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Measuring volatile emissions from biosolids: A critical review on sampling methods

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ABSTRACT

As a by-product of wastewater treatment, biosolids are a source of volatile emissions which can lead to community complaints due to odours and other pollution risks. Sampling methods play a significant role in collecting gas emissions from biosolids-related sources (i.e., pure biosolids, landfilling, land application and composting of biosolids). Though a range of different sampling techniques (flux hood, wind tunnel, static chamber, headspace devices) have been explored in many published papers, the management and best practice for sampling emissions from biosolids is unclear. This paper presents a comprehensive review of sampling methods for collecting gaseous emissions from biosolids. To account for the inconsistent terminologies used to describe sampling devices, a standard nomenclature by grouping sampling devices into five categories was proposed. Literature investigating emission sampling from biosolids-related sources was reviewed. Subsequently a critical analysis of sampling methods in terms of design, advantages, and disadvantages were compiled based on literature findings and assumed mechanistic understanding of operation. Key operational factors such as the presence of fans, purge gas flow rates, insertion depth, and incubation conditions were identified and their level of influence on the measurement of emissions were evaluated. From the review, there are still knowledge gaps regarding sampling methods used to collect gases from biosolids-related sources. Therefore, a framework for the management of emission sampling methodologies based on common sampling purposes was proposed. This critical review is expected to improve the understanding of sampling methodologies used in biosolids-related sources, by demonstrating the potential implications and impacts due to different choices in sampling methods.

1. Introduction

Due to the rapid development of urbanization and the improvement in sanitation, the production of biosolids continues to grow worldwide along with the increasing demand for sustainable waste control and management. For instance, total annual dry biosolids production reported in Australia has increased by about 25% from 2010 to 2019, rising to approximately 371,000 dry tonnes a year ANZBP (2020). In this paper, the term *biosolids* referred exclusively to dewatered sewage sludge, either as a pure "cake" immediately after dewatering or at subsequent stages of its management (storage, transport, or end use). The distinction between the terms *sludge* and *biosolids* was based on the water content, where moisture content in sludge typically ranged from 95% to 99% while that varied from 70% to 85% in biosolids after dewatering treatments like centrifuge, filter or press (Fisher et al., 2017a; Wang et al., 2008). The management of biosolids as resources with potential beneficial uses have been increasingly explored, with practices such as application on agricultural land being consolidated (Barth et al., 2010). On average 38% of biosolids (totally 6.63×10^6 kg/year) was applied to agricultural soils in Western European countries during 1990s, due to the high nutrient and water content that improve the crops and plants growth (Chang et al., 2001). However, along with the biosolids treatment, concerns such as soil or ground water contamination have been the focus of traditional management practices. Moreover, these resources can also be associated with major emission issues related to malodorous, greenhouse gases, micropollutants, or bioaerosols. For example, the nuisance odour emissions have been identified to limit the potential markets for biosolids reuse by impacting local communities (Hayes et al., 2014, 2017a, 2017b). On the other hand, the Intergovernmental Panel on Climate Change (IPCC) reported that the greenhouse gases (GHGs; e.g., methane and carbon dioxide) emissions estimated from agricultural activities, including land application of biosolids, account for about 20% of the total human induced global warming budget (Parry et al., 2007). The accurate measurement of gaseous emissions

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from biosolids is critical to support the assessment of environmental impacts such as nuisance and human health as well as climate change. From a management perspective, consistent terminology and sampling approaches are needed to support odour and GHGs management on site and in the biosolids supply chain, and realise benefits associated with GHG offsets from land application.

Accordingly, there has been an increasing emphasis on measuring emissions from the processing of the biosolids in recently decades. The reported common analytes were complex mixtures of volatile organic compounds (VOCs) (Byliński et al., 2018; Fisher et al., 2017b; Glindemann et al., 2006; Gruchlik et al., 2013), volatile sulfur compounds (VSCs) (Higgins et al., 2006; Kim et al., 2002a; Rosenfeld and Henry, 2000), volatile fatty acids (VFAs) (Rosenfeld et al., 2004a), greenhouse gases (GHGs) (Donovan et al., 2011; Rosenfeld et al., 2001a), and nitrogen oxides (NOx) (Chantigny et al., 2013; Roelle and Aneja, 2002b; Tabachow et al., 2001). Moreover, the emissions control and management from biosolids have been the focus of research initiatives internationally, particularly in USA (Glindemann et al., 2006; Higgins et al., 2005; Kim et al., 2002a; Turkmen et al., 2004a), Australia (Fisher et al., 2017a, 2017b, 2019; Lomonte et al., 2010; Majumder et al., 2014), Canada (Chantigny et al., 2013; Visan and Parker, 2004), Poland (Byliński et al., 2017, 2018, 2019b) and Spain (Maulini-Duran et al., 2013).

Emission sampling is an essential step in the entire emission assessment process, where sampling methods may seek to simulate the real emission scenario and minimize the biases (Fisher et al., 2017a). Compared to ambient measurement, sampling devices are more flexible as they are able to attribute to sources and remove errors from back calculations to emission rates. According to current publications, a range of wind tunnels (Mannheim et al., 1995; Parker et al., 2009; Sommer and Misselbrook, 2016) and closed chambers (Fisher et al., 2016; Prata et al., 2016b) have been used to collect emitting gases from area sources like passive liquid surfaces (Gholson et al., 1991, Prata et al. 2016a, 2016b), composting sites (Hudson et al., 2009; Kumar et al., 2011; Rosenfeld et al., 2004b; Sánchez et al., 2015), agriculture soil (Rochette et al., 1992; Roelle et al., 1999, 2001; Roelle and Aneja, 2002a), and biosolids (Fisher et al., 2016). The use of various chambers or wind tunnels, which in turn may be operated under different conditions, along with the different numeric values obtained from industrial or laboratory settings created difficulties in being able to compare and evaluate approaches and findings. It has been recurrently discussed that different design and operational conditions of sampling devices are likely to bias emission measurements for various area sources, such as open-lot feed yard (Rhoades et al., 2005), liquid surfaces (Prata et al., 2018a, 2018b), manure (Parker et al., 2008) and land after manure application (Parker et al., 2013a, 2013b). However, there is a lack of a systematic understanding of the impact of sampling methods on volatile emissions from biosolids-related sources, which leads to the debate in terms of methods employment. Therefore, there is a strong need to compile, collate and compare different emission sampling approaches and to investigate how various factors affect the emission fluxes measured from biosolids-related sources. A former review summarized the physical dimensions and typical operating conditions of some devices used to collect samples of volatile emissions emitted from solid and liquid surfaces (Hudson and Ayoko, 2008). However, it was not clear to how and to what extent the design and operation of sampling methods and operating parameters will influence the measurements, especially for biosolids. Moreover, some of the common methodologies such as headspace devices were not included in that review, which did not have a specific focus on biosolids.

This review provides state-of-the-art knowledge regarding the published sampling procedures for measuring gaseous emissions from biosolids and associated sources. By understanding the role of sampling methods on emissions, it enables comparison and communication of different sampling methods and establishes a foundation to model emission from biosolids and relevant sources. Moreover, this study reviewed papers that investigated factors affecting emission measurements from biosolids, and proposed a framework that identified the main parameters and suggested actions to mitigate and manage these factors.

2. Principles for literature search, nomenclature, and characteristics of sampling methods

2.1. Principles for literature search

To systematise the collection of published literatures and aim at being as comprehensive as possible, the series of keywords have been input for initially searching based on the types of biosolids, analytes and sampling techniques. For example, various keywords related to types of biosolids such as “biosolids”, “dewatered sludge”, “stabilized sludge”, “aged sludge” have been utilized, then more collections about the relevant citation were considered. According to these searching criteria, around 50 biosolids-oriented publications were collected, reviewed, and compiled, including journal papers, conference papers, theses, technical reports and books. Considering the limitation of publications talking about biosolids, other area surfaces (passive liquid surface, soil, landfill site, compost surface) have been expanded and searched with totally reaching over 150 papers.

2.2. Nomenclature and classification of sampling methods

Over past decades, a range of sampling methods have been used in scientific research and engineering practice to estimate gas exchanges at the porous media-atmosphere or the water-atmosphere interfaces (Gao and Yates, 1998b; Hudson and Ayoko, 2008), hence, potentially applicable for sampling emissions from biosolids can be found. However, there has not been a consistent nomenclature to represent the different sampling methods. For instance, a variety of terms have been used to denote an enclosed sampling chamber without constant sweep flush flow: “closed flux chamber” (Gollapalli and Kota, 2018; Senevirathna et al., 2006), “closed static chamber” (Abichou et al., 2006; Majumder et al., 2014; Szyłak-Szydłowski, 2017), “passive chamber” (Gao and Yates, 1998b), “static flux chamber” (Gaetano et al., 2011; Schroth et al., 2012), “static hood” (Lucernoni et al., 2017b), or “non-flow-through chamber” (Chantigny et al., 2013). On the other hand, it was also common in the literatures that the same term was used to denote sampling devices that operate with different basic mechanisms. For example, Rosenfeld et al. (2001b) named a container-like device without sweep flush flow as “flux chamber”; by contrast, Byliński et al. (2019a, 2019b) adopted the same term to describe a bottom-opened apparatus with constant sweep flush flow. Although there was no point in arguing for “correct” terms, it became clear that the inconsistent naming can be a source of confusion and misunderstanding. Therefore, this review established a basis for consistent nomenclature.

In this review, two main categories of sampling methods were established, as illustrated in Fig. 1, namely: direct sampling methods, characterized by the use of bottom-opened devices which enclosure parts of the emitting surface and directly sample the emissions in lab or field; and headspace methods, which are closed containers in which the sample is incubated, with emissions being captured in the container’s headspace. According to the presence of sweep flow, direct sampling methods can be further divided into static chamber (SC: without sweep gas flow) (Chantigny et al., 2007, 2013; Majumder et al., 2014), and dynamic devices (with sweep gas flow) which are common in the shapes of flux hood (FH: mixed gas inside; also called “dynamic flux chambers”) (Fisher et al., 2016; Wang et al., 2011) and wind tunnel (WT: directional flow inside, predominantly parallel to the sampled surface) (Capelli et al., 2009; Jiang et al., 1995). On the other hand, headspace techniques (HS) can be classified either as dynamic headspace (dynamic HS), which is flushed by a flow-through system capturing the entire exhaust as sample (Kim et al., 2005b; Murthy et al., 2003; Turkmen et al., 2004b;

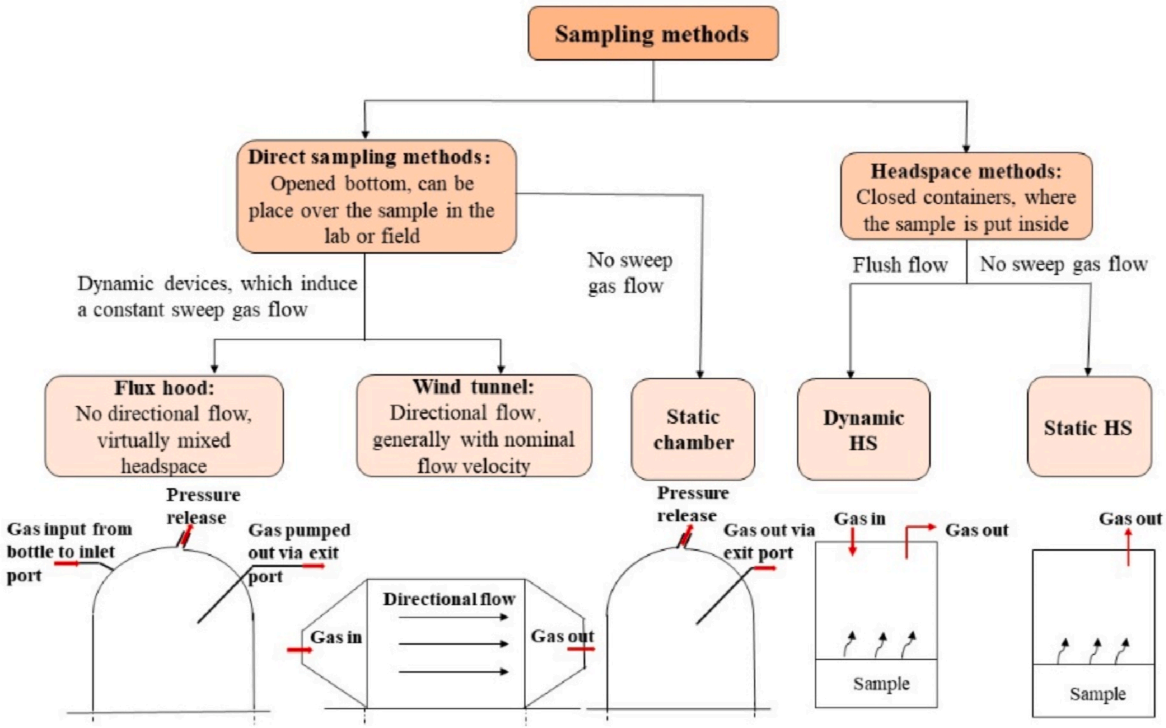

Fig. 1. The characteristics of different sampling methods in this review underlying the proposed nomenclature.

Visan and Parker, 2004); or static headspace (static HS), which does not have a sweep flush flow, with samples being taken directly from the equilibrated headspace gas.

2.3. Characteristics and operation of sampling methods

Generally, each device should be produced from odourless and inert materials and kept clean before sampling in order to avoid the background contamination. Based on literature findings and assumed mechanistic understanding of operation, direct sampling methods, like flux hoods and wind tunnels, will be inserted into solid sources (or fixed to a float over liquid surfaces), to sample the emission flux from the enclosed area. In contrast, for headspace methods, an entire sample (usually small amounts) of the emitting material is placed inside the vessel before sealing. Then, for both direct sampling methods or headspace methods, it is necessary to ensure a good seal condition prior to starting collection, which can minimize the losses of emitted gas flux or unwanted influx of ambient atmosphere. During the sampling process, it is common to wait for a "stabilisation period" before collecting real gas sample from outlet point, to ideally achieve steady state for dynamic devices which produce a constant sweep gas. As for static chamber and static headspace, gas collection is done after an accumulation or incubation period. When the sampling process is completed (into sampling bags, sorbent tubes or injected directly into the analytical equipment), the collected gas can be analysed by a range of techniques (e.g. gas chromatography, sensory analysis, etc.), depending on the different purposes. Operational procedures of each method based on best practice from the published papers were summarized in Table 1.

For dynamic sampling methods with sweep flush flow, the emission rate, $F_{dynamic}$ (kg/(s·m^2)), is calculated as (Gao and Yates, 1998b; Gholson et al., 1991; Hudson et al., 2009; Rolston, 1986; Smith et al., 2007):

$$Fdynamic = \frac{Q}{A}(c_i - c_0) \quad (1)$$

where Q is the flush flow rate through the device (m^3/s); A is the enclosure surface area (m^2); c_i and c_0 are the target gas concentrations of exhaust and air intake (kg/m^3), respectively. In the case of static devices without gas flow, a diffusion-driven flux F_{static} is related to time, which can be given as (Gao and Yates, 1998b):

$$Fstatic(t) = \frac{V}{A}\frac{dC(t)}{dt} \quad (2)$$

where V is the volume of sampling devices (m^3); A is the enclosed surface area (m^2); C (t) is the concentration in the device's headspace (kg/m^3) at time t (s).

3. Implications of sampling devices for gas sampling from biosolids

The reported literatures identified several common area sources that were generally reported in terms of gas emission and odour concerns. From Figure S1, these sources can be grouped into biosolids-related sources and other area sources. As for other area sources, sampling methods like flux hood have been applied to estimate emission from organic green waste/manure composting (Hudson et al., 2009; Kumar et al., 2011; Rosenfeld et al., 2004b; Sánchez et al., 2015), land application of swine slurry (Lovanh et al., 2010), agriculture soil (Rochette et al., 1992; Roelle et al., 1999, 2001; Roelle and Aneja, 2002a), landfill surface (Lucernoni et al., 2017a; Monster et al., 2019; Park and Shin, 2001), lagoons and other passive liquid surfaces (Fu et al., 2017; Gholson et al., 1991; Prata et al., 2016b). Emissions and trends in gas sampling from biosolids and biosolids-related sources were discussed and summarized in the following sections.

3.1. The development of sampling methods applied to biosolids-related sources

Apart from other area sources showed in Figure S1, volatile emissions associated with biosolids are normally related to three main source types: dewatered and stabilized biosolids cake, sometimes aged ("pure biosolids"); amended grass land, agricultural soil or forests with

Table 1
Generalised procedures of each sampling method.

	Flux hood	Wind tunnel	Static chamber	Dynamic HS	Static HS
Prior to sampling	(1) Clean the hood; (2) Place the hood onto the surface and ensure a good seal condition; (3) Insert the hood into a recorded depth, or fix to a float, which depends on the source types;	(1) Clean the wind tunnel; (2) Place the wind tunnel onto the surface and ensure a good seal condition; (3) Fix the tunnel to a float or place into a recorded depth with reaching good seal conditions;	(1) Clean the chamber; (2) Insert the chamber into a recorded depth, or fix to a float; ensure a good seal condition; (3) Wait for a set period for gas accumulation in the headspace;	(1) Clean the vessel**; (2) Sample is placed in headspace vessels with known volume or mass and sealed with caps containing septum;	(1) Clean the vessel**; (2) Place the sample into vessels with known volume or mass and sealed with caps containing septum; (3) Wait for a set incubation time;
Sampling processing	(4) Connect the hood to a purge gas flow and let the system equilibrate for a set period of time; (5) Samples are collected from outlet point; (6) Record metrological factors during sampling process.	(4) Induce a constant flow in the tunnel and continuously flush the surface until steady state; (5) Samples are collected from outlet point into collection vessel, or even for direct analysis; (6) Record metrological factors during sampling process.	(4) Gas samples are collected with syringe (or pumped out with vacuum pump) from the headspace and stored for analysis or directly fed into analyser; (5) Record the factors during sampling process.	(3) Connect the vessel to a steady purge flow until equilibrium; (4) Gas samples are collected with syringe, tubes, bags or SPME* from the headspace. (5) Record incubation time, temperature, flow rate and metrological factors during sampling process.	(4) Gas samples are collected with syringe, tubes, bags or SPME* from the headspace; (5) Record incubation time, temperature, and metrological factors during sampling process.

Note: SPME*-solid phase microextraction, vessel**- The containers used to seal a quantity of sample, which are common to see as PET bottle, glass vials, Nalophan bags etc.

biosolids ("biosolids-land application"); and composted biosolids ("biosolids-composting"). These three types will be hereinafter collectively referred as "biosolids-related sources". Table S1 summarized the benefits and risks associated with application of biosolids for end use. Commonly, all the application scenarios related to utilization of biosolids have the concern of gas emission, which highlights the importance of gas sampling and analysis, which may have implications for the product usage.

As an initial product of biosolids treatment process, pure biosolids accounted for the largest proportion in research, followed by biosolids land application and composting. For understanding the current status of applying sampling methods to measure emission from biosolids-related sources, Fig. 2 showed the cumulative number of papers using at least one of the above sampling methods to evaluate gas emissions from biosolids-related sources. To 2020, the total quantity reached 60, allowing comparison of the popularity of different sampling methods. Static HS was the most popular method and has been consistently used within biosolids projects, reportedly due to its simplicity (25 papers in the past 20 years) (Adams and Witherspoon, 2004; Glindemann et al., 2006; Higgins et al., 2006; Mangus et al., 2006; Novak et al., 2006; Verma et al., 2006). It is worthy to note that Glindemann et al. (2006) developed a sampling protocol based on static headspace to evaluate the odour potential of stored biosolids, representing potential emissions when piles of stored biosolids were distributed for transport from the wastewater treatment plant. As such, static headspace sampling method has been widely used to measure emissions from pure biosolids products in the past decade. Furthermore, static chamber also became largely adopted in the latest decade, which was due to the ability to measure low concentrations of analytes like CH_4, CO_2. Moreover, flux hood has also become more common in the past 5 years as it is alleged to represent and quantify ambient emissions from the surface of biosolids and related sources at various stages during biosolids management such as storage and land application (Barczak et al., 2019; Byliński et al., 2018; Fisher et al., 2016, 2017a, 2018a, 2018b; González et al., 2019). Primarily, flux hood was widely used for measuring other types of non-point source emissions (Fisher et al., 2016) and had a uniform EPA guideline to follow (Kienbusch, 1986). In comparison, limited number of sources reported the gas sampling from biosolids using wind tunnels.

As discussed in section 4, the choice of sampling methods must be predominantly related to the objectives of the studies, and more clarity and comparison between the choices of sampling methods will inform users of the implications of sampling choices.

3.2. Analytes collected from biosolids-related sources

Fig. 3 summarized the common types of gases from biosolids-related sources reported in the literatures, highlighting the different sampling

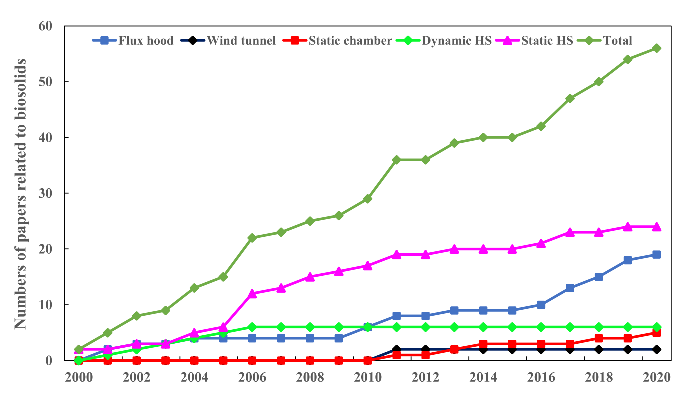

Fig. 2. Cumulative number of publications using different sampling methods for evaluating emissions from biosolids-related sources between 2000 and 2020.

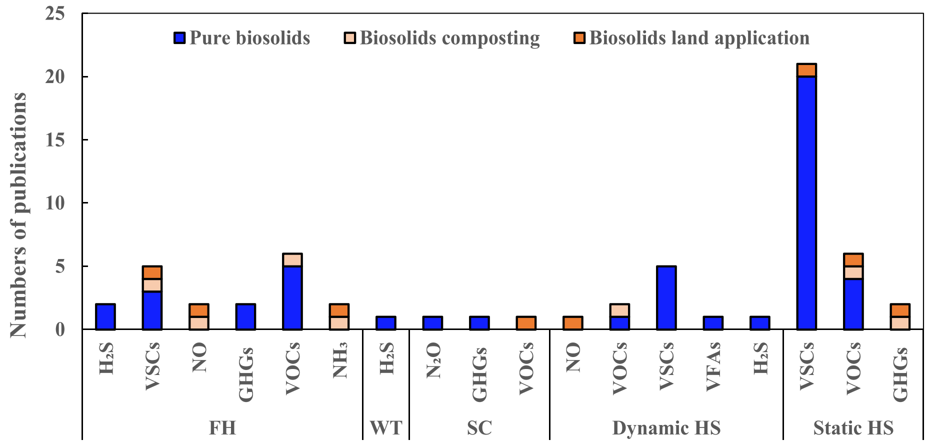

Fig. 3. Summary of analytes from biosolids-related sources by using different sampling methodologies.

methods across the variety of analytes. A wide range of compounds have been reported through different sampling devices, such as VSCs (Fisher et al., 2017a, 2017b; Rosenfeld and Henry, 2000; Rosenfeld et al., 2001b), CO_2 (Andrés et al., 2012; Donovan et al., 2011), VOCs (Byliński et al., 2017, 2018; González et al., 2019), N_2O and NO (Chantigny et al., 2013; Roelle and Aneja, 2002b). Specifically, VSCs were the focus of most studies, probably due to their importance as nuisance odorants. Measurement of VSCs using static HS appeared in 20 papers, more frequent than the usage of flux hood and dynamic HS. However, it was not concluded that static HS was suggested as the best practice for collecting VSCs emission, when considering the area surface, specific analytes and research purposes. Moreover, VOCs have also been identified throughout biosolids processing, which contributed to perceivable odour character of emission streams; however these were not routinely measured as they were judged less sensorially important compared to VSCs (Fisher et al., 2017a). It was worth noting that flux hood was the dominant method in VOC sampling, rather than static HS. This may be due to an attempt to perform the sampling with conditions closer to the real emission environment (Fisher et al., 2017a), which in many cases will be exposed to some degree of sweep air flow. However, it is very unlikely that any sampling device will be able to reproduce all the relevant features of the mass transfer processes observed in full-scaled scenarios (Prata et al., 2018a; Witherspoon et al., 2002; Eklund, 1992).

Furthermore, there was no published material which provided recommendations for the decision making about sampling methods selection in measuring one or several kinds of target gases. For instance, H_2S is a common gas detected in wastewater treatment systems which has a significant odour characteristic defined as “rotten egg” (Dincer and Muezzinoglu, 2007). Since the H_2S has been collected by flux hood (Fisher et al., 2018b; Wang et al., 2011), wind tunnel (Wang et al., 2011), dynamic HS (Kim et al., 2005b) and static HS (Tepe et al., 2008), the possibility that results are differentiated along with the variation of sampling devices cannot be excluded.

Therefore, according to the above discussion on application of sampling devices for gas sampling from biosolids-related sources, the understanding about each sampling method needs to be deepened, in order to reinforce the communication of various emissions result in terms of the biosolids products and sampling devices.

4. Critical analysis of sampling devices

According to the above discussion on various of sampling methods applied to measure volatile emissions from biosolids-related sources (Figs. 2 and 3), there was always a question about what led to the diversity of sampling techniques being used. For instance, if a specific sampling technique was selected because of its lower cost, popularity or mechanistic advantages in being more suitable for using in biosolids-related sources. To better clarify these motivations, 50 papers focused on biosolids-related sources have been reviewed. Of these biosolids-related papers, 65% did not provide a rationale for the choices of sampling method. Reasons for the choice of sampling method provided by the remaining papers can be grouped into three categories: Reason I- mechanistical reasons, such as wind tunnels trying to simulate the wind action; Reason II- practical reasons, like static chamber is selected because of its simplicity and lower running cost; Reason III- mainstream reasons, for instance, flux hood is popular and widely used because US EPA has a guideline for its operation. Based on this classification, the summary of sampling rationale for biosolids-related sources and other area sources is shown in Fig. 4a and b, respectively. Clearly, mechanism and popularity were the two most significant reasons attributed to the choice of sampling devices for capturing gases from both biosolids-related sources and other area surfaces.

However, differences appeared in more detail in Fig. 4c between biosolids-related sources and other area sources. The number of papers that gave reasons for their choice of sampling method corresponded to a smaller proportion in biosolids-related sources, compared to other area sources, which indicated the discussion about sampling methods applied to biosolids-related sources was relatively less developed. Considering the limited data from biosolids-related sources, the following section (where various aspects of sampling methods are discussed) will also involve other area sources (shown in Fig. 4b), to build a wider understanding about sampling techniques.

4.1. Advantages and limitations of sampling methods

4.1.1. Flux hood

Table 2 has summarized the benefits and limitation of every sampling method as pointed out by the literature sources. Flux hoods as a chamber-based sampling method was reported to be popular due to their simplicity and relatively low running cost according to Rolston (1986), Gao and Yates (1998a, 1998b) and Yates et al. (1996). They were thought to be an economical and flexible technique (Eun et al., 2007), known for their field portability and versatility over flat surfaces (Lin et al., 2012). However, Sarkar and Hobbs (2003) pointed out the lack of reliability when flux chambers were used over a relatively rough surface such as the daily operational area of a municipal solid waste (MSW) landfill site. Uncertainty related to the effect of roughness on emission fluxes measured by flux hoods may also be relevant for biosolids-related sources, which may vary from a flat to a rough surface depending on the characteristics of sample.

One advantage often associated with flux hoods was the induction of a constant flush flow that is trying to maintain controlled conditions within the chamber (Eklund, 1992; Fisher et al., 2016; Gustin et al., 1999). However, depending on the processes governing the emissions

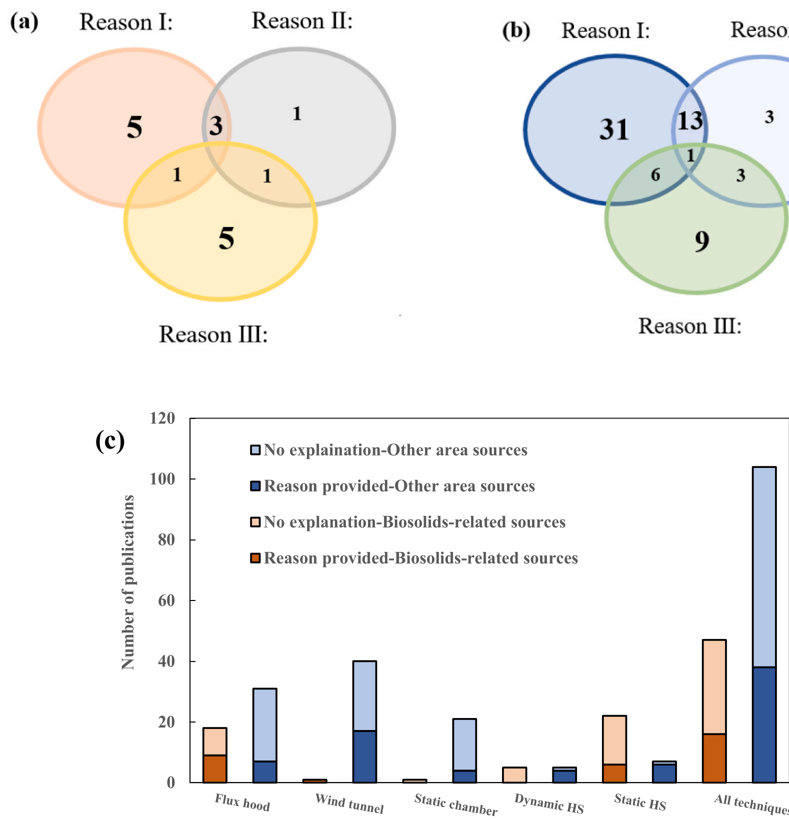

Fig. 4. The classification of reasons for using sampling methods in biosolids-related sources (Figure a) and other area sources (Figure b), where Reason -mechanistical reasons, Reason II-practical reasons, Reason III-mainstream reasons; (c) Summary of sampling methods used in biosolids-related sources and other area sources.

for certain source types, those controlled conditions may introduce a bias in the emission measurements, since they did not reproduce the environmental conditions outside the chamber. In passive liquid surfaces, for instance, turbulence and vapor-phase concentration will be affected by the sweep flow and lead to differences between inside and outside (Gholson et al., 1991; Prata et al., 2018c). Some of the changes verified over liquid surfaces, such as reduced turbulent transport (Andreao et al., 2019; Prata et al., 2018b), non-uniform shear stress (Prata et al., 2016a) and concentration build-up (Prata et al., 2018c), can potentially affect emission rates from solid/porous surfaces (Eklund, 1992). However, Gao and Yates (1998a, 1998b) did not observe a significant concentration build-up of target gas when a flux hood was placed on the soil surface. In addition, the flush flow may contribute to an additional convective mass flow of the target analytes from the covered soil surface, which has been verified to have effects on emission rates (Gao and Yates, 1998a, 1998b; Gao et al., 1997). As a consequence of convective mass flow in the chamber, the pressure gradient between the sample gas phase and the chamber interior have changed and shear stress over the enclosure surface was non-uniform any more (Denmead and Raupach, 1993; Gustin et al., 1999; Kanemasu et al., 1974; Rolston, 1986). Other reported concerns were possible pressure differential from outside the chamber (Gao and Yates, 1998b; Lucernoni et al., 2016b) and temporary disturbance of the surface (Abichou et al., 2006; Monster et al., 2019).

4.1.2. Static chamber

Besides the reported low operational cost, similar to the flux hood method, static chambers were simple to set up, operate, and maintain a high sensitivity at detecting even small fluxes (Gao and Yates, 1998b; Monster et al., 2019; Schroth et al., 2012; Senevirathna et al., 2006; Wang et al., 2019). Based on these advantages, static chambers were commonly used to collect trace gases like CH_4 and CO_2 from landfill surface (Abichou et al., 2006; Lucernoni et al., 2017a), and N_2O from soil (Christensen et al., 1996; Pedersen et al., 2001). However, a major concern was that the enclosed microenvironment above the area surface became different from that outside the chamber due to the chamber isolation, especially the pressure (Gao and Yates, 1998b). Hudson and Ayoko (2008) argued that the increasing in pressure led to the underestimation of the pollutant's emission rates, which was in line with an existing relationship between the pressure and the source surface emission (Frechen et al., 2004). To minimize errors caused by increased pressure and concentration build-up inside the chamber, actual measurements tried to shorten the sampling time to no more than a few minutes (Monster et al., 2019; Rolston, 1986). Moreover, pressure differentials can also be prevented by installing a long and small-diameter vent tube, which equalise the pressure inside the chamber with the surrounding air, but with little loss of the sampled gases (Lucernoni et al., 2016a, 2017b).

In addition, the physical disturbance of the chamber collars to the emitting surface and the possibility of leaks between the enclosure collar and the chamber after the chamber placement, may lead to errors in the measured fluxes (Gao and Yates, 1998b). To avoid gas leakage, following insertion of static chamber, an incubation or stabilisation time was used to ensure there was a good seal condition prior to sampling. The generalised procedures of operating static chamber were summarized in Table 1.

4.1.3. Wind tunnel (Gao and Yates, 1998a, 1998b)

In comparison to indirect sampling methods like microgeological methods, wind tunnels as a direct sampling technique resulted in smaller plots, making it possible to carry out more replicates (Pedersen et al., 2020). However, when wind tunnels are applied to large-scale field

Table 2
Summary of sampling methods for measurement of gas emission, as pointed out by literature sources.

Advantages	Drawbacks	Sources	Analyte	Ref.
Flux hood (FH)				
The equipment with fan or vent tube can increase mixing and reduce pressure.	1. Smaller chamber measurements may underestimate emissions if emissive areas are heterogeneous and larger than the base area of these chambers. 2. Smaller chamber will underestimate the emission level especially for active gases of flux.	Landfill	VOCs, H_2S	Bihan et al. (2020)
Accessing the CH_4 oxidation with combining vertical soil gas concentration profiles.	Not suitable to quantify very low fluxes due to dilution of chamber gas with carrier gas.	Landfill	CH_4	Monster et al. (2019)
Widely used in a range of meteorological or topographical conditions	1. Measured fluxes may suffer biases from variable chamber flow rates and the presence of mixing fans. 2. Changing temperatures of the surface and the air above it due to the sun, especially if they are transparent.	Liquid surface	Organic compounds	Lyman et al. (2018)
N/A	Use of traditional fixed flux hood may lack spatial and temporal representativeness.	Ponds surface	CO_2, CH_4	Fu et al. (2017)
Used to predict the levels of emissions expected in ambient conditions of biosolids storage sheds, or when applied to land.	N/A	Biosolids	VSCs	Fisher et al. (2016)
Satisfactory in recovery rate, precision and repeatability	1. In case of insufficient flow rate, the concentration of target gases in the headspace may be artificially increased. 2. Difficult to reproduce relevant features of the atmospheric flow to which the water surface is exposed in the absence of the enclosure device.	Liquid surface	H_2S	Prata et al. (2016b)
Providing a standardized method for measuring emissions of industrial chemicals from area sources where wind effects are negligible	N/A	Biosolids	Odour, H_2S	Wang et al. (2011)
It is possible to calculate the emission rate of the surface by using the sweep air volume per time and per area, and the emission concentration.	It's difficult to obtain knowledge or even control over the flow pattern inside the chamber due to the shape of the chamber and the air inlet, except for the information that it is turbulent.	Area sources	Odour	Frechen et al. (2004)
It is the most economical and flexible technique.	Having tendency to underestimate emission rates.	Landfill	H_2S	Eun et al. (2007)
1. Simplicity and relatively low running cost. 2. Simulating the field conditions.	1. Extra sweep gas tank is needed. 2. Being sensitive to pressure changes inside the chamber caused by induced airflow, which may create artificially high fluxes.	Landfill	CH_4	Senevirathna et al. (2006)
N/A	Lack of reliable when flux chamber is used over a relatively rough surface like the daily operational area of a MSW landfill site.	MSW** landfill	Odour	Sarkar and Hobbs (2003)
1. They are portable and relatively inexpensive. 2. The influence of soil properties on gas fluxes is allowed to be ascertained.	N/A	Soil	Hg	Gustin et al. (1999)
1. High possibility of maintaining conditions within the chamber to represent those in the surrounding field. 2. A significant concentration build-up of target gas can be avoided due to the flush flow.	The introducing of an airstream through the chamber may create a pressure deficit within the chamber headspace.	Soil	VOCs	Gao and Yates (1998b)
Flux hood is probably one of the simplest methods for measuring emission flux.	1. Covering only a small area cause the estimated flux rate to be highly variable. 2. The presence of chamber can affect the temperature and relative humidity over the sampled surface.	Soil	MeBr	Yates et al. (1996)
A nonintrusive technique and offering advantages of accuracy, simplicity, and flexibility.	Potential underestimation the actual emission rate.	MSW** landfill	CO_2, CH_4	Reinhart et al. (1992)
Wind tunnel (WT)				
The wind tunnel provides a controlled environment that can be reproduced and simulated.	Difference of pressure in- and outside of the chamber.	Waste piles	Odour	Szyłak-Szydłowski (2017)
Suitable to use for comparing treatments under similar environmental conditions	Difference between microenvironment inside the tunnel and ambient conditions, which in turn may modify the emission rate.	Land application	NH_3	Sommer and Misselbrook (2016)
Having advantage in intuitive nature, field portability and versatility over flat surfaces	1. Lack of understanding on the internal flow mechanics that causes the bias. 2. Using measured flux for scale-up estimate could produce significant uncertainty.	Soil	Hg	Lin et al. (2012)
Simulate wind action on the surface (parallel flux without vertical mixing).	N/A	Soil	VOCs	Capelli et al. (2012)
1. Possibility to make unbiased replications if designed correctly 2. Compared to micrometeorological method, wind tunnels require smaller plots and is easy to have more replicates.	1. Overestimate ammonia emissions via modifying the measurement environment. 2. The design of the wind tunnels strongly affects results, with air flow or air velocity being recognized as the most important factor in several studies, as higher values result in higher measured fluxes.	Land application	NH_3	Pedersen et al. (2020)
Simulate the wind action on the surface (parallel flux without vertical mixing)	N/A	Lagoon and tanks	NH_3	Zilio et al. (2020)
	N/A	Pure biosolids	Odour, H_2S	Wang et al. (2011)

(continued on next page)

Table 2 (*continued*)

Advantages	Drawbacks	Sources	Analyte	Ref.
WT provides uniform horizontal air stream inside the tunnel and is suitable for environments where wind varies.				
Easy to work under controlled conditions, changing as few as possible of the ambient environment.	N/A	Land	NH_3	Loubet et al. (1999)
Static Chamber (SC)				
System is easier to use and less costly than open chamber.	Build-up of pressure with time will distort the gas flow pathways in the soil and decrease the flow into the chamber, underestimating gas fluxes.	Soil	CO_2, N_2O	Healy et al. (1996)
N/A	Operating an equilibrium state and not suitable for producing odour emission rates, as equilibrium conditions are not reality in real case.	Area sources	Odour	Frechen et al. (2004)
Simplicity in fabrication and operation.	The enclosed microenvironment above the soil surface varies from that outside the chamber following chamber placement.	Landfill	VOCs	Gao and Yates (1998b)
N/A	1. Not an appropriate method for measuring convective fluxes 2. Temporary disturbance of soil surface can influence the emission	Soil	Gas	Rolston (1986)
Static chamber measurement is inexpensive, simple, and highly sensitive at detecting even small fluxes	Influenced by the increase in pressure due to chamber installation, leading to the underestimation of pollutant's emission rates.	Landfill	VOCs	Wang et al. (2019)
1. Simple to deploy with low limits of detection for fluxes. 2. The only technique that can measure both the emission and the uptake of CH_4	1. Time- intensive and laborious. 2. Not appropriate for either measuring convective fluxes or whole-site CH_4 emission quantification. 3. Temporary disturbance of surface when the chamber is placed.	Landfill	CH_4	Monster et al. (2019)
Relatively cheap, simple, and highly sensitive at detecting even small fluxes	N/A	Landfill	CH_4	Lucernoni et al. (2017b)
N/A	Different pressure insider and outsider of chamber may lead to errors, as there is a relationship between the pressure and the surface emission.	Waste heaps	Odour concentration	Szyłak-Szydłowski (2017)
N/A	Higher height of the chamber would cause the concentration profile inside the chamber to be very inhomogeneous along the hood height.	Landfill	CH_4	Lucernoni et al. (2016b)
1. An easy and cheap technique 2. Versatile to adapt to a wide range of situations, and the capability to measure very low gas fluxes.	A rise in the temperature caused by chamber may influence gas diffusion rates.	Composting	GHGs	Sánchez et al. (2015)
Relatively inexpensive, simple to set up and operate, and highly sensitive at detecting even small fluxes	N/A	Soil	CH_4	Schroth et al. (2012)
In-expensive, simple to use and integrating the flux over time (hours to days).	The placement of the chamber on the soil surface may disturb the soil microclimate.	Soil	CO_2	Rochette (2011); Rochette et al. (1992)
Much easy to use and less costly.	Probable underestimation of gas fluxes due to pressure build-up distorting the gas flow pathways in the soil and decreasing the flow into the chamber.	Landfill	CH_4	Senevirathna et al. (2006)
1. Widely used for measuring various trace gases. 2. Relatively simple to measure gas exchange between the soil and the atmosphere.	1. Possibility of leaks between the enclosure collar and the chamber. 2. Pressure differences between the enclosure sample and above air can alter the fluxes measured.	Landfill	N_2O	Rinne et al. (2005)
Dynamic Headspace (Dynamic HS)				
Easy to perform, and highly reproducible.	1. Sweep gas flows dilute the odorous compounds. 2. Fluxing also removes the gases from contact with the biosolids, so less opportunity to investigate how the microbes in biosolids transform the odorous compounds over time.	Biosolids	MT, DMDS, DMS***	Glindemann et al. (2006)
Static Headspace (Static HS)				
1. Providing reliable and accurate result by eliminating the problems associated with flux chambers. 2. Representative of the storage pile interior, easier to perform, and highly reproducible. 3. Keeping contact between headspace and biosolids, ease of sampling, and less equipment needs.	N/A	Biosolids	MT, DMDS, DMS***	Glindemann et al. (2006)
Relatively rapid, inexpensive, easily automated and solvent-free allowing for minimal sample handling	N/A	Biosolids	Sulfur compound, OVACs****	Gruchlik et al. (2012)

Methods*: FH-Flux hood; WT-Wind tunnel; SC-Static chamber; Static HS-Static headspace; Dynamic HS-Dynamic headspace; MSW** landfill: Municipal solid waste landfill; MT, DMDS, DMS***: MT-Methanethiol; DMDS-Dimethyl disulfide, DMS-Dimethyl sulfide; OVACs****- Odourous volatile aromatic compounds.

measurement they will only enclose a small area footprint (Sommer and Misselbrook, 2016). Wind tunnels have been widely used for collecting gas emissions from solid and liquid area surfaces (Capelli et al., 2009, 2012; Lucernoni et al., 2018). The aerodynamics of the wind tunnels may be controlled using diffusers or a perforated baffle to adjust the wind action on the surface (Szyłak-Szydłowski, 2017; Wang et al., 2011; Zilio et al., 2020). However, wind tunnels were not able to reproduce all the factors controlling emissions under natural condition. Hudson and Akoyo (2008) compared the conditions within wind tunnels with different sizes, flushing rates, velocity of wind, and exchange rates. One of the influencing factors was the air flow rate and the corresponding air velocity, which was recognized as the most important factor for a wind tunnel, as higher values resulted in higher emission fluxes (Eklund, 1992; Sommer and Misselbrook, 2016). The flow velocity was also related to the tunnel height; Frechen et al. (2004) reported that the lower height of tunnel, the higher sweep flow rate and consequently the higher emission rate of target gases from a passive liquid surface. Because the emission rate will be modified by the sweep flow, wind tunnels may not be suitable for determining absolute emission under a range of natural conditions (Pedersen et al., 2020). Furthermore, the thickness and the stability of the boundary layer created by the wind tunnel will rarely represent the respective atmospheric flow outside the tunnel. However, the importance of these factors for the measurement of emissions from biosolids-related sources have not yet been explored.

4.1.4. Headspace methods

Due to benefits, such as easy accessibility, flexibility in shapes, low cost and rapid set-up, there has been broad adoption of headspace devices in an attempt to represent potential emissions from sludge, biosolids, soil and water (Byliński et al., 2019a; Gruchlik et al., 2012, 2013; Psillakis et al., 2000; Zhang et al., 2020). Common applications of the headspace methods were to represent internal portions of solid/porous sources with limited atmospheric exchanges (e.g., the interior of a biosolids stockpile) (Glindemann et al., 2006). Nevertheless, one of the concerns with headspace methods was the common use of manual injections of the headspace gases into the GC inlet, which was laborious, time-consuming, and can introduce uncertainties.

The materials of headspace devices that have been widely used were plastic PET beverage bottles (Novak et al., 2006), glass bottles/vials (Sekyiamah and Kim, 2009; Turkmen et al., 2004a; Verma et al., 2006), serum bottle or vials (Chen et al., 2011; Gabriel et al., 2005; Higgins et al., 2005, 2006), Nalophan bags (Fisher et al., 2016), Tedlar bags (Krach et al., 2008) or Summa canisters (Kim et al., 2005a). The volume of the incubation vessels (vials, bottles and bags) also spanned a wide range, from 20 to 1000 mL (Glindemann et al., 2006).

Compared to static headspace, dynamic headspace could result in dilution of the emitted compounds by the relatively continuous sweep flow (Fisher et al., 2016). Due to the flushing gas in the headspace, the influence of microbes on the composition of emissions may be less apparent, which can be related to the incubation and reaction times (Glindemann et al., 2006). However, systematic evaluations of the significance of sweep flow rate on emission rate in dynamic HS have not yet been conducted.

4.2. Comparison among different sampling methods

Besides the above evaluation of each sampling technique, several comparisons between different sampling methods have been summarized in Table 3. Several comparisons between flux hood and wind tunnel have been conducted when measuring the emission from various area sources, including biosolids (Hudson et al., 2009; Leyris et al., 2005; Lucernoni et al., 2017a; Navaratnasamy et al., 2009; Parker et al., 2009; Wang et al., 2011). Wang et al. (2011) reported that no statistically significant differences were found for measuring H_2S emission rates from freshly dewatered biosolids. However, Parker et al. (2009) reported that NH_3 emission rate from animal feedlots measured by a flux hood was only 49% of what they measured with a small wind tunnel. A similar conclusion was made by Hudson et al. (2009), who compared the odour emission rates from compost windrows, open liquor surface and manure pad surface, and determined that emission rates measured with a flux chamber were consistently lower than the ones measured with a wind tunnel, although there was no correlation between wind tunnel and flux chamber at compost windrows. Therefore, differences between emission fluxes measured using flux hoods and wind tunnels can vary according to the target gases and source types.

Static chambers have also been compared against flux hoods or wind tunnels, albeit not for biosolids-related sources. Previously, Jiang and Kaye (1996) recommended wind tunnel as a preferred method to collect VOCs emissions from liquid surfaces after comparing to static chamber, because static chamber was seen to cause different degrees of underestimation from prepared VOCs standards. Similar underestimation driven by static chamber was also reported by Smith et al. (2007) in the

Table 3
Studies conducting comparisons of different sampling methods for a range of sources and analytes.

Sampling Methods*		Source	Analyte	Conclusion	Ref.
FH	WT	Agriculture odour sources	Odour	Odour concentrations were consistently higher in samples collected with a flux chamber, whereas odour emission rates were consistently larger when derived from wind tunnels.	Hudson et al. (2009); Hudson and Ayoko (2009)
FH	WT	Feedlots	VOCs, odour, NH_3	Flux chamber is suitable for simulating calm condition without wind, while the wind tunnel would overestimate under same situation.	Parker et al. (2009)
FH	WT	Prepared chemical	n-butanol	Flux chamber is a better sampling method for odour emission rate assessment because it provides more consistent, less variable results.	Navaratnasamy et al. (2009)
FH	WT	Biosolids	Odour, H_2S	The difference between emission rates of odour and hydrogen sulphide measured with the two methods was not statistically significant.	Wang et al. (2011)
FH	WT	Animal feeding operations	VOC	Both the EPA flux chamber and small wind tunnel underestimated the field emissions of VOC by using water evaporative flux ratio correction factor.	Parker et al. (2013a)
FH	WT	Liquid surface	Diethyl sulphide	Tests in the wind tunnel showed good accuracy and precision than flux chamber.	Leyris et al. (2000)
FH	WT	Solid and liquid surface	Odour	This study confirms the great influence of the sampling device on the final results, which is higher than previous report.	Guillot et al. (2014)
FH	SC	Soil	CO_2	The static chamber method consistently produced lower soil respiration values than did the dynamic closed system and the difference was larger at higher CO_2 fluxes.	Rochette et al. (1992)
FH	SC	Landfill surface	Odour	A precise chain of actions for both the static chamber and flux hood measurements needs to be determined.	Lucernoni et al. (2017a)
WT	SC	Land application of swine manure	NH_3	Static chambers were found to underestimate NH_3 emissions (by 95–99%), compared with the wind tunnel.	Smith et al. (2007)
WT	SC	Aerobic stabilized pile	Odour	Odour in the samples using the static chamber were consistently higher than those from the samples measured in the wind tunnel.	Szyłak-Szydłowski (2017)
WT	SC	Liquid surface	VOCs	The static chamber caused different degrees of underestimation of the emission rate, but the wind tunnel in suitable for sampling all VOC emissions.	Jiang and Kaye (1996)
FH	Static HS	Biosolids	VSCs	A greater variety of volatile sulfur compounds (VSCs) were observed using the headspace method	Fisher et al. (2016)

Methods*: FH-flux hood; WT-wind tunnel; SC-static chamber; Static HS- static headspace; Dynamic HS- dynamic headspace.

measurement of NH_3 emission rate from land application of swine manure. However, Lucernoni et al. (2017a) used a static chamber for the determination of CH_4 from landfill surfaces due to its better performance in representation of greenhouse gases emission, while flux hood was used to measure odours. Intercomparisons among headspace methods with direct sampling devices were also reported but limited. Fisher et al. (2016) compared the emissions from biosolids by using flux hood and static HS techniques, where greater variety of VSCs concentrations were detected in static HS due to the accumulation of volatile compounds in the headspace. However, it was unclear if emission rates from flux hood or from static HS were more accurate or suitable for different scenarios.

4.3. Influencing factors of sampling methods on emissions

Differences in operational factors chosen when conducting different emission sampling methods can also account for inconsistency and variations. Table 4 presented the typical ranges of the commonly reported factors. Outcomes of evaluations on these factors have been summarized in Table 5 where the degree of influence was also classified within three levels of relevance, namely, high, low and unknown. For example: if the emission rates increased and decreased obviously in accordance with the changes of one factor, this factor was grouped into high relevance level; if no clear or strong relation was found, it was belonged to low relevance. The description unknown was adopted for the factors without existing investigation or evidence about the effect on emission rates for a particular sampling device.

4.3.1. Sweep gas flow rate (SGFR)

The rate of sweep gas in flux hood, wind tunnel and dynamic HS related to the inside pressure, compounds removal rate, evaporation rate and emission rates. Due to the lack of standard sweep gas flow rate (SGFR) for usage in dynamic measurement methods, many different values have been adopted in the literatures (shown in Table 4). It was worth noting that one of the most commonly adopted designs, the flux hood commissioned by the U.S. Environmental Protection Agency (Kienbusch, 1986; Rosenfeld et al., 2004a), entailed the use of a flushed neutral gas flow at 5 L/min (equivalent to a volumetric exchange rate of approximately 0.167 exchanges per minute). Following the guideline of EPA, the same flow rate has been reported in many papers (Fisher et al., 2017a; Hudson et al., 2009; Kumar et al., 2011; Parker et al., 2009; Rosenfeld et al., 2004a, 2004b).

SGFR has been proven the most important operating factor for dynamic sampling techniques on many types of sources. From an experimental perspective, the slower flow rate meant it took longer to reach steady-state concentrations within the devices. In contrast, a higher flow rate can reach steady state in a short time, but also resulted in a short residence time of emitted compounds inside the devices (Lindhardt and Christensen, 1996). A higher SGFR can contribute to over-dilution and consequent low concentrations in the samples, which may increase the uncertainty for chemical and sensorial analysis (Frechen et al., 2004; Lucernoni et al., 2017a). Moreover, higher SGFR also can cause lower relative humidity within the sampling device, which may change the moisture content of the atmosphere above the enclosed sample. Therefore, researchers can vary the SGFR to achieve the desired analytical sensitivity.

On the aspect of mechanism, the concern was that a low SGFR was insufficient to ensure good mixing and turbulent transport in the headspace, which can lead to a point where gas-phase resistance artificially control the emission process (Eklund, 1992). It has been previously identified that the flux of gas-phase limited compounds increased with the growth of airflow configurations (Parker et al., 2008, 2009).

As discussed above, both experimental evidence and mechanistic considerations indicated that SGFR is a factor with the most relevance for affecting emissions. However, since current studies mainly investigated the changes of SGFR on emission from liquid surfaces and solid sources like landfill sites, soils or animal feeding operations, the conclusions obtained for these sources may not be appropriate for biosolids-related sources. Accordingly, future research into the effect of SGFR variation on emission from biosolids-related sources, for various sampling methods would help define suitable sampling conditions for estimating meaningful emission rates. Furthermore, due to the gas-side mass transfer being typically much lower in laboratory than under normal conditions expected for the field, the comparison of gas measurement from biosolids-related sources between lab-scale and field-scale requires a systematic investigation.

4.3.2. Fan (impeller)

An impeller was commonly used in some sampling devices, to mix the headspace homogeneously (AS/NZS, 2009; Das et al., 2008; Roelle and Aneja, 2002b; Woodbury et al., 2011). This acknowledged that how gases were mixed is equally an important factor, besides the SGFR itself. In previous studies, the presence of a fan was shown to have a significant effect on liquid surfaces (Eklund et al., 1987). Parker et al. (2011) observed that the water evaporative flux (which, in their case, was positively correlated with NH_3 flux) more than doubled after installing a small 12 V fan within the chamber. Besides, the impeller size and rotation speed may also influence emissions. Eklund (1992) showed a growth in emission fluxes from liquid surfaces by increasing impeller rotation. In contrast, the same result was not obtained from land surfaces, where less turbulence was produced by the fan rotation speed (Kienbusch et al., 1986); even when the speed of a motor-driven Teflon impeller was increased from 20 to 100 revolutions per minute there was still no significant change in NO flux (Roelle et al., 1999, 2001). The effect that the presence of fan and its rotational speed may have on biosolids is presently unclear and should be further explored in future studies.

4.3.3. Incubation conditions in headspace methods

Emission measurements performed with headspace techniques can be modulated by the incubation temperature, incubation time, and ratio of sample to the total bottle volume. The published factors related to incubation were summarized in Tables 4 and 5 The incubation temperature was generally regarded as a critical factor, not only for ensuring comparability of emission rates between samples but also for explaining the spatial and seasonal variations in emissions from fields (Glindemann et al., 2006; Kumar et al., 2011; Roelle and Aneja, 2002b). The variations of incubation temperature may affect the gaseous compound production patterns as incubation temperature has a strong relationship with the activity of microbial communities (Fisher et al., 2019). This is in agreement with field observations that more odorous compounds were released from dewatered sludge during the summer, when higher temperature occurred (Kim et al., 2002a), as compared to those in winter (Arispe, 2005). Apart from the difference in microbial communities led by temperature variation, this factor also influenced the characteristics of solid sample such as texture, water-filled-pores-space (WFPS), diffusion coefficients and nutrient mineralization (Roelle and Aneja, 2002b); however, those correlations have not been reported linked to biosolids-related sources.

Incubation time affected the physical and chemical parameters that regulate gaseous transport. However, there was lack of a unified incubation time in previous published papers, where a range of times have been listed in Table 4. One reason for the dependency of gas emissions on incubation time can be attributed to microbial activity. For instance, the general trend of VOSCs (volatile organosulfur compounds) from biosolids, such as methanethiol, dimethyl sulfide (DMS), and dimethyl disulfide (DMDS), was observed to peak in the initial days then declined to nearly undetectable levels over several days to several weeks (Fisher et al., 2016; Novak et al., 2006). The tendency was caused by microbial degradation of sulfur-containing amino acids or methanethiol, which can be converted to either hydrogen sulfide from DMS or through DMDS to DMS and hydrogen sulfide (Novak et al., 2006). However, the effect of incubation time on odorous volatile aromatic compounds (OVACs: e.g.,

Table 4
Summary of the typical ranges of common operational factors used in gaseous emission sampling methods from the reviewed literature Detailed parameters are shown in Table S3 and S4.

Factor	Flux hood	Wind tunnel	Static chamber	Factor	Dynamic HS	Static HS
Insert depth	<1 cm[e,f,aa] 2.5 cm[b,d] 5 cm[w] 10 cm[g,k,m,n,z]	1 cm[e, tt] 2.5 cm[cc, dd] 4 cm[hh] 5 cm[nn]	1 cm[yy] 5 cm[ll] 5–10 cm[zz] 10 cm[aaa,bbb,ccc,ddd]	Incubation ratio**	1.1/0.001[iii]	100/1[hhh, mmm, nnn]-200/1[ooo] 50-80/20[jjj] 140/0.5[lll] 3/20[rrr] 126/3.78[sss]
Fan	20–100 rpm[m,n,p] On[f,g,u] 1.5–1.6 m/s[t]	On[cc,dd,ll,ww]	Not reported	Incubation Time	0.5 h[kkk]	0–18 days[mmm] 1–50 days[lll,nnn] 1h[rrr] >1h (24h)[ooo,sss]
Sweep gas flow rate	<4 L/min[o,u,v,x,y,z] 4 L/min[k,l,m,n] 5 L/min[a,b,c,d,e,f,g,h,i,j] 5–10 L/min[o,p,q,t] >10 L/min[r,s]	0.1–1 m/s[ff,hh,mm,rr,ss, ww] 1–5 m/s[cc,ii] 1–10 L/min[bb,dd,ee,gg,jj,ll,uu,xx] >10 L/min[kk,tt,vv] 19 m^3/h[nn,oo,pp,qq]	Not related	Incubation temperature	20 °C[eee, fff] 21 °C[ppp] 35 °C[kkk]	Around 22 °C[jjj,lll,mmm,nnn,sss,] Room temp.[hhh] 30 °C[rrr]
				Sweep gas flow rate	72 mL/min[eee,fff,ggg] 1.25L/min[ppp,qqq] 0.076 L/min[kkk]	

Noted: Incubation ratio ** = Sample mass(g)/container (L). Reference details in Table 4 are showing as follows.

[a] Fisher et al. (2017a, 2017b).
[b] Rosenfeld et al. (2004b).
[c] Hudson et al. (2009)
[d] Rosenfeld et al. (2004a).
[e] Kumar et al. (2011).
[f] Parker et al. (2013a).
[g] Roelle and Aneja (2002a).
[h] Gholson et al. (1991).
[i] Prata et al. (2016b).
[j] Parker et al. (2009).
[k] Roelle and Aneja (2002b).
[l] González et al. (2019).
[m] Roelle et al. (1999).
[n] Roelle et al. (2001).
[o] Gholson et al. (1991).
[p] Lyman et al. (2018).
[q] Eun et al. (2007).
[r] Yates et al. (1996).
[s] Reinhart et al. (1992).
[t] Bihan et al. (2020).
[u] Park and Shin (2001).
[v] Gholson et al. (1991).
[w] Gallego et al. (2014).
[x] Lucernoni et al. (2017b).
[y] Cardellini et al. (2003).
[z] Rochette et al. (1992).
[aa] Lucernoni et al. (2016b).
[bb] Parker et al. (2013b).
[cc] Taha et al. (2005).
[dd] Taha et al. (2004).
[ee] Zilio et al. (2020).
[ff] Pampuro et al. (2016).
[gg] Taha et al. (2007).
[hh] Pedersen et al. (2020).
[ii] Sommer and Misselbrook (2016).
[jj] Rochette et al. (2001).
[kk] Loubet et al. (1999).
[ll] Smith et al. (2007).
[mm] Frechen et al. (2004).
[nn] Liu et al. (2016).
[oo] Liu et al. (2017).
[pp] Liu et al. (2019).
[qq] Liu et al. (2018).
[rr] Liu et al. (2015).
[ss] Szyłak-Szydłowski (2017).
[tt] Parker et al. (2008).
[uu] Woodbury et al. (2015).
[vv] Gao and Yates (1998a).
[ww] Sohn et al. (2005).

[xx] Lin et al. (2012).
[yy] Senevirathna et al. (2006).
[zz] Schroth et al. (2012).
[aaa] Gollapalli and Kota (2018).
[bbb] Rinne et al. (2005).
[ccc] Donovan et al. (2011).
[ddd] Chantigny et al. (2007).
[eee] Kim et al. (2002a).
[fff] Kim et al. (2002b).
[ggg] Kim et al. (2003).
[hhh] Fisher et al. (2016).
[iii] Glindemann et al. (2006).
[jjj] Gruchlik et al. (2017).
[kkk] Turkmen et al. (2004b).
[lll] Novak et al. (2006).
[mmm] Verma et al. (2006).
[nnn] Wilson et al. (2006).
[ooo] Psillakis et al. (2000).
[ppp] Jousset et al. (2001).
[qqq] Cho and Peirce (2005).
[rrr] Laor et al. (2011).
[sss] Rosenfeld et al. (2001a).

indole and skatole) or NH_3 showed no tendency, which suggests the influence also depended on characteristics of gases (Fisher et al., 2016; Turkmen et al., 2004a). As an additional point supporting the importance of incubation time, increasing in residence time can lead to pressure gradient build up in static HS devices, affecting flux measurements and compounds concentration in the headspace (Gholson et al., 1991).

The ratio of sample mass to container volume in headspace methods may also impact emission, but this factor is rarely studied. Based on published literature, the ratios varied from 0.1 to 0.5 g/mL when evaluating emission rate from pure biosolids, biosolids stabilized with lime and coal fly ash, biosolids composting and green waste composting (Fisher et al., 2016; Kumar et al., 2011; Laor et al., 2011; Maulini-Duran et al., 2013). Glindermann et al. (2006) sampled pure biosolids and found that less DMDS was formed when biosolids volume occupied more than 20% of the bottle volume. It was suggested that the microbial degradation in the higher volume of biosolids will consume most of the oxygen inside to form H_2S or DMS, rather than DMDS (Novak et al., 2006). The ratio variations could affect the emission fluxes by varying the amount of oxygen retained in the vessels, which was then used in biotic or abiotic processes. However, no further research on the influence of sample/bottle volume ratio in both static HS and dynamic HS devices has been published.

4.3.4. Other potential influencing factors

Apart from the above common factors, efforts were also made to analyse the influence of other parameters. For example, the insertion depth of direct sampling devices (i.e., how much of the device's walls is inserted into the sample), which typically varied between 1 cm and 10 cm, would be expected to have an effect on the transport of compounds at the surface (Gholson et al., 1991; Kumar et al., 2011; Nay et al., 1994; Pedersen et al., 2001; Rinne et al., 2005; Roelle et al., 1999; Roelle and Aneja, 2002b; Schroth et al., 2012). The insertion of an US EPA flux hood into soil did have a measurable effect on the emission process, where the increase of VOCs flux has been found to be from 80% up to 250% with more insertion (Eklund, 1992). The difference led by the insertion depth can be explained by the changes in boundary layer conductance, the degree of mixing, and the diffusion distance of compounds in the headspace (Gholson et al., 1991).

Furthermore, the relative humidity (RH) of sweep flow was identified as capable of influencing the water evaporative flux from liquid surfaces, as well as the flux of gas-film controlled gases like NH_3 (Parker et al. 2013a,b).

The shapes and dimensions of wind tunnels has been reviewed by Hudson and Ayoko (2008), which showed variations in emission rate provide by different devices. Considering the knowledge currently available is often based on liquid surfaces or soil, the translation to biosolids-related sources should be critically assessed. Therefore, a framework was proposed corresponding to gases, area sources, sampling methods and operational factor to provide the best practice for method application from biosolids-related sources.

5. Future work and management recommendations

5.1. Future research direction

Over past decades, various sampling methods have been employed to measure gas emissions from area surfaces. These tended to relate to the purpose of sampling (analytes, sources), as well as commonly used practices. However, there were still many unsettled issues regarding the measurement of volatile emissions from biosolids-related sources. As discussed above, the influencing factors of sampling methods did affect the results of measurement and their application, but future research should focus on how to understand the influences on the biosolids-related sources. By assessing the roles of influencing factors, the methodology itself can be improved, in addition to elucidating the mechanism driving the difference in emission flux, which is also a future research prospect. Based on these works, an integrated and communicated sampling methods system can be reached, explained and reported. Then, a reliable database can be established and applied to emission inventories or emission models.

5.2. Development of a framework for the management of sampling devices for biosolids-related sources

In response to the current understanding of emission sampling methods for biosolids-related sources, a detailed framework has been developed to support decision making. In Table 6, the framework listed the management of sampling devices in terms of investigating purposes, analytes, practical locations, factors and limitations. Based on the targets of research, examples including the type of biosolids product and analytes have been given. Following the provided information, the rationale for the decision making was suggested in addition to the sampling method. After determining the sampling technique, operational factors that have effects on measurement need to be considered, as they may affect the interpretation of results. In addition, in order to manage the impact of affecting factors on emissions, Fig. 5 summarized the actions to mitigate the effect of various affecting factors, sampling

Table 5
The summary of influencing factors of sampling methods.

Device	Factors	Source	Analytes	Flush flow gas	Details	Level[f]	Findings	Reference
FH	Flow rate	Soil	Hg	Air	1.5–15.6 L/min	High	The chamber flushing rate appears to have a very significant impact on the measured fluxes and on the response behaviour to environmental change.	Wallschläger et al. (1999)
FH	Flow rate	Landfill	NMOC[a]	50% CH_4+50% CO_2	13–18.5 L/min	High	Biasing shifts from positive to negative as the sweep air flow rate is increased. At low flow rates, diffusion of methane is enhanced by turbulent reduction of a laminar film boundary above the soil.	Reinhart et al. (1992)
FH	Fan rate	Soil	NO	Air	20–100 rpm	Low	Varying the speed between 20 and 100 revolutions per minute (rpm) did not produce any significant changes in the NO flux.	Roelle et al. (1999); Roelle et al. (2001)
FH	Flow rate	Soil, liquid	VOCs	Air	Review	High	The sweep air flow rate must be high enough to ensure that good mixing occurs and to promote turbulent reduction of any laminar film boundary above the soil surface.	Eklund (1992)
FH	Flow rate	Soil	VOC	Air	10–1000 L/min	High	The chamber system reaches the steady state more quickly at higher airflow rates. After reaching the steady state, higher air flow rate is corresponding to a lower concentration.	Gao and Yates (1998b)
FH	Insert depth	Soil, liquid	VOCs	Air	Review	High for soil	The increase in measured emission flux due to the soil disturbance from the insertion of the chamber has been found to be 80%–250%.	Eklund (1992)
FH	Insert depth	Landfill	NMOC[a]	50% CH_4+50% CO_2	1.6–8.9 cm	High	Positive biasing apparently is encouraged with increasing penetration depth as a result of soil disruption. Insertion depth, therefore, should be minimized as much as possible, while still maintaining a good seal.	Reinhart et al. (1992)
FH	Fan	AFO[b]	VOCs, NH_3	Air	Installing or not	High	The water evaporative flux was more than doubled from disturbing the boundary layer at the water-air interface when the fan is existing.	Parker et al. (2013a)
FH	Fan rate	Soil, liquid	VOCs	Air	Land or liquid	Low for soil	Impeller rate has no significant effect on emissions from land surfaces, while has significant effect on liquid surfaces (increased rate resulted in increased emission fluxes).	Eklund (1992)
FH	Sampling time	Soil, liquid	VOCs	Air	Review	Un-known	The minimum sampling time necessary is that time required to approach a steady-state concentration within the flux chamber (3–4 residence times).	Eklund (1992)
FH	Air temp.	Covered field	MeBr	Air	In- or outside	High	The emission of MeBr was highly correlated with the diurnal variation in incoming solar radiation and the diffusion through polyethylene film was found to be strongly dependent on the temperature.	Yates et al. (1996)
FH, WT	Flow rate	AFO[b]	VOCs, NH_3	Air	1–20 L/min	High	A strong linear relationship was observed across the range of flow with the EPA flux chamber; however, a logarithmic relationship was observed for the wind tunnel.	Parker et al. (2013a)
FH, WT	Sweep air RH[c]	AFO[b]	VOCs, NH_3	Air	0–100%	High	Sweep air RH greatly influences the water evaporative flux and fluxes of gas-film controlled compounds like NH_3 and VOC.	Parker et al. (2013a)
WT	Flow rate	AFO[b]	VOCs, NH_3	N/A	1.1–85 L/min	High	Fluxes for all eleven VOCs increased with increasing wind velocity ($R^2 = 0.7–0.99$), and the linear relationship occurred for both liquid and solid samples.	Parker et al. (2008); Parker et al. (2009)
WT	Flow rate	Livestock manure	NH_3	Air	0–3.5 m/s	High	The NH_3 emission measured with an air flow of 1 m/s deviated significantly ($P < 0.05$) from the emission measured using other methods, which means air flow cause turbulent convection in the air layers above the emitting surface.	Sommer and Misselbrook (2016)
WT	Flow rate	Soil	CH_2Cl	N/A	0–140 L/min	High	The flux in steady-state behaviour increases with the air flow rate, and high airflow rates the effect of the pressure deficit.	Gao and Yates (1998a)
WT	Flow rate	Landfill	Total C	N/A	0.5–3.5 m^3/h	High	Odour flowrate is influenced by the wind speed inside the wind tunnel, mass-flow rates are a function of the velocity to the power of 0.383.	Frechen et al. (2004)
WT	Flow rate	Landfill	VCs	N_2	0.1–1.07 m/s	High	The emission rates of VCs increased linearly with sweeping velocity where oxygenated compounds were dominant at low sweeping velocities (<0.5 m/s), which accounted for over 50% of the total VCs emission.	Liu et al. (2015)
WT	Flow rate	Liquid	n-butanol	N/A	0.1–17.6 L/s	Unknown	The behaviour of the wind tunnel both at high and low air flow rates should be studied more in	Capelli et al. (2009)

(continued on next page)

Table 5 (*continued*)

Device	Factors	Source	Analytes	Flush flow gas	Details	Level[f]	Findings	Reference
							detail, with the aim of defining the best sampling conditions.	
WT	Flow rate	Soil and liquid	CO	Air	0.07–1.69 m^3/min		Gas recovery efficiencies in the tunnel were consistently high at the higher wind speeds indicating it will give accurate estimates of odour emission rates.	Sohn et al. (2005)
SC	Dimension	Landfill	CO_2, CH_4	None		High	The errors largely depend on the chamber dimensions, including chamber height, surface gas flux rate, and time interval over which the data are collected.	Senevirathna et al. (2006)
SC	Time	Soil	VOC	N/A	0–100min	High	The flux at the covered soil surface decreases with time after the chamber is placed. But the concentration increases faster, lead to a faster decrease of concentration gradient across the interface, which in turn leads to a faster decrease of the flux.	Gao and Yates (1998b)
Static HS	Ratio[d]	Dewater sludge	MT[e], DMDS	None	0–100%	High	At higher than 20% sample/bottle volume ratio, less DMDS is formed.	Glindemann et al. (2006)
Static HS	Incubation time	Dewater sludge	MT[e], DMS	None	1–49 Days	High	No matter MT or DMS, the VOS odorants peaking and then disappearing within 14 days of incubation.	Glindemann et al. (2006)
Static HS	Incubation temp.	Dewater sludge	VOS	None	5,20,25 °C	High	A decrease in the incubation temperature from 25 to 20 °C causes the necessary odour curing time" to rise from 12 to 31 days. Low temperature incubation at 5 °C inhibit the biochemical formation of MT in the cake.	Glindemann et al. (2006)
Dynamic HS	Flow rate	Waste system	SCs	N_2	0–500 mL/min	High	The detector signal for the sulfur compounds increased 70–150% when the flow rate was raised from 0 to 50 mL/min. It appears that the influence of flow rate on adsorption is greatest for less volatile chemicals.	Kim et al. (2002b)
Static HS	Incubation time	Waste system	TMA, SCs, VFA	N_2	0–4 h	High	The peak of VFAs appear at 2 h of incubation, while that of the SCs and TMA is after 4 h.	Kim et al. (2002b)
Static HS	Incubation time	Biosolids	NH_3 VSCs	None	1–14 days	High	VSC emissions decrease after the initial days of storage until they were undetectable, signified by the ammonia emissions, isn't apparently affected.	Fisher et al. (2016)
Static HS	Incubation time	Biosolids	OVACs	None	1–21days	Unknown	The production of VOSCs is due to microbial degradation of sulfur-containing amino acids. While the OVACs, like indole and skatole, start to accumulate after the VOSCs have been depleted.	Gruchlik et al. (2017)
Static HS	Incubation time	Biosolids	DMDS, MT, DMS	None	0–16 days	Low	Averaged headspace concentrations of DMS, DMDS, and MT can be generated with the precursors like amino acids.	Turkmen et al. (2004a)
Static HS	Incubation time	Sludge cake	DMDS, MT, DMS, H_2S	None	1–49 days	High	The MT, DMS followed a predictable pattern of production, peaking between day 1 and day 7, then declining to nearly undetectable levels over several days to several weeks. Hydrogen sulfide often peaks later than the organo-sulfur compounds.	Novak et al. (2006)
Dynamic HS	Incubation time	Soil	NO	Air	1day, week, month	High	NO flux reaches the peak after 1 week incubation and gradually decrease in the following one month.	Cho and Peirce (2005)

Note: NMOC[a]- Nonmethane organic compounds; AFO[b]- Animal feeding operations; Sweep air RH[c]- Sweep air relative humidity; Ratio[d]- The ration of sample mass over the volume of containers; MT[e]- Methanethiol; Level[f]- Influence level on emissions.

devices and significance levels. More details about the management of affecting factors can be found in Table S4. Finally, the scope and limitation of each technique were considered, highlighting boundaries for the use of sampling methods.

For instance, with the purpose for calculating emission fluxes or concentration of VSCs or VOCs from biosolids storage sheds, flux hood was suggested since the emission measurements can be used for odour profiling and risk assessment and, with limitation, as an input to emission modelling. For a flux hood, however, several operational parameters have been identified as high influencing level, such as sweep gas flow rate. Even though actions can be taken to mitigate the impact of gas flow rate (shown in Fig. 5), the limitation of flux hood would still exist if the emission process depends on the wind boundary layer over the source, which can cause biased results. Moreover, if a sample is taken to laboratory for emission collection by a flux hood, the changes of moisture content and oxygen content in the pores, compared to field, will also lead to the limitation of the sampling method. In the perspective of other sampling methods, more detailed information was discussed and listed in Table 6.

According to the above discussion, the influencing factors play important roles on the performance of sampling methods and the result of measurement. Therefore, some actions were suggested to manage the impact according to the significance level of the factors respective to each sampling method. In an effort to avoid confusion, these measures were summarized based on the common practice, journal publications or guidelines. The sweep gas flow rate affected emissions significantly in all dynamic techniques. Thus, it was recommended to maintain a proper flow rate with the aim of avoiding over-dilution by too high flow rate or

Table 6
Framework for the management of sampling devices based on common sampling purposes for biosolids-related source.

Purpose	Examples		Sampling method and rationale	Affecting Factors***	Limitations****
	Sample type	Analytes			
◆ Emission rates (or concentration of steady state) from the passive area sources in ambient conditions. ◆ Sampling may be in field or in a laboratory.	Biosolids in storage sheds or ambient conditions	Range of odour emissions (VSCs, VOCs), likely to impact surrounding community	Flux hood: Emission rates could be used for odour profiling, risk assessment or as input to emission modelling	SGFR; Insertion depth; Fan rate; Presence of fan	◆ If emission process is dependent on wind boundary layer over the source, emission rates will likely be biased by the sampling device. ◆ Samples taken to the laboratory may have other conditions (moisture content, oxygen content in the pores, etc.) altered compared to the field.
◆ Emission rates from passive area sources in environmental conditions. ◆ Sampling may be in field or in a laboratory with large samples.	Biosolids in environmental conditions, e.g. land application, external stockpiles, exposed liquid surfaces (ponds, lagoons)	Range of odour emissions (VSCs, VOCs), likely to impact surrounding community	Wind tunnel: Emission rates could be used for odour profiling, risk assessment or as input to emission modelling	SGFR; Inserted depth; Fan rate	◆ If emission process is dependent on wind boundary layer over the source, emission rates will likely be biased by the sampling device. ◆ Samples taken to the laboratory may have other conditions (moisture content, oxygen content in the pores, etc.) altered compared to the field.
◆ Low concentration analytes from passive or active area sources. ◆ Sampling most commonly in field.	Biosolids in field. Active surfaces of landfills or compost piles	Low concentrations of GHGs or VOCs emissions	Static Chamber: Capture low concentrations of analytes emitted from surfaces.	Insertion depth; Insertion time	◆ Not applicable to use in evaluating the emission rate of volatile gases. ◆ The long-term of placement for measuring low-concentration gases may overestimate the concentration captured than the real conditions.
◆ Emissions rates and/or concentrations compared from different materials in controlled laboratory conditions	Comparing different biosolids products or treatments	Odours	Dynamic HS: Compare emission rates and/or concentrations from materials	SGFR; Time*; Ratio*; Temp.*	◆ If emission process is dependent on wind boundary layer over the source, emission rates will likely be biased by the sampling device.
◆ Emission concentration in headspace generated from stored samples. ◆ Less number of samples in lab	Potential emission levels from the centre of stored biosolids piles on out loading	VSCs emissions, key odorants	Static HS: Potential emissions from a sealed system.	Time*; Ratio*; Temp.*	◆ Not applicable to use in evaluating the emission rate of volatile gases. ◆ The incubation condition may affect the concentration accumulated.

Note:Time, ratio, temp.*- Incubation time, incubation ratio of mass over volume, incubation temperature, respectively; RH**- Relative humidity of sweep gas; Affecting Factors***- The factors reported to influence the emissions; Limitations****-Limitations must be explicitated when reporting results and guide their interpreation.

gas accumulation led by low flow rate. Moreover, the insertion depth should be minimized as long as a good seal condition can be formed, since it has been observed to have impact on changes of boundary layer conductance, mixing and diffusion of compounds. Furthermore, the presence of fan can ensure that the headspace is adequately mixed, so the specifications and operation of the fan must be recorded during the process.

Furthermore, as indicated in Fig. 5 and Table S4, the effect of incubation conditions can also be mitigated by several steps. For example, the ratio of incubated mass of sample over the volume of vessel can highly affect the emission fluxes or concentrations. Therefore, this value needs to be consistent during the sampling process for controlling the bias. Additionally, an incubation ratio of less than 20% was recommended according to the common practice. Besides, a series of screen tests should be conducted, in order to find out the most suitable incubation ratio and ensure an ideal sensitivity for each analysis, along with the proper recording.

6. Conclusion

This paper has reviewed sampling methods used to measure volatile emissions from biosolids and related sources, in order to provide a systematic understanding of the impact that sampling methods have on emissions. Since sampling devices varied widely as reported in the literatures, this paper provided a classification of sampling devices based on common nomenclature. These uniform categories enabled the consolidation and comparison of sampling methods for further detailed discussions. However, large knowledge gaps still exist in interpreting analytical influence due to limited reporting of experiment details for biosolids-related sources. A critical analysis of advantages and drawbacks of sampling methods and comparisons between them was conducted. Influencing factors of sampling methods as well as their significance level in relation to their likely effect on emission concentrations and fluxes were elucidated. Based on the analysis of sampling methods, a framework for the management of affecting factors was proposed to support decision making. This framework will be helpful in deciding the best practice for sampling devices regarding the research purpose, sample types, analytes, influencing factors and limitations of methods, and actions for managing these influencing factors. By addressing the potential limitations and influences of different sampling methods on emission measurement, better practice for pollutant management and resource recovery of biosolids can be achieved.

Credit author statement

Lisha Liu: Conceptualization; Methodology; Formal analysis, Writing – original draft; Writing – review & editing. **Ademir Abdala Prata Junior**: Conceptualization; Writing – review & editing;

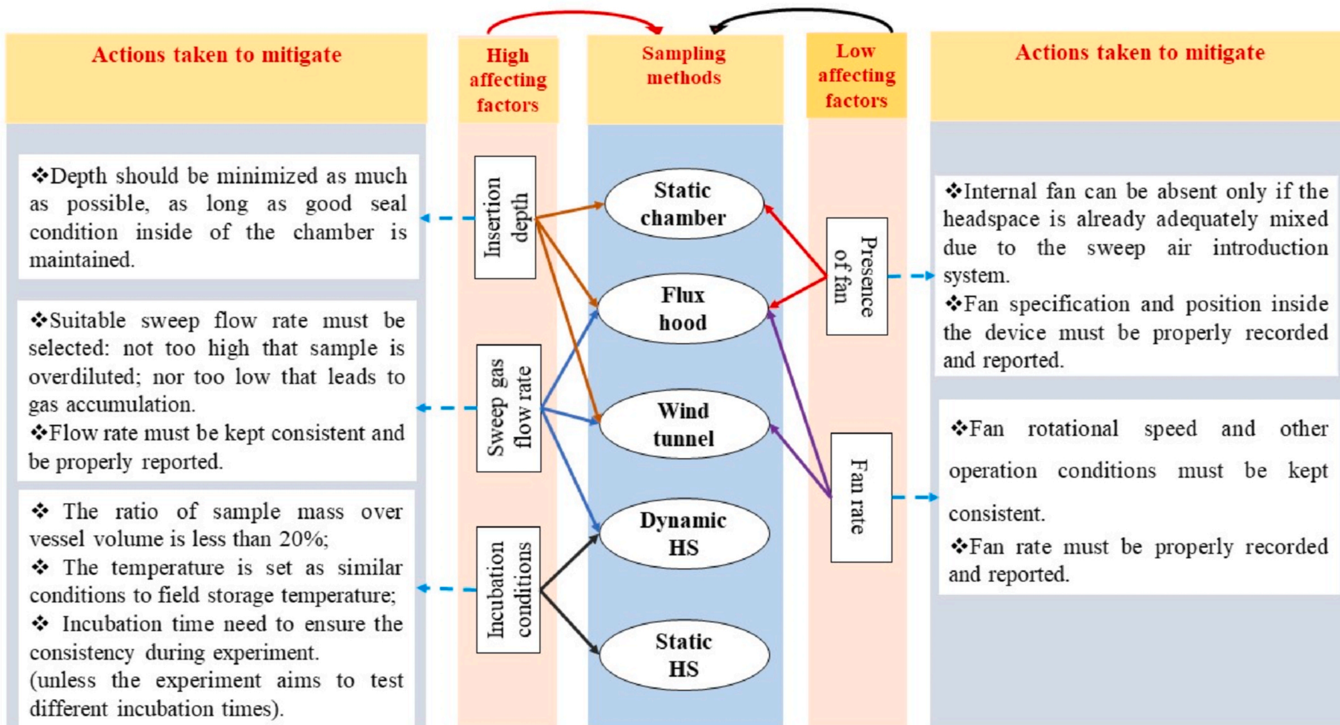

Fig. 5. Framework for the management of affecting factors based on common sampling methodology.

Supervision. **Ruth Fisher**: Conceptualization; Writing – review & editing; Supervision; Resources. **Richard Stuetz**: Conceptualization; Writing – review & editing; Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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References

Abichou, T., Powelson, D., Chanton, J., Escoriaza, S., Stern, J., 2006. Characterization of methane flux and oxidation at a solid waste landfill. J. Environ. Eng. 132, 220–228.

Adams, G.M., Witherspoon, J., 2004. Identifying and Controlling Municipal Wastewater Odor Phase II. IWA publishing.

Andreao, W.L., Santos, J.M., Reis Jr., N.C., Prata Jr., A.A., Stuetz, R.M., 2019. Effects of flux chamber configuration on the sampling of odorous gases emissions. Int. J. Heat Mass Tran. 140, 918–930.

Andrés, E.F.d., Tenorio, J.L., del Mar Albarran, M., Walter, I., 2012. Carbon dioxide flux in a soil treated with biosolids under Semiarid conditions. Compost Sci. Util. 20, 43–48.

Anzbp, 2020. Biosolids Production and Use Survey Summary 2010 to 2019. Australian & New Zealand Biosolids Partnership.

Arispe, S.C., 2005. Measuring and Developing a Control Strategy for Odorous Gases from Solids Handling Processes of a Large Wastewater Treatment Plant. University of Maryland.

Barczak, R.J., Fisher, R.M., Le-Minh, N., Stuetz, R.M., 2019. Importance of 2,4,6-Trichloroanisole (TCA) as an odorant in the emissions from anaerobically stabilized dewatered biosolids. Chemosphere 236, 124340.

Barth, E.F., Forbes, R., Clark, P., Foote, E., McConnell, L.L., 2010. Evaluation of odors associated with land application of biosolids. J. Residuals Sci. Technol. 7, 73–80.

Bihan, L.Y., Loranger-King, D., Turgeon, N., Pouliot, N., Moreau, N., Deschênes, D., Rivard, G., 2020. Use of alternative cover materials to control surface emissions (H_2S and VOCs) at an engineered landfill. Multidisciplinary Journal for Waste Resources & Residues 118–126.

Byliński, H., Barczak, R., Dymerski, T., Gębicki, J., Namieśnik, J., 2017. Direct Measurements of Odorous Volatile Organic Compounds Present in Biosolids Cakes by Proton Transfer Reaction–Mass Spectrometry Technique. The 13th ISCModern Analytical Chemistry, Prague.

Byliński, H., Dymerski, T., Gębicki, J., Namieśnik, J., 2018. Complementary use of GCxGC–TOF–MS and statistics for differentiation of variety in biosolid samples. Monatshefte für Chemie-Chemical Monthly 149, 1587–1594.

Byliński, H., Aszyk, J., Kubica, P., Szopińska, M., Fudala-Książek, S., Namieśnik, J., 2019a. Differences between selected volatile aromatic compound concentrations in sludge samples in various steps of wastewater treatment plant operations. J. Environ. Manag. 249, 109426.

Byliński, H., Barczak, R.J., Gębicki, J., Namieśnik, J., 2019b. Monitoring of odors emitted from stabilized dewatered sludge subjected to aging using proton transfer reaction–mass spectrometry. Environ. Sci. Pollut. Res. 26, 5500–5513.

Capelli, L., Sironi, S., Del Rosso, R., Céntola, P., 2009. Design and validation of a wind tunnel system for odour sampling on liquid area sources. Water Sci. Technol. 59, 1611–1620.

Capelli, L., Sironi, S., Barczak, R., Grande, M., Del Rosso, R., 2012. Validation of a method for odor sampling on solid area sources. Water Sci. Technol. 66, 1607–1613.

Cardellini, C., Chiodini, G., Frondini, F., Granieri, D., Lewicki, J., Peruzzi, L., 2003. Accumulation chamber measurements of methane fluxes: application to volcanic-geothermal areas and landfills. Appl. Geochem. 18, 45–54.

Chang, A.C., Pan, G., Page, A.L., Asano, T., 2001. Developing Human Health-Related Chemical Guidelines for Reclaimed Waster and Sewage Sludge Applications in Agriculture. World Health Organization.

Chantigny, M.H., Angers, D.A., Rochette, P., Bélanger, G., Massé, D., Côté, D., 2007. Gaseous nitrogen emissions and forage nitrogen uptake on soils fertilized with raw and treated swine manure. J. Environ. Qual. 36, 1864–1872.

Chantigny, M.H., Pelster, D.E., Perron, M.H., Rochette, P., Angers, D.A., Parent, L.E., Masse, D., Ziadi, N., 2013. Nitrous oxide emissions from clayey soils amended with paper sludges and biosolids of separated pig slurry. J. Environ. Qual. 42, 30–39.

Chen, Y.C., Higgins, M.J., Beightol, S.M., Murthy, S.N., Toffey, W.E., 2011. Anaerobically digested biosolids odor generation and pathogen indicator regrowth after dewatering. Water Res. 45, 2616–2626.

Cho, K.D., Peirce, J.J., 2005. Nitric oxide emissions from the soil to lower levels of the troposphere. Environ. Eng. Sci. 22, 46–57.

Christensen, S., Ambus, P., Arah, J.R.M., Clayton, H., Galle, B., Griffith, D.W.T., Hargreaves, K.J., Klemedtsson, L., 1996. Nitrous oxide emission from an agricultural

field: comparison between measurements by flux chamber and micrometerological techniques. Atmos. Environ. 30, 4182–4190.
Das, P., Kim, K., Sa, J., Bae, W.S., Kim, J., Jeon, E., 2008. Emissions of ammonia and nitric oxide from an agricultural site following application of different synthetic fertilizers and manures. Geosci. J. 12, 177–190.
Denmead, O., Raupach, M., 1993. Methods for measuring atmospheric gas transport in agricultural and forest systems. Agricultural ecosystem effects on trace gases and global climate change 55, 19–43.
Dincer, F., Muezzinoglu, A., 2007. Odor determination at wastewater collection systems: olfactometry versus H_2S analyses. Clean 35, 565–570.
Donovan, S.M., Skartsila, A.M., Head, M.K., Voulvoulis, N., 2011. An initial investigation into the use of a flux chamber technique to measure soil-atmosphere gas exchanges from application of biosolids to UK soils. Applied and Environmental Soil Science 2011, 1–10.
Eklund, B., 1992. Practical guidance for flux chamber measurements of fugitive volatile organic emission rates. J. Air Waste Manag. Assoc. 42, 1583–1591.
Eklund, B., Kienbusch, M., Ranum, D., Harrison, T., 1987. Development of a sampling method for measuring VOC emissions from surface impoundments. In: EPA/APCA Symposium on Measurement of Toxic and Related Air Pollutants..
Eun, S., Reinhart, D.R., Cooper, C.D., Townsend, T.G., Faour, A., 2007. Hydrogen sulfide flux measurements from construction and demolition debris (C&D) landfills. Waste Manag. 27, 220–227.
Fisher, R.M., Barczak, R.J., Alvarez Gaitan, J.P., Stuetz, R.M., 2016. Comparing static headspace and dynamic flux hood measurements of biosolids odour emissions. Chem. Eng. Trans. 54, 43–48.
Fisher, R.M., Barczak, R.J., Alvarez Gaitan, J.P., Le-Minh, N., Stuetz, R.M., 2017a. Odorous volatile organic compound (VOC) emissions from ageing anaerobically stabilised biosolids. Water Sci. Technol. 75, 1617–1624.
Fisher, R.M., Le-Minh, N., Sivret, E.C., Alvarez-Gaitan, J.P., Moore, S.J., Stuetz, R.M., 2017b. Distribution and sensorial relevance of volatile organic compounds emitted throughout wastewater biosolids processing. Sci. Total Environ. 599–600, 663–670.
Fisher, R.M., Barczak, R.J., Stuetz, R.M., 2018a. Identification of odorant characters using GC-MS/O in biosolids emissions from aerobic and anaerobic stabilisation. Water Sci. Technol. 2017, 736–742.
Fisher, R.M., Le-Minh, N., Alvarez-Gaitan, J.P., Moore, S.J., Stuetz, R.M., 2018b. Emissions of volatile sulfur compounds (VSCs) throughout wastewater biosolids processing. Sci. Total Environ. 616–617, 622–631.
Fisher, R.M., Alvarez-Gaitan, J.P., Stuetz, R.M., 2019. Review of the effects of wastewater biosolids stabilization processes on odor emissions. Crit. Rev. Environ. Sci. Technol. 49, 1515–1586.
Frechen, F.B., Frey, M., Wett, M., Löser, C., 2004. Aerodynamic performance of a low-speed wind tunnel. Water Sci. Technol. 50, 57–64.
Fu, L., Huda, Q., Yang, Z., Zhang, L., Hashisho, Z., 2017. Autonomous mobile platform for monitoring air emissions from industrial and municipal wastewater ponds. J. Air Waste Manag. Assoc. 67, 1205–1212.
Gabriel, S.A., Vilalai, S., Arispe, S., Kim, H., McConnell, L.L., Torrents, A., Peot, C., Ramirez, M., 2005. Prediction of dimethyl disulfide levels from biosolids using statistical modeling. J. Environ. Sci. Health 40, 2009–2025.
Gaetano, D.B., Daniele, D.T., Gaspare, V., 2011. Evaluation of methane emissions from Palermo municipal landfill: comparison between field measurements and models. Waste Manag. 31, 1820–1826.
Gallego, E., Perales, J., Roca, F., Guardino, X., 2014. Surface emission determination of volatile organic compounds (VOC) from a closed industrial waste landfill using a self-designed static flux chamber. Sci. Total Environ. 470, 587–599.
Gao, F., Yates, S.R., 1998a. Laboratory study of closed and dynamic flux chambers: experimental results and implications for field application. J. Geophys. Res. Atmos. 103, 26115–26125.
Gao, F., Yates, S.R., 1998b. Simulation of enclosure-based methods for measuring gas emissions from soil to the atmosphere. J. Geophys. Res. Atmos. 103, 26127–26136.
Gao, F., Yates, S.R., Yates, M.V., Gan, J.Y., Ernst, F.F., 1997. Design, Fabrication, and application of a dynamic chamber for measuring gas emissions from soil. Environ. Sci. Technol. 31, 148–153.
Gholson, A.R., Albritton, J.R., Jayanty, R., Knoll, J., Midgett, M., 1991. Evaluation of an enclosure method for measuring emissions of volatile organic compounds from quiescent liquid surfaces. Environ. Sci. Technol. 25, 519–524.
Glindemann, D., Murthy, S.N., Higgins, M.J., Chen, Y.C., Novak, J.T., 2006. Biosolids incubation method for odorous gas measurement from dewatered sludge cakes. J. Residuals Sci. Technol. 3, 153–160.
Gollapalli, M., Kota, S.H., 2018. Methane emissions from a landfill in north-east India: performance of various landfill gas emission models. Environ. Pollut. 234, 174–180.
González, D., Colón, J., Sánchez, A., Gabriel, D., 2019. A systematic study on the VOCs characterization and odour emissions in a full-scale sewage sludge composting plant. J. Hazard Mater. 373, 733–740.
Gruchlik, Y., Heitz, A., Joll, C., Driessen, H., Fouché, L., Penney, N., Charrois, J.W., 2012. Laboratory scale investigations of potential odor reduction strategies in biosolids. Odour Management 58–64.
Gruchlik, Y., Heitz, A., Joll, C., Driessen, H., Fouché, L., Penney, N., Charrois, J.W., 2013. Odour reduction strategies for biosolids produced from a Western Australian wastewater treatment plant: results from Phase I laboratory trials. Water Sci. Technol. 68, 2552–2558.
Gruchlik, Y., Fouché, L., Joll, C.A., Heitz, A., 2017. Use of alum for odor reduction in sludge and biosolids from different wastewater treatment processes. Water Environ. Res. 89, 2103–2112.
Guillot, J.M., Clincke, A.S., Guilleman, M., 2014. Odour emission from liquid and solid area sources: a large intercomparison of sampling devices. Chemical Engineering Transactions 40, 151–156.
Gustin, M.S., Lindberg, S., Marsik, F., Casimir, A., Ebinghaus, R., Edwards, G., Hubble-Fitzgerald, C., Kemp, R., Kock, H., Leonard, T., London, J., Majewski, M., Montecinos, C., Owens, J., Pilote, M., Poissant, L., Rasmussen, P., Schaedlich, F., Schneeberger, D., Schroeder, W., Sommar, J., Turner, R., Vette, A., Wallschlaeger, D., Xiao, Z., Zhang, H., 1999. Nevada STORMS project: measurement of mercury emissions from naturally enriched surfaces. J. Geophys. Res. Atmos. 104, 21831–21844.
Hayes, J., Stevenson, R., Stuetz, R., 2014. The impact of malodour on communities: a review of assessment techniques. Sci. Total Environ. 500, 395–407.
Hayes, J., Fisher, R., Stevenson, R., Mannebeck, C., Stuetz, R., 2017a. Unrepresented community odour impact: Improving engagement strategies. Sci. Total Environ. 609, 1650–1658.
Hayes, J., Stevenson, R., Stuetz, R., 2017b. Survey of the effect of odour impact on communities. J. Environ. Manag. 204, 349–354.
Healy, R.W., Striegl, R.G., Russell, T.F., Hutchinson, G.L., Livingston, G.P., 1996. Numerical evaluation of static-chamber measurements of soil-atmosphere gas exchange: Identification of physical processes. Soil Sci. Soc. Am. J. 60, 740–747.
Higgins, M., Hamel, K., Chen, Y.C., Murthy, S., Barben, E., Livadaros, A., Travis, M., Maas, N., 2005. Part II of field research: impact of centrifuge torque and polymer dose on odor production from anaerobically digested biosolids. In: Proceedings of the Water Environment Federation, vol. 2005, pp. 1068–1083.
Higgins, M.J., Chen, Y.C., Yarosz, D.P., Murthy, S.N., Maas, N.A., Glindemann, D., Novak, J.T., 2006. Cycling of volatile organic sulfur compounds in anaerobically digested biosolids and its implications for odors. Water Environ. Res. 78, 243–252.
Hudson, N., Ayoko, G., 2008. Odour sampling. 2. Comparison of physical and aerodynamic characteristics of sampling devices: a review. Bioresour. Technol. 99, 3993–4007.
Hudson, N., Ayoko, G.A., 2009. Comparison of emission rate values for odour and odorous chemicals derived from two sampling devices. Atmos. Environ. 43, 3175–3181.
Hudson, N., Ayoko, G., Dunlop, M., Duperouzel, D., Burrell, D., Bell, K., Gallagher, E., Nicholas, P., Heinrich, N., 2009. Comparison of odour emission rates measured from various sources using two sampling devices. Bioresour. Technol. 100, 118–124.
Jiang, K., Kaye, R., 1996. Comparison study on portable wind tunnel system and isolation chamber for determination of VOCs from areal sources. Water Sci. Technol. 34, 583–589.
Jiang, K., Bliss, P.J., Schulz, T.J., 1995. The development of a sampling system for determining odor emission rates from areal surfaces: Part I. Aerodynamic performance. J. Air Waste Manag. Assoc. 45, 917–922.
Jousset, S., Tabachow, R.M., Peirce, J.J., 2001. Soil nitric oxide emissions from nitrification and denitrification. J. Environ. Eng. 127, 322–328.
Kanemasu, E., Powers, W., Sij, J., 1974. Field chamber measurements of CO2 flux from soil surface. Soil Sci. 118, 233–237.
Kienbusch, M., 1986. Measurement of Gaseous Emission Rates from Land Surfaces Using an Emission-Isolation Flux Chamber. Radian Corp., Austin, TX (USA). User's guide.
Kienbusch, M., Balfour, W., Williamson, S., 1986. The Development of an Operations Protocol for Emission Isolation Flux Chamber Measurements on Soil Surfaces. In: 79th Annual Meeting of the Air Pollution Control Association (Minneapolis, US).
Kim, H., Murthy, S., McConnell, L., Peot, C., Ramirez, M., Strawn, M.J.W.S., 2002a. Characterization of wastewater and solids odors using solid phase microextraction at a large wastewater treatment plant. Water Sci. Technol. 46, 9–16.
Kim, H., Nochetto, C., McConnell, L.L., 2002b. Gas-phase Analysis of Trimethylamine, Propionic and Butyric acids, and sulfur compounds using solid-phase microextraction. Anal. Chem. 74, 1054–1060.
Kim, H., McConnell, L.L., Millner, P., 2003. Characyerization of odourants from products of 14 different commercial composts using solid phase mircroextraction. Air Pollution from Agricultural Operations III, 26–33.
Kim, H., McConnell, L., Millner, P., 2005a. Comparison of odorous volatile compounds from fourteen different commercial composts using solid-phase microextraction. Transactions of the ASAE 48, 315–320.
Kim, H., McConnell, L., Ramirez, M., Abu-Orf, M., Choi, H.L., Peot, C., 2005b. Characterization of odors from limed biosolids treated with nitrate and anthraquinone. Journal of Environmental Science and Health 40, 139–149.
Krach, K.R., Li, B., Burns, B.R., Mangus, J., Butler, H.G., Cole, C., 2008. Bench and full-scale studies for odor control from lime stabilized biosolids: the effect of mixing on odor generation. Bioresour. Technol. 99, 6446–6455.
Kumar, A., Alaimo, C.P., Horowitz, R., Mitloehner, F.M., Kleeman, M.J., Green, P.G., 2011. Volatile organic compound emissions from green waste composting: characterization and ozone formation. Atmos. Environ. 45, 1841–1848.
Laor, Y., Naor, M., Ravid, U., Fine, P., Halachmi, I., Chen, Y., Baybikov, R., 2011. Odorants and malodors associated with land application of biosolids stabilized with lime and coal fly ash. J. Environ. Qual. 40, 1405–1415.
Leyris, C., Maupetit, F., Guillot, J.M., Pourtier, L., Fanlo, J.L., 2000. Laboratory study of odour emissions from areal sources: evaluation of a sampling system. Analusis 28, 199–206.
Leyris, C., Guillot, J.M., Fanlo, J.L., Pourtier, L., 2005. Comparison and development of dynamic flux chambers to determine odorous compound emission rates from area sources. Chemosphere 59, 415–421.
Lin, C.J., Zhu, W., Li, X., Feng, X., Sommar, J., Shang, L., 2012. Novel dynamic flux chamber for measuring air-surface exchange of Hg(0) from soils. Environmental Science & Technology 46, 8910–8920.
Lindhardt, B., Christensen, T.H., 1996. Volatilisation of aromatic hydrocarbons from soil: Part II, Fluxes from coal tar contaminated soils residing below the soil surface. Water, Air, Soil Pollution 92, 375–389.

Liu, Y., Lu, W., Li, D., Guo, H., Caicedo, L., Wang, C., Xu, S., Wang, H., 2015. Estimation of volatile compounds emission rates from the working face of a large anaerobic landfill in China using a wind tunnel system. Atmos. Environ. 111, 213–221.
Liu, Y., Lu, W., Guo, H., Ming, Z., Wang, C., Xu, S., Liu, Y., Wang, H., 2016. Aromatic compound emissions from municipal solid waste landfill: emission factors and their impact on air pollution. Atmos. Environ. 139, 205–213.
Liu, Y., Lu, W., Dastyar, W., Liu, Y., Guo, H., Fu, X., Li, H., Meng, R., Zhao, M., Wang, H., 2017. Fugitive halocarbon emissions from working face of municipal solid waste landfills in China. Waste Manag. 70, 149–157.
Liu, Y., Lu, W., Wang, H., Huang, Q., Gao, X., 2018. Odor impact assessment of trace sulfur compounds from working faces of landfills in Beijing, China. J. Environ. Manag. 220, 136–141.
Liu, Y., Lu, W., Wang, H., Gao, X., Huang, Q., 2019. Improved impact assessment of odorous compounds from landfills using Monte Carlo simulation. Sci. Total Environ. 648, 805–810.
Lomonte, C., Fritsche, J., Bramanti, E., Doronila, A., Gregory, D., Baker, A.J.M., Kolev, S. D., 2010. Assessment of the pollution potential of mercury contaminated biosolids. Environ. Chem. 7, 146–152.
Loubet, B., Cellier, P., Flura, D., Génermont, S., 1999. An evaluation of the wind-tunnel technique for estimating ammonia volatilization from land: Part 1. Analysis and improvement of accuracy. J. Agric. Eng. Res. 72, 71–81.
Lovanh, N., Warren, J., Sistani, K., 2010. Determination of ammonia and greenhouse gas emissions from land application of swine slurry: a comparison of three application methods. Bioresour. Technol. 101, 1662–1667.
Lucernoni, F., Rizzotto, M., Tapparo, F., Capelli, L., Sironi, S., Busini, V., 2016a. Use of CFD for static sampling hood design: an example for methane flux assessment on landfill surfaces. Chemosphere 163, 259–269.
Lucernoni, F., Tapparo, F., Capelli, L., Sironi, S., 2016b. Evaluation of an Odour Emission Factor (OEF) to estimate odour emissions from landfill surfaces. Atmos. Environ. 144, 87–99.
Lucernoni, F., Capelli, L., Sironi, S., 2017a. Comparison of different approaches for the estimation of odour emissions from landfill surfaces. Waste management 63, 345–353.
Lucernoni, F., Rizzotto, M., Capelli, L., Busini, V., Del Rosso, R., Sironi, S., 2017b. Sampling method for the determination of methane emissions from landfill surfaces. Waste Manag. Res. 35, 1034–1044.
Lucernoni, F., Capelli, L., Busini, V., Del Rosso, R., Prata Jr., A., Stuetz, R., Sironi, S., 2018. Investigation of mass transfer phenomena affecting emission rate of gaseous compounds from porous solids. Process Saf. Environ. Protect. 116, 516–526.
Lyman, S.N., Mansfield, M.L., Tran, H.N.Q., Evans, J.D., Jones, C., O'Neil, T., Bowers, R., Smith, A., Keslar, C., 2018. Emissions of organic compounds from produced water ponds I: characteristics and speciation. Sci. Total Environ. 619–620, 896–905.
Majumder, R., Livesley, S.J., Gregory, D., Arndt, S.K., 2014. Biosolid stockpiles are a significant point source for greenhouse gas emissions. J. Environ. Manag. 143, 34–43.
Mangus, J.J., Li, B., Burns, B.R., Butler, H.G., Cole, C.A., 2006. Impact of lime dose and mixing quality on odor generation by lime-stabilized biosolids. In: Proceedings of the Water Environment Federation, vol. 2006, pp. 6848–6857.
Mannheim, T., Braschkat, J., Marschner, H., 1995. Measurement of ammonia emission after liquid manure application: II. Comparison of the wind tunnel and the IHF method under field conditions. Z. für Pflanzenernährung Bodenkunde 158, 215–219.
Maulini-Duran, C., Artola, A., Font, X., Sánchez, A., 2013. A systematic study of the gaseous emissions from biosolids composting: raw sludge versus anaerobically digested sludge. Bioresour. Technol. 147, 43–51.
Monster, J., Kjeldsen, P., Scheutz, C., 2019. Methodologies for measuring fugitive methane emissions from landfills - a review. Waste Manag. 87, 835–859.
Murthy, S., Kim, H., Peot, C., McConnell, L., Strawn, M., Sadick, T., Dolak, I., 2003. Evaluation of odor characteristics of heat-dried biosolids product. Water Environ. Res. 75, 523–531.
Navaratnasamy, M., Edeogu, I., Feddes, J., 2009. Comparison of a wind tunnel and vented flux chamber in measuring odour emission rates. Can. Biosyst. Eng. 51, 61–68.
Nay, S.M., Mattson, K.G., Bormann, B.T., 1994. Biases of chamber methods for measurement soil CO_2 efflux demomstrated with a laboratory apparatus. Ecology 75, 2460–2463.
Novak, J.T., Adams, G., Chen, Y.C., Erdal, Z., Forbes Jr., R.H., Glindemann, D., Hargreaves, J.R., Hentz, L., Higgins, M.J., Murthy, S.N., 2006. Generation pattern of sulfur containing gases from anaerobically digested sludge cakes. Water Environ. Res. 78, 821–827.
Nzs, A.S./, 2009. Stationary Source Emission, Methods 4: Area Source Sampling-Flux Chamber Technique.
Pampuro, N., Dinuccio, E., Balsari, P., Cavallo, E., 2016. Evaluation of two composting strategies for making pig slurry solid fraction suitable for pelletizing. Atmos. Pollut. Res. 7, 288–293.
Park, J., Shin, H., 2001. Surface emission of landfill gas from solid waste landfill. Atmos. Environ. 35, 3445–3451.
Parker, D.B., Caraway, E.A., Rhoades, M.B., Cole, N.A., Todd, R.W., Casey, K.D., 2008. Effect of wind tunnel air velocity on VOC flux from standard solutions and CAFO manure wastewater. Environ. Sci. Technol. 53, 831–845.
Parker, D.B., Cole, N.A., Casey, K.D., Galvin, G., Ormerod, R., Paris, C.S., Caraway, E.A., Rhoades, M.B., 2009. In: Wind Tunnels vs Flux Chamber- Area Source Emission Measurments and the Necessity for VOC and Odour Correction Factors, Proceeding 19th Internation Clean Air and Environment Conference (Perth, Australia).
Parker, D., Jay, H., Bryan, W., Lingshuang, C., Mindy, S., Marty, R., Steve, T., Rick, T., Andy, C., 2013a. Standardization of flux chamber and wind tunnel flux measurements for quantifying emissions from area sources at animal feeding operations. Atmos. Environ. 66, 72–83.
Parker, D.B., Cole, N.A., Casey, K.D., Galvin, G., Ormerod, R., Paris, C.S., Caraway, E.A., Rhoades, M.B., 2013b. Odorous VOC emission following land application of swine manure slurry. Atmos. Environ. 66, 91–100.
Parry, M., Parry, M.L., Canziani, O., Palutikof, J., Van der Linden, P., Hanson, C., 2007. Climate Change 2007-impacts, Adaptation and Vulnerability: Working Group II Contribution to the Fourth Assessment Report of the IPCC. Cambridge University Press.
Pedersen, A.R., Petersen, S.O., Vinther, F.P., 2001. Stochastic diffusion model for estimating trace gas emissions with static chambers. Soil Sci. Soc. Am. J. 65, 49–58.
Pedersen, J.M., Feilberg, A., Kamp, J.N., Hafner, S., Nyord, T., 2020. Ammonia Emission Measurement with an Online Wind Tunnel System for Evaluation of Manure Application Techniques. Atmospheric Environment, p. 117562.
Prata Jr., A.A., Calvo, M.J., Boncardo, G., Sivret, E.C., Santos, J.M., Timchenko, V., Stuetz, R.M., 2016a. Influence of the fetch parameter on results from empirical correlations for estimating odorous emissions at passive liquid surfaces. Water Sci. Technol. 74, 2384–2391.
Prata Jr., A.A., Santos, J.M., Beghi, S.P., Fernandes, I.F., Vom Marttens, L.L., Neto, L.I.P., Martins, R.S., Reis Jr., N.C., Stuetz, R.M., 2016b. Dynamic flux chamber measurements of hydrogen sulfide emission rate from a quiescent surface–a computational evaluation. Chemosphere 146, 426–434.
Prata Jr., A., Felder, S., Flocard, F., Santos, J., Stuetz, R., 2018a. Changes in the Near-Surface Hydrodynamics of a Monochromatic Gravity Wave Field Due to a Cylindrical-Shaped Intrusion in the Liquid Surface. In: 21st Australasian Fluid Mechanics Conference (Adelaide, Australia).
Prata Jr., A.A., Lucernoni, F., Santos, J.M., Capelli, L., Sironi, S., Le-Minh, N., Stuetz, R. M., 2018b. Mass transfer inside a flux hood for the sampling of gaseous emissions from liquid surfaces–Experimental assessment and emission rate rescaling. Atmos. Environ. 179, 227–238.
Prata Jr., A.A., Santos, J.M., Timchenko, V., Stuetz, R.M., 2018c. A critical review on liquid-gas mass transfer models for estimating gaseous emissions from passive liquid surfaces in wastewater treatment plants. Water Res. 130, 388–406.
Psillakis, E., Naxakis, G., Kalogerakis, N., 2000. Detection of TNT-contamination in spiked-soil samples using SPME and GC/MS. Global Nest: Int. J. 2, 227–236.
Reinhart, D.R., Cooper, D.C., Walker, B.L., 1992. Flux chamber design and operation for the measurement of municipal solid waste landfill gas emission rates. J. Air Waste Manag. Assoc. 42, 1067–1070.
Rhoades, M.B., Parker, D.B., Auvermann, B., Cole, N.A., Perscbacher-Buser, Z., DeOtte, R. E., 2005. Factors Affecting Emission Measurements with Surface Isolation Flux Chambers, 2005 ASAE Annual Meeting. American Society of Agricultural and Biological Engineers.
Rinne, J., Pihlatie, M., Lohila, A., Thum, T., Aurela, M., Tuovinen, J.P., Laurila, T., Vesala, T., 2005. Nitrous oxide emissions from a municipal landfill. Environmental science technology 39, 7790–7793.
Rochette, P., 2011. Towards a standard non-steady-state chamber methodology for measuring soil N_2O emissions. Anim. Feed Sci. Technol. 166, 141–146.
Rochette, P., Gregorich, E., Desjardins, R., 1992. Comparison of static and dynamic closed chambers for measurement of soil respiration under field conditions. Can. J. Soil Sci. 72, 605–609.
Rochette, P., Chantigny, M.H., Angers, D.A., Bertrand, N., Côté, D.J.C.J.o.S.S., 2001. Ammonia Volatilization and Soil Nitrogen Dynamics Following Fall Application of Pig Slurry on Canola Crop Residues, vol. 81, pp. 515–523.
Roelle, P.A., Aneja, V.P., 2002a. Characterization of ammonia emissions from soils in the upper coastal plain,North Carolina. Atmos. Environ. 36, 1087–1097.
Roelle, P.A., Aneja, V.P., 2002b. Nitric oxide emissions from soils amended with municipal waste biosolids. Atmos. Environ. 36, 137–147.
Roelle, P., Aneja, V.P., O'connor, J., Robarge, W., Kim, D.S., Levine, J.S., 1999. Measurement of nitrogen oxide emissions from an agricultural soil with a dynamic chamber system. J. Geophys. Res. Atmos. 104, 1609–1619.
Roelle, P.A., Aneja, V.P., Gay, B., Geron, C., Pierce, T., 2001. Biogenic nitric oxide emissions from cropland soils. Atmos. Environ. 35, 115–124.
Rolston, D., 1986. Gas Flux Methods of Soil Analysis: Part 1 Physical Mineralogical Methods, vol. 5, pp. 1103–1119.
Rosenfeld, P.E., Henry, C.L., 2000. Wood ash control of odor from biosolids application. J. Environ. Qual. 29, 1662–1668.
Rosenfeld, P.E., Henry, C.L., Bennett, D., 2001a. Wastewater dewatering polymer affect on biosolids odor emissions and microbial activity. Water Environ. Res. 73, 363–367.
Rosenfeld, P.E., Henry, C.L., Dill, R.L., Harrison, R.B., 2001b. Comparision of odor emissions ffom three different biosolids applied to forest soil. Water Air Soil Pollut. 127, 173–191.
Rosenfeld, P., Grey, M., Sellew, P., 2004a. Measurement of bosolids compost odor emissions from a windrow, static Pile, and biofilter. Water Environ. Res. 76, 310–315.
Rosenfeld, P., Grey, M., Suffet, I., 2004b. Compost odor control using high carbon wood ash. Water Sci. Technol. 49, 171–178.
Sánchez, A., Artola, A., Font, X., Gea, T., Barrena, R., Gabriel, D., Sánchez-Monedero, M.Á., Roig, A., Cayuela, M.L., Mondini, C., 2015. Greenhouse Gas from Organic Waste Composting: Emissions and Measurement, CO_2 Sequestration, Biofuels and Depollution. Springer.
Sarkar, U., Hobbs, S.E., 2003. Landfill odour: assessment of emissions by the flux footprint method. Environ. Model. Software 18, 155–163.
Schroth, M.H., Eugster, W., Gómez, K., Gonzalez-Gil, G., Niklaus, P., Oester, P., 2012. Above-and below-ground methane fluxes and methanotrophic activity in a landfill-cover soil. Waste Manag. 32, 879–889.

Sekyiamah, K., Kim, H., 2009. Biosolids odor reduction by solids inventory management in the secondary activated sludge treatment system. Water Sci. Technol. 59, 241–247.
Senevirathna, D.G.M., Achari, G., Hettiaratchi, J.P.A., 2006. A laboratory evaluation of errors associated with the determination of landfill gas emissions. Can. J. Civ. Eng. 33, 240–244.
Smith, E., Gordon, R., Bourque, C., Campbell, A., 2007. Comparison of three simple field methods for ammonia volatilization from manure. Canada Journal of Soil 87, 469–477.
Sohn, J., Smith, R., Hudson, N., Choi, H., 2005. Gas sampling efficiencies and aerodynamic characteristics of a laboratory wind tunnel for odour measurement. Biosyst. Eng. 92, 37–46.
Sommer, S.G., Misselbrook, T.H., 2016. A review of ammonia emission measured using wind tunnels compared with micrometeorological techniques. Soil Use Manag. 32, 101–108.
Szyłak-Szydłowski, M., 2017. Validation of odor concentration from mechanical-biological treatment piles using static chamber and wind tunnel with different wind speed values. J. Air Waste Manag. Assoc. 67, 1046–1054.
Tabachow, R.M., Peirce, J.J., Jousset, S., 2001. Nitric oxide emissions from fertilized and biosolids-amended soil. J. Environ. Eng. 127, 517–523.
Taha, M.P.M., Pollard, S.J.T., Sarkar, U., Longhurst, P.J., 2004. The Influence of Process Variables on Bioaerosol Emission Flux and Exposure-Estimating Fugitive Bioaerosol Releases from Static Compost Windrows. In: 1st UK Conference and Exhibition on Biodegradable and Residual Waste Management (Harrogate, UK).
Taha, M., Pollard, S.J., Sarkar, U., Longhurst, P., 2005. Estimating fugitive bioaerosol releases from static compost windrows: feasibility of a portable wind tunnel approach. Waste Manag. 25, 445–450.
Taha, M.P.M., Drew, G.H., Tamer, A., Hewings, G., Jordinson, G., Longhurst, P.J., Pollard, S.J.T., 2007. Improving bioaerosol exposure assessments -comparative modelling of emissions from different compost ages and activities. Atmos. Environ. 41, 4504–4519.
Tepe, N., Yurtsever, D., Duran, M., Mehta, R., Bruno, C., Punzi, V., 2008. Odor control during post-digestion processing of biosolids through bioaugmentation of anaerobic digestion. Water Sci. Technol. 57, 589–594.
Turkmen, M., Dentel, S.K., Chiu, P., Abu-Orf, M., Hepner, S., 2004a. Odor production in biosolids: Mechanisms and optimal control strategies. In: Proceedings of the Water Environment Federation, vol. 2004, pp. 445–451.
Turkmen, M., Dentel, S.K., Chiu, P., Hepner, S., 2004b. Analysis of sulfur and nitrogen odorants using solid-phase microentration and GC-MS. Water Sci. Technol. 50, 115–120.
Verma, N., Park, C., Novak, J.T., Erdal, Z., Forbes, B., Morton, R., 2006. Effects of anaerobic digester sludge age on odors from dewatered biosolids. Proceedings of the Water Environment Federation 1119–1141.
Visan, M., Parker, W.J., 2004. An evaluation of solid phase microextraction for analysis of odorant emissions from stored biosolids cake. Water Res. 38, 3800–3808.
Wallschläger, D., Turner, R.R., London, J., Ebinghaus, R., Kock, H.H., Sommar, J., Xiao, Z., 1999. Factors affecting the measurement of mercury emissions from soils with flux chambers. J. Geophys. Res. Atmos. 104, 21859–21871.
Wang, H., Brown, S., Magesan, G., Slade, A., Quintern, M., Clinton, P., Payn, T., 2008. Technological options for the management of biosolids. Environ. Sci. Pollut. Res. Int. 15, 308–317.
Wang, T., Sattayatewa, C., Venkatesan, D., Noll, K.E., Pagilla, K.R., Moschandreas, D.J., 2011. Comparison of two dynamic measurement methods of odor and odorant emission rates from freshly dewatered biosolids. J. Environ. Monit. 13, 1746–1752.
Wang, Q., Zuo, X., Xia, M., Xie, H., He, F., Shen, S., Bouazza, A., Zhu, L., 2019. Field investigation of temporal variation of volatile organic compounds at a landfill in Hangzhou, China. Environ. Sci. Pollut. Control Ser. 26, 18162–18180.
Wilson, C.A., Murthy, S.M., Fang, Y., Novak, J.T., 2006. The effect of digester temperature on the production of volatile organic sulfur compounds associated with thermophilic anaerobic biosolids. In: Proceedings of the Water Environment Federation, vol. 2006, pp. 6830–6847.
Woodbury, B.L., Parker, D.B., Eigenberg, R.A., Spiehs, M.J., 2011. Flow Characteristics of the Dynamic "EPA Flux Chamber". American Society of Agricultural and Biological Engineers, Louisville, Kentucky.
Woodbury, B.L., Gilley, J.E., Parker, D.B., Marx, D.B., Eigenberg, R.A., 2015. VOC emissions from beef feedlot pen surfaces as affected by within-pen location, moisture and temperature. Biosyst. Eng. 134, 31–41.
Yates, S., Gan, J., Ernst, F., Wang, D., 1996. Methyl bromide emissions from a covered field: III. Correcting chamber flux for temperature. J. Environ. Qual. 25, 892–898.
Zhang, L., Jiang, R., Li, W., Muir, D.C., Zeng, E.Y., 2020. Development of a solid-phase microextraction method for fast analysis of cyclic volatile methylsiloxanes in water. Chemosphere 250, 126304.
Zilio, M., Orzi, V., Chiodini, M., Riva, C., Acutis, M., Boccasile, G., Adani, F., 2020. Evaluation of ammonia and odour emissions from animal slurry and digestate storage in the Po Valley (Italy). Waste Manag. 103, 296–304.