A thesis submitted to the Department of Environmental Sciences and Policy of Central European University in part fulfillment of the Degree of Master of Science

Top-down and bottom-up landfill methane emissions estimates: a comparative study of the European Union and the United States

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Author’s declaration

No portion of the work referred to in this thesis has been submitted in support of an application for another degree or qualification of this or any other university or other institute of learning.

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ABSTRACT OF THESIS submitted by: Charles GIORDANO

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Landfills are a significant source of anthropogenic methane (CH_4) emissions. Methane is a potent greenhouse gas (GHG) with a global warming potential (GWP) 25-28 times that of carbon dioxide. This means environmental benefits of potential reductions could be realized in a much shorter time frame. The IPCC includes a set of models for estimating landfill methane emissions in its *Guidelines* for compiling bottom-up (B-U) national inventories. Top-down (T-D) studies using atmospheric-based measurements generate estimates independent of bottom-up (B-U) inventory estimates. This study includes analysis of more than 50 T-D studies conducted in both the United States (US) and European Union (EU) to determine context-dependency of the accuracy of inventory landfill emissions estimates. These findings are leveraged by information supplied through a comprehensive literature review of research regarding landfill emissions quantification, in order to draft meta-inferences explaining discrepancies observed between estimates. Both strands of knowledge inform application of a model framework for inferring T-D emissions estimates in three different scenarios. Results indicate that landfill emissions are more likely underestimated in the US than in the EU. This is potentially the result of US operators wrongly accounting for efficiency of landfill gas (LFG) recovery equipment in models for estimating emissions. As well, a lack of transparency in terms of reporting protocols applied may contribute to the lack of overall representativeness of accepted figures. Areas for further research include use of T-D and other measurement-based methods in landfill emissions inventory compilation, validation of EU landfill emissions estimates, and evaluation of the model framework applied here.

Keywords: landfill, methane, emission, inventory, top-down, bottom-up, emissions reporting, US, EU, EPA, waste management

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GHG – Greenhouse gas

CH_4 – Methane

GWP – Global warming potential

T-D – Top-down

B-U – Bottom-up

LFG – Landfill gas

O/NG – Oil/natural gas

CE – LFG gas collection efficiency

OX – landfill cover soil methane oxidation

FOD – First-order decay

WIP – Waste in place

L_0 – Methane generation potential

k-value – Waste half-life

AMB – Aircraft mass-balance

EU – European Union

US – United States

UK – United Kingdom

IPCC – Intergovernmental Panel on Climate Change

EPA – United States Environmental Protection Agency

GHGI – Inventory of US Greenhouse Gas Emissions and Sinks

GHGRP – Greenhouse Gas Reporting Program

NAEI – UK National Atmospheric Emissions Inventory

LAEI – London Atmospheric Emissions Inventory

SWICS – Solid Waste Industry for Climate Solutions

SoCAB – South Coast Air Basin

MD/VA – Maryland/Virginia

SFBA – San Francisco Bay Area

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1. Introduction

Methane (CH_4) is a potent GHG with a global warming potential (GWP) 25-28 times that of carbon dioxide (IPCC 2007, 2014). As of 2007, atmospheric concentrations were 156% that of pre-industrial levels (Bahor et al. 2009). Mitigating anthropogenic emissions of methane provides a more economically feasible means of reducing overall GHG emissions compared to carbon dioxide (Shindell et al. 2012). Moreover, the benefits of reducing methane emissions could be realized within a much shorter time frame given its GWP value (Barlaz et al. 2004, Abichou et al. 2006, Spokas et al. 2011). This is especially relevant considering the IPCC's Fifth Assessment Report (2014) indicating the need for immediate action globally in order to limit effects of global warming.

Landfills represent a significant source of anthropogenic methane emissions, through the process of anaerobic digestion of organic matter contained in waste. Estimates vary, but landfills contribute roughly 8% of an approximate total amount of anthropogenic methane emissions equal to 500 Tg CH_4 y^{-1} (Barlaz et al. 2004). Landfill methane emissions are currently reported in national inventories according to the IPCC *Guidelines for National Greenhouse Gas Inventories* (2006). The *Guidelines* (IPCC 2006) include prescribed methods for emissions quantification. IPCC-prescribed methods consist of a set of models. All are based on a number of assumptions corresponding with the complexity of the model used. A tiered approach is used to classify IPCC-compliant models, ranging from Tier 1 to Tier 3 with complexity increasing with each tier. Furthermore, a Tier 1 complicit model includes the least amount of assumptions, a Tier 3 model the most.

The fundamental assumption inherent to all IPCC-prescribed methods is that methane generation and ultimately emissions are proportionate to the amount and quality of waste deposited in a landfill (SWICS 2009). This describes first-order kinetics, which in this case supports use of the first-order decay (FOD) model. The FOD model (Method) is prescribed by the IPCC as the base model at all three tiers. Higher tiered methods, that specifically require more intensive efforts to quantify emissions, are permissible with validation from the IPCC (2006). Several countries have created their own models for landfill emissions quantification to be used for compiling national GHG inventories (Scharff & Jacobs 2006, Mou et al. 2015, Oonk 2010).

Modeled approaches to quantification of landfill emissions are generally part of larger bottom-up (B-U) approaches to establishing regional, national, and global methane emissions estimates. B-U methods rely on site-specific data in order to estimate overall emissions. The data necessary in order to fill out the models usually includes the amount of WIP, as well its composition. In contrast to B-U methods, top-down (T-D) approaches generate quantitative emissions estimates via atmospheric measurements. T-D approaches are used by researchers to validate or evaluate GHG inventories at every scale from city-specific to estimating the global methane budget (McKain et al. 2015, Fernandez-Amador et al. 2020). Some T-D studies, conducted in both the US and EU, observe discrepancies between their emissions estimates and values reported and accepted by the IPCC (Plant et al. 2019, Helfter et al. 2016).

In certain contexts, the same T-D studies have implicated landfill-specific emissions estimates as potentially over or underestimated in regional and/or national GHG inventories considered technically compliant with the IPCC (Peischl et al. 2013, Ren et al. 2018, Jeong et al. 2017). Despite such findings, landfills are not actually the usual target of these same studies in which their reported emissions are evaluated. Instead, recent T-D studies have been focused either on discerning overall inventory uncertainty, or as is often the case in the US specifically – accounting for fugitive emissions from natural gas production and distribution (Alvarez et al. 2018, Plant et al. 2019).

Landfill methane emissions have been researched and scrutinized by academic researchers, waste industry operators, and regulators themselves for decades, in both the US and EU (Czepiel et al. 1996, Peer et al. 1993, Huitric et al. 2007, Oonk 2010, Mønster et al. 2015). Generally high level of interest in researching landfill-specific methane emissions estimates is the result of a number of factors, chief among them the overall uncertainty of accepted figures (Mosher et al. 1999). Landfills are considered an area source of emissions, which are variable both spatially and temporally (Oonk & Boom 1995, Scharff et al. 2005). Moreover, a large portion of the literature aims at better characterizing the effects of specific factors on emissions, which are each largely uncertain.

Therefore, this study aims at agglomerating knowledge at the point the two noted branches of research regarding landfill methane emissions quantification meet. In other words, the focus of this work is T-D study results as they relate to endeavors to estimate landfill-specific methane emissions. This is done to contribute a more clear image of limitations inherent to existing methodologies employed for landfill emissions quantification. To summarize, this paper's explicit research questions include:

1. What implications do T-D estimates hold for B-U methods of compiling GHG inventories currently used for landfill methane emissions quantification?
2. Does evidence from literature regarding landfill-specific emissions estimates support any theories for why discrepancies are noted in certain T-D studies?
3. To what extent is the observed phenomenon of discrepancies between T-D and B-U estimates context dependent; are inventory landfill methane emissions estimates more likely inaccurate in the US than in the EU?

Accomplishing the aforementioned task of this paper involves a comprehensive review of literature concerning landfill emissions specifically. There is an apparent distinction here between conclusions reached from research funded by industry in the US, or the EU, as well as from academia. This review is meant to provide the reader an appropriate level of context with which to understand the nature of landfill methane emissions quantification. Additionally the *Literature review* serves to allow readers to better gauge overall significance of any implications found from T-D studies for reported landfill emissions estimates, which is often limited in actual reporting (Scharff & Jacobs 2006). Following the *Literature review*, which includes an overview of relevant policy measures including the IPCC *Guidelines* themselves (2006), analysis of T-D studies conducted over the last 20 years is presented. Accompanying this analysis are results of applying a model framework to infer emissions for individual landfills based on individual landfill methane emissions estimates separately generated from three T-D studies (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018).

All T-D studies analyzed for this paper are separated between the US and EU, for the purposes of better understanding context-dependency of accuracy via comparison of findings. The same delineation is made on an ongoing basis throughout the *Literature review* so as to allow the reader insight into any potential patterns in author opinion corresponding to geographic location. Analysis of T-D estimates and the implications rendered regarding landfills specifically, should elucidate to what extent current efforts toward landfill methane emissions quantification meet expectations regarding accuracy and transparency, among other choice criteria set by the IPCC (2006) as crucial for overall inventory quality. This study does not to any significant extent aim at reviewing the individual methods applied by each T-D study. It instead proceeds from an observed pattern to better discern to what extent the apparent trend in discrepancies between B-U estimates and T-D estimates both generally and with specific regard to landfills, holds.

2. Literature review

The focus of this research is T-D studies as they relate to landfill emissions estimates in the US and EU. That said, for these findings to prove relevant it is important to first visit the seminal international policy framework and research surrounding reported emissions estimates appearing in national GHG inventories. There are a number of policies that either directly or indirectly affect landfill emissions reporting, including several versions of the *EU Landfill Directive* (1999), and the EPA *New Source Performance Standards* (1996) That said, the instrumental piece of legislation analyzed to fulfill the purpose of this *Literature review* is the IPCC's *Guidelines for National Greenhouse Gas Inventories* (2006). The most relevant piece of information to take from this analysis is that IPCC-prescribed methods for reporting landfill methane emissions in national inventories rely on the FOD Method. The FOD Method is based primarily on the assumption that the amount of methane generated and then eventually emitted by a landfill is proportionate to the amount of WIP.

A selection of literature is then reviewed from researchers and landfill operators concerning methods and effective variables for estimating landfill emissions. This subsection takes up the largest portion of the *Literature review*, though main findings are summarized in the openings and closings. Industry opinion is somewhat divided between EU and US landfill operators regarding conservative values chosen for several parameters included by the IPCC in its prescribed FOD Method. Additionally, several decades of research have yet to yield a single method for measuring emissions in-situ, which is considered the most cost-effective and accurate for predicting an annual methane emissions rate. There is overwhelming consensus that more research is required for increasing certainty regarding conservative values for the FOD Method's application.

2.1. IPCC Guidelines for National Greenhouse Gas Inventories

The IPCC *Guidelines* (2006) mandate all UNFCCC-compliant nations report GHG emissions in a national inventory. To that end the *Guidelines* (IPCC 2006) prescribe specific compliant methodologies to be used for estimating landfill methane emissions. According to research done by the IPCC itself, landfills contribute 3-4% of total anthropogenic GHG emissions, nearly all of which consisting of methane (IPCC 2000). The latest version of the *IPCC Guidelines* was published in 2006, and included two major changes from the previous version published in 1996 (IPCC 1997).

The 1996 *Revised Guidelines* (IPCC 1997) include a three-tiered approach to establishing methods for emissions quantification, Tier 1 being the simplest, Tier 3 the most complex. The same tiered structure is used in the 2006 *Guidelines* (IPCC), however the Tier 1 methods were altered from a mass-balance approach to a simplified FOD Method, originally considered a Tier 2 method in the 1996 *Guidelines* (IPCC 1997). The FOD Method relies on the theory of first-order kinetics, in this case applied as a first-order reaction. A first-order reaction maintains that the amount of a product is proportionate to the amount of the reactant. In the case of landfills, this means that the FOD Method used, based on a first-order reaction, assumes that the amount of methane emitted is ultimately proportionate to the amount of WIP (IPCC 2006). This is a rough translation of the model ultimately applied for estimating emissions. Although some details will be expanded upon, this central assumption is crucial to consider as one encounters numbers generated using different models and measurements for estimating landfill methane emissions.

The FOD Method is considered Tier 1 methodology, using mainly default activity data and default parameter values. Tier 2 methodologies include use of the FOD Method with some country-specific parameters and data based on current and historical waste disposal.

Country-specific data relevant for applying a Tier 2 FOD Method would include amounts of WIP for example. Tier 3 methods are characterized by country-specific data (like Tier 2) but with parameter values established nationally either through actual measurements or some other means. Tier 3 methods are encouraged by the IPCC to include certain parameters: a) half-life (*k-values)* of landfilled waste and b) methane generation potential (L_0) or through Degradable Organic Carbon (DOC) and the fraction of DOC that degrades (DOC_f) combined (2006) For a basic summary of the process prescribed by the IPCC to national regulators for determining which Tier (1-3) to use for modeling landfill emissions, see Figure 1 below.

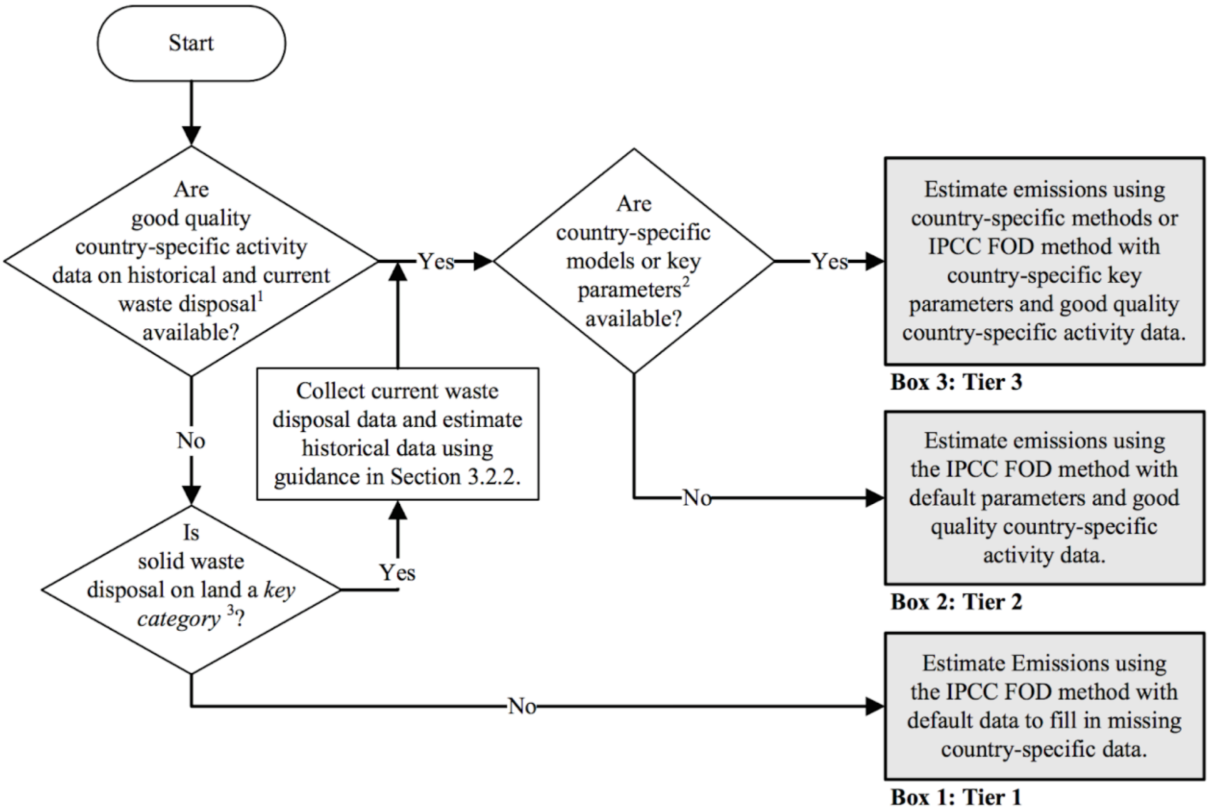

Figure 1. Decision-making guide for regulators regarding IPCC-prescribed models

Source: (IPCC 2006)

The *Guidelines* (IPCC 2006) provide a Waste Model used for calculating the total amount of DOC landfilled, and allows for two options: single-phase and multi-phase. The single-phase option is based on the assumption that different waste types and categories degrade in a *dependent* fashion. This allows for use of a single half-life or *k*-value when completing the model because the assumption renders the bulk of waste in mass the only critical factor, rather than the separate degradability of different types of waste co-landfilled at the same site. The multi-phase model assumes that the waste degradation of different WIPs is an *independent* reaction, in which different WIP types require separate *k*-values in order to account for their marginally different degradation rate. The IPCC asserts that, "at the time of writing these guidelines, no evidence exists that one approach is better than the other," (2006). That said, the *Guidelines* acknowledge that this is dependent on the relative stability of waste composition. Banning disposal of waste high in organic content (i.e. food waste) can lead to rapid changes in composition and by extension, the k-value of landfilled waste. This affects the relative accuracy of the Waste Model for predicting DOC.

The *Guidelines* (IPCC 2006) note a number of methodological choices as 'good practice'. These include using DOC_f values specific for different WIP types only when adequate data is available, as well as using direct measurements from landfill gas (LFG) collection systems to develop country-specific parameter values. In general, measurements are not discouraged from use in generating country-specific values for waste input data, as well as parameter values. That said, the *Guidelines* (IPCC 2006) caution use of any measurements without fully accounting for limitations of methods typically used for measuring landfill emissions in-situ.

The IPCC (2006) notes methane emissions measurement at landfills is extremely difficult given spatial and temporal variability, as well as a lack of standardized best practices. A number of factors are noted in the *Guidelines* (IPCC 2006) as influencing this level of difficulty, however generally it is appropriately characterized by noting that on a landfill surface, "emissions at locations a few meters away from each other can vary over a factor 1000." Moreover, none of the recommended methods typically used for landfill methane emissions measurements are considered both affordable and accurate.

Despite uncertainty of in-situ landfill methane emissions measurements, the IPCC notes its own FOD Method as itself highly uncertain (2006). Several measurement methods are noted in the literature as worthy of research for delivering precise, accurate results. Yet these options remain very expensive, while the FOD Method offers a simplified approach to establishing emissions from a, “very complex and poorly understood system,” (IPCC 2006). The FOD Method’s uncertainty can be attributed first to the fact that all DOC may not degrade according to the pivotal assumption of the first-order reaction. As well, that all solid waste disposal sites (SWDS) are heterogeneous, and the fact that neither k-values or historic WIP amounts are understood or adequately researched. This begs the question: given all the noted uncertainties in the prescribed FOD Method(s), why was any overestimation using the former Tier 1 *mass-balance* method (IPCC 1997), considered less desirable? Especially given GHG inventories are in theory meant to inform policymakers where promoting emissions reductions could be most beneficial.

In summation, IPCC-prescribed methods include a three-tiered approach (2006). Each tier up indicates increasing overall complexity and the number of assumptions inherent to (any) emissions estimate generated. The change from prescribing a mass-balance method for Tier 1 to simplified IPCC FOD Method in the latest IPCC *Guidelines* (2006) is justified for the potential overestimation of emissions resulting from using mass-balance. This in spite of inherent uncertainties noted for the newly prescribed FOD Method, the main assumption of which bears a large degree of uncertainty.

The IPCC *Guidelines* (2006) do provide ample supporting research as well as accounting of apparent limitations and potential for errors. As well, numerous warnings are issued regarding use of data or measurements without accounting for potential underrepresentation and/or limitations. The *Guidelines* (IPCC 2006) also note, “inventory compilers should study significant discrepancies to determine if they represent errors in the calculation or actual differences,” among other advice regarding reporting protocol. This idea will be revisited later on given the aim of this paper to investigate discrepancies attributed to landfill emissions by T-D studies.

2.2. Landfill methane emissions estimation

Waste management, and by extension landfills, represents a potentially excellent route for mitigating overall GHG emissions in both developed and undeveloped countries (Bahor et al. 2009). Bahor et al. (2009) expanded on the concept of 'stabilization wedges' (Pacala and Socolow 2004) to demonstrate via life-cycle analysis (LCA) more than 1 Gt C potentially avoided via altering waste management practices. This was shown to be mainly achievable through reducing the amount of waste landfilled globally on an annual basis. Policy measures drafted to take advantage of this potential emissions reduction necessitate reliable methods for quantifying landfill methane emissions (Scharff & Jacobs 2006). To that end, a large selection of literature is available concerning methodologies for and variables affecting the accurate quantification of landfill methane emissions. A number of subtopics have been researched within this field. Some included are: half-life or k-values for waste, carbon sequestration & storage in landfills, methane generation potential (L_0-values), efficiency of gas collection systems (CE), microbial soil oxidation in soil cover layers (OX), and comparison of measurement methods used for emissions quantification.

Methanogenesis is the final segment of a multi-phase process that occurs in landfills during which DOC in waste is converted to methane (Scharff et al. 2005). A number of connected processes affect the end emission rate from landfills, in turn affected by complex variables that vary by location (Scharff et al. 2005). Models used to deal with variability in emissions are best categorized as, "simplifications of the actual world," and indeed, "striking the balance between simplification and thoroughness is difficult," (Peer et al. 1993). The basic process through which landfills emit methane is demonstrated in Figure 2, below.

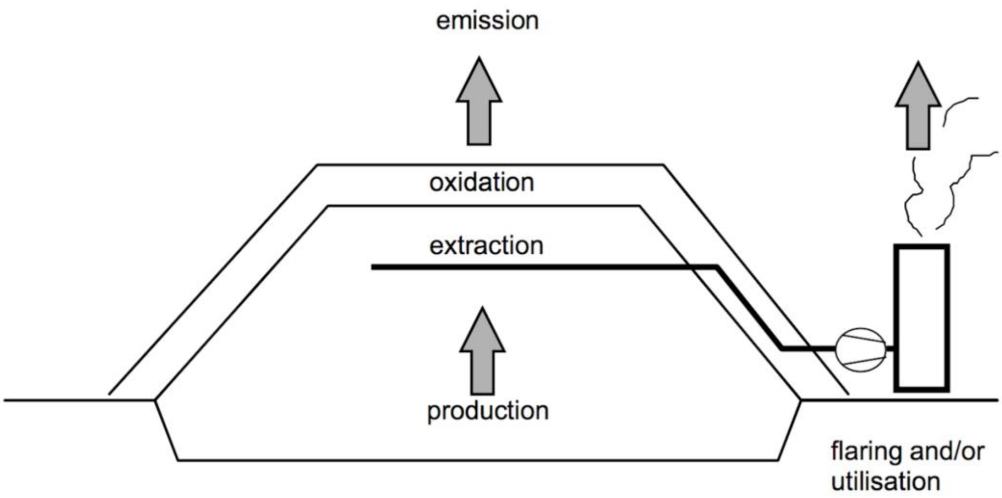

Figure 2. Formation process of landfill methane emissions

Source: (Scharff et al. 2001)

While under ideal circumstances this *Literature review* would enable a comprehensive understanding of the complex factors affecting quantification of methane emissions, they are indeed many in number and brevity is a necessity. That in mind, three topics are explored from the literature in great detail: CE, OX, and direct emissions measurement techniques. The relative prevalence of each within the literature supports this decision. Additionally, CE and OX are noted as the two main options landfill operators have at their disposal for effective emissions 'control' (Spokas et al. 2011a). Measurement techniques remain at this time the alternative means for landfill methane emissions quantification, in most cases used for model validation. Moreover, there is currently no consensus on preferred methodology and measurements remain scarce in number (Oonk 2010, Lohila et al. 2007). A range of topics typically receive attention in literature, all of which will not be discussed at length in this paper for reasons already mentioned. These topics include k-values, methane generation potential (L_0-values), and carbon sequestration/storage among several others.

K-values are mainly a concern with regard to the effects of different waste compositions on half-life and by extension, methane generation rate(s), of waste itself. There is slight division, especially evident from certain industry points of view, regarding the assumed higher half-life or reduced k-values for waste categories low in DOC (Kong et al. 2012). For the purposes of this paper, it suffices to make note of the fact EU waste policies that limit disposal of waste high in DOC seems to lead to lower k-values and lowered methane generation rate (Mou et al. 2015), as well that overall the k-values used in most models are rough estimates at best (Börjesson et al. 2009).

There is evidence supporting the relative importance of the assumed methane generation potential (L_0) for quantifying overall landfill methane emissions. It's even been claimed that L_0-values are more crucial for determining accurate emissions quantification than the choice of model to be applied (Peer et al. 1993). As well, L_0-values have been suggested as highly important for the operation of landfill gas-to-energy (LFGTE) projects. This affects Clean Development Mechanism (CDM) planning that make use of LFGTE technology for emissions reductions (Cho et al. 2012). However, there is also evidence suggesting the quantity of waste is ultimately the optimal indicator for methane generation potential, despite other factors known to have *some* effect on L_0-values (Cho et al. 2012, Börjesson et al. 2009). This division is considered sufficient for this paper. If anything, one should remain aware of the documented importance of L_0-values when quantifying landfill methane emissions.

CE and OX are ultimately the subjects of separate subtopics within this paper for the prospects they offer for garnering enhanced understanding of the landfill industry perspective regarding emissions quantification. Both are widely researched within and outside the industry. This is due both to the relative importance of their designation in most models including the IPCC FOD (2006), as well as the perceived 'control' they offer operators for emissions (Spokas et al. 2011a, SWICS 2009, Huitric & Kong 2006). Carbon sequestration appears throughout the literature, alongside CE and OX (SWICS 2009), and is as well noted in the IPCC *Guidelines* (2006). This is a result of speculation landfills should be credited for the carbon stored temporarily beneath cover soils that is not emitted over the short term.

Certain US landfill operators, also claim a 'carbon sequestration factor' (CSF) be applied that would entirely negate any modeled or measured emissions from landfills (SWICS 2009).

This work recognizes landfills (environmental externalities aside, Eshet et al. 2005) as still the most economically feasible waste management option in a number of countries including the US (Amini et al. 2013). Considering landfill industry position, which calculates CSF-values that would in some cases balance landfill methane emissions (SWICS 2009, Weitz et al. 2002) and contrasting findings stating its effect as potentially 'trivial' when factored into models (Amini et al. 2013), the latter will remain the position of this paper. The reader should consider CSF a valid topic worthy of further research. This paper is constructed accepting any due criticism for ruling out further discussion of a theoretical CSF in favor of focusing more on widely agreed upon influential variables; CE and OX.

To reinforce a point already raised, the task of estimating landfill emissions is inclusive of a range of interrelated, interdependent, observable, nonetheless uncertain variables. This section and all its component constitute a brief exploration of the variability of landfill emissions so as to harness available knowledge toward offering a sense of how conspicuous actual quantitative emission estimate truly are. This is not done to declare inventory figures wholly inaccurate. Instead this *Literature review* should illustrate the uncertainty inherent to the task of generating comparable landfill methane emissions estimates, regardless of methods chosen to do so.

Some experts from industry and academia recommend harmonization of models to allow for more useful comparisons and better consistency among emissions estimates (Oonk 2010, Oonk 2012, Scharff & Jacobs 2006, Scharff et al. 2005). Still, variables that differ between countries including waste composition and climate, make it extraordinarily difficult to create a single model that balances, "simplification and thoroughness," which is applicable everywhere (Peer et al. 1993). Using US methane recovery data for example, would by one estimate result in global overestimation of methane production from landfills (Peer et al. 1993).

Beyond quality of models applied, data availability and quality even among developed countries directly inhibits efforts to improve accuracy of landfill emissions estimates (Bogner & Matthews 2003). Fundamentally, there is widespread consensus that individual landfills 'behave' quite differently from one another in terms of their emissions (Börjesson et al. 2009, Aronica et al. 2009, Bogner et al. 2007, SWICS 2009, Scharff et al. 2005). Therefore, it seems inevitable certain T-D studies of overall methane emissions would implicate landfills in their scope as poorly accounted for in GHG inventories. This is explored in great detail further down in the *Analysis & results*. Presently, effects related to chosen values for CE and OX, as well as strengths and weaknesses of different measurement techniques will be reviewed toward granting insight into the scientific mystery that is landfill methane emissions.

2.2.1. Landfill gas collection efficiency

LFG collection was pioneered at the Palos Verdes Landfill in California (US) in 1975 (Spokas et al. 2006). The technology itself was likely not developed as an 'engineered control' on emissions (Bogner et al. 2007) but instead at least partially as a means for mitigation of odors and any explosion hazards (Cosulich et al. 1992). LFG collection matured through the 1980s and was largely implemented throughout Northwest Europe and the US during the 1990s (Oonk 2012). At the time of publication, Spokas et al. (2006) documented 1100 LFG collection systems in place globally, with 465 split between Denmark, Italy, and the UK, and more than 350 within the US alone. These numbers have likely grown significantly in the years since, given increasing interest in the technology for potential economic and environmental benefits (Cho et al. 2012).

Simultaneous to its development as a technology, LFG collection modeling began in the 80s as well, at first as a means of determining the profitability of LFGTE projects. A shift took place in the 1990s when LFG collection modeling instead began to emphasize landfill methane emissions quantification as its main purpose (Oonk 2010). Some argue profitability remains the true dominant motivation for implementing LFG collection (Oonk 2012). Others, usually within the landfill industry particularly in the US, differentiate projects implemented for profits from those they regard as geared toward reducing emissions (SWICS 2009, Huitric & Kong 2006). In general, both the need for alternative, non-fossil, sources of energy and to reduce overall carbon emissions are reasons cited for growing interest in implementing LFG collection systems globally (SWICS 2009, Spokas et al. 2006, Cho et al. 2012).

Motivation for implementation aside, LFG collection is largely regarded as an extremely influential factor affecting overall landfill methane emissions (Bogner & Matthews 2003, Lohila et al. 2007, Fjelsted et al. 2020). This finding comes from field observations from both the US and EU. It is more or less agreed upon that presence and effectiveness of LFG collection directly impact overall methane emissions. However, the collection efficiency (CE) values suggested for use in models to estimate emissions are the subject of a large amount of scrutiny. The general conclusion from US landfill operators US is that the EPA's default 75% CE used in their prescribed model for reporting emissions, is too low and negates the operation of collection systems operating at or above 90% CE (Huitric & Kong 2006, SWICS 2009, Huitric et al. 2007). Similar CE values are supported by a study conducted at three landfill sites in France, which estimated CE in excess of 90% at all but one cell, based on measurements (Spokas et al. 2006).

Despite the findings of a study conducted in France (Spokas et al. 2006), the trend in the EU appears more toward lower CE values estimated from field measurements. Mønster et al. (2015) estimated average CE for 5 Danish landfills at 41-81%. Börjesson et al. (2009) calculated an average CE for all the landfills in Sweden to be 51 ± 5%. Aronica et al. (2009) quantified CE for a single landfill in Palermo (Italy) of 23.5-43.1%. Dutch authorities report values equal or less than 15%, while the UK uses a 75% default (similar to the US) based on the assumption that incentivized improvements for landfill operators directly influences CE (Oonk 2012). Generally, literature supports country-specific average CE values from measurements to be lower. Among EU countries with available national average CE values estimated from measurements, Finland ranks among the highest at 69% (Lohila et al. 2007). This remains significantly lower than CE values between 90-100% US landfill operators claim as justified based on their own studies.

Collection itself is a matter of extracting LFG using a pump via pipes that penetrate the surface of a landfill. Collected LFG is either flared, converting methane into carbon dioxide and water through oxidation that occurs while burning, or it is used to generate electricity. Numerous factors affect CE and the overall functions of this process, including OX (discussed in the next section), seasonality (Spokas et al. 2006), as well as the age of the landfill and cover material properties (Barlaz et al. 2004, Oonk 2012). These are highly variable at each individual landfill reporting to compile a national inventory of landfill methane emissions. However, given use of models for estimating emissions that are reported, the IPCC (2006) informs us of the critical nature of establishing a representative default CE value. Overestimation of CE leads to global underestimation of methane emissions, which could delay or prevent implementation of policies and measures aimed at reducing overall landfill emissions (Oonk 2012). Therefore, CE is a crucial parameter to be aware of when viewing reported or otherwise estimated landfill methane emissions, perhaps bearing in mind, "as a rule of thumb, inaccuracy of methane emissions increases when the efficiency of LFG recovered increases," (Oonk 2010).

Operators and researchers, within certain spheres, may deny increased CE will lead to inaccurate landfill methane emissions estimates. They would instead likely assert that methane generation is often overestimated leading to lower measured CE values (Bogner &

Matthews 2003), which should otherwise regularly reach 100% (Huitric et al. 2007). Whilst accepting relative plausibility of such findings being true, it is imperative to reiterate that most measurement results indicate lower average CE values (Oonk 2012, Hensen & Scharff 2001). That most of the separate findings supporting this notion come from EU countries may seem counterintuitive. After all, EU waste policies largely restrict the amount and overall DOC of landfilled waste, reducing overall landfill methane generation (Scharff & Jacobs 2006). Bahor et al. (2009) quantified a ton of MSW in Northern and Southwestern Europe as equivalent to 0.05 tons carbon emissions, compared to 0.15-0.20 tons carbon per ton MSW for the rest of the world, inclusive of the US. It is equally important to note that similar to overall landfill emissions estimates, values for CE from models are reflective of assumptions, while measurements reflect the actual site-specific practices for LFG collection (Amini et al. 2013). Those that support CE values contrary to typical prescribed defaults and other measured values, which are in excess of 90%, do usually base their findings on measurements of some kind. These are however, in some cases reliant on measurement approaches (Spokas et al. 2006, Huitric et al. 2007) lacking spatial resolution to account for variability of landfill methane emissions (Oonk 2010, Börjesson et al. 2007).

All that said, relative strengths of different measurement techniques are the subject of a later subsection within this *Literature review*. For this subsection with regards to values found for CE of LFG collection systems, it is important to make note of several observations:

- LFG collection's presence is an important factor when considering the magnitude of landfill methane emissions (Bogner & Matthews 2003, Fjelsetd et al. 2020).
- Measurements usually lead to lower average CE values, particularly in the EU compared to the US. This, despite policies affecting decreasing DOC and overall amounts of waste landfilled in the EU (Scharff & Jacobs 2006).
- Landfill operators in the US support use of CE values in excess of 90% (Huitric & Kong 2006, SWICS 2009).
- Overestimating CE directly results in underestimation of landfill methane emissions (Oonk 2012).

This in mind, one is able to better understand the variability of accuracy of landfill methane emissions estimates based on CE values used in models. The purpose of this is to express the variability of a crucial parameter contained within the models typically used (CE, FOD or IPCC). This hopefully illustrates the assumption-laden nature of the models themselves, and by extension any landfill methane emissions estimates they are used to generate.

2.2.2. Cover soil methane oxidation

The IPCC (2006) prescribes a default 10% soil methane oxidation (OX) value in their FOD Model for estimating landfill methane emissions. This is meant to account for the process by which a certain portion of methane generated in a landfill is oxidized by bacteria as it travels through cover soil, converting it to carbon dioxide before emitted. The 10% figure is meant to characterize the percentage of generated methane that is oxidized and thus removed from total emissions estimates. The actual process for OX in landfill covers occurs due to the presence of a special group of bacteria called methanotrophs. Methanotrophs are found anywhere methane concentrates at or above atmospheric levels and oxygen remains accessible, since they are aerobic (SWICS 2009). Thus, methanotrophs are limited in their distribution within landfill cover soils, by the upward diffusion of methane and the downward diffusion of atmospheric oxygen. It is based on this that a 'methanotrophic active zone' can be described, including roughly the uppermost 30-40 cm. Within this active zone exists a 'maximum oxidation zone' 15-20 cm below the surface where a large proportion of methane oxidation in landfill cover soils takes place (Jones & Nedwell 1993, Czepiel et al. 1996).

Landfill cover soil oxidation (OX) receives a lot of attention from researchers and landfill industry for the potential it offers as an emissions 'control.' Specifically, research focuses on application of biofilters or biocovers (Barlaz et al. 2004) in circumstances when LFG collection is not cost-effective (Scheutz et al. 2009). Despite the potential cited for optimizing OX toward mitigating landfill methane emissions, the actual rate or percentage of methane oxidized by landfill covers is “rarely known with any degree of certainty,” (Boeckx et al. 1996, Scharff et al. 2005). Furthermore, the uncertainty in accurately accounting for oxidation through the variable OX, contributes disproportionately more to the overall uncertainty for national or global landfill methane emissions estimates (SWICS 2009).

The majority of the uncertainty associated with quantifying OX in landfill cover soils is due to the range of factors limiting its effects including:

a) spatial homogeneity of (methane) flux,
b) flux rate,
c) cover material,
d) moisture content,
e) ambient temperature, and
f) climate and seasonal variability as it relates to effects on moisture and temperature of soils (Oonk 2010).

A number of these factors are difficult to characterize and/or are largely uncertain in terms of the extent to which they affect OX. For reference, ‘flux’ is the transport of methane through the cover layer, and can be described both spatially and temporally, which are differentiated in terms of effects of each on OX. Methane is transported from the anaerobic digestion zone in a landfill where waste generates methane and other components of LFG via advection, as a result of a pressure gradient, or diffusion, due to a concentration gradient (SWICS 2009). These processes often occur simultaneously, creating a major obstacle for estimating OX (Scheutz et al. 2009). The importance of the delineation between advection and diffusion is also variable site-to-site (SWICS 2009), partially related to different cover properties. Of vital importance, there is no method for apportioning flux between advection and diffusion (Börjesson et al. 2007).

Flux rate also influences OX in landfill cover soils (Oonk 2010, Chanton & Liptay 2000). Most landfill cover layers possess a relatively uncertain ‘capacity’ for LFG, beyond which ‘overloading’ occurs and OX significantly decreases (SWICS 2009). Field measurements for OX in actual landfill covers may not represent the full potential for oxidation in landfill cover but the localized in-situ capacity (Spokas & Bogner 2011). Defining the flux rate however, for even a single landfill, requires a number of assumptions. Furthermore, attempting to define this rate quantitatively creates a sort of paradox. Characterizing OX for model use, in order to estimate overall landfill methane emissions, requires an estimate for the flux rate, which implies a known emissions rate (SWICS 2009).

Soil moisture content of landfill cover is a crucial element for determining OX, affecting both gaseous transport and microbial activity (Boeckx et al. 1996, Bogner et al. 1997). Moisture content, similarly to flux rate, is optimized for OX at a certain level that changes depending on several related factors including cover material. Beyond this threshold, soil (or other cover material) saturation actually inhibits transport and limits overall OX (Scheutz et al. 2009). Ambient temperature generally has a positive relationship with OX, increasing temperature corresponding with increased OX (Oonk 2010). A Q10 variable is often described, which quantifies the rate of increase for OX with each 10 degree C ambient temperature increase (Scheutz et al. 2009). This effect is largely associated with the effect ambient temperature has on overall microbial activity, especially pronounced in soils at lower temperatures where microbial activity slows considerably (Scheutz et al. 2009). There is evidence that of the two distinct types of methanotrophs, one continues to perform at colder temperatures (Börjesson et al. 2004). Notably, the vast majority of the literature still associates generally colder climates with reduced overall OX (Oonk 2010).

Given the range of factors already mentioned, it is perhaps not so surprising that like CE, an appropriate default OX value for model use is widely disputed. Beside the IPCC (2006), default 10% OX is prescribed by the EPA based on the findings of Czepiel et al. (1996), along with some EU regulators (Hensen & Scharff 2001). Voices within the landfill industry speculate this number is low and requires updating to accurately reflect, “current engineering technology,” and, “more recent research,” (SWICS 2009, Schmeltz 2017b, Bogner 2020, personal communication). Other, 'more recent research', shows mixed results largely inconclusive regarding the ongoing validity of a 10% default OX (Oonk 2010). This study is inclined more toward agreement with the latter, though it is the position of landfill industry itself that is useful for gaining contextual understanding of analysis of T-D studies.

A report from the Solid Waste Industry for Climate Solutions (SWICS 2009) describes a mean 35±4% OX from in-situ measurements at several sites with final clay-sand covers. Fjelsted et al. (2020) observed 63% OX at Hedeland Landfill (Denmark) during the time in-situ measurements were taken to quantify overall methane emissions. Chanton & Liptay (2000) measured OX at a landfill in Florida (US) and divided their results seasonally, estimating OX for winter = 3-5%, and for July = 43±10%. Börjesson et al. (2007) took measurements for OX at several Swedish landfills. Results were split between active and closed landfills, rather than establishing different seasonal rates. That study found closed landfills to exhibit higher OX (36.7-42.8%) compared to operational landfills (6.0-24.8%). Comparatively both Chanton & Liptay (2000) and Börjesson et al. (2007) suggest SWICS' findings are at least plausible (2009). Although, the fact Chanton & Liptay (2000) conducted their study at a site in a warm, moist climate and still found winter OX rates less than that of the 10% OX suggests SWICS' findings may not account for seasonality (2009).

Czepiel et al. (1996) is the sole piece of literature cited supporting EPA's reinforcement of the IPCC-prescribed (2006) 10% default OX. Czepiel et al. (1996) took in-situ measurements at a landfill in New Hampshire (USA), representative of a relatively cold climate. OX was found to be 20% in October. This figure was then used to generate an annual 10% OX estimation based on fluctuations in soil temperature & moisture content as a function of the seasons. Chanton et al. (1999) took measurements at the same landfill later, and estimated a similar overall OX (0-23.6%). The justification for a 10% OX is old and perhaps unrepresentative of different climates. Still, considering lower bounds of estimated OX from several studies conducted in warm weather areas, 10% appears at least conservative if not tangible for use in generally uncertain models to estimate landfill methane emissions.

There are several measurement methods typically used in order to quantify estimated overall OX. Stable isotope methods are, “generally regarded as the most accurate,” (Scharff et al. 2005), performed by measuring the ratio of $^{13}C/^{12}C$ isotopes in emitted methane before and after passing through landfill cover (SWICS 2009). This method is possible due to general methanotrophic preference for consumption of ^{12}C isotopes compared to ^{13}C (Scheutz et al. 2009). This leads to unusally enriched ^{13}C LFG post-soil oxidation. OX estimates based on use of such stable isotope methods should still be considered cautiously due to the necessity for use of a fractionation factor (α), which is largely uncertain (Borjessson et al. 2007). The fractionation factor describes rate constants for $^{13}C/^{12}C$ ratios in methane. In other words, α represents an attempt to classify the $^{13}C/^{12}C$ ratio for methane not yet oxidized and for methane already partially oxidized. Therefore, while measurements are considered instrumental for estimating OX, the actual estimates should be considered with knowledge of their uncertainty due to assumptions necessary for their quantification. While OX estimates based on in-situ measurements are not uncommon, that OX is cited as capable of ranging from negligible to “>100%” in the field (Spokas & Bogner 2011, Scheutz et al. 2009) should serve as the basis for a reluctant acceptance for using the model variable at all.

There are those within industry and academia who support considering the upper bound of this proposed range as, “expected,” under the right circumstances, which generally precludes a highly efficient LFG collection system (SWICS 2009, Bogner et al. 1997, 1995). With regard to the fact industry voices suggesting this also come from the US, where similar positions have been noted for CE and other variables (SWICS 2009, Huitric & Kong 2006, Huitric et al. 2007, Kong et al. 2012), such testimony should be considered tentatively, if at all. This is due to the apparent motivation of these industry members to nullify emissions from certain landfills.

This section explored studies aimed partially or exclusively at characterizing OX quantitatively for landfill cover soils in various geographical contexts. Considering the 10% default OX for model use, which is prescribed by the IPCC (2006) and most other regulators including in the US (EPA) and EU, review of this specific literature is demonstrative of scattered and mixed results. This is overall reflective of the variability of OX in landfill cover soils. Such is largely due to influence by many interrelated, still poorly understood effective variables.

Certain research suggests a default 10% OX be reconsidered toward actually raising it or allowing for different values depending on cover material (Spokas et al. 2011a). It is the choice of this paper to highlight instead the large uncertainty of modeled emissions estimates from landfills globally (Spokas et al. 2011a) as evidence that safety in potential overestimation would seem prudent. This option is especially attractive when compared to potential underestimation on the grounds OX is poorly accounted for, or that landfill covers extract atmospheric methane from surrounding environs (Bogner et al. 1997).

It seems wise to reiterate the merits of further exploring the influence of OX, given the potential of biocovers and biofilters as cost-effective overall emissions reduction (Barlaz et al. 2004, Scheutz et al. 2009). Still the importance of this endeavor for characterization remains secondary to resolving CE values for enhanced overall landfill methane emissions quantification (Börjesson et al. 2007). This suggestion is informed by the previous subsection, the findings and conclusions expressed here, not to mention evidence OX only accounts for a small amount of the total LFG emitted (Börjesson et al. 2009, Amini et al. 2013). That OX receives so much attention in literature seems a byproduct of its use in IPCC-prescribed models (2006). As well, due to often reiterated potential for simple emissions reductions, as well as its pre-emptive role as one of only two possible 'controls' on emissions (Bogner et al. 2007).

2.2.3. Measuring landfill methane emissions

The only way to gain landfill-specific information on either of the model parameters explored, or methane flux itself, is to measure emissions directly on-site (Lohila et al. 2007). Measurements have been suggested as potentially pivotal for validating modeled emissions (Aronica et al. 2009) and by extension for actually 'tuning' models used for inventory compilation of landfill methane emissions (Scharff & Jacobs 2006). The merits of measurement for use in landfill methane emissions estimation will yet be discussed, though it is important to note while numerous methods are used, no single method is recognized unanimously as preferred (Oonk 2010). This is the result of a number of factors limiting the actual use of measurements, though two appear foremost in the literature: costs and difficulties.

Cost is a major limiting factor, resulting in far fewer emissions estimates from measurement methods. This is also largely related to potential disinterest among landfill operators in investing in something that is not technically mandatory (Huitric & Kong 2006). Furthermore, how much a particular method of measurement costs influences how frequently it is applied overall (Oonk 2010). Low costs however, hardly guarantee accuracy. This becomes especially problematic considering measuring landfill methane emissions is made extremely difficult by the various sources of potential uncertainty (Oonk 2010, Spokas et al. 2006, Lohila et al. 2007, Peer et al. 1993). Sources of uncertainty for measurement methods are the same as those noted for modeling landfill methane emissions; multi-dimensional spatial and temporal variability (Oonk & Boom 1995, Bogner et al. 2007).

Notably, the size of a modern landfill presents an added obstacle to conducting emissions measurements (Mosher et al. 1999), especially given the method used must somehow account for the entire landfill (Oonk 2010). The typical pathways for emissions, which are often cracks and fissures in the cover or leaks in the gas collection system, collectively known as 'hotspots', experience concurrent flux and must be simultaneously measured (Spokas et al. 2006). Hotspots are also apt to relocate (Börjesson et al. 2009). Such movements occur over time, often in daily, weekly, and seasonal cycles (Spokas et al. 2006) that must somehow be represented in the estimate from measurements.

Landfill-specific issues creating temporal variability in emissions necessitates 4-6 one-day measurements to support a representative annual methane emissions estimate (Oonk 2012). Generally this may necessitate several measurements be taken throughout the year (Hensen & Scharff 2001). This creates added complications when one considers the original issue; costs, which are generally exorbitant for methods demonstrating high levels of accuracy (Oonk 2010). Though carbon sequestration is disproportionately emphasized by some researchers compared to its influence on landfill emissions, it may affect short-term variations in flux. This leads to added uncertainties in emissions measurements in accounting for temporal variation (Scheutz et al. 2009).

Regarding difficulties inherent to landfill emissions measurements, owed to, "variability in landfill design, construction, and operation," some posit two methods be used simultaneously in order to accurately measure emissions (Peer et al. 1993, Bogner et al. 1997). This again however, brings into question the costs associated with generating valid landfill methane emissions estimates if more than one measurement technique are potentially needed for a single landfill. To that end, the most commonly applied method is also the cheapest; closed (flux) chamber measurements (Amini et al. 2013, Oonk 2010). Boxes or 'chambers' are placed partially submerged atop the surface of a landfill, measuring flux as it passes through the top of the cover into the surrounding environment. By averaging the measured flux from the chambers, researchers are able to estimate emissions for a specific cell, or in some cases, the entire landfill. This is a simplified explanation, but it is crucial one note although it is quite inexpensive to execute in the field, chamber measurements lack spatial resolution. Moreover, estimates likely significantly underestimate emissions by missing a portion of the overall flux from hotspots (Oonk 2010). This is theoretically amended through qualitative measurements to locate cracks and fissures on the surface, but cracks are again, likely to relocate over time (Börjesson et al. 2009). Unless every crack and fissure is accounted for with a flux chamber then this method will always miss an uncertain amount of overall methane emissions. Moreover, Mosher et al (1999) concluded that at distances greater than seven meters, "adjacent chamber flux measurements were essentially independent."

Micrometeorological methods use a horizontal plane extending several meters above the surface of a landfill to measure flux and wind concurrently. An emissions estimate is generated from the measured covariance between the two variables. The method is is generally considered reliable but is not usually applicable for the largest landfills or landfills with less than perfect topography (i.e. sloped or otherwise uneven, Oonk 2010). Several other methods use *vertical* planes, either along the edges of the landfill or located both upwind and downwind. Mass-balance methods fall within the latter. Estimates are made based on the difference between concentrations of methane measured as the plume travels through the plane downwind, and the background measured from the opposite side. This method is also noted as robust though is again limited to landfills with relatively even topography. Mass-balance methods are also considered expensive (Oonk 2010). Tracer plume methods rely on the release of a known amount of 'tracer' gas upwind of a landfill, and measurement of the plume downwind to compare the mixing ratio and calculate the influence from the landfill. This method is noted for its accuracy but is again expensive. Tracer methods are also prone to influence by nearby sources of methane that infiltrate the plume downwind (Oonk 2010).

Furthermore, tracer methods require available roads to drive the measuring equipment on, which are situated along a transect downwind from the plume.

Description of methods here is not meant as an exhaustive list for explanation or review, but to expose the general limitations and logic inherent to using them. This is relevant given numerous authors touting measurement techniques as useful for tuning or validating models in use for estimating landfill methane emission (Bogner & Matthews 2003, Scharff & Jacobs 2006, Aronica et al. 2009, Oonk 2012, Hopkins et al. 2016). From here several experiments are explored, which test modeled emissions estimates against measurement-based estimates. Notice should be given to the fact measurement techniques are noted for their relative consistency for estimating emissions compared to different models (Oonk 2010, Scharff & Jacobs 2006).

Mønster et al. (2015) estimated emissions from 15 Danish landfills based on measurements using tracer methods and compared the results to inventory estimates reported to the IPCC. The measurement-based estimates indicated overestimation in the inventory for 12 of 15 landfills. Börjesson et al. (2009) conducted a similar study to calculate an annual emissions rate for all Swedish landfills and compared their results to modeled estimates from IPCC-prescribed methods and those prescribed by Swedish environmental authorities. Their results show solid agreement between measurements and the Swedish regulator model-based estimates, but overestimation from the IPCC-prescribed methods compared with the measurement-based estimates. Scharff & Jacobs (2006) had similar results. Their measurements from two Dutch landfills include modeled estimates accounted for up to 570% of the measured emissions at one landfill, and up to 520% of the measurement-based estimate from the other.

In the US, Amini et al. (2013) compared modeled and measured results for OX, CE, and overall methane emissions for three landfills using EPA-prescribed, IPCC FOD-derived modeled methods and measurements. Results indicate slight overestimation of average annual emissions using the EPA-prescribed modeled approach. These are similar results previously mentioned from studies conducted in three different EU countries (Mønster et al. 2015, Börjesson et al. 2009, Scharff & Jacobs 2006). In the EU this discrepancy (overestimation) can be attributed to poor accounting for the low organic content of waste typically landfilled in the EU as a result of legislation regarding national waste disposal policies (Mou et al. 2015). The same reasoning however, does not explain the discrepancy demonstrated by Amini et al. (2013) between modeled and measured results for three US landfills. Spokas et al. (2011a) and Cambaliza et al. (2017) also conducted measurements to establish emissions estimates at two different US landfills. Spokas et al. (2011a) however, did not compare their results to modeled results from EPA-prescribed methods, but to their own custom model (CALMIM), which was being tested for validation. Regardless, the results from both indicate far larger estimates achieved from measurement-based methods than from models.

At this point focus shifts from the estimation and reporting of landfill emissions specifically, to implications of top-down atmospheric measurement-based estimates for landfills occurring within the scope(s) of certain studies. T-D studies for overall emissions constrainment are noted as a viable component for inclusion in inventory compilation methods as means of both validation and evaluation of models (Hopkins et al. 2016). This *Literature review* was meant as a primer of some sort; an introductory exploration of the various factors that make estimating methane emissions from landfills complex, difficult, and seemingly political. The reader take from it a better knowledge of the relative deficiencies of methods currently used for estimating methane emissions from landfills with models, the relative strengths of measurements with mind to the many limitations inherent to their use. Implications noted from the *Analysis & results* should now be understood more fully regarding why the original estimate reported for a respective regional or national inventory, was incorrect in the first place.

3. Methodology

3.1. Research design

This study is constructed using both quantitative and qualitative methods, in a sequential format so as to enhance our understanding regarding a single, critical environmental impact of interest; landfill methane emissions (Tashakkori & Teddlie 2003). This study can be classified as an 'explanatory' mixed methods research study, for the fact qualitative data is used to enhance comprehension of a trend evident within quantitative data for landfill methane emissions (Cresswell & Clark 2007). These efforts include a novel framework for modeling landfill methane emissions estimates at individual sites, based on estimates generated by three T-D studies (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al 2018). The research design, including all elements described here, is displayed below in Figure 3.

The decision to employ a mixed methods approach is largely symptomatic of the problem itself. Landfill methane emissions appear quantified in the form of estimates, from both T-D & modeled inventory estimates. Quantitative estimates are affected by inherently more qualitative attributes. These attributes include cover material, on-site LFG collection, as well as more abstract indicators including the model chosen, especially considering the specific parameters included and how they are described. Such parameters include for example soil cover methane oxidation (OX) and landfill gas collection efficiency (CE). Thus, the decision to follow a mixed methods approach is justified for use in the context of landfill methane emissions for several, more conceptual reasons. These include the fact mixed methods research enables addressing both exploratory and confirmatory questions via a single 'research inquiry,' (Venkatesh et al. 2013).

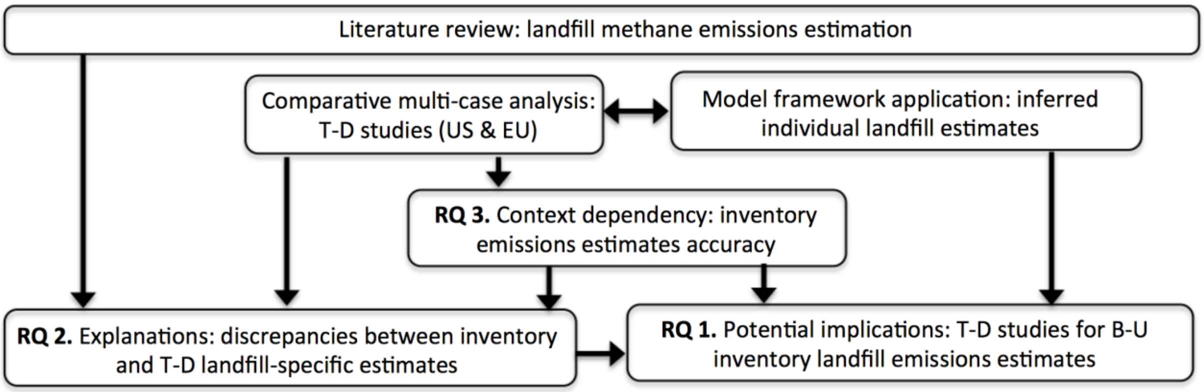

Figure 3. Research design

Source: (Venkatesh et al. 2013)

The 'confirmatory' research question, is with regard to whether the frequency of documented discrepancies between T-D and landfill-specific inventory reported estimates is context dependent between the US and EU. This mainly requires quantitative data analysis; comparison of results from T-D studies within different contexts and reported inventory estimates for the same landfills. 'Exploratory' questions include the remaining two stated research questions. These regard how B-U inventory estimation and reporting methods could be impacted by such discrepancies, and why or why not discrepancies appear at all. Attempting to offer responses to this second set of questions relies mostly on qualitative information, mainly supplied via the *Literature review*. Answering why discrepancies may appear evident and how they implicate inventory compilation methods is decidedly more worthwhile, if done through enhanced characterization of inventory emissions estimates themselves. Furthermore, connections are made through examining research used to establish the B-U methods in question, including the modeling approaches currently utilized to estimate landfill methane emissions.

The model framework is applied to draft independent methane emissions estimates for individual landfills. All figures generated are based on singular estimates generated as a feature of three T-D studies conducted in the US (Pesichl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018). The framework allows for depicting potential findings of further T-D research to evaluate inventory landfill methane emissions estimates, particularly from the EPA GHGRP. Provision of the framework and results of its application, is aimed at helping to provide additional information for the latter two stated research questions.

The overarching goal of this study is to 'leverage' findings from the *Literature review* regarding landfill methane emissions, to add overall richness to analysis of quantitative landfill methane emission estimates provided by T-D studies (Venkatesh et al. 2013). This entire inquiry stems from an inductive line of reasoning. An observed pattern leads eventually through several steps, to theorizing; using established trends, "to predict the unknown," (Heit 2000). The final product is a multiple case study including an applied model framework, informed by a literature review, all to characterize the phenomena itself; landfill methane emissions estimation.

The two case studies included in the *Analysis & results* both have extraordinarily large geographic scopes (the entire US & EU). This is justified by the fact so few studies actually fit the criteria of T-D inclusive of landfill-specific estimates, within each. A central tenet of this mixed methods multiple case study is comprehensive collection of all relevant data, as Yin (1994) prescribes. So while the scope appears massive, it should be viewed as a 'catchment', in that it provides a filter to narrow down existing studies capable of producing data for quantitative analysis. This ultimately leaves 55 T-D studies contributing to the *Analysis & results*, US and EU combined.

Beside the apparent inductive line of reasoning inherent to this study of landfill methane emissions, there is also a retroductive element. This is a side effect of the nature in which the phenomena itself was observed. Although this study appears sequential, in which literature review informs quantitative analysis, the former is actually conducted in order to help explain regularities witnessed from the latter (Blaikie & Priest 2019). That said, the format of this paper still follows the general procedure of a sequential mixed methods study.

Qualitative evidence from the *Literature review* describing landfill methane emissions and emissions estimates, informs quantitative analysis of figures supplied by T-D studies and application of a model framework (Tashakkori & Teddlie 2003, Casebeer & Verhoef 1997).

Although inductive reasoning is typically meant to generate 'grounded theory' (Glaser 1992), this study stops short of definitive explanation beyond forming tentative possibilities to answer 'why' questions regarding its findings. Conclusions drawn from this study are meant as 'meta-inferences' (Venkatesh et al. 2013). These appear in various forms. In all cases meta-inferences are, "inferred from an integration of findings from quantitative and qualitative strands of mixed methods research," (Venkatesh et al. 2013). Thus, this research inquiry is designed as a mixed methods multi-case comparative study with the rigor of any study owed to inductive reasoning, in terms of data collection. However, the end goal is meta-inference generation for answering stated research questions, rather than more typical theory generation.

The model framework is applied in three different settings using individual T-D methane emissions estimates from landfills within each, supplied by three US-based T-D studies included in the multi-case study analysis (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018). Two separate emissions estimates are generated for three whole states, and the largest metropolitan area in the US - inclusive of a combined 124 landfills. Separate emissions estimates are also generated for three of these regions, which include only landfills reporting two different estimates from different equations. This allows for comparison between equations, the estimate actually reported, as well as both inferred estimates from the model framework. The areas chosen for applying this framework correspond to the T-D studies noted above. These include the South Coast Air Basin (SoCAB) encompassing the Greater Los Angeles area of California (Peischl et al. 2013), the state of Indiana (Cambaliza et al. 2015), and the states of Maryland and Virginia (MD/VA) combined (Ren et al. 2018).

These three areas selected are each vastly different in terms of geographic location, climate, and population (AQMD 2020, US Census Bureau 2019).

The model framework offers a hypothetical alternative to current inventory landfill methane emissions estimates. Its application is based on limited T-D information rather than limited information typically used for B-U modeling. The estimates inferred for SoCAB, Indiana, and MD/VA are not used to draw conclusions from regarding the inventory-reported estimates' accuracy. Rather, the estimates, and vast amount of data collected and analyzed toward their generation, directly enhances capabilities granted this work to draft more meaningful, rich meta-inferences toward answering the three stated research questions.

3.2. Data collection

Data was collected primarily from exhaustive analysis of 55 T-D studies conducted within the last two decades (2000-2020). Any studies that generated landfill-specific emissions estimates, alongside the general T-D estimate offered for the region included within each respective scope, are especially relevant. Both landfill-specific T-D estimates, where available, as well as the overall T-D estimates, were compared with inventory estimates for corresponding regions and/or landfills. These figures were taken largely from the published materials, though in several cases outside reference was required either to enable additional comparisons, or to allow comparison at all (Peischl et al. 2013, Cambaliza et al. 2015, Johnson et al. 2014, Jeong et al. 2017).

For a number of studies included for the US case, as well as for application of the model framework, EPA GHGRP data was retrieved online. This data is used for comparison with inventory values referenced by T-D studies. EPA GHGRP-reported data on WIP and reported emissions estimates for individual was also collected for model framework application. The EPA GHGRP does not include every landfill in its estimate for regional totals. Values reported for entire regions through the EPA GHGRP are still valid for the fact they represent the total emissions from landfills modeled to generate the most amount of methane, which is conveyed to the public in an available online format. The EPA estimates 89% of waste disposed of in US landfills, or roughly 74% of US landfill methane emissions, are accounted for through the GHGRP (Schmeltz 2017a).

The *Literature review* supplies evidence for positing meta-inferences to explain discrepancies observed in data from T-D studies and inferred estimates from model framework application. Although in the scheme of this mixed methods study, this evidence is considered qualitative data, this delineation is largely subjective and ultimately the purpose of their inclusion is for adding richness and depth to findings from *Analysis & results*.

3.3. Data analysis

Analysis done for the quantitative data accumulated from T-D studies was cursory and largely a work of compilation and conversion of figures expressed in various formats. The purpose of conversion was to somehow standardize expressed rates of methane emissions from entire study areas, as well as from specific landfills. This is in order to allow for more convenient comparison between inventory and measured values. It is anticipated this will help the scientific community in future comparisons and model development. There are limitations to conducting such conversions in certain cases, which are elucidated in the following subsection; *Limitations*.

Applying the model framework relies upon emissions estimates provided for 18 individual landfills, part of larger T-D studies, using aircraft mass-balance (AMB) methods (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018). Two different rates are created to infer potential methane emissions from landfills in relative proximity to those the T-D studies supply individual estimates for. One is able to quantify an emissions rate: t CH_4/t WIP, as the quotient of the mean annual emission rate, divided by the total amount of waste disposed of in the landfill throughout its history. The average rate, t CH_4/t WIP, is gathered from every landfill emissions estimated for in each T-D study. This rate is then applied to infer an independent methane emissions estimate for any individual landfill for which EPA GHGRP data exists from the specific year in question including reported WIP. EPA GHGRP data used for each area corresponds to the year AMB measurements were taken.

The second rate used for comparison with EPA GHRP-reported values is a difference-rate multiplier applied to the EPA GHGRP-reported value for a given individual landfill. The quotient is found for each individual landfill in each scenario, for which a T-D AMB emissions estimate is generated, divided by the EPA GHGRP-reported value. The average from each scenario becomes the multiplier applied to all other landfills within the area. A new total landfill methane emissions estimate is once more produced for all three scenarios.

Individual landfill methane emissions estimates were used for 2 landfills in SoCAB (Peischl et al. 2013), 5 landfills in Indiana (Cambaliza et al. 2015), and 11 in the Greater Baltimore-Washington D.C. (Ren et al. 2018). The two rates were applied to the other 26 landfills within SoCAB that reported emissions to EPA GHGRP for the year Peischl et al. (2013) conducted AMB measurements (2010). The two rates derived from results of Cambaliza et al. (2015), were applied to every landfill in Indiana that reported its emissions to the EPA GHGRP for 2011 (35), the year airborne measurements were originally taken. Ren et al. (2018) generates independent emissions estimates for landfills in both Maryland and Virginia (MD/VA). The average rate (t CH_4/t WIP) and multiplier is applied to every landfill emissions reported for in both states corresponding to the year airborne measurements were completed (2015).

All landfill estimates offered by Peischl et al. (2013) and Ren et al. (2018) include uncertainty ranges. Thus for both, separate upper and lower bound rates (t CH_4/t WIP) and multipliers are also calculated based on these ranges, and are applied to the rest of the landfills reported for SoCAB and MD/VA, respectively. Cambaliza et al. (2015) do not provide uncertainty ranges for the 5 Indiana landfills for which AMB-based estimates are used. Therefore, upper and lower bound rates are reached by applying a default 30% error margin to T-D estimates for the 5 original landfills. A 30% uncertainty margin is noted in relevant literature as a potential upper bound for uncertainty of individual estimates using AMB methods (Cambaliza et al. 2014). While a 30% uncertainty margin is valid in the context of Indiana, results of Ren et al. (2018) indicate uncertainty of AMB estimates of landfill emissions could be significantly higher.

Under EPA Reporting Protocol, any landfill with installed active LFG recovery must include separate emissions estimates from two different equations; HH-6 and HH-8. HH-6 is based on the IPCC FOD Method and formerly included a default 10% value for OX (EPA 2019). Since 2013, landfills have had the opportunity to change the default OX value used to estimate emissions. As of 2017, a default OX value of 25% was used for GHGRP reporting by 52.1% of landfills using equation HH-6, and 41.2% of landfills using HH-8 (EPA 2019). While both HH-6 and HH-8 require LFG recovery data, HH-8 relies on it for the equation's use and generally produces lower emissions estimates (Bogner 2020, personal communication). Estimates from both, when available, are visible in the GHGRP Facility Level Information of Greenhouse Gases Tool (FLIGHT). Thus, two additional estimates are calculated from the reported emissions from HH-6 and from HH-8, separately. SoCAB is the only scenario for which every landfill reports two estimates to the EPA GHGRP. Thus, the total EPA-reported values from HH-6 and HH-8 for Indiana and MD/VA are the sums of the portion of landfills that do include both estimates in their report for the EPA GHGRP. Corresponding values are computed for the other three categories of estimate achieved from this framework (t CH_4/ t WIP rate, multiplier, and EPA GHGRP-reported) from the same smaller samples in both Indiana and MD/VA. This permits valid comparison separate from the totals calculated statewide in both Indiana and MD/VA.

3.4. Limitations

A great limitation of this study is the lack of an actual self-constructed T-D methane emissions estimate. T-D studies referenced and used for the purposes of establishing findings in this study generally required highly sophisticated methods inclusive of technical expertise. This type of study would not be feasible, given several practical limitations related to funding, time, labor, access, and indeed expertise. Most T-D studies are the product of intensive measurement campaigns, involving use of piloted or more recently, even unmanned aircraft (Cusworth et al. 2020). T-D studies are often funded by large, established research organizations including the Environmental Defense Fund (EDF), National Oceanic and Atmospheric Administration (NOAA), and the National Aeronautics and Space Administration (NASA, Wennberg et al. 2012, Wecht et al. 2014b, McKain et al. 2015).

Another major limitation is the lack of review of methods used for each individual study from which data is mined for reference in the *Analysis & results*. Overall uncertainty of emissions estimates is not evaluated on individual, "study-by-study", basis, but instead should be generally regarded as variable. Generally, this work is meant more to fill a gap. Few existing studies accurately portray the uncertainty of accepted and reported landfill methane emissions inventory values via characterization of estimation methods and exploration of alternative T-D estimates.

Conversions made to T-D estimates provided, as well as inventory figures, to standardize units used for their expression, could lead to slightly greater uncertainty than is initially accounted for. Teragrams methane per year (Tg CH_4/y) are used to describe all emissions estimates provided in the analyzed studies, both T-D and inventory reported, and are unanimously rounded to the third decimal place. Uncertainty owed this choice can be considered negligible for the fact relevant discrepancies are generally large enough to show clear doubling or tripling of the inventory reported estimate (Manning et al. 2011, Jeong et al.

2013, McKain et al. 2015, Ren et al. 2018). Ultimately Tg CH_4/y is a useful unit for this type of study, since it allows for easy comparison with anthropogenic methane emissions estimates generated for entire countries, continents, or even studies to constrain global anthropogenic source totals (Miller et al. 2013, Turner et al. 2015, Maasakkers et al. 2016, Bergamaschi et al. 2009, 2018). As an exception, several figures referenced instead use gigagrams methane per year (Gg CH_4/y) to describe emissions from individual landfills. Contextually this is reasonable given the same figures expressed in Tg CH_4/y would appear miniscule and thus the standardization of units would no longer serve a useful purpose. Figure 5 specifically is another exception to this rule, and expresses its values in tons methane (t CH_4).

Other limitations are related to methodological choices. Primary among this strand is the nature of 'conclusions' drawn. These are 'meta-inferences' rather than actual theories, as typical inductive reasoning would seek to generate (Venkatesh et al. 2013, Glaser 1992). The greatest weakness of providing meta-inferences, rather than a theory for why discrepancies are evident between T-D studies and inventory values for landfill methane emissions, is inability for clear validation. Validation is difficult in any mixed methods study (Venkatesh et al. 2013). Validity of findings is more a product of inference quality, related to the rigor of the experiment, and the research design itself (Venkatesh et al. 2013). Thus, all conclusions drawn are a product of inferring and are not meant as definitive answers to 'why' questions.

A number of limitations are specifically related to the model framework application. Several are symptomatic of using T-D AMB-based landfill-specific methane emissions estimates. T-D studies, including those used for applying the framework (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018), must somehow overcome homogeneity of sources that affect measured emissions (Spokas 2020, personal communication). Furthermore, T-D methods including AMB must somehow make up for a lack of spatial resolution, as a result of measuring emissions downwind of major sources. This is especially relevant for landfills given the existence of hotspots through which a majority of methane is emitted, as well as general temporal variability of flux due to a number of confounding factors (IPCC 2006, Oonk 2010, Oonk 2012, Börjesson et al. 2007). These are among other limitations, which create potential for gross over or underestimation of landfill methane emissions estimated using T-D methods (Spokas 2020, personal communication). One true advantage of T-D estimates for landfills methane emissions is to provide an upper limit estimate for methane emissions (Spokas 2020, personal communication).

Given only two landfill estimates provided by Peischl et al. (2013) are used to generate inferred estimates for the 26 other landfills in SoCAB, overall representativeness may be limited. Moreover, the two landfills (Olinda Alpha and Puente Hills) are among the most highly engineered and controlled landfill systems in the US, perhaps the world (Spokas 2020, personal communication). In a sense this could add legitimacy for using T-D observed methane emissions estimates to infer emissions from other, less well-managed landfills, by limiting possible overestimation. By the same token, this could also result in actual underestimation of methane emissions from inferred T-D estimates, by wrongly representing this same class of less managed landfills. It could also lead to wrongful comparison with T-D inferred emissions for the other scenarios; Indiana (2011) and MD/VA (2015). It is assumed here the lack of estimates for SoCAB is somewhat made up for by the fact the T-D observed methane emissions estimates agree with reported state inventory estimates from CARB (Peischl et al. 2013).

Cambaliza et al. (2015) do not divulge the identities of 4 of 5 landfills T-D estimates generated, which are used for application to the Indiana scenario. Information provided for each includes a designated abbreviation ("name"), total landfill surface area and the county in which the landfill is located (Cambaliza et al. 2015). Through comparison of these data with corresponding data from the EPA GHGRP-reported Indiana landfills for 2011, the identities of these landfills are safely assumed (see Table 1). Furthermore, for each of the four Indiana counties, only one landfill actually reports methane emissions to the EPA GHGRP for 2011 (EPA 2020). Regardless, concealing of the true identities of these four landfills in the study (Cambaliza et al. 2015), means all inferred methane emissions estimates are made assuming estimates used to the derive the applied rate and multiplier, are actually from the four landfills mentioned.

Table 1. Evidence supporting assumed identities of landfills for Indiana scenario

Cambaliza et al. (2015)			**This study (EPA GHGRP)**		
Landfill surface area (x1000 m^2)	**County**	**Abbrevi-ation**	**Waste disposal area (x1000 m2)**	**County**	**Name**
702	Hendricks	TBLF	701.781	Hendricks	Twin Bridges
795	Newton	NCLF	855.919	Newton	Newton County
455	Randolph	RFLF	480.767	Randolph	Randolph Farms
324	Shelby	CLF	337.901	Shelby	CGS Inc.

Sources: (Cambaliza et al. 2015, EPA GHGRP)

Calculating a rate (t CH_4/t WIP) requires available data on the amount of waste disposed of annually in every landfill for which T-D emissions are inferred, as well as for the landfills from which the average rate for each study area is generated. US landfills that generate more than 25,000 t CH_4/year are required to report emissions as well as several other data points to the EPA GHGRP. These data include quantity of disposed waste. This allows for calculating total WIP at an individual landfill for application of the model framework. However, the EPA GHGRP was only established in 2010, and all data regarding the amount of waste disposed of before then is subject to scrutiny regarding overall representativeness. Uncertainty is manifest if one views data reported to the EPA GHGRP from certain landfills that report identical amounts of waste disposed of every year prior to 2010 (see Table 2). The EPA requires data regarding the year a landfill started receiving waste, its closure year, and either the WIP or Waste Acceptance Rate in order to estimate WIP (Schmeltz 2017a). More than 400 landfills were missing two or more of these data points and WIP could therefore not be estimated. More than 250 landfills (15%) were missing one data point and the EPA 'force filled' the data to account for the missing information (Schmeltz 2017a). The latter is likely the source of such scenarios as depicted in Table 2. Therefore, inferred emissions based on individually calculated rates (t CH_4/t WIP) are also subject to scrutiny due to uncertainty of reported waste disposal quantities reported to the EPA GHGRP.

Table 2. Newton County Landfill (IN) GHGRP-reported annual waste disposal

Year(s)	**1995-2010**	**2010**	**2011**
Amount (t WIP)	1,614,726	2,689,515.203	2,501,559.224

Source: (EPA GHGRP)

This study is unable to validate or evaluate the quantified totals using conventional means, namely via further T-D studies. This is for the same reasons mentioned to explain why no actual T-D study is included; costs, time, labor, expertise, access. However, the value of inferred methane emissions estimates generated using this model framework does not lay in the accuracy of figures, but in its application. The framework is itself reliant only on existing T-D individual landfill methane emissions estimates. This is largely related to the aim to generate 'meta-inferences' regarding methods used for generating inventory-reported B-U landfill methane emissions estimates in the US. Accomplishing this does not necessarily include definitive evaluation of individually reported landfill methane emissions estimates. Figures expressed do help to grant this ability for drafting meta-inferences. However, this is not a focus but a byproduct of applying a useful framework that holds potential for offering a novel way of drafting alternative landfill methane emissions estimates.

3.5. Ethical considerations

There are several details originating from how this study was begun that should be made transparent for ethical consideration. The initial research that ultimately resulted in this study was conducted while I was an intern with the Sustainability Department of Covanta Holding Corporation. Covanta is primarily a waste-to-energy utility. I am now contracted to begin employment with Covanta, within a different department from the one I interned with, beginning July 2020. This employment was made without any regard to the specific nature of my thesis work beside the academic merits it provided as credit to my ability.

Advising from Dr. Marco Castaldi of City College of New York, is the result of a connection made via my former supervisor at Covanta, with whom Dr. Castaldi maintains a professional relationship for supporting a piece of the research he conducts. All of that said, I did not receive funding of any kind for completing this study, nor any career incentive, either from Covanta or Dr. Castaldi. All of the advice I received was purely academic, for the benefit of this research, which is pertinent for the findings it delivers regarding regulator reporting of landfill methane emissions. Finally, this work is a result almost entirely of my own work and diligence. The flaws inherent to the methods selected for analysis, as well as any conclusions reached, are a result of decisions I made as the author. In no way was I at any point encouraged to alter any findings to suit my own or anyone else's assumptions, either while an intern with Covanta, or in the time since.

4. Analysis & results

This section includes results from analysis and comparison of data collected from 55 T-D studies from within the US and EU. Figures and details from all 55 studies are not included in this section. For data from all T-D studies included in the US case, see Appendix Table A1, and for the EU Case, please see Table A2. Regarding the substance of this section and all subsections contained herein, results are primarily expressed quantitatively. That said, in line with the mixed methods approach of the entire study, qualitative details are offered as well, especially when necessary or relevant to better understanding of the choice topic; landfill methane emissions. Still, the majority of "answers" posited for the stated research questions, in the form of meta-inferences (Venkatesh et al. 2013), appear first in the *Discussion* section, including qualitative features of any evidence supporting their foundation.

T-D studies are designed as a means for generating estimates of methane emissions from areas of varying size, based on actual atmospheric measurements. This is as opposed to models, typically designed for calculating emissions by source in B-U emissions quantification schemes. Of those T-D studies used as part of this study for data collection and analysis, a variety of different approaches are taken by researchers to eventually reach emissions estimates. In some cases, T-D studies include separately generated emissions estimates for the landfill(s) specific to their scope. Despite differences inherent to the actual approaches taken to reach these estimates, all are considered within the larger umbrella of T-D studies. Beneath this "umbrella", there are several subcriteria to further classify T-D studies according to how measurements are taken, as well as the methods chosen for deriving actual emissions estimates from measurements. The latter is especially important to consider. This is less due to any need to categorize T-D studies - but because it is important one grasps T-D studies also include some assumptions. Furthermore, T-D estimates should not be unilaterally considered the truest depiction of reality simply because they include in-situ measurements as a basis for establishing emissions estimates.

Limitations related to assumptions inherent to all T-D studies are quite different from B-U efforts. As has already been discussed, the main limitation of B-U estimates is reliance on models with uncertain methods for establishing emissions estimates. These models contain most of the assumptions limiting overall accuracy for B-U estimates. Namely, that the amount of waste disposed in a landfill clearly indicates how much methane it can emit, which is impacted by several crucial factors including CE. T-D studies are both labor and cost intensive, and often struggle to account for overall variability of emissions within a respective study area, temporally especially. Estimates generated from T-D studies can be the product of only several days of measurements, sometimes even a single day (Ryoo et al. 2019). Furthermore, a large sample of studies use an inversion model to generate an annual emissions rate estimate from measurements, without attaining continuous measurements over the course of an entire year (Wennberg et al. 2012, Jeong et al. 2013, Manning et al. 2011). Inversions, although based on detailed quantitative information, are themselves assumption-laden, in order to somehow account for seasonal, as well as finer resolution temporal variability of methane emissions.

T-D studies that generate landfill-specific emissions estimate, of special significance, in some cases do so via measurements taken separate from those used to generate an overall emissions estimate (e.g. Lamb et al. 2016). In such cases, uncertainties inherent to measuring methane emissions from individual landfills, explored within the *Literature review*, must also be considered. This is all to say T-D studies and the estimates generated, including those referenced in this work, while based on measurements, cannot be considered ubiquitously "superior" for accuracy without fully considering their own relative uncertainties.

This work is not meant to specifically evaluate individual methodological approaches of studies that fall beneath the umbrella of T-D studies generating methane emissions estimates. Nor does this study evaluate methods chosen for the T-D studies used for data collection and analysis. *Analysis* is the product of compiling and analyzing quantitative data in the form of T-D methane emissions estimates, as well as those reported in corresponding national or regional GHG inventories. What follows is a detailed overview of the results of T-D studies conducted within both the US and EU, inclusive of specific emissions estimates generated for landfills.

4.1. Case study: US top-down studies

Media attention surrounding one T-D study encompassing six major US cities was the original impetus for the research supporting this entire work (Plant et al. 2019, Perkins 2019, Harvey 2019). The original research question regarding the findings of Plant et al. (2019) was confirmatory: are there any implications for landfills in the region observed for this study? From that point, the study presented here was developed, mainly as a result of observing a pattern. T-D studies within the US seemed to contradict reported inventory estimates for cities, regions, and even the entire nation (see Figure 1). The emphasis of most of these studies, including Plant et al. (2019), is unaccounted for fugitive emissions from the oil/natural gas (O/NG) sector (Alvarez et al. 2018, McKain et al. 2015, Karion et al. 2015, Kort et al. 2014, Brandt et al. 2014). However, several studies implicate landfills (Ren et al. 2018, Jeong et al. 2016, Johnson et al. 2014, Peischl et al. 2013). Thus, the essential research question has remained mostly the same throughout the process. Results presented here are primarily in response to an updated version of that simple confirmatory inquiry.

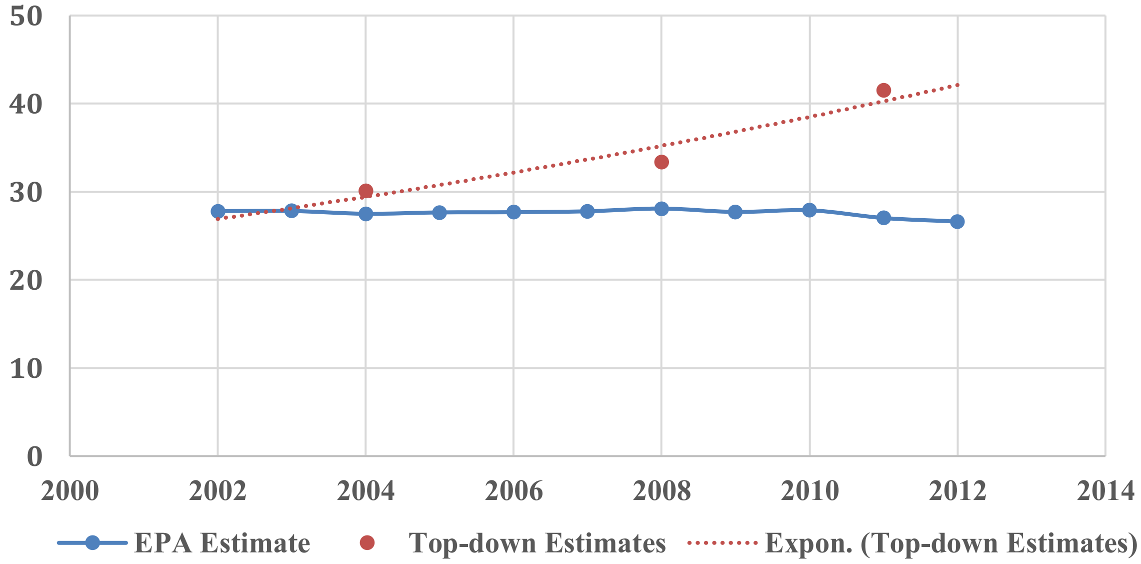

Figure 4. US anthropogenic methane emissions (Tg CH_4)

Sources: (Turner et al. 2016, Goodfriend et al. 2017, Turner et al. 2015, Wecht et al. 2014, Miller et al. 2013)

T-D studies conducted in the US do not unanimously implicate landfills, even if portraying overall underestimation of methane emissions in inventories (Plant et al. 2019, McKain et al. 2015, Wunch et al. 2016, Zhao et al. 2009). However, there is a fairly large selection of T-D studies conducted in the US that do hold implications for landfills - a third of those used for this study (11 of 33 total). Furthermore, results of analyzing 33 T-D studies conducted within the US can be summarized: indication of general underestimation of methane emissions in reported inventories, with limited incidence of underestimated landfill-specific emissions as well.

While the importance of T-D studies for evaluating overall accuracy of inventory methane emissions estimates is apparent from these findings, such is not the expressed topic of this study. The fact T-D studies conducted in the US indicate underestimation of methane emissions overall, validates exploring instances of landfill emissions implicated therein. This is because landfill emissions are reported without atmospheric measurements informing their estimation (EPA 2019). Peischl et al. (2013) conducted several flight campaigns in SoCAB, which includes Los Angeles (LA). Their results culminated in an overall methane emission estimate, as well as individual estimates for two landfills. Peischl et al.'s (2013) results, when compared with the statewide inventory compiled by the California Air Resources Board (CARB), indicate overall underestimation of methane emissions in SoCAB; 0.411 ± 0.37 Tg CH_4/y from measurements compared with 0.301 Tg CH_4/y from CARB's inventory.

The combined landfill-specific emissions estimate for both landfills measured from Peischl et al. (2013), are in agreement with values reported in CARB's inventory (0.047 ± 0.013 Tg CH_4/y and 0.050 Tg CH_4/y, respectively). However, this landfill-specific estimate (0.047 ± 0.013 Tg CH_4/y) from Peischl et al. (2013) is significantly larger than the combined reported value from the EPA GHGRP (0.031 Tg CH_4). These and all other discrepancies observed between T-D and B-U EPA GHRP inventory-reported landfill estimates are displayed in Table X, below. While a third of T-D studies used that were conducted in the US do implicate landfills in some way, not all observed implications include indication of inventory-underestimated methane emissions. In this case CARB's inventory appears to have adequately constrained landfill emissions according to the results from Peischl et al. (2013). The same cannot be said for the EPA GHGRP. Peischl et al. (2013) concluded the main source of the discrepancy between their overall methane emissions estimate for SoCAB and CARB's, is unaccounted for losses in the O/NG infrastructure for LA. Cui et al. (2015) support these findings, estimating combined emissions from landfills and O/NG for SoCAB in excess of CARB's inventory reported estimate. However, this speculatively due more to fugitive emissions from O/NG infrastructure than from landfills.

In contrast to findings for SoCAB, Jeong et al. (2013) calculate a mean statewide California landfill methane emissions estimate more than double CARB's (0.687 ± 0.387 Tg CH_4/y and 0.314 Tg CH_4/y). Jeong et al. (2013) used continuous measurements from a tower network situated in Central California. The EPA GHGRP reports methane emissions from landfills in California for both years Jeong et al. (2013) conducted their airborne measurements (2010 & 2011) equal to 0.319 Tg CH_4 (113 facilities) and 0.341 Tg CH_4 (115 facilities), respectively. Similar to CARB's statewide inventory, these values drastically underestimate emissions according to the T-D estimate from Jeong et al. (2013). The location of the tall towers is cited as potentially leading to a lower overall emissions estimate, due to unaccounted for landfill emissions from coastal urban areas. This notion is supported by findings from Wecht et al. (2014a), which estimates total California landfill emissions from airborne measurements of 1.05 Tg CH_4/y compared with CARB and the EPA's (GHGRP) corresponding inventory values of 0.390 Tg CH_4/y and 0.319 Tg CH_4(112 landfills), respectively. In line with their landfill-specific estimate, Wecht et al. (2014) estimate total California emissions nearly double the CARB inventory reported value (2.860 ± 0.210 Tg CH_4 /y and 1.510 Tg CH_4/y, respectively).

Table 3. Discrepancies between T-D & EPA GHGRP inventory landfill-specific estimates

Study	Scope	EPA GHGRP-reported Value (Tg CH_4/y)	Measured Value (Tg CH_4/y)	Difference Factor
Peischl et al. (2013)	SoCAB	0.031	0.047 ± 0.013	1.1-1.9x
Wecht et al. (2014)	California	0.319	1.05	3.3x
Johnson et al. (2014)	California	0.319	0.821	2.6x
Jeong et al. (2016)	California	0.310	0.435	1.4
Jeong et al. (2017)	San Francisco Bay Area (SFBA)	0.054	0.116	2.1x
Cambaliza et al. (2015)	Indianapolis	0.014	0.023 ± 0.007	1.1-2.1x
Ren et al. (2018)	Baltimore/DC	0.035	0.065 ± 0.038	0.8-2.9x
LF Average				**1.8-2.3x**

Sources: described in table

A contemporary study echoes findings of Wecht et al. (2014) regarding the same version of CARB's annual GHG inventory. Johnson et al. (2014) estimate statewide California emissions of 1.930 Tg CH_4/y, 0.821 Tg CH_4/y from landfills, compared with CARB inventory values (again equal to 1.510 Tg CH_4/y and 0.390 Tg CH_4/y). The same landfill methane emissions value reported to the EPA GHGRP mentioned above (0.319 Tg CH_4) are used for comparison with figures from Johnson et al. (2014). A later study supports these findings regarding statewide overall emissions in California. Jeong et al. (2016) estimates emissions of 2.420 ± 0.490 Tg CH_4/y compared with an updated CARB value of 1.650 Tg CH_4/y. Jeong et al. (2016) also indicate underestimation of landfill emissions, though to a lesser degree than previous studies; estimating statewide landfill emissions of 0.435 Tg CH_4/y compared with a CARB inventory value of 0.335 Tg CH_4/y. Comparatively, the EPA GHGRP reports California statewide emissions for the two years measurements are used by Jeong et al. (2016) equal to 0.310 Tg CH_4 from 116 landfills (2013) and 0.306 Tg CH_4 from 117 landfills (2014).

Regarding specific California urban municipalities, the trend from T-D studies conducted for SoCAB (Peischl et al. 2013, Cui et al. 2015) does not apply to the San Francisco Bay Area (SFBA). Fairley & Fischer (2015) do not express a specific value for estimated SFBA landfill emissions. That said, Fairley & Fischer (2015) attribute a significant portion of the discrepancy observed between their T-D estimate (0.24 ± 0.06 Tg CH_4/y) and a regional inventory compiled by the Bay Area Air Quality Management District (BAAQMD, 0.126 Tg CH_4/y), to landfills. Jeong et al (2017) quantitatively express this finding in their study, observing landfill emissions of 0.116 Tg CH_4/y compared with a BAAQMD inventory-reported value of 0.068 Tg CH_4/y. Total landfill methane emissions reported to the EPA GHGRP from the nine counties included in the SFBA for the year Jeong et al. (2017) conducted measurements (2015), is about 0.054 Tg CH_4 (19 facilities). Likewise, Jeong et al. (2017) estimate vast underestimation of overall emissions for the SFBA in BAAQMD's inventory; estimating total emissions of 0.225 ± 0.051 Tg/y compared with 0.126 Tg CH_4/y. A total value of all methane emissions for the SFBA (2015) is available via the EPA GHGRP, however this number does not include emissions from a number of sources included within the O/NG sector, and so is not included in these results.

Elsewhere in the US besides California, there are far fewer T-D studies that implicate inventory landfill methane emissions estimates. Of those used for this study, three are concentrated in Indianapolis; capital city of the state of Indiana. Cambaliza et al. (2015) used AMB methods to quantify methane emissions from the entire city, and from the Southside Landfill (SSLF) specifically. Their estimate for SSLF (0.023 ± 0.007 Tg CH_4/y) significantly exceeds the value reported to the EPA via the GHGRP (0.014 Tg CH_4). Notably, the inventory value expressed here is not included directly in Cambaliza et al. (2015), but was taken from the EPA GHGRP website (2020).

Lamb et al. (2016) also generate T-D methane emissions estimates for Indianapolis, including a SSLF-specific estimate. Despite indicating inventory underestimation (0.041 ± 0.012 Tg CH_4/y and 0.029 Tg, respectively), Lamb et al.'s (2016) SSLF-specific emissions estimate is in agreement with the corresponding value from the EPA GHGRP (0.0145 ± 0.007 Tg CH_4/y and 0.015 Tg CH_4, respectively). Heimburger et al. (2017) generated their own overall Indianapolis and SSLF-specific estimate, though the latter is inferred from a percentage of the total estimated by Cambaliza et al. (2015). Similar to Lamb et al. (2016), Heimburger et al. (2017) find general agreement between their T-D estimate for the SSLF (roughly 0.016 Tg CH_4/y) and the EPA GHGRP-accepted value for the same year measurements were conducted (2014, roughly 0.0184 Tg CH_4).

From the three T-D studies for Indianapolis discussed here, it appears evident more measurements are likely required for both SSLF and the entire area. Cambaliza et al. (2015) completed the only T-D study that generated an estimate for the SSLF that stands as significantly greater than the EPA GHGRP reported value, however they also report the only estimate generated from airborne measurements. Lamb et al. (2016) report an estimate reached via mobile-measurements taken from an automobile, the uncertainties of which can be potentially significant (Oonk 2010). Heimburger et al. (2017) generate an SSLF-specific estimate based on application of the percentage attributed to SSLF from the total, originally expressed by Cambaliza et al. (2015). Moreover, all three predict overall underestimation of methane emissions from Indiana in the inventory. Differences lie mainly in the source(s) to which the observed discrepancy is credited.

The final study discussed in this section was conducted on the East Coast of the US. Ren et al. (2018) conducted a number of airborne measurement campaigns to generate T-D emissions estimates for the Baltimore-Washington D.C. Area. Similar to several studies from California and Indianapolis, results indicate both overall and landfill-specific underestimation of emissions in the corresponding inventory (EPA GHGRP).

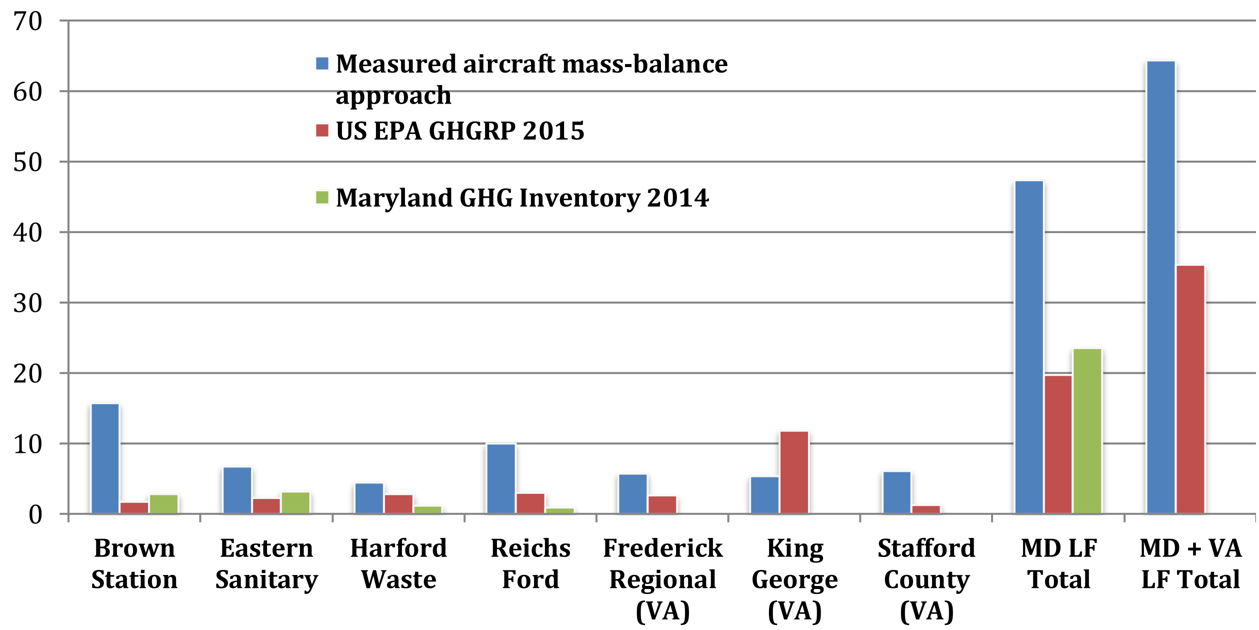

Figure 5. Baltimore-Washington D.C. landfill emissions estimates (2015-2016)

Source: (Ren et al. 2018)

Ren et al. (2018) conducted airborne measurements for 11 landfills within the study area. An overall landfill-specific estimate is generated based on these results, and an accompanying estimate that also includes individual estimates from 3 other landfills inferred from the measurements taken elsewhere (see Figure 2). The landfill-specific emissions estimate not including those three landfills (0.065 ± 0.038 Tg CH_4/y) exceeds the EPA GHGRP total (0.035 Tg CH_4) by a factor near 2. For one landfill in particular, Brown Station, Ren et al (2018) estimate emissions near 9 times that reported to the EPA GHGRP (see Figure 2). Moreover, considering only Maryland landfills, Ren et al. (2018) estimate emissions in excess of the EPA GHGRP-accepted total by a factor of roughly 2.5. These findings represent the largest observed discrepancy between T-D estimates for landfill methane emissions and reported inventory values, in this study.

Similar to the other discrepancies of this kind (regarding landfill methane emissions), Ren et al. (2018) likewise observe far greater overall methane emissions (0.278 ± 0.066 Tg CH_4/y) than reported in the corresponding inventory (0.097 Tg CH_4). Thus, roughly a third of the T-D studies used for the US case, implicate landfill-specific inventory emissions estimates. Of these, slightly less observe emissions significantly greater than inventory-reported values (Jeong et al. 2013, Wecht et al. 2014, Cui et al. 2015, Jeong et al. 2017, Johnson et al. 2014, Cambaliza et al. 2015). Ren et al. 2018 also indicate a large discrepancy between observed landfill methane emissions and EPA GHGRP-reported figures. Reasons potentially explaining the results of Ren et al. (2018), as well as other discrepancies or related findings are discussed further in the *Discussion* and *Conclusion & recommendations* sections.

Definitively, based on the T-D studies analyzed, landfill methane emissions do not appear in any way ubiquitously, or even usually, underestimated in reported inventories within the US. Rather, within certain contexts in the US, landfill methane emissions estimates are sometimes implicated in T-D studies. The meaning of these findings is more difficult to determine. Regardless, it would seem a foregone conclusion to attempt to say that current estimation and reporting standards using B-U methods are uniformly adequate in the US. This topic specifically is explored further here in the final subsection, as well as later in the *Discussion* and *Conclusion & recommendations* sections.

4.2. Case study: EU top-down studies

Presented here are all relevant findings concerning landfill methane emissions from 22 T-D studies conducted at varied scales within the EU. Although technically these studies are taken from the last 20 years (2000-2020), 20 of the total 22 T-D studies reviewed were published 2010 or later. Moreover, of the remaining two studies, only one study was published before 2005 (Kuc et al. 2003). All to say that the results presented here are truly representative of the period extending from 2009 to the present. This corresponds well with the studies used for the previous subsection concerning results from analyzing 33 T-D studies conducted in the US. Before elucidating the results in full, several considerations require attention.

The UK is at the time of this work's creation, no longer a EU Member State, though it was at the time every study mentioned in this work conducted in the UK, was published. Moreover, while political context plays an important role in comparing results of T-D studies conducted in the EU, with those from the US, it is not the sole focus. Brexit will not receive more attention except to say that it could theoretically prove influential on landfill methane emissions in the UK, considering the role EU Waste Policy has likely played in reducing actual emissions (Mønster et al. 2015). Furthermore, considering T-D studies from the UK reviewed for this work, several combined national estimates are discussed that include Ireland, which is both an independent state and a *current* EU Member State. This is meant to convey to the reader that different from B-U inventory estimation methods for methane emissions, T-D approaches are capable of ignoring borders and politics except for rationalizing the emissions estimates they generate from observations, as well as any discrepancies they find between these estimates and inventory counterparts.

It is also imperative the reader is informed that wholly different from the US, despite presence of 27 Member States in the EU, fewer T-D studies generating methane emissions estimates are available. Slighly limited data, especially in the form of T-D landfill-specific methane emissions estimates, does not necessarily diminish the value of these results, though it does affect associated findings. The nature of this effect is explored more in the next subsection and in the *Discussion* section.

A third crucial point to make concerning these results is with regard to the inventories referenced for comparison with T-D estimates independently generated. From results from analysis of US-based T-D studies, it is apparent in different contexts at different scales, different GHG inventories from different regulators (i.e. BAAQMD, CARB, and the EPA) are referenced, often simultaneously. This definitely limits the validity of drawing conclusions about the crucial inventory values, ultimately reported to national regulating bodies for compiling national GHG inventories for the IPCC (EPA in the US). This issue becomes even more prevalent in this subsection considering how many nations and regulating bodies exist, with some overlap, within the EU. All of this in mind, one should consider this entire work, as useful for enhancing overall comprehension of generic inventory methods for estimating landfill methane emissions, which regardless of source, are considered B-U approaches.

Concerning results of T-D studies conducted within the EU, 6 of 22 used do not generate an overall methane emissions estimate for their study area. Of 16 that do, 3 do not cite an inventory value for comparison, from any source whatsoever. Notably however, these constitute 3 of 4 T-D studies used for this work that were conducted in Poland, where inventory data below overall national estimates is still largely unavailable, despite the country's status as an Annex-I party to the UNFCCC (2020). Furthermore, 2 of 4 Polish T-D studies used for this work were conducted in Krakow (Kuc et al. 2003, Zimnoch et al. 2010), where a more recent study indicates agreement between its generated overall methane emissions estimate and a cited inventory from the European Database for Global Atmospheric Research (EDGAR, Zimnoch et al. 2018). Therefore it would be imprudent to suggest the previous emissions estimates for Krakow by Kuc et al. (2003) and Zimnoch et al. (2010) would likely indicate underestimation in the corresponding inventory, were such data available. Regardless, 13 of 22 T-D studies conducted within the EU contain both a generated overall methane emissions estimate for the study area, and a reported inventory value for comparison. However, only one study meets these criteria, and also generates a landfill-specific methane emissions estimate (Pison et al. 2018).

Pison et al. (2018) generate overall and landfill-specific national methane emissions estimates, for France, via tower-based measurements. When compared with the overall inventory national estimate (3.108 Tg CH_4/y) from a French regulator (CITEPA), their results (3.011-3.934 Tg CH_4/y) indicate relative agreement, especially observing the lower bound (3.011 Tg CH_4/y). These figures, as presented here, were altered to remove natural emissions (an estimated 0.259 - 0.559 Tg CH_4/y). Natural emissions are not included in the inventory, but are included in Pison et al.'s (2018) original results for overall measured methane emissions. Pison et al. (2018) depict overestimation of landfill methane emissions in the French national inventory (CITEPA) based on their own estimate from measurements (0.522 Tg CH_4/y and 0.460 Tg CH_4/y, respectively). Thus, the only quantitative figure expressed to account for landfill methane emissions from a given study area, from 22 T-D studies from the EU that are analyzed for this work, depicts actual overestimation in the corresponding inventory.

Findings from Pison et al. (2018) regarding landfill-specific methane emissions estimates, provide little from which to draw meaningful conclusions. That said, a lack of evidence is in some ways meaningful on its own, pending the question it is used to answer. Considering again, the 13 T-D studies that contain a generated overall estimate and a corresponding inventory estimate for comparison, only 3 indicate significant underestimation of emissions in inventories. This group includes Bergamaschi et al. (2015) predicted EU-wide underestimation of emissions in UNFCCC-reported national inventories. This, despite the fact the estimate from Bergamaschi et al. (2015) does not explicitly delineate natural emissions, which are cited as a potentially significant source of the discrepancy. A later study (Bergamaschi et al. 2018) quantifies natural wetland emissions in the EU, which when subtracted from the overall measured emissions, indicate agreement between inventory and measured figures for the entire EU (18.8-21.3 Tg CH_4/y and 12-21.4 Tg CH_4/y, respectively).

Wunch et al. (2019) provide another continental study with a slightly smaller scope stretching across France, Germany and Poland. Results depict overestimation in the corresponding inventories compared with their estimates based on continuous tower-based measurements (3.0 Tg CH_4/y and 2.4 ± 0.3 Tg CH_4/y, respectively). Four studies that generate overall methane emissions estimates for comparison with inventory-reported values, come from the UK & Ireland. Of these, 2 depict relatively significant underestimation in corresponding inventories. Connors et al. (2018) estimate emissions for East Anglia (UK, 0.311 ± 0.063 Tg CH_4/y) in solid agreement with the inventory total (0.278 Tg CH_4/y), which falls within the measured uncertainty range. The same is true of Helfter et al.'s (2019) combined estimate for the UK and Ireland excluding Scotland, and the combined national inventory totals (2.55 ± 0.48 Tg CH_4/y and 2.29 Tg CH_4/y, respectively). Helfter et al. (2019) based their estimates on measurements taken regularly along freight ferry regularly traveling between Belgium and Scotland along the East Coast of the UK.

Findings of Helfter et al. (2019) regarding combined UK and Ireland methane emissions are supported by two earlier studies. Ganesan et al. (2015) depict overestimation of emissions for the UK and Ireland from a tower-based measurement-derived estimate (roughly 2.71 Tg CH_4/y) compared with the combined inventory-reported value (2.995 Tg CH_4/y).

Manning et al. (2011) derive an overall estimate for the UK alone, which portrays overestimation when compared with the inventory value (2.429 Tg CH_4/y and 3.631 Tg CH_4/y, respectively).

Two T-D studies not yet discussed, which both indicate inventory underestimation of overall emissions, were both conducted in London (Helfter et al. 2016, Pitt et al. 2019). Pitt et al. (2019) describe an emissions estimate using an AMB approach they speculate may overestimate flux, which is higher than the reported value from a regional London inventory (0.092 Tg CH_4/y and 0.075 Tg CH_4/y, respectively). Pitt et al. (2019) do not explicitly mention source influences, however a 'qualitative' study by Zazzeri et al. (2017) notes likely unaccounted for methane emissions in London stem from O/NG infrastructure leaks. This is similar to findings of some T-D studies conducted in urban municipalities in the US (Plant et al. 2019, McKain et al. 2015). Helfter et al. (2016) also indicate underestimation in London's regional inventory (LAEI) based on their T-D estimate. These figures cannot be expressed in units comparable to findings from other studies given their original formatting in t/km^2/y (29 and 72 ± 3, respectively) and the fact no actual size of the study area is communicated. Helfter et al. (2016) speculate the observed discrepancy is potentially due to unaccounted for emissions from urban wastewater treatment (WWT) and commuting dynamics.

In summary, of 22 T-D studies conducted within the EU 2009-2020, 2 depict overall underestimation of methane emissions in reported inventories, both from London (Pitt et al. 2019, Helfter et al. 2016). Landfills are not cited in either case as a potential source of the observed discrepancy. Only one study generates a landfill-specific estimate in quantitative terms, which in fact predicts inventory *over*estimation of landfill methane emissions for France (Pison et al. 2018). Considering the entire EU, several studies confirm this trend otherwise noted on the regional/national scale - indicating agreement with compiled national inventory values for the UNFCCC excluding natural emissions (Bergamaschi et al. 2015, Wunch et al. 2019).

Thus little evidence appears to indicate underestimation of landfill methane emissions in EU inventories, based on 22 T-D studies reviewed for this work. This could be interpreted as symptomatic of a lack of evidence from T-D studies to verify current landfill methane emissions in inventories. However, given considerable evidence indicating widespread agreement between B-U inventory and T-D estimates, this would seem unsupported. More supporting evidence from T-D studies in the EU at various scales is important for drawing potentially more clear conclusions. Regardless, evidence accumulated from 22 T-D studies clearly does not implicate inventory landfill methane emissions as potentially underestimated or otherwise incorrect in any significant fashion.

4.3. Comparison: US & EU-based top-down studies

A total of 55 T-D studies are analyzed for this work, split roughly 60:40 between those conducted in the US and EU (33 and 22, respectively). When comparing the results from the two cases (US and EU), several key similarities and differences are apparent. These similarities and differences are key for drafting meta-inferences toward answering one of this work's stated research questions; is the accuracy of a landfill-specific emission estimate context-dependent? Specifically, is a landfill methane emission estimate more or less likely accurate according to results of T-D studies, if it regards landfills in the US, or the EU? Notably, this question is not meant for establishing whether overall, inventory landfill methane emissions estimates are more accurate in the US, or the EU.

Comparing overall accuracy of landfill methane emissions measurements in the EU and US would require vastly different methods. These would most likely include in-situ measurements from landfills for each case. To compare accuracy of landfill-specific inventory estimates using data analyzed here, would incur a number of bold assumptions regarding certainty of T-D methane emissions estimates. All that said, it should be made clear to the reader that comparison of results from a large quantity of T-D studies split between the US and EU offers insight into the potential validity of inventory estimates in each.

The most obvious difference between results of analysis of T-D studies from the US and EU, is the amount of landfill-specific methane emissions estimates generated for comparison with B-U inventory estimates. Combined, 13 T-D studies analyzed generate landfill-specific emissions estimates, constituting about 24% of the total (55). Of that 13, 12 (roughly 92%) are from the US case study. As is stated previously, this lack of landfill-specific data from T-D studies conducted in the EU does not diminish the value of conclusions reached by this work, but it does alter their nature. When one observes data from studies conducted within the EU, overall emissions estimates largely agree with B-U inventory estimates. A lack of evidence makes definitive conclusions invalid regarding current inventory landfill methane emissions estimates. However, the relationship between overall and landfill-specific emissions estimates observed in the US case, grants validity to deeming EU-based inventory landfill methane emissions estimates as more likely accurate. This is according to results from a number of T-D studies that indicate agreement between their overall methane emissions estimates and corresponding inventory-reported values. Further explanation of this designation is provided in the *Discussion* section.

It is imperative one recognizes more data is available from T-D studies conducted within the US for comparison of methane emissions estimates. Of the 55 total T-D studies analyzed between both cases, 47 generate their own overall methane emissions estimate. This includes about 91% (30 of 33) studies used for the US case, and about 73% (16 of 22) of those used for the EU case. Perhaps even more relevant, are notable differences in findings from T-D studies conducted in the EU or US that generate overall methane emissions estimates.

Of 13 studies conducted within the EU that generate overall methane emissions estimates and cite inventory estimates for comparison, 2 indicate underestimation in the corresponding inventory (Helfter et al. 2016, Pitt et al. 2019). A third (Bergamaschi et al. 2015) is not counted for the large contribution of its overall T-D estimate it cites from natural wetland methane emissions, which are not included in the comparable inventory figure. This, in contrast to the US case, where of 30 estimates for overall methane emissions in a given area, 25 indicate underestimation in corresponding inventories. This difference is pronounced when comparing T-D estimates on the continental, or in the case of the US, national scale. Miller et al. (2013), Kort et al. (2014), and Turner et al. (2015), support consistent underestimation of overall anthropogenic methane emissions for the entire US by the EPA in its annual national GHG inventory (see Figure 1 above). Bergamaschi et al. (2015, 2018) for the entire EU, and Wunch et al. (2019) for a large region, depict agreement between B-U and T-D estimates of anthropogenic methane emissions.

Basic analysis of results from both the EU and US shows similar clustering of T-D studies within several specific regions for each respective case. For the US, T-D results are conducted mostly for California at varying scales, the city of Indianapolis, several large East Coast cities (McKain et al. 2015, Ren et al. 2018, Plant et al. 2019), and regions supporting the US O/NG sector (Alvarez et al. 2018, Kort et al. 2014). For the EU, T-D studies are concentrated mostly within the UK (Manning et al. 2011, Pitt et al. 2019) and Ireland (Ganesan et al. 2015, Helfter et al. 2019), as well as France (Pison et al. 2018, Lopez et al. 2015) and Poland (Kuc et al. 2003, Zimnoch et al. 2010, 2018). T-D studies appearing clustered around the same foci within specific regions for each respective case can be indicative of the need for more T-D studies in areas not yet researched. In the US, this could mean pinpointing where methane emissions are potentially underestimated in inventories, including from landfills potentially. For the EU, increasing research efforts to generate T-D estimates in different contexts could help to further validate existing B-U inventory reporting methods, overall and landfill-specific.

4.4. Model framework application for inferring top-down landfill methane emissions estimates

The model framework for inferring T-D landfill methane emissions estimates required EPA GHGRP-reported data for WIP and emissions. The framework was applied to 106 landfills based on pre-existing T-D estimates for 18 individual landfills. Different rates (t CH_4/t WIP) and multipliers were established for use in three scenarios: a) SoCAB (2010), b) Indiana (2011) and c) MD/VA (2015). Because T-D studies used to describe representative rates were conducted in different years, EPA GHGRP data for each area was extracted for different years in order to correspond with the studies. Peischl et al. (2013) conducted airborne measurements in 2010, so EPA GHGRP data used for SoCAB landfills is from 2010. Cambaliza et al. (2015) conducted their measurements in 2011, so EPA GHGRP data is from 2011. Ren et al. (2018) conducted airborne measurements in both 2015 and 2016, but compared results to EPA GHGRP-reported figures from 2015. Thus, this work also uses data for 2015 from the EPA GHGRP for Maryland and Virginia landfills.

Results indicate general underestimation of landfill methane emissions reported to the EPA GHGRP in all three scenarios (see Figure 3). However, the discrepancy between inferred emissions estimates and those reported to the EPA GHGRP are significantly different for each. This is in spite of the fact Reported Total emissions are nearly the same for all three scenarios (see Table 3). Notably, the lower bounds for the T-D inferred estimates for both SoCAB and MD/VA are roughly in agreement with the total reported to the EPA GHGRP for each. That said, the discrepancy between the mean inferred emission estimates and the uncertainty range are both far larger in the case of MD/VA. This is attributed foremost to the disproportionate amount of emissions measured at eleven MD/VA landfills by Ren et al.

(2018), especially compared to the amount of WIP (see Figure 2 for results, Table 4 for WIP). Themelis & Ulloa (2007) provide roughly 3.6 kg CH_4/t WIP as a conservative estimate of the amount of methane *generated* per ton of waste landfilled (converted from original expressed form in Nm^3/t). The inferred emission rates described in Table 4 seem feasible according to this standard. This is without at all accounting for site-specific variability regarding methane generation and emissions.

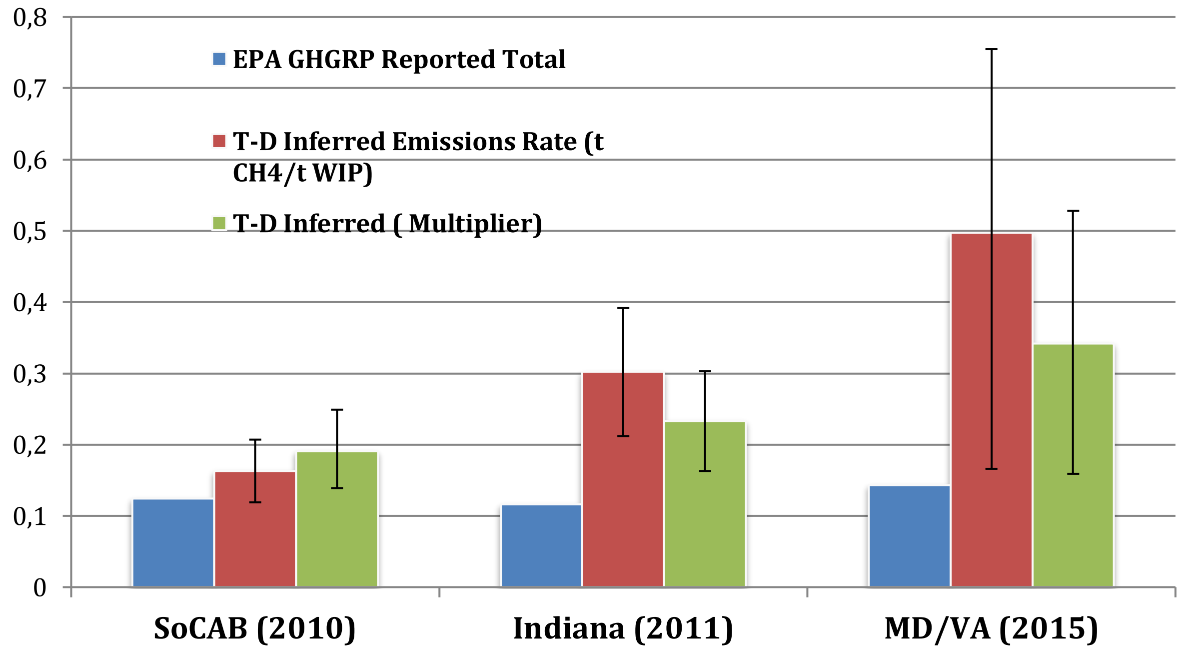

Figure 6. EPA GHGRP & T-D inferred landfill methane emissions estimates (Tg CH_4)

Sources: (EPA GHGRP, Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018)

Table 4. EPA GHGRP & T-D inferred landfill methane emissions estimates (Tg CH_4)

Scenario	EPA GHGRP Reported	T-D Inferred (t CH_4/t WIP)	T-D Inferred (Multiplier)
SoCAB (2010)	0.124	0.163 (0.119 - 0.207)	0.191 (0.139 - 0.249)
Indiana (2011)	0.116	0.302 (0.212 - 0.392)	0.233 (0.163 - 0.303)
MD/VA (2015)	0.143	0.497 (0.166 - 0.755)	0.342 (0.159 - 0.528)

Sources: (EPA GHGRP, Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018)

Discrepancies noted by Ren et al. (2018) between T-D estimates based on AMB methods, and EPA GHGRP-reported figures, are far greater than those expressed by either

Peischl et al. (2013) or Cambaliza et al. (2015). That said, it is important to note the high level of uncertainty associated with the foundational estimates from Ren et al. (2018). By extension, this applies to inferred T-D estimates for MD/VA landfills from applying the model framework (see Figure 3). Two additional estimates are generated for each scenario from only the landfills in each area, that report emissions estimates based on both equations HH-6 and HH-8 (see Figure 4). For SoCAB, this criterion fits every landfill that reported emissions to the EPA GHGRP for 2010 anyway, though some are excluded for both Indiana and MD/VA.

Table 5. Emission Rates (kg CH_4/t WIP) based on inferred T-D estimates

Scenario	Number of Landfills	Total Waste Disposal Area (km^2)	Total inferred emissions based on calculated rate (Tg CH_4)	Total WIP (t)	Emission Rate (kg CH_4/t WIP Landfilled)
SoCAB (2010)	28	21.383	0.163	652,660,090	0.250
Indiana (2011)	35	14.633	0.302	236,039,541.2	1.280
MD/VA (2015)	61	21.774	0.497	323,702,671.4	1.536

Sources: (EPA GHGRP, Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018)

Mean total estimates from HH-6 are nearly in agreement with the inferred estimate based on the multiplier for Indiana, and with both T-D inferred mean estimates for SoCAB, though in that case so is the Reported Total to some extent. While the reported estimate from HH-6 is significantly more in agreement with inferred estimates for MD/VA, it is perhaps more relevant to note the vast discrepancy from the Reported Total. Moreover, the Reported Total emissions for MD/VA (2015) are roughly half that of the Reported Total using HH-6, and roughly a fifth the mean T-D inferred emissions based on the rate (t CH_4/t WIP). In all cases the reported total using HH-6 is within the uncertainty margin of either one or both T-D inferred estimates generated from the model framework.

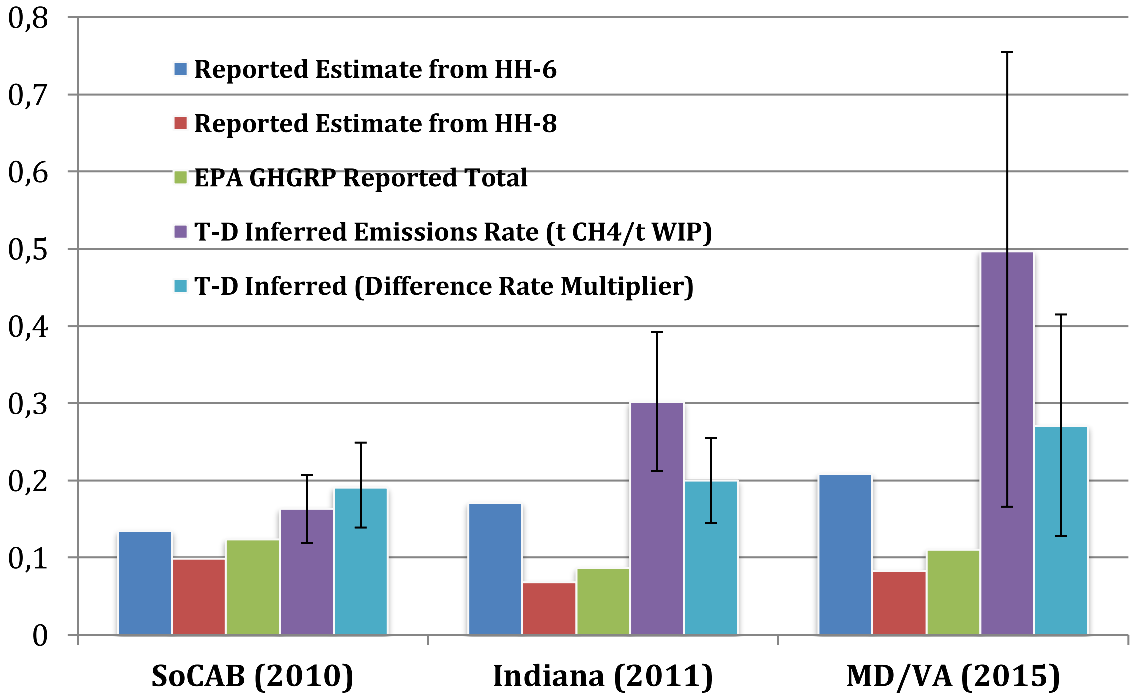

Figure 7. EPA GHGRP & T-D inferred landfill methane emissions estimates incl. reported estimates from HH-6 and HH-8 (Tg CH4)

Sources: (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018)

For SoCAB, it would appear enforcing the supposed EPA protocol mandating larger estimates be reported by all landfills generating two would largely resolve the discrepancy between EPA GHGRP-reported and T-D inferred methane emissions estimates. This roughly translates to an assumed validity of emissions modeled using the IPCC Waste Model for SoCAB. In contrast, results for Indiana and MD/VA show the multiplier is far exceeded by the estimate based on a derived rate (t CH_4/t WIP). There are several possible explanations for the apparent difference between T-D inferred emissions based on the rate (t CH_4/t WIP) and the multiplier, which are explored mainly in the *Discussion*. It remains crucial to note the uncertainties of either group of T-D inferred estimates, for all three scenarios. The reader should also bear in mind landfill size does not have a uniformly positive relationship with methane emissions amounts (Spokas 2020, personal communication). This model framework does primarily rely on the reported amount of WIP at individual landfills to infer emissions. While WIP is likely a better indicator of methane generation and emissions than landfill size

(Bogner 2020, personal communication), this is more or less the same assumption the IPCC FOD Method (2006) is based on. The difference, however, is the fact the actual numbers used to infer the amount of methane emitted per ton waste (t CH_4/t WIP) come from AMB campaigns, rather than assumed default values for certain model parameters. Essential parameters of this model framework are the rates themselves, and WIP of every landfill for each scenario. The inferred estimate from the multiplier relies solely on the difference rate multiplier itself, and the estimate reported to the EPA GHGRP for the corresponding year. The multiplier is essentially a scaling up of the reported estimates to match the extent to which the three foundational studies (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018) indicate underestimation of methane emissions for individual landfills in each scenario.

Several details should be taken from the results of applying this model framework in all three scenarios (SoCAB, Indiana, and MD/VA). Totals reported to the EPA GHGRP do not correspond with estimates based on generation alone (HH-6), the difference being roughly double for both Indiana and MD/VA. In all cases, this reported total estimate from HH-6 is in better agreement with the two T-D inferred methane emissions estimates. This represents roughly half of the landfills the framework is applied to for both Indiana and MD/VA individually, and the entire SoCAB group. The T-D inferred estimate based on a derived emissions rate (t CH_4/t WIP) is higher than that from using the derived multiplier, for both Indiana and MD/VA. The opposite is true for SoCAB, though in that case T-D inferred estimates are also the most in agreement of all three scenarios, both with each other, and the reported estimates from the EPA GHGRP (see Figure 4).

5. Discussion

The *Discussion* that follows is divided into three parts to correspond with the three stated research questions. All answers made available in this section are the result of the preceding mixed methods multi-case study, model framework application, and comprehensive *Literature review*. This work has thus far moved backwards. Answers were \ sought for the third stated research question regarding context dependency of landfill methane emissions estimates accuracy for the EU and US. This choice was made in line with the inductive, quasi-retroductive nature of the work, outlined explicitly in the *Methodology*. To allow for implications of T-D studies for B-U landfill methane emissions estimation to materialize, meta-inferences were first posited to the other two, more specific research questions (Venkatesh et al. 2013). This *Discussion* is structured in the same fashion, such that the first section concerns the third stated research question, and the second concerns the second. The final section regards the first research question and stated topic of this work; implications of T-D studies for B-U inventory methods for estimating landfill methane emissions.

5.1. Relative accuracy of bottom-up inventory landfill emissions estimates

Thus far, this paper has addressed two topics somewhat separately; landfill methane emission estimation for B-U inventories & T-D studies that generate landfill-specific methane emissions estimates. Application of the model framework involved combining the two. Meta-inferences have already been stated which help to answer the third research question stated in the introduction;

> To what extent is the observed phenomena of discrepancies between T-D and B-U estimates context dependent; are inventory landfill methane emissions estimates more likely inaccurate in the US than in the EU?

Inventory landfill methane emissions estimates, as reported to the IPCC and in localized inventories, appear more likely inaccurate in the US than in the EU, according to results from analyzing 55 T-D studies. This end is met from observing a trend in the US, which through extrapolation to the EU predicts agreement between B-U and T-D landfill methane emissions estimates, with several assumptions. Application of the model framework is based on a segment that contributes to these findings, and is itself supporting these findings. T-D studies from the US indicate underestimation of overall methane emissions both at the national level (Miller et al. 2013, Wecht et al. 2014b, Turner et al. 2015), and at the statewide or regional level, within certain contexts (McKain et al. 2015, Plant et al. 2019, Peischl et al. 2013, Ren et al. 2018, Cambaliza et al. 2015). This pattern, observed for overall methane emissions estimates, corresponds with T-D landfill-specific estimates indicating underestimation in B-U inventories (Peischl et al. 2013, Jeong et al. 2017, Cambaliza et al. 2015, Ren et al. 2018).

Although T-D overall estimates include landfill-specific figures, meaning underestimation of the latter mandates that of the former, only one T-D study attributes 100% of its observed overall discrepancy to landfills (Johnson et al. 2014). Thus, the vast majority of the time T-D studies indicate underestimated landfill methane emissions in the US, other sources are also underestimated in inventories. Underreporting of landfill emissions cannot be deduced from overall underestimated emissions alone given not every instance T-D studies indicate underestimation of methane emissions, includes implications for landfill-specific inventory estimates. Such is apparent when one considers results from T-D studies in several dense urban areas (McKain et al. 2015, Plant et al. 2019), as well as regions vital to the US O/NG sector (Kort et al. 2014, Alvarez et al. 2018). This is all to say that given limited incidence of inventory underestimated *overall* methane emissions from T-D studies conducted in the EU, it appears likely EU inventory landfill emissions estimates are relatively accurate. It is not possible to say at this time whether results of T-D studies render landfill-specific methane emissions estimates likely inaccurate on a large scale. Instead, what can be said is that such is more likely possible for the US than the EU, if one considers T-D studies relatively more valid for quantifying methane emissions from various sources. Moreover, application of the model framework would suggest this fact as at least true on a regional scale within certain sections of the US.

T-D studies do however indicate inventory *over*estimation of overall methane emissions within certain EU contexts (van der Laan et al. 2009, Manning et al. 2011, Ganesan et al. 2015, Wunch et al. 2019). This is potentially meaningful considering the only study that generates an independent landfill-specific estimate likewise indicates overestimation of landfill methane emissions in the corresponding national inventory (Pison et al. 2018). Of the stated research questions mentioned, the confirmatory (one) regarding accuracy of landfill-specific methane emissions estimates is meant to be answered primarily through analysis completed in the previous section. However, from the aforementioned pattern of potentially overestimated overall methane emissions in certain EU contexts (the Netherlands, the UK, Ireland, France), an alternative narrative appears feasible.

Several studies conducted in the EU measured emissions from individual landfills and found varied degrees of overestimation of emissions in national inventories (Mønster et al. 2015, Scharff & Jacobs 2006, Börjesson et al. 2009). Scharff & Jacobs (2006) in particular, found overestimation from modeled emissions estimates for three Dutch landfills according to their measurements. Van der Laan et al. (2009) predict overestimation of overall emissions for the Netherlands in a corresponding national inventory. Mønster et al. (2015) find overestimation in inventories for Danish landfills, at a similar level to that found by Scharff & Jacobs (2006). Börjesson et al. (2009) find agreement between their estimate for Swedish landfill emissions and the national regulator's, but overestimation in the IPCC-reported total for Swedish landfills. One partial explanation for this trend is the lack of accounting for lesser organic content in landfills as a result of EU Waste Policy (Mou et al. 2015).

From evidence cited above, one might then conclude EU landfills are potentially misrepresented in inventories similar to the manner in which such appears the case for the US, per T-D studies analyzed in either case. This is to say T-D studies from the EU depicting overestimated overall emissions (Ganesan et al. 2015, van der Laan et al. 2009, Manning et al. 2011, Wunch et al. 2019), are potentially the end result of overestimated landfill methane emissions. This would be assumed from evidence indicating such in more isolated circumstances (Mønster et al. 2015, Börjesson et al. 2009, Scharff & Jacobs 2006), perhaps as a result of EU policies even (Mou et al. 2015). This is certainly an appropriate perspective from which to gear further research. It is not, however, an expressed meta-inference for potentially answering whether or not landfill methane emissions estimates are more or less likely accurate in the EU or US according to results of T-D studies. This is mostly related to the nature of evidence compiled in this work that would be cited for justifying this inference.

The evidence referenced here from the *Literature review* comes from 3 countries neither geographically, nor economically representative of the entire EU: the Netherlands, Sweden, and Denmark (Scharff & Jacobs 2006, Börjesson et al. 2009, Mønster et al. 2015). It remains valid to cite EU Waste Policy as contributing to lower landfill methane emissions (Mou et al. 2015), perhaps even to corroborate findings from overlapping T-D and B-U efforts that find overestimation from modeled emissions via measurements (van der Laan et al. 2009, Scharff & Jacobs 2006). Still, this is not inferred as indicative of a continent-wide phenomenon. Furthermore, literature indicates larger amounts of organic waste are still landfilled in some EU countries despite EU Waste Policy (Stanisavljevic et al. 2012, Tatsi & Zouboulis 2002).

Toward formulating meta-inferences, evidence cited from T-D studies to support inferring landfill methane emissions estimates less likely accurate, would not be used if results did not include landfill-specific estimates. EU Waste Policy coupled with evidence of individual landfill-specific overestimation from models (Mønster et al. 2015, Börjesson et al. 2009, Scharff & Jacobs 2006), does not allow for inferring from several instances of overestimation of overall methane emissions estimates (Ganesan et al. 2015, van der Laan et al. 2009, Manning et al. 2011, Wunch et al. 2019) that landfill-specific emissions estimates are likely overestimated. At least not according to T-D studies. Therefore, results from analyzing 55 T-D studies from the EU and US supports inferring US landfill-specific methane emissions estimates as less likely accurate. Still, it is important one considers some flaws inherent to this inference.

The evidence presented, though representative of a range of landscapes, and densely populated regions of the US (LA, Indianapolis, East Coast cities), is not geographically representative of even a large portion of the country. What is presented is enough to constitute a trend; of T-D studies that indicate underestimation of overall methane emissions estimates, a significant number portray landfills as specifically underestimated (i.e. Johnson et al. 2014). This trend allows for making meta-inferences about the applicability of a single prescribed model for estimating landfill methane emissions throughout the US, and the likelihood those generated estimates are of adequate accuracy, overall. This does not imply a given individual inventory-reported modeled methane emissions estimate can itself be assumed inaccurate.

T-D studies used to assert US landfill emissions estimates less likely accurate do not necessarily represent every waste management strategy applied throughout the US. Individual states and cities often develop their own plans for waste management based on various overarching goals of government (CalRecycle 2000, Mass ECDP 2013, Sharp 2019). This leads to a wide range of strategies and plans, often reflective of different ideas surrounding waste management and disposal itself. This also relates to findings from application of the model framework, which depicts vastly different levels of methane emissions from landfills in different regions of the country. The same can be said about the EU to a certain extent.

However per the Landfill Directive (1999), all EU Member States are obliged to work to reduce overall amounts of organic waste (the most potent source of landfill methane emissions) disposed of in landfills. The fact differing waste management strategies often result in different levels of landfill methane emissions actually bolsters this paper's claim landfill emissions estimates are less likely accurate in the US, compared to the EU.

Potentially variable methane emissions levels from landfills in different cities and/or states, increases the importance of instances in which inventory emissions estimates are observed to be inaccurate. In the EU case, inferring landfill estimates as likely inaccurate based on available evidence would involve ignoring, a) an absence of supporting evidence from 22 EU T-D studies analyzed for this work and, b) the unrepresentative nature of evidence available from the literature. For the US, the same meta-inference is based on evidence from T-D studies characterizing landfill emissions as low in certain settings (i.e. Peischl et al. 2013). Methane generation is also largely uncertain for landfills elsewhere in the US where management strategies may differ significantly from regions covered in this study.

5.2. Discrepancies between top-down and bottom-up landfill methane emissions estimates

Several points of discussion are framed by the assumption T-D generated estimates are inherently more accurate than B-U inventory estimates used for comparison. This is largely due to the fact this study does not review individual methodologies chosen for completing T-D studies. This study also does not purport to make definitive conclusions regarding accuracy of individual B-U landfill methane emissions estimates. Regardless, several caveats should be announced, which may limit viability of meta-inferences. More than caveats, the following are alternative explanations for observed discrepancies, made plausible by the generally high degree of uncertainty inherent to T-D methane emissions estimates.

It is important to first consider the influence of background methane concentration on T-D estimates, especially landfill-specific estimates. Similar to tracer methods for landfill emissions measurement (Oonk 2010), T-D estimates for specific landfills may be impacted by background methane concentrations and/or nearby ulterior sources (i.e. other landfills, wastewater treatment plants, agriculture, etc.) This is part of the reason some experts recommend that within the field of landfill methane emissions measurement, two methods should be employed simultaneously (Peer et al. 1993, Bogner et al. 1997). Some prescribe at least 4-6 one day measurements for a single landfill to account for temporal variability (Oonk 2012), which T-D studies do not always provide (i.e. Ryoo et al. 2019). Often T-D studies include numerous unsuccessful flyovers as part of airborne measurement campaigns. The data from these flyovers might still hold value for understanding variability of the methods themselves, though are generally discarded (Spokas 2020, personal communication). The same limitations impact landfills emissions estimates inferred from applying the model framework, which relies on individual landfill methane emission estimates from T-D studies.

Similar to flaws in accounting for seasonal fluctuation of OX and CE using B-U models (Spokas et al. 2006, Oonk 2010, Chanton & Liptay 2000), accounting for general seasonal variability can be difficult for T-D studies without measurements taken throughout the year (Ren et al. 2018). For these and other reasons, T-D estimates are usually the result of an inversion of measurement data to account for general variability of emissions in generating annual estimates (i.e. Wennberg et al. 2012, Jeong et al. 2013, Manning et al. 2011).

The reader should not envision either T-D or B-U methane emissions estimates, landfill-specific or otherwise, as infallible - or even ideal for dealing with the vast uncertainty associated with landfill emissions. Landfills are themselves a, “very complex and poorly understood system,” (IPCC 2006), and to generate an annual estimate of their emissions will always preclude certain assumptions. Moreover, Both B-U and T-D methods of landfill methane emissions estimation remain somewhat reliant on different models. The difference is that B-U methods use models emphasizing characterization of specific variables that differ for every landfill (i.e. CE and OX), and T-D methods use models to deploy measurements of actual methane emissions. The applied framework relies on a somewhat separate set of assumptions, based on the idea the use of measurements in T-D methods generates relatively more realistic emissions figures for the individual landfills used to generate rates (t CH_4/t WIP).

Nearly every instance of a T-D study finding a pronounced discrepancy between their landfill-specific generated estimate and that provided from a B-U inventory comes from the US case. In fact, only one study matches this description from the EU case (Pison et al. 2018). That said, evidence used for potentially explaining these discrepancies comes from research conducted at individual landfills in both the EU and US. Using both remains valid for the fact the same document steers B-U landfill methane emissions estimation and relevant research in both the US and EU; the IPCC *Guidelines* (2006). Moreover, evidence provided in the *Literature review* that comes from the EU provides a) context for results from the US, and b) information to help explain the lack of discrepancies between B-U and T-D estimates from the EU case.

The most obvious explanation for noted discrepancies between B-U and T-D landfill methane emissions estimates is uniform application of the B-U models using default values for certain parameters, namely CE and OX. It is important to note that this, as well as any other posited meta-inferences should not be considered theories for explaining T-D/B-U discrepancies on an individual landfill basis. That said, with regard to the use of default values for model parameters specifically, potential shortcomings might arise from uniform application, as a result of the vastly heterogeneity of landfills themselves (IPCC 2006). The 75% default CE value in particular, gives rise to a great deal of scrutiny, via numerous independent researchers quantifying this parameter at values both significantly higher and lower for landfills located in different geographic and political contexts (Mønster et al. 2015, Lohila et al. 2007, Spokas et al. 2006, Huitric & Kong 2006, SWICS 2009). Research from industry operators and researchers in the EU indicates far lower assumed CE values (Oonk 2010). This is manifest in the results of several individual landfill measurement campaigns (Lohila et al. 2007, Mønster et al. 2015, Aronica et al. 2009, Oonk 2012, Börjesson et al. 2009, Fjelsted et al. 2020). That said, some maintain a 75% default value is still potentially reasonable for use in certain countries (Börjesson 2020, personal communication).

In both the EU and US, researchers tend to criticize model use for establishing CE, maintaining that measurements are essential for establishing verifiable CE values (Lohila et al. 2007, Amini et al. 2013, Huitric & Kong 2006, Scharff & Jacobs 2006). That said, the results from measurements are totally different for each. Börjesson et al. (2009) find a CE value for all Swedish landfills equal to 51±5%. Mønster et al. (2015) find CE values in a range 41-81% for 15 Danish landfills. At the individual facility level, Fjelsted et al. (2020) found CE at a Danish landfill in a range of 8-21%, Aronica et al. (2009) one Italian landfill 25.2-43.1%. While a 75% default value for CE may still be considered valid in certain circumstances (Börjesson 2020, personal communication), no evidence exists of EU researchers or industry operators lamenting 75% as too low a number. In contrast, certain individual US landfill operators support CE values in excess of 90% (Huitric & Kong 2006, SWICS 2009).

Through the EPA GHGRP, operators of landfills generating 25,000 t CH_4 or more are obliged to report individual facility-level methane emissions. While technically this data is meant for use in compiling the US GHGI for the IPCC (EPA 2019), it is at this point 'hard to say' to what extent EPA GHGRP data is actually used for this purpose (Bogner 2020, personal communication). Regardless, its use is warranted as part of this study for its application of several different modeled approaches to estimating landfill methane emissions, using equations partially based on the IPCC FOD Method (HH-6) and on LFG recovery data (HH-8). As well, the availability of data on the individual landfill level is primarily what allows for application of the model framework toward better answering the stated research questions.

There are several instances in which the EPA GHGRP-reported value for a given landfill or group of landfills, is far lower than the estimate generated from a T-D study (Peischl et al. 2013, Jeong et al. 2016, Cambaliza et al. 2015, Ren et al. 2018). In the case of Peischl et al. (2013), this is actually found independent of a separate regional inventory (CARB) value for the same landfill or landfills, which is in agreement with the T-D generated landfill-specific methane emissions estimate. Jeong et al. (2013) and Wecht et al. (2014a) both estimate California's total statewide landfill methane emissions roughly double corresponding EPA-reported values. There are several issues rendering the use of EPA GHGRP data a bit frail at the regional or state level, which speculatively make cited discrepancies not necessarily the "fault" of the EPA or the self-reporting program (GHGRP) itself. This however, is explored more in the next subsection regarding implications of T-D estimates for B-U landfill methane emissions estimates overall.

Disregarding aforementioned discrepancies from T-D studies at the statewide level (Jeong et al. 2013, Wecht et al. 2014a), those observed by T-D studies for individual landfills (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018) occur in separate, largely different contexts, geographically and otherwise. Results of applying the model framework show reported estimates generated from LFG recovery data (HH-8) are significantly lower than those reported estimates from generation (HH-6). Furthermore, there is a tendency of operators to claim a 75% default CE value as low, the EPA GHGRP employs a self-reporting structure, and evidence indicates inaccuracy of methane emissions estimates increases with increasing CE (Oonk 2010). This study then infers assumed CE characterization based on LFG recovery data could have a negative effect on overall accuracy of US landfill methane emissions estimates.

Results from applying the model framework show generally that estimates from LFG recovery data (HH-8) are potentially contributing to a higher degree of underestimation of emissions in the US, according to GHGRP data. This seems apparent compared to a scenario in which the still-flawed, nonetheless higher estimates from modeled generation (HH-6) were used instead (see Figure 4). Such could be the case not just on the individual landfill level in the three scenarios included, but also on a large, perhaps national scale. State and regional-specific inventories do not seem to always follow this trend. Consider that CARB and T-D individual landfill-specific estimates from Peisch et al. (2013) are in agreement. Regardless, a correlation is apparent from the results of this study and evidence from the *Literature review*. US industry independently testifies observing CE values higher than 75% (SWICS 2009). T-D observed emissions estimates indicate underestimation on both regional and individual-landfill levels in certain regions. Inferred estimates from model framework application show far more agreement with EPA GHGRP estimates from modeled generation (HH-6), than from those actually reported. More recent industry opinion is not readily available, that would indicate whether higher CE values are still assumed by US landfill industry operators. There is however, still evidence from the GHGRP of vast discrepancies between emissions from modeled generation and from LFG recovery data as recently as 2018 (see Figure 6 below).

Given all the above evidence, it is especially important one considers overestimated CE tends to directly lead to underestimated emissions (Oonk 2012). This does not negate the merits of installing LFG collection equipment to reduce landfill methane emissions. Rather, the reader should now have a clearer image of potential drawbacks inherent to modeled landfill methane emissions estimates, considering emphasis placed on individual parameters -

in this case CE. Moreover, the purpose of emissions controls should be to reduce emissions, not to enable reducing the amount of emissions reported, as evidence from applying the model framework as well from several T-D studies from the US case suggests may be the case (Peischl et al. 2013, Johnson et al. 2014, Wecht et al. 2014a, Cambaliza et al. 2015, Ren et al. 2018).

In a similar fashion to CE, focus is often directed toward the value of OX for estimating landfill methane emissions. This is likely due to its assigned value in the IPCC *Guidelines* (2006) for use in B-U models worldwide. As well, results from studies conducted within the US and EU find vastly different ranges for OX via measurements taken at individual landfills. However, a clear division of opinion and/or findings between research from the US and EU, is not distinctly visible. This does not rule out the possible effect OX has on accuracy of methane emissions, positive or negative, for either case. Furthermore, the main applicable finding from the *Literature review* regarding OX, which is worth noting here, is its overall variability at different landfills (Scharff et al. 2005). Still, the fact OX is regarded as the other main 'control' on emissions and its prominence in IPCC-prescribed models (2006) may contribute to OX proving more influential on landfill methane emissions estimates accuracy than seems probable based on results of this study.

Similar to the nature of actual landfill cover soil methane oxidation (OX being the model representation of this phenomenon), OX's effect on actual emissions estimate accuracy appears largely uncertain. The fact the IPCC (2006) still prescribes a 10% default OX value based on somewhat dated research (Czepiel et al. 1996), is cited by US landfill industry as reason for use of alternative models for overall emissions estimation (Bogner 2020, personal communication, Schmeltz 2017b). Of the choice 'alternative models', HH-8 appears preferred by industry and is indicated as directly resulting in underreported emissions according to application of the model framework, as well as foundational evidence from T-D studies (Ren et al. 2018). Thus, although it seems reliance on CE may be directly linked to underestimated emissions from landfills in the US, the evidence supporting a 10% default OX value may be the avenue through which US operators are able to leverage the EPA. Reporting emissions for landfills with active recovery systems using HH-8 based on the idea HH-6 is outdated, implies reliance on a single model parameter, CE, is somehow preferable. Furthermore, by swapping out the traditional model equation (HH-6), already assumption-laden, for HH-8, the landfill industry does not seem to be improving the overall accuracy of reported emissions. This is clear based on evidence from both T-D studies (Peischl et al. 2013, Johnson et al. 2014, Cambaliza et al. 2015, Jeong et al. 2016, Ren et al. 2018, etc.) and application of the model framework.

Therefore, evidence from literature, results of T-D studies, and results of applying the model framework, indicate CE characterization as a potential cause for observed discrepancies between T-D and B-U landfill methane emissions estimates in the US. By extension, less frequent misrepresentation of CE may partially explain the lack of noted discrepancies of this kind from the EU case. This is a valid standpoint from which to focus further research.

5.3. Implications of top-down estimates for bottom-up landfill emissions estimation

The third and final portion of this *Discussion* is aimed at answering the first stated research question. Mulit-case analysis of T-D studies conducted in both the US & EU has allowed for pinpointing the occasions in which discrepancies are noted between B-U and T-D landfill methane emissions estimates. Review of literature regarding B-U landfill methane emissions estimation allowed for documenting potential reasons to explain why. Here this study seeks to use this information, in addition to inferred estimates from model framework application, to contribute knowledge if and how B-U methods for landfill methane emissions estimation are implicated by T-D studies.

T-D approaches for estimating methane emissions require a place in emissions estimation and reporting. How this might manifest itself in actual reporting structures is more the subject of the *Recommendations* section, but is also not a question this study can definitively answer. That said, alternatives to B-U methods (T-D) exist and are viable for use in national GHG emissions reporting. This is exemplified by model framework application to infer upper limit T-D methane emissions estimates for a region containing nearly 15 million people, as well as for three whole states within the US. The IPCC *Guidelines* (2006) encourage investigation of discrepancies between modeled and measured landfill emissions estimates, in order to help discover the source of error and/or disagreement. Certainly the basic FOD method (IPCC 2006) remains a simple, relatively robust way of developing emissions estimates for a, "very complex and poorly understood system." Achieving balance of simplicity and thoroughness is difficult for any model to estimate landfill methane emissions (Peer et al. 1993). However, the fact there are some apparent discrepancies between T-D and B-U landfill methane emissions estimates (i.e. Ren et al. 2018), indicates that improved thoroughness could be beneficial for overall accuracy.

Measurements are often touted in literature for gathering useful information on a micro-scale relevant to specific model parameters (CE and OX) as well as overall emissions for individual landfills (Lohila et al. 2007). The same types of measurements are likewise cited for model validation (Aronica et al. 2009) and potentially model 'tuning' (Scharff & Jacobs 2006). Despite this, B-U research includes little discussion of observed discrepancies from T-D studies and likewise, little attention is paid within T-D research to individual variables used in models, which may contribute to observed discrepancies. That said, T-D studies are not meant to fully discern the 'why' inherent to their findings. Some posit explanations for observed discrepancies, though in general this extends only as far as pointing to which sector emissions are generated from (McKain et al. 2015, Plant et al. 2019). Even then, a lack of spatial resolution for most methods makes accurate source apportionment an obstacle (Spokas 2020, personal communication). It is then imperative for research aimed at enhancing B-U landfill emissions estimation to consider T-D studies that indicate emissions reported in certain inventories as underestimated for individual landfills (Peischl et al. 2013, Ren et al. 2018).

For several studies used for data collection and analysis, T-D generated landfill methane emission estimates significantly exceeded reported values published by the EPA GHGRP (e.g. Wecht et al. 2014a). Of the 8 studies that fit these criteria, 3 are especially valid for inferring findings of potential implications for EPA GHGRP itself, which are themselves the basis for application of the model framework (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018). The other 5 studies, for which discrepancies are noted between the landfill-specific T-D estimates and the EPA GHGRP-reported equivalent, are considered less valuable for this study since they do not generate individual landfill-specific emissions estimates.

Within these 5 studies as well as Peischl et al. (2013), no actual reference or use of the EPA GHGRP reported value appears for the specific landfill(s) mentioned. Instead, because each was conducted in California, the inventory estimates referenced are from CARB. Therefore, the values from the GHGRP come from the EPA website and were collected separately. All of that said, the 5 studies without individual landfill T-D emissions estimates, all indicate significant underestimation by the EPA GHGRP for given areas in given years, even more so than from CARB, in several cases (Wecht et al. 2014a, Johnson et al. 2014, Jeong et al. 2016, 2017). Furthermore, Peischl et al. (2013), Cambaliza et al. (2015), and Ren et al. (2018) all depict emissions from individual landfills that significantly exceed the amount reported through the EPA GHGRP. In the case of Peischl et al. (2013), this is especially intriguing given the equivalent from CARB is actually in marginal agreement with their T-D generated estimate for the two individual landfills the study estimates methane emissions for.

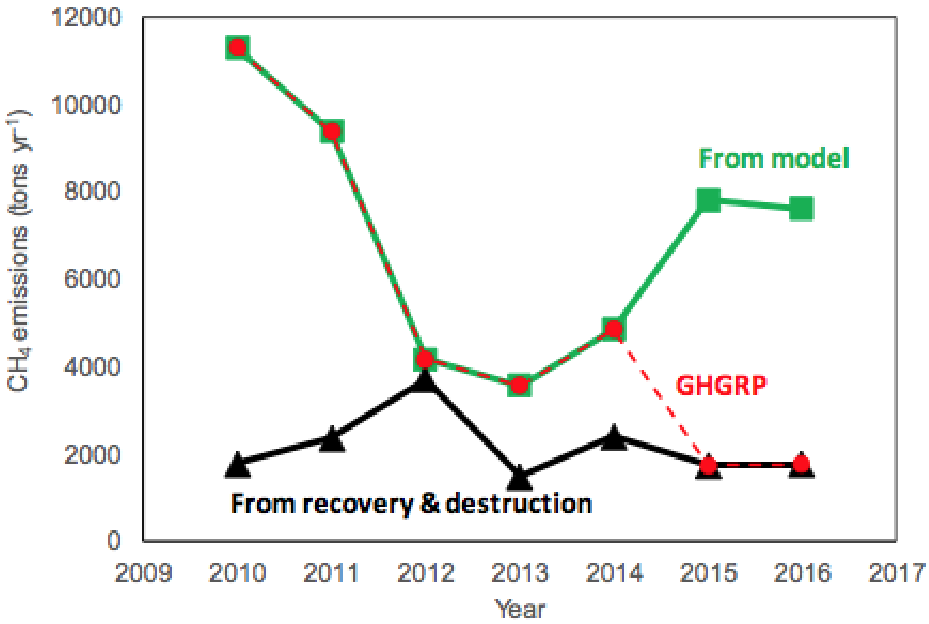

Figure 8. EPA GHGRP-reported emissions from Brown Station Road Landfill (Maryland)

Source: Ren et al. 2018

It seems despite cited opinion from some US landfill operators that default values for model parameters including CE and OX (SWICS 2009, Huitric & Kong 2006), are unfair for landfills that operate with a focus on emissions 'control', GHGRP-reported values may still be underestimated. This is an especially crucial point to raise, again considering findings from Peischl et al. (2013), Cambaliza et al. (2015), and Ren et al. (2018). These studies indicate underestimation of methane emissions on an individual landfill basis in three different geographic and political contexts within the US. The idea T-D estimates wrongly indicate underestimated emissions appears unlikely given Peischl et al. (2013) also references an alternative inventory that does not underestimate emissions (CARB). Given the way the EPA GHGRP is structured, permitting emissions reporting by landfill operators themselves, the indicated level of underestimated methane emissions may be a result of poor reporting practices.

According to the EPA GHGRP protocol, facilities with active recovery systems must report two emissions estimates; one using a traditional model based on methane generation (HH-6) and one from LFG recovery data (HH-8). Both estimates are made publicly available, but only one is actually reported and used for inventory compilation. The larger one should be used according to the EPA, however this is not always the case, indicating a potentially less-than-perfect level of transparency. Evidence supporting this entire line of thinking comes from Ren et al. (2018) in particular, which specifically notes the lower of these two estimates is used by Brown Station (Road Sanitary Landfill) for reporting to the EPA GHGRP, from 2014 on (see Figure 5). Why this decision was made is not clear, and is especially confusing given the fact the larger estimate should be used according to the GHGRP, not to mention T-D estimates indicating the higher estimate from a typical model is often still too low (Ren et al. 2018).

Further investigation as a part of data collection for applying the model framework, reveals of 11 landfills for which a T-D methane emissions estimate is generated by Ren et al. (2018), 9 use active LFG recovery systems and therefore report two separate estimates to the EPA GHGRP. Of these 9 landfills, 4 including Brown Station actually report the lower of these two estimates for the most recent year data is available (2018); Quarantine Road, King George, Stafford County, and Brown Station. Though in all cases, no reasons are explicitly given to support this decision, for King George at least, the difference between the two estimates is less than 200 t CH_4. For reference, 200 t is equal to 0.0002 Tg. The other three however, again including Brown Station, report emissions from LFG recovery data significantly lower than the estimate produced using HH-6 (see Figure 6). This is relevant considering findings related to application of the model framework, namely that despite stated EPA GHGRP Reporting Protocol, the lower estimate based on the HH-8 equation is still oftentimes used. That the same remains true for 2018 (the most recent year EPA GHGRP data is available for) is a solid indicator of the fact HH-8 is still most likely being used for estimating and reporting landfill methane emissions through the EPA GHGRP.

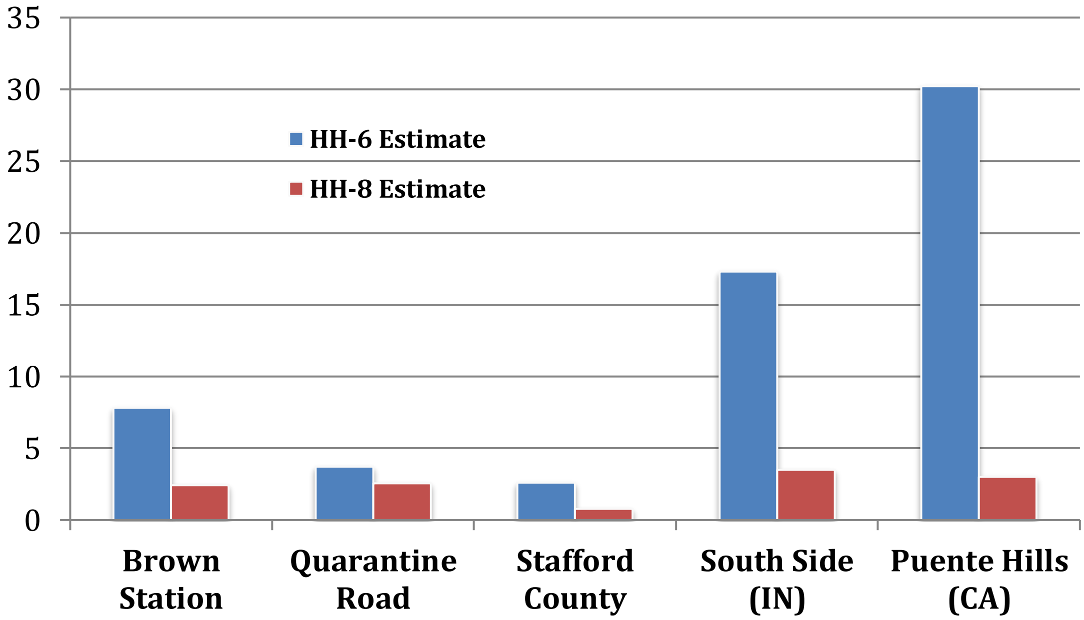

Figure 9. 2018 EPA GHGRP-reported emissions estimates from 3 Maryland landfills (Gg CH_4)

Source: (EPA GHGRP)

Figure 6 also includes 2018 EPA GHGRP-reported emissions from two other landfills referenced in this work. Both also report an emissions estimate using HH-8 as opposed to the HH-6 equation. As it were, despite the nature of the stated EPA Reporting Protocol that appears on the EPA GHGRP's website, industry guidance recommends landfill operators, "use the equation they feel is most appropriate based on their facility operations," (EPA 2019). Moreover, individual landfills are not even obligated to use the same equation in consecutive reporting years.

None of the landfills referenced in Figure 6 or through application of the model framework, support the choice to use HH-8 with explicit reasoning. What is important to take from this observation, in terms of implications of T-D studies for B-U landfill methane emissions estimation, is that the lower estimate for all of these landfills directly contrasts T-D generated estimates. According to T-D estimates portraying underestimated emissions in the EPA GHGRP-reported inventory, B-U methods for this process may need to consider enhancing transparency in reporting procedures. It otherwise appears a deliberate choice of landfill operators to always report the lower of two emissions estimates, especially considering independent T-D estimates that support higher methane emissions figures from certain individual landfills (i.e. Brown Station, Ren et al. 2018).

Generally, the most supported meta-inference to draw for B-U landfill methane emissions estimation from T-D studies is the apparent need for validation, and in some cases revision. A number of factors make landfill methane emissions estimation rather uncertain, not the least of which being the nature of landfill methane emissions themselves; highly variable both spatially and temporally (Oonk & Boom 1995, Scharff et al. 2005). From analyzing results of T-D studies and application of the model framework, certain inventories in the US appear flawed in their attribution of landfill methane emissions, namely the GHGRP. This appears in some cases for reasons besides issues related to the traditional FOD model itself (HH-6). Such findings do also indicate context dependency of the likelihood of accuracy of B-U emissions estimates between the US and EU. Generally they support the need for enhanced methods of accounting for landfill methane emissions in GHG inventories not reliant on 'regulatory calculations,' usually, "more simplistic than they should be," (Bogner 2020, personal communication).

Ideas have been posited in literature, both for increased use of individual landfill measurements in B-U emissions estimation (Aronica et al. 2009, Scharff & Jacobs 2006), and for inclusion of T-D methods in national methane emissions reporting schemes (Leip et al. 2018). The latter prescribes use of enhanced B-U reporting according to Tier 2 or 3 *Guidelines* (IPCC 2006), which in the case of landfills would mean enhanced models including measurements, either for tuning (Scharff & Jacobs 2006) or validation (Aronica et al. 2009). Evidence exists supporting highly engineered models potentially more accurate for accounting for methane emissions from individual landfills, particularly from the US (Spokas et al. 2011a, Cambaliza et al. 2017). T-D measurements including AMB methods can provide value in a number of ways including to establish upper limits, and for semi-quantitative information on local sources (Spokas 2020, personal communication, Bogner 2020, personal communication). Frankly, it seems beyond the scope of this study to suggest any sort of comprehensive overhaul of methods used for B-U landfill methane emissions estimation. This is especially true given the lack of evidence from the EU indicating definitive misrepresentation of landfill methane emissions in inventories compiled using current methods compliant with the IPCC *Guidelines* (2006).

The fundamental recommendations prescribed here are more to do with research requirements, which could serve as a prerequisite for supporting more sweeping changes to the status quo in B-U landfill methane emissions estimation and reporting. These are contained in the *Recommendations* section, still to follow. From this comprehensive analysis, the clearest meta-inference is that the application of a semi-uniform set of *Guidelines* (IPCC 2006) globally to prescribe emissions from complex, heterogeneous sources (landfills), certainly falls short in terms of accuracy in certain specific contexts. While evidence of this is mostly with regard to a singular regulator, the EPA GHGRP, the notion implies the logic supporting one model and one set of assumptions may lead to incorrect methane emissions estimates elsewhere, not only within the US and not only considering one model (FOD).

Although conclusive evidence from T-D studies of potential overestimation of landfill methane emissions has not surfaced from the EU case, there is little indication this is not still possible and would not come to light with more focused research. Though seemingly less concerning than unaccounted for or underestimated methane emissions, such could be a byproduct of missing emissions from another source, including perhaps WWT in urban areas like London (Zazzeri et al. 2017), or even more likely, leaks in O/NG infrastructure, given existing results from T-D studies (McKain et al. 2015, Plant et al. 2019). Therefore, considering everything mentioned, the main meta-inference drawn from this study for B-U landfill methane emissions estimation methods, is a general need for enhancement. This should be accomplished through continued research, given findings suggesting known uncertainties already contribute to incorrect estimates in the US, and potentially in the EU as well. Reliance on a single method, namely the use of models, is alone not enough to ensure more representative figures are circulated regarding global landfill methane emissions.

Conclusion & recommendations

The aim of this thesis was to better understanding of landfill methane emissions. Specifically, outcomes include determining context-dependency on accuracy of emissions estimation, and posited explanations for discrepancies between inventory and measurement-based estimates. Completing tasks necessary to achieve these allowed for positing meta-inferences regarding the overall implications of T-D studies for typical B-U methods of landfill methane emissions estimation. Landfill emissions quantification and management, and T-D atmospheric measurement-based studies have to this point largely been separate research topics. This work approached completing the tasks stated above at the point these fields meet. Furthermore, the preceding work is classified as a mixed methods multi-case study inclusive of a novel framework for generating inferred estimates of individual landfill emissions. A comprehensive literature review of research regarding landfill emissions estimation supports analysis of 55 T-D studies conducted in the US and EU, and application of the model framework.

It was determined overall, reported inventory landfill methane emissions estimates are more likely accurate in the EU than the US. Discrepancies between inventory and T-D emissions estimates are noted in several different geographic contexts in the US (Peischl et al. 2013, Wecht et al. 2014a, Jeong et al. 2017, Cambaliza et al. 2017, Ren et al. 2018) and are not in the EU. Although the same *Guidelines* (IPCC 2006) steer B-U landfill emissions estimation in both, research indicates estimation is largely uncertain due to significant spatial and temporal variability of landfill emissions themselves (Scharff & Jacobs 2006, Bogner et al. 2007, Oonk & Boom 1995). Furthermore, the US landfill industry demonstrates apparent discrediting of default values for CE and OX in model use (SWICS 2009, Huitric & Kong 2007). Increased reliance on CE in particular, for determining emissions, is shown to deteriorate accuracy of estimates reached (Oonk 2010, 2012).

The EPA GHGRP appears particularly likely to underestimate emissions as a result of HH-8, a relatively new equation for estimating emissions at the individual facility level. HH-8 is considered more up-to-date than its predecessor, HH-6, though not necessarily more realistic (Bogner 2020, personal communication). HH-8 relies on data collected by installed LFG recovery equipment, and often produces estimates significantly lower than those from HH-6 (see Figure 6), which is partially based on the IPCC FOD Method (EPA 2019). Data supplied by the EPA GHGRP is at this point likely not integrated into the annual US GHGI, which is reported to the IPCC. However, GHGRP data does provide an image of the state of landfill emissions estimation in the US, as well as self-reported emissions totals from individual landfills on an annual basis. These appear low when compared with results of contemporary T-D studies in certain locations (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018). Based on these findings the model framework was applied in three different scenarios. Results further support the notion overestimating CE could be resulting in underestimated landfill emissions in the US, as previous research would indicate (Oonk 2010, 2012).

The need for further validation of models used in both the US and EU for B-U estimation of landfill methane emissions is implied from findings described above. Though emissions appear underreported in the US per results of T-D studies and application of a model framework here, ramifications extend onto the methods themselves. While discrepancies between inventory and T-D emissions estimates are not noted from the EU case, IPCC-prescribed models for landfill emissions quantification do not guarantee accuracy. While EU Waste Policy is in general linked to lower methane generation (Mou et al. 2015), municipal waste management is itself fragmented; different EU countries apply different waste management schemes (Antonioli & Massarutto 2012). A lack of discrepancies observed from analysis of T-D studies conducted in the EU may indicate landfill emissions estimates are largely accurate, or at least more accurate than in the US, which is inferred by this paper. It could also mean however, that T-D studies have not been conducted in areas where emissions remain poorly quantified. Therefore, the main implication of current B-U methods of landfill methane emissions estimation, from this work, is the clear need for more evaluation and validation.

From comparative analysis with the US case, the lack of AMB or other atmospheric-measurement based studies for individual landfills in the EU is made apparent. Discrepancies observed in the US between estimates using such methods and inventory figures, particularly from the GHGRP, are also lacking in overall representativeness. It is therefore necessary for research to generate more methane emissions estimates using methods other than models currently used for B-U reporting, for individual landfills. Such efforts would be valuable in any location, though given the amount of existing research conducted in Northern European countries regarding landfill emissions (Börjesson et al. 2007, 2009, Scharff & Jacobs 2006, Lohila et al. 2007 Scheutz et al. 2009, Mønster et al. 2015, Mou et al. 2015), for the EU it appears necessary to test the viability of the assumed impact of EU Waste Policy on methane emissions in southern and eastern European locations. In the US, existing measurement-based landfill emissions estimates are disproportionately representative of several locations (California, Indianapolis, East Coast cities). Therefore, further research is warranted in areas not well represented for climate and other factors including landfill management practices, by areas already included in T-D studies.

Transparency, regarding emissions estimation formulation, appears lacking in the US based on findings of T-D studies and from application of the model framework. It would be valuable for research to understand clearly the reasons for the difference between the GHGRP stated reporting protocol that appears while using FLIGHT, and its messaging to the landfill industry (EPA 2019). Specifically, this refers to the prescribed use of the “higher” estimate for reporting to the GHGRP between HH-6 and HH-8, while in reality landfills are recommended to use the equation that best fits operations. Use of HH-8 appears questionable based both on research regarding the value of default CE values for estimating emissions (Oonk 2010, 2012), and findings from T-D studies depicting significant underestimation of emissions quantified by its use (Peischl et al. 2013, Cambaliza et al. 2015, Cambaliza et al. 2017, Ren et al. 2018). Thus, to understand why HH-8 was adopted for use in compiling the EPA GHGRP's inventor would be valuable for fostering a positive relationship with research efforts to improve landfill emissions estimates,.

The model framework applied in this study provides a useful tool for inferring methane emissions from landfills for which measurement-based estimates do not already exist. Its application could be improved through selectively choosing landfills to infer emissions for that are more accurately represented by the landfills used in the foundational studies (Peischl et al. 2013, Cambaliza et al. 2015, Ren et al. 2018). This could be done for a number of landfill-specific characteristics including cover soil material properties, site-specific climate, operational practices, and WIP, either separately or combined. Such an application, if validated by measurement-based estimates, could provide a cost-effective alternative means for estimating emissions from other individual landfills. Regardless, its application here is novel and is one of several notable contributions made by this study to the field of landfill methane emissions estimation, which overall include:

- Comprehensive review of literature supporting current methodologies for B-U national and regional landfill methane emissions estimation and reporting
- Mixed methods approach for studying landfill methane emissions, which joins two fields of research that primarily exist apart: landfill methane emissions estimation, and T-D studies generating emissions estimates at larger scales
- Separate analysis of 55 T-D studies conducted in both the US and EU
- Comparative multi-case analysis joining findings from literature review concerning B-U landfill emissions estimation and T-D studies conducted in the US and EU
- Built novel, replicable model framework for inferring emissions from individual landfills based on WIP, reported emissions, and T-D estimates from individual landfills in close proximity

- Generated inferred estimates for three different regions within the US (124 landfills in total) via application of the model framework

- Confirmed context dependency of inventory landfill emissions accuracy based on relative accuracy of EU landfill emissions, compared with the US, per results of multi-case analysis

- Results allow determination that:

    - Assumed values for CE as likely contributing to discrepancies observed between T-D estimates and B-U inventory-reported figures in US, especially considering EPA GHGRP
    - Dated nature of literature cited to support IPCC-prescribed default OX values as avenue used by US landfill industry to levy EPA for use of HH-8 equation
    - Per results of multi-case analysis and model framework application, HH-8 appears to directly contribute to underreported emissions from some US landfills

- Posited meta-inferences concerning B-U inventory methods of landfill emissions estimation in general:
    - Revision is likely needed for some accepted figures
    - T-D methods for emissions estimation are likely required in some capacity for validation or evaluation of B-U inventory estimates
    - Models used remain assumption-laden and opportunity for use of highly-engineered models validated in different climates appears a viable option for improving B-U emissions estimation while remaining cost-effective

In terms of landfill methane emissions estimation, methods other than IPCC-prescribed models should be integrated into the current system in place for reporting emissions in national inventories. There are various methods supported by extensive research for use in estimating landfill emissions at the individual landfill-level (i.e. Lohila et al. 2007) as well as on larger regional and even national scales (Johnson et al. 2014, Ren et al. 2018, Pison et al. 2018). While IPCC-prescribed models provide a relatively robust way for accounting for landfill methane emissions, their use would be significantly improved within more cohesive system including T-D atmospheric measurement-based methods (i.e. Leip et al. 2018).

Landfill emissions quantification remains largely uncertain, despite extensive research aimed at better understanding spatial and temporal variability. For this reason, it remains equally important to continue research for cost-effective means of mitigating landfill methane emissions at the operations level. Landfilling ultimately remains the most economically attractive means of disposing waste in countries around the world including the US (Amini et al. 2013). Therefore, it is imperative to consider improving management strategies for means of controlling methane emissions (Spokas 2020, personal communication). That said, one key purpose of accurate GHG emissions quantification in the case of landfills is to understand the potential environmental benefits for making improvements. Methane (CH_4) is a GHG 25-28 times more potent than carbon dioxide (IPCC 2007, 2014). Benefits of limiting sources of anthropogenic emissions, including from landfills, could be realized in a much shorter timeframe than reductions in carbon dioxide emissions (Barlaz et al. 2004, Abichou et al. 2006, Spokas et al. 2011). Underreported methane emissions create long-term environmental impacts that remain invisible to the public eye. When properly accounted for, landfill emissions figures could promote policy changes for improved waste management strategies that reduce overall GHG emissions.

Several options exist for disposing organic waste aside from landfills, including biogas generation (Budzianowski 2014), composting (Van Fan et al. 2016), and incineration for waste-to-energy (Bahor et al. 2009). All of these options occur further up the waste hierarchy and are therefore preferred for limiting environmental impacts (EC 2008). Landfills are a significant source of methane emissions, the magnitude of which will likely remain uncertain given spatial and temporal variability. Acknowledging costs associated with alternatives mentioned, it is imperative this source is not understated due to this associated uncertainty. Moreover, results of this study indicate despite reported estimates, landfill methane emissions are likely underreported in the US. It is not clear to what extent this inference should extend within and beyond the US. Thus without researching the nature of discrepancies observed between T-D and B-U estimates, the long-term environmental costs of delaying sweeping changes in overall waste management will remain unknown.

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Personal Communication

Bogner, J. Professor of Earth and Environmental Sciences, University of Illinois at Chicago. Email communication, subsequent informal interview. 20 May 2020.

Börjesson, G. Researcher at Department of Soil and Environment, Swedish University of Agricultural Sciences. Email communication. 7 May 2020.

Spokas, K. Research Soil Scientist, United States Department of Agriculture, Agricultural Research Service. Email communication, subsequent informal interview. 8 May 2020.

Appendix

Please note: blank spaces in Table A1 and Table A2 indicate data that was not available from analysis of the T-D study indicated.

Table A1. Top-down studies conducted in US

Study	Year	Scope	Notes on measurement methods	Inventory Source	Inventory Overall (Tg CH_4/y)	Inventory Landfill (Tg CH_4/y)	Measured Overall (Tg CH_4/y)	Measured Landfill (Tg CH_4/y)
Zhao et al.	2009	Central California	Tower-based	EDGAR/CARB*	1.308	0.293		
Wunch et al.	2009	SoCAB	Ground-based FTS	CARB	~ 0. 26	0.6 (CA Total)	0.6 ± 0.1	
Mays et al.	2009	Indianapolis	Airborne					
Hsu et al.	2010	SoCAB	Ground-based monitoring	CARB	1.43		2.0 ± 0.056	
Wennberg et al.	2012	SoCAB	Airborne	CARB	0.212		0.44 ± 0.15	0.086 (entire SoCAB)
Peischl et al.	2013	SoCAB	Airborne	CARB	0.301	0.05	0.411 ± 0.037	0.047 ± 0.013
Miller et al.	2013	Continental US	Tower-based	EPA	~ 22.1		33.4 ± 1.4	
Jeong et al.	2013	California	Tower-based network	CARB	1.33	0.314	1.57 ± 0.1	0.687 ± 0.187
Kort et al.	2014	Four Corners Region (US)	Satellite (SCIAMACHY)	GHGRP	0.33		0.59	
Johnson et al.	2014	Northern California (SFBA + SJV)	Airborne	CARB	1.51	0.39	1.93	0.821
Wecht et al.	2014	North America	Airborne					
Wecht et al.	2014	California	Airborne	CARB	1.51	0.39	2.86 ± 0.21	1.05
Fairley & Fischer	2015	SFBA	Tower/ground-based	BAAQMD	0.126		0.24 ± 0.06	
Cui et al.	2015	SoCAB	Airborne	NEI	0.23	0.197	0.406 ± 0.81	0.347 ± 0.071
Cambaliza et al.	2015	Indianapolis	Airborne	EPA GHGRP		0.014	0.068 ± 0.029	0.023 ± 0.007
McKain et al.	2015	Greater Boston	Tower/ground-based	Custom	0.132	0.063	0.333	
Karion et al.	2015	Barnett Shale	Airborne	EPA GHGRP	0.666 ± 0.114		0.158	
Turner et al.	2015	Continental US	Satellite (GOSAT)	EDGAR	25		42.8	
Wong et al.	2015	Los Angeles	Ground-based FTS	CARB	0.28		0.39 ± 0.06	
Wunch et al.	2016	SoCAB	Ground-based FTS	CARB	0.453 ± 0.910		0.413 ± 0.086	
Jeong et al.	2016	California	Tower-based network	CARB	1.64	0.335	2.42 ± 0.49	~ 0.435
Wong et al.	2016	SoCAB	Tower-based	CARB			~ 0.342	
Lamb et al.	2016	Indianapolis	Airborne & Tower-based	Custom/EPA GHGRP	0.029	0.015	0.041 ± 0.012 & 0.081 ± 0.011	0.015 ± 0.007
Cui et al.	2017	San Joachin Valley	Airborne	CARB	0.701		1.183 ± 0.245	
Jeong et al.	2017	SFBA	Tower-based network	BAAQMD	0.126	0.068	0.225 ± 0.051	0.116
Heimburger et al.	2017	Indianapolis	Airborne	Custom/EPA GHGRP	0.029	0.0139	0.034 ± 0.020	0.016

Ren et al.	2018	Baltimore/DC	Airborne	EPA GHGRP/GHGI	0.097	0.035	0.278 ± 0.066	0.064 ± 0.038 -- 0.071 ± 0.042***
Hedelius et al.	2018	SoCAB	Tower-based network & satellite	CARB/EPA GHGRP		0.144	0.360 ± 0.090	
Alvarez et al.	2018	US O/NG Sector	Airborne measurements for validation of facility-level data	EPA GHGI	8.1 (6.8-10.0)		13 ± 2.1	
Ryoo et al.	2019	Sacramento, California	Airborne	EPA GHGI**	0.135 ± 0.064		0.087 ± 0.009	
Kuwayama et al.	2019	SoCAB	Tower/ground-based	CARB	0.16	~ 0.080	0.181	
He et al.	2019	SoCAB	Tower/ground-based FTS	CARB			0.275 ± 0.013	
Plant et al.	2019	East Coast Cities	Airborne	EPA GHGI**	0.37		0.89	

*For landfill inventory estimate
**Gridded version of 2012 EPA GHGI by Maasakkers et al. (2016)
***Second estimate included landfills emissions not estimates using AMB measurements

Table A2. Top-down studies conducted in EU

Study	Year	Scope	Notes on measurements methods	Inventory Source	Inventory Overall (Tg CH_4/y)	Inventory Landfill (Tg CH_4/y)	Measured Overall (Tg CH_4/y)	Measured Landfill (Tg CH_4/y)
Kuc et al.	2003	Krakow (Poland)	Tower/ground-based				0.012	
Van der Laan et al.	2009	the Netherlands	Tower-based	Dutch National Inventory (MNP)	0.69 ± 0.124		0.573 ± 0.015	
Zimnoch et al.	2010	Krakow (Poland)	Tower/ground-based				0.0045	
Manning et al.	2011	UK	Tower-based	NAEI	3.631		2.429	
Gioli et al.	2012	Florence (Italy)		Regione Toscana Inventory (IRSE)				
Schmidt et al.	2014	Órleans Forest (France)	Tower-based	French National Inventory (CITEPA)				
Ganesan et al.	2015	UK & Ireland	Tower-based network	NAEI	2.995	1.008	~2.71	
Lopez et al.	2015	Central France	Tower-based	French National Inventory (CITEPA)	0.156		0.182 ± 0.104	
Bergamaschi et al.	2015	EU-28	Tower/ground-based	National Inventory values reported to IPCC	12.08	1.17-4.94	16.0-19.4	
Helfter et al.	2016	London (UK)	Tower-based	LAEI	0.001		~ 0.003	
Palmer et al.	2018	UK	Tower-based network & airborne & satellite & ship-based	NAEI				
Pawlak & Fortuniak	2016	Łódź (Poland)	Tower-based				0.001	
Zazzeri et al.	2017	London (UK)	Tower-based	NAEI				

Pison et al.	2018	France	Tower-based network	IER/ French National Inventory (CITEPA)	3.108 (IER) 2.43 ± 0.637 (CITEPA)	0.522	3.570-4.193	0.46
Bergamaschi et al.	2018	EU-28	Tower-based network	National Inventory values reported to IPCC	18.8 (2006) -21.3 (2012)		19.7 (12-27.4)	
Connors et al.	2018	East Anglia (UK)	Tower-based network	NAEI	0.278		0.311 ± 0.063	
Zimnoch et al.	2019	Krakow (Poland)	Tower-based	EDGAR	0.006		0.004	
Pitt et al.	2019	London (UK)	Airborne	NAEI	0.075		0.0921	
Helfter et al.	2019	UK & Ireland	Ship-based	BEIS (UK) & EPA (IRL)	2.29 (UK:1.76 + IRL: 0.53)	0.929 (UK only)	2.55 ± 0.48	
Wunch et al.	2019	Region of EU (France, Germany, Poland)	Ground-based	EDGAR	3		2.4 ± 0.3	
Xueref-Remy et al.	2020	Paris (France)	Mobile-measurement	Paris Inventory (AIRPARIF)	0.037	0.016		
Venturi et al.	2020	Florence (Italy)	Tower-based	Regione Toscana inventory (IRSE)	0.001 - 0.003	0		