~13 and 47 at Vankiva) emission rate in comparison to the TDM and the UAV methods. It was not applicable to the controlled release test.

The surface flux chamber method detects emissions close to the landfills' surface, hence provides the opportunity to detect spatial emission variations (hotspots) in great detail. On the other hand, the full landfill emission plume is never detected, and the method relies on the performance of a very high number of measurements to gain a result that is statistically reliable. The sensitivity analysis with the presented minimum and mean values show that the methodology for the calculations of the total emission has a significant impact on the overall emission rate.

The TDM and the UAV methods are remote sensing techniques detecting the whole methane plume at a distance and thus monitor the total emission from the landfill including emissions from surfaces, leaks, on-site storages etc. However, if there are other methane emission sources close to the landfill, these might also be included in the plume. At all three landfills there are several potential sources of methane emissions other than the landfill e.g. composting areas, leachate lagoons, sludge and soil storage and food waste storage, food waste pretreatment and biogas plant. These kinds of other potential methane emission sources are mainly present at Filborna and Vankiva. The emission rates based on TDM and UAV are thus expected to provide higher emission rates than the flux chamber method, which only detects fluxes in single points at the landfill surface. At Filborna where several on-site as well as off-site methane sources were seen during the screening, the flux chamber method gave the highest emission rate. This was most likely due to the flux chamber locations identifying and pinpointing hotspots on the surface and quantifying a very high flux in these spots, which impacted the overall result. The low emission rate observed by the flux chamber method at Vankiva was most likely due to the limited amount of chamber measurements that were performed – less than half of the theoretical amount of chambers were used and a quantifiable methane flux was recorded only in a total of four chamber measurements, which leads to a high uncertainty for the collected data. Also, no prior walk-over had been undertaken, and a higher wind speed might have influcenced the recorded fluxes, where gusts led to decreasing methane concentrations in the flux chamber and thus to underestimations of fluxes.

Methodological constraints and potential improvements

The TDM method is currently the best-documented method and it has been applied at large number of landfills as well as other emission sources. The method has been applied in many research studies but has also been commercialised. The advantage of the method is the use of a tracer gas, which provides information about atmospheric dispersion and modelling based on assumptions about atmospheric stability and wind information (wind speed and direction) is thus not necessary. The quatification should be accompanied with an initial screening of the area surrounding the emission source to rule out other methane emission sources, which could influence the measurement. Futhermore a on-site screening should be performed, which will provide valuable information about important on-site emission sources or hotspots. The TDM method can be scaled down to quantify emission rates from on-site emission sources like vents, lagoons, leachate wells etc. The TDM method requires wind speeds of 3-6 m s^{-1}, which is often the case in Denmark. The method constraints are access to drivable roads and at some sites specific wind directions are needed for quantification. The use of tracer gas significantly reduces the uncertainty of quantification, however, the downside is that site access is needed and for larger sites several trace release points are required. The method is not sensitive to rainfall but high solar radation can cause plume rising and high vertical mixing hindering downwind plume detection. However, for Danish conditions the latter is a rare event. Following best practice when performing measurements, the overall error of a TDM measurement is less than 20%.

The application of UAV for landfill emission quantification has only recently been introduced and the method is still under development. The UAV method relies on precise and accurate windspeeds and extrapolation of this to higher altitudes based on a logarithmic transformation. Two ground-based anemometers were used in this research, a mast-mounted sonic anemometer and a tripod-mounted Kestrel 5500. Unfortunately, Filborna was the only site where it was possible to rely completely on mast-mounted measurements; at Slite, some data was lost on both days, and at Vankiva there was not sufficient time to set up the mast. In these cases the Kestrel was used – however, it averages over 2 minutes and is positioned closer to the ground, which greatly increases the size of possible error in determining windspeed at altitude. It is plausible that there was some systematic over- or under-estimation due to this; future work will use a taller mast. Positioning of the anemometer also plays a role – at Vankiva due to time constraints the transect was flown extremely close (above an adjacent road on-site) which meant that measurements were taken in the lee of the landfill. This likely caused the estimated windspeed to be too high, as it assumes that windspeed at ground level is similar at the point of measurement and the point of the anemometer.

It was not possible to use the UAV-mounted anemometer for live windspeed due to unexplained systematic error, but this seems likely to be possible to correct in the near future. In any case, a taller mast with a dedicated anemometer will be used in future work, which will reduce uncertainty. In addition, the results of this study show the importance of repeated methane concentration measurements. Furthermore, variability in future will be reduced by increasing sampling density in the area of the plume after the plume has been located; this will be possible with visual feedback from the measurement sensor to the UAV control systems, and this is currently being tested. Currently, only expensive heavy-payload sensors provide the precision and frequency necessary for this work, meaning greater logistical needs for batteries and limiting flying time, but miniaturised sensors are likely to appear of the market within the next two years. Any UAV method will be sensitive to rainfall and high winds (in this case, above 10 m s^{-1}) but as the measurement takes only 15 minutes, it is possible to measure for only part of the day.

For the UAV, no formal uncertainty analysis was conducted. However, measurement errors were quantified and described according to the method. In lay terms, the method is less sensitive to windspeed error at higher windspeeds, and less sensitive to other errors at higher emissions rates, all else being equal.

The OTM 33A method is a well described method designed for measuring of emissions from point sources. The method has primarily been used to monitor methane leakages from oil and gas activites. The method cannot be used for landfill emission quantification due to the large area and often complex emission pattern. In this study, the method was instead used to monitor emissions from localised on-site sources and emission hotspots. Emissions measured by OTM 33A are reliable when the application of this method is fulling the requirements stated in the guideline. The most variable

and important parameter is wind conditions. If the main wind direction change during a measurement, the data collection will suffer as a result. Additionally, any barrier between a measurement point and a source will affect the dispersion of the plume. If the measurement is performed in relatively far distance to the source, this can have implication on the results and result in underestimated emissions. This was the situation during the second measurement at the controlled release test were the error of the measured emission was -80%. The determination of the atmospheric stability class can influence the emission rate. In Table 6.3 differences in emission is presented, when the stability class were close to two different values. Measurement variations connected with this method can come from instruments malfunctions. Before emission calculations, raw data has to be checked, to remove unrealistic values, e.g. negative methane readings can cause a problem with background estimation and fitting Gause curve to data. The uncertainty analysis for OTM 33A has been done and presented by Edie et al. 2019. Increasing distance from source by ~5%, increased flux estimation by 10%, but compared to others factor is negligible. Influence of wind speed is not clear, however minimum wind speed for this method should be higher than 2 m s^{-1}. The method is sensitive to the source strength. For sources with emissions closer to the detection limit, the results are less accuarate.

The flux chamber technique is the most widely used technique employed at landfills to quantify methane emissions, and is developed as a standard method in the UK. One of the limitations of the surface flux chamber is that it is intended for measuring diffusional surface fluxes. At landfills, surface methane emissions are governed by both diffusion and advection, and if the latter is dominating, which is often the case at emission hotspots, the measured flux will be underestimated due to a fast pressure build-up in the static chamber. A proper measurement often requires that vegetation is cut down and that the chamber's contact with the surface is sealed with, for example, clay or bentonite. The main limitation of using surface flux chambers is however that the measured emission represents only the small area covered by the flux chamber itself (often 0.5-1 m^2). At many landfills, the majority of the combined emission occur from small localised cracks and fissures in the landfill cover or smaller areas where the landfill cover is weak e.g. on landfill slopes. Idenfifying and quantifying these many emission hotspots is a significant challenge, and hence this method should be accompanied by a walk-over survey to complement the flux chamber monitoring . A proper screening with a gridsize of 20-50 m takes place prior to the flux measurement. When performing a large number of chamber measurements, the chance of identifying all emission hotspots is low, especially if a random sampling approach is needed. Even if an intial walk-over screening is performed before flux chamber monitoring is undertaken, the area selection of the measured fluxes is complex. If flux chamber monitoring is primarly conducted at emission hotspots – identified during the initial walk-over screening or by visual inspections - and assumed to be valid for the whole landfill surface area, the total emission may be overestimated. Flux chambers cannot be used to quantify emission flux from landfill installations such as leachate wells, leaking gas collection wells, etc but these emissions are quantified during the walk-over survey, in concentrations rather than flow rates.

A proposed attempt to obtain a more reliable estimate of the total methane emission from a landfill or landfill section, is to apply a systematic testing strategy where flux chamber measurements are performed in grid points followed by geo-statistic models to calculate the total landfill methane emission. However the potential improvement of the results remain unvalidated. Flux chambers can however be an important tool for obtaining emission rates for smaller areas, such as an engineered biocover for methane oxidation or smaller scale landfill cover studies (e.g. covered vs. non-covered cells, slopes vs. none-sloped area, etc.). Combined with cover soil gas profiles information about gas transport and methane oxidation can be be obtained. A walk-over survey (potentially combined with flux chamber monitoring), with its exact pin-point identification of leaks and areas of weak cover, has the advantage of providing the landfill operator with an direct action plan on where and how to reduce emissions from the landfill.

There is no standard method for uncertainty analysis of the surface flux chamber method in LFTGN07 (Environment Agency, 2010). In this case, the min, mean and max values are calculated according to LFTGN07. Additionaly, four ways of calculated the total flux has been performed for one of the sites, where measurement data from earlier campaigns has been used for two of the four calculations. A better model for sensitivity analysis could be developed more suited to the measurement data at hand.

Temporal resolution of the measurement methods
All four methods included in this study measure the emission rate over a relative short time frame; the UAV has a time resolution of 10-15 minutes (or 1-2 hours if 3-4 consecutive flights are performed), the TDM about 1-2 hours (the time is takes to perform 10-20 transects) and the OTM 33A, which in accordance to the standard should be carried out over a period of minimum 20 minutes. The time required to perform one flux chamber measurement is normally about 5-30 minutes. However, due to the large number of measurements needed, a campaing can take one to several days to perform. Landfill methane emissions show high temporal variability especially due to changes in barometric pressure. Several studies have shown how a drop in the barometric pressure can lead to a significant increase in the landfill emission or the reverse how an increase in the barometric pressure can dampen the emission. Obviously methane emission rates measured using flux chambers are likely to be affected if measurements are carried out during a period of atmospheric pressure changes. As an example, if fluxes are measured prior and during a period with a pressure drop, emissions measured during the pressure drop will be higher than the emissions measure prior to the pressure drop causing a bias in the obtained flux rates.

The only method currently available for continuous landfill emission measurements is the eddy covariance method. The method has a number of constraints (not applicable in complex terrain, or areas with unhomogenous emission patterns, etc.) why it is not used at landfills for whole site quantifications.

8 CONCLUSIONS AND FUTURE RECOMMONDATIONS

8.1 CONCLUSION

This study suggests that whole-site methane emission quantifications are best performed using methods measuring downwind of the landfill, such as tracer gas dispersion or UAV plume measurements. This study also, for the first time. clearly demonstrates that the latter method can give similar results to the currently-accepted tracer dispersion method based on controlled releases and results from actual landfills, offering an alternative or supplementary method of measurement. Obtaining proper wind information (wind speed profiles) when applying the UAV method is important as the emission is a direct result of the the applied wind speed through the vertical plume plane. Further validation and documentation of the method compared to controlled releases and known sources is required, as well as proper documentation. It would be beneficial to conduct further controlled release tests in different topographies and wind conditions in order both to verify and to assess whether it is possible to further constrain windfield precision by better use and analysis of anemometers and improved UAV flight patterns.

Ground-based method OTM 33A is not suitable for quantification emission from all landfills. It can however be used as an additional method for investigation methane leaks from infrastructure at the landfill or localised area sources e.g. smaller lagoons, or storages. Combined with whole-site methane emission quantifications the importance of on-site emission sources can be determined. This is important knowledge, because it shows where effort should be put to minimize methane emissions.

This study demonstrates that heterogeneous emissions from landfills make it difficult to quantify total methane emissions using the surface flux chamber approach. In spite of several studies using the approach to quantify whole landfill site emissions, this approach remains unvalidated in terms of its appropriateness for accurately calculating total methane emissions, and there is a need for further documentation (including sufficient numbers of measurements and statistical methods for data interpretation) and testing against other measuring methods. Currently, surface flux chambers cannot be recommended for whole landfill site methane emission quantification. However, the method can, when combined with walk-over surveys, provide valuable information about leak patterns and point sources at the landfill, which the landfill operator can use as an action plan for reducing the surface emissions from the site. The method can also be undertaken by trained site personnel and although more time consuming than the other methods, requires less expensive and readily available equipment.

This study also demonstrated that UAV can be used to identify areas with elevated methane concentrations providing information about areas with higher emissions. Due to the larger measuring distance, specific emission hotspots and locations can be difficult to pinpoint. For this, a walkover methane screening survey combined with visual surface inspection is more appropriate. Knowledge

about emission hotspots are important knowledge to the landfill operator for emission mitigation. Screening surveys could be combined with OTM 33A and/or surface flux chamber measurements to obtain more precise information about emissions in specific areas of the landfill.

No method for continuous monitoring of landfill gas emissions is available today. Reporting of landfill emissions is often required on an annually for the environmental report which is why landfill emission monitoring is undertaken. The methods available today all provide a quantification of the emission at the day and time the measurement was performed. Landfill emissions often show a high temporal variability (daily and seasonally) caused by barometric pressure changes, seasonal changes in precipitation and temperature, landfill activities, etc. To obain a representative annual emission rate, several measurements must be performed. The number of measurements needed and the conditions under which mesurements should be performed remains unsolved. Future studies should therefore focus on how annual emission data from landfills should be measured and reported.

8.2 FUTURE RECOMMENDATION

In general, there are two main purposes for carrying out methane emission monitoring at landfills. One purpose is to identify installations with methane leakages or areas with emissions on the landfill cover for the purpose of maintenance and reduction of emissions. Another purpose is to quantify the total methane emission from the landfill, for example for reporting to the authorities or as verification of implementation of major emission reduction measures at the landfill.

Qualitative and semi-qualitative methane survey

For qualitative landfill surveying it is recommended to screen installations and the landfill surface for methane emissions by performance of walkovers using a handheld gas analyser (such as a portable flame ionisation detector) gauging concentrations in ppm just above the landfill cover or close to leaking installations. The screening campaign should be performed using a combination of systematic approach (dividing the landfill into sections, whereby methane recordings are taken systematically at grid points), knowledge about the site (where are emissions expected) and visual inspections (bare soil spots, cracks, slopes, etc.).

Qualitative emission surveys can to some extent also be performed using mobile analytical platforms such as the ones used in the TDM or OTM 33A measurements combining recorded methane concentrations and information about wind directions. However, the method is limited by road accessibility, and as a result the specific area giving rise to the recorded elevated methane concentrations cannot be pinpointed. UAV based “Birdseye” emission mapping can potentially be a strong instrument in locating emission hotspots at the landfill surface. In order to obtain detailed information, which can be used to identify specific leakages and emission hotspots the UAV must measure close to the ground (< 10 m). The main advantage is that large areas and inaccessible areas can be surveyed.

Qualitative surveys can be complemented with flux chamber measurements, which can provide information about emission rates in the identified areas.

Equipment to perform walkover surveys and surface flux chamber measurements is available on the market; the instruments are relatively cheap and the monitoring is suitable for landfill operators to perform themselves. UAV based surveys are available on the market, but are still quite rare. In this study, the methane sensor used is relatively expensive. Cheaper methane sensors are available, but their application for landfill emission screenings has not yet been assessed.

Quantification of total landfill emission

For whole-site methane emission quantifications, the most suitable methods are remote sensing methods measuring downwind of the landfill, such as TDM and the UAV method. The TMD method is the most widely tested and documented in total methane flux emission applications from landfills; in this study the method provided the most accurate data compared. The TDM is commercialised, although there are not many companies in Sweden offering TDM measurements. In this study, the UAV based plume measurements showed similar results to the TDM method, which makes it suitable for monitoring total methane flux. Improved wind speed measurements and lighter equipment will improve the precision of this method rapidly in the coming years. Currently, UAV based quantifications are not commercially available in Sweden and the method still needs further improvement and validation. This study showed that whole site emission quantification can be supplemented by

quantification of individual localised emission sources like gas wells, lagoons, storages and similar using the OTM 33A method. At the moment the OTM 33A is not commercialised; it is only used in research applications. However, the OTM 33A method is more applicable for the monitoring of emissions from point sources like gas wells, biogas plants and similar. The TDM method can also be used for quantification of individual on-site sources. Performance of TDM, UAV and OTM 33A requires relative expensive instruments and experienced personnel.

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