The impact of landfilling and composting on greenhouse gas emissions – A review

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ABSTRACT

Municipal solid waste is a significant contributor to greenhouse gas emissions through decomposition and life-cycle activities processes. The majority of these emissions are a result of landfilling, which remains the primary waste disposal strategy internationally. As a result, countries have been incorporating alternative forms of waste management strategies such as energy recovery from landfill gas capture, aerobic landfilling (aerox landfills), pre-composting of waste prior to landfilling, landfill capping and composting of the organic fraction of municipal solid waste. As the changing global climate has been one of the major environmental challenges facing the world today, there is an increasing need to understand the impact of waste management on greenhouse gas emissions. This review paper serves to provide an overview on the impact of landfilling (and its various alternatives) and composting on greenhouse gas emissions taking into account streamlined life cycle activities and the decomposition process. The review suggests greenhouse gas emissions from waste decomposition are considerably higher for landfills than composting. However, mixed results were found for greenhouse gas emissions for landfill and composting operational activities. Nonetheless, in general, net greenhouse gas emissions for landfills tend to be higher than that for composting facilities.

1. Introduction

Prior to the mid-1990s, waste management (WM) issues tended to revolve around public heath and safety issues (Marshall, 1972; Tchobanoglous and O'Leary, 1994), and the optimization of landfill gas (LFG) capture and utilization (Komilis et al., 1999a). The subject of its impact on greenhouse gases (GHG) has only been addressed recently, making it a relatively young study. This reflects an increasing acknowledgement of wastes contribution to GHG which is reflected in recent development in GHG mitigation policies internationally (SoE, 2007; Themelis and Ulloa, 2006; European Communities, 2001).

The waste sector is a significant contributor to greenhouse gas (GHG) emissions accountable for approximately 5% of the global greenhouse budget (IPCC, 2006). This 5% consist of methane (CH_4) emission from anaerobic decomposition of solid waste and carbon dioxide (CO_2) from wastewater decomposition (IPCC, 2006). Life cycle activities associated with these WM strategies are not included in this estimation. However, for a more holistic approach, streamline life cycle activities should also be accounted when quantifying a WM strategy impact on GHG emissions.

The most common WM strategy today is landfilling (Hogg et al., 2008; Pipatti and Wihersaari, 1998) and is expected to increase due to developing countries movement away from open dumping to landfilling (Meadows et al., 1997). However, as a result of increased environmental awareness, there has been an emergence of alternative and "add-on" features to landfilling such as energy recovery from LFG recovery, aerox landfills, pre-composting of waste prior to landfilling and compost capping; all of which have various impacts on GHG emissions. More importantly, there has been a movement to divert waste from landfills in order to reduce the negative environmental impact of landfills such as leachate contamination, GHG emissions and space limitation (Slater and Frederickson, 2001; Norbu et al., 2005). Composting has thus been widely acknowledged as an alternative to landfills.

In this review, the impact of landfilling and some of its more common mitigation applications, alongside with the impact of composting on GHG emissions are reviewed. This summary of information allows for the start and/or a further understanding of the consequences different WM strategies has on climate change; thus allowing decision makers to make more climate responsible decisions with regard to WM.

2. Landfill

The impact of landfills on climate change lies in their emissions of CO_2 and CH_4, along with several other gaseous components. These gases are the by-product of anaerobic decomposition of organic waste, characteristic of conventional landfills, and tend to accumulate within the landfill (USEPA, 1998). Only CH_4 is accounted for in the estimation of GHG emissions from landfills

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and not CO_2 despite its global warming potential (GWP) upon release (IPCC, 2006; DCC, 2007). This is due to the general consensus that CO_2 from waste decomposition is of biogenic origin and hence does not add to the overall GHG emissions that contribute to global warming (IPCC, 2006).

There are two life stages in a landfill, its operating stage, where municipal solid waste (MSW) is being disposed of, and its closed stage, where storage capacity is reached. Operating landfills emits more CH_4 than closed landfills due to the majority of degradation occurring in the first few years following disposal with decreasing emission rates with time after closure (Fourie and Morris, 2004; Humer and Lechner, 1999a). Following closure, a landfill continues to emit GHG, possibly for several hundreds of years (Borjesson et al., 2004). The general trend of GHG emissions from landfills can be seen in Fig. 1. It can be observed from Fig. 1 that even if landfills were to be removed right now, the issue of CH_4 generating in current landfills will still remain for many years. Hence, landfill managers today are faced with the challenge of dealing with the GHG emissions from current and old landfills.

Various independent theoretical and experimental studies suggest a large variation of GHG generation from 1 ton of waste, ranging from $40 \, m^3$ to $250 \, m^3$ (Humer and Lechner, 1999a; Ayalon et al., 2000; Bogner et al., 1997; Themelis and Ulloa, 2006). This is understandable as LFG generation is highly dependent on a variety of factors which are covered in Komilis et al. (1999a). One of the most crucial factors determining GHG emissions is waste composition. The organic fraction of waste stream is responsible for the emission of GHG and different types of organic waste have their own associated degradable organic carbon (DOC) and rate constants which affect the total amount emitted, and the rate at which is generated, respectively (DCC, 2008). The rate of emissions can be controlled with strategies such as waste compaction, leachate recirculation (Cabaraban et al., 2008) and even aerobic landfilling (Read and Hudgins, 2000). Despite the rate of emissions, the potential emissions from a conventional landfill from MSW, from Australian default values, is approximately 1.287 ton CO_{2-e}/ton of waste. This value can be lowered by adopting different mitigation strategies that converts CH_4 to CO_2 such as waste-to-energy, and pre-composting of waste prior to landfilling.

CH_4 accounting aside, GHG emissions from landfilling activities are also an important contribution to the overall impact of landfilling on GHG emissions. Emissions are a result of transportation, excavation, compaction and soil spreading. Baky and Eriksson (2003) estimated a fuel consumption of 1.036×10^{-3} kL/ton waste landfilled

for a conventional landfill while WARM, designed by the USEPA (2006) estimated 0.72 ton CO_{2-e}/ton waste landfilled and McDougall et al. (2001) provided a fuel consumption estimate of 0.6 L/m^3 of void space. Different landfill operations will result in different operational emissions; hence, the choice of landfill operations will affect the net emissions. This raises the question of whether certain landfill mitigation strategies, despite their ability to reduce GHG emissions by CH_4 conversion, will be able to achieve net lowered GHG emissions, when operational activities are considered too.

2.1. Landfill gas capture

The most common mitigation strategy is the capture of LFG for flaring or combustion to recover energy as this present significant environmental, economic and energy benefits (El-Fadel and Sbayti, 2000). As of 2001, there were approximately 955 landfills collecting LFG worldwide (Themelis and Ulloa, 2006) as compared with an estimate of 400 sites in 1995 (Meadows et al., 1997). This shows that countries are taking a step in the right direction towards mitigating GHG from landfills. Despite an increase in LFG capture in recent years, the number of landfill sites recovering energy in Australia is still relatively low, ranging from 4% to 19% (WMAA, 2008). However, with legislative requirements such as the National Greenhouse and Energy Reporting Act 2007 that specifies an emission threshold of 25 kilo-ton of carbon dioxide equivalent units (expressed as CO_{2-e}) for a landfill facility (Wong, 2008), LFG capture and utilization will become a more promising option to fulfil that requirement.

There are two conventional fates that entail captured LFG: the biogas is flared or combusted to generate electricity (aka waste-to-energy i.e. WtE). The primary objectives of these practices are to recovery energy and protect the people and the environment from the volatile nature of LFG, respectively. GHG reduction, through the conversion of CH_4 to CO_2, is a secondary effect as a result of these actions. Recovery efficiency from landfill depends largely on the operating conditions and the age of the waste, though independent studies have yielded varying results. Pipatti and Wihersaari (1998) stated efficiencies to vary between 50% and 100% while Oonk and Boom (1995) reported 24% to 60% and Humer and Lechner (1999b) reported 40% to 60% efficiency. Laboratory experiments by Bogner and Spokas (1995) yielded values ranging 25–50%. This range of recovery efficiency is largely dependent on the waste composition and moisture content. Waste streams with a low organic fraction, e.g. food waste, will result in
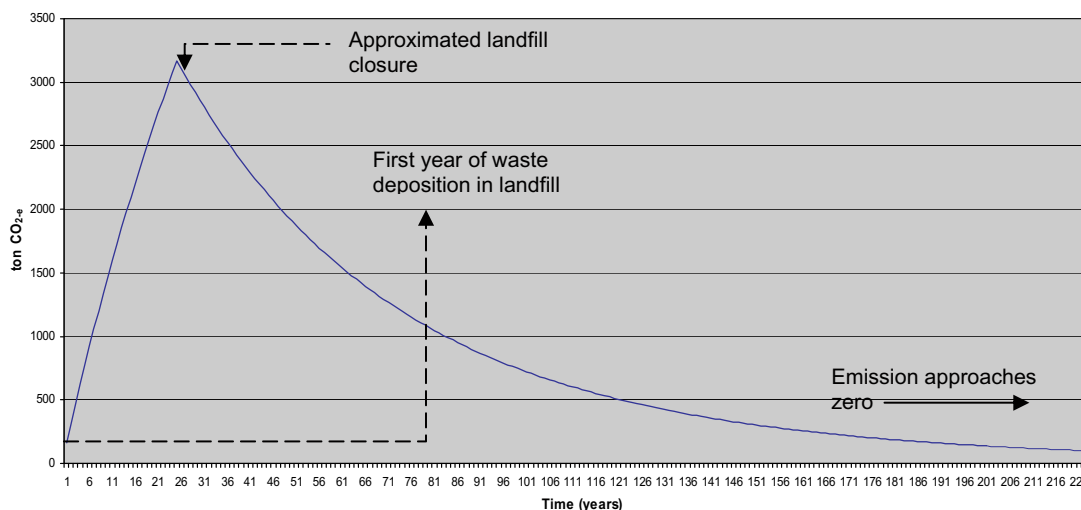

Fig. 1. General trend of CH_4 emission from landfills in their operating post closure years (calculated using the IPCC 1st order decay model).

lower capture efficiency, which others with a higher DOC such as paper or waste sludge are able to achieve higher capture efficiencies due to the amount of biogas produced (DCC, 2008).

Knowing the efficiency of biogas capture can have enormous implications for the GHG mitigation potential. This is illustrated by a study in Phuket, where a 50% recovery of LFG was recorded which led to a GWP reduction around 58% of the total landfill's GWP (Liamsanguan and Gheewala, 2008). This study suggests the possible large mitigation potential a landfill can bring if gas recovery efficiency was improved. Energy recovery in particular was found to have a net negative GWP (Themelis and Ulloa, 2006) and offered the most cost-effective method in reducing GHG emissions from landfills making it a promising WM option with regards to GHG mitigation (Pipatti and Wihersaari, 1998).

2.2. Aerobic landfilling (Aerox landfills)

As oppose to the "dry tomb" approach of traditional landfills where MSW decomposition is inhibited, some have opted to speed up the rate of decomposition in situ by introducing air into landfills (USEPA, 1998; Cossu, 2003). This effectively converts the anaerobic site to an aerobic state, acting like a composter. Aerobic respiring bacteria converts the organic fraction of MSW to mostly CO_2 and H_2O instead of CH_4, reducing the GWP of the landfill (Read et al., 2001a). Studies have shown a reduction of CH_4 concentration by at least 80% at the start of the aerobic process and remained consistently below 15% (Read et al., 2001b), with 90% reduction achieved in some cases (Cossu, 2003).

Aerobic landfilling was initiated with the principle objectives of: (1) leachate control (Read et al., 2001a), and (2) rapid waste stabilization (Stessel and Murphy, 1992; Read et al., 2001b). One application of aerobic landfills is bioreactors which incorporate leachate recirculation within the landfill to accelerate waste decomposition (Read and Hudgins, 2000). This can increased the LGF collection efficiency by 10% from LFG capture without leachate recirculation (Barlez et al., 2003). However, despite a greater rate of CH_4 production and energy recovery, prolong recirculation of leachate has been shown to trigger nitrous oxide (N_2O) production even when recirculation has ceased (Watzinger et al., 2005; Lee et al., 2002). With N_2O having a GWP 310 time of CO_2, this would be significant. The amount of leachate and the rate of its circulation will have to be monitored closely to ensure N_2O emission is minimized. Watzinger et al. (2005) has reported the threshold for leachate application to avoid increased GHG concentration is approximately 200 mm. The amount of leachate application will have to be balanced to ensure that maximum biogas recovery is obtained without causing extensive N_2O production.

Due to increased rate of waste stabilization and LFG gas capture efficiency, energy recovery from aerobic landfills is noted to be higher than anaerobic landfills (Cabaraban et al., 2008). Cabaraban et al. (2008) accounted and energy recovery of 9.6 MJ/kg of waste at a certain point of measurement, while Lou (2008) reported a potential average of 4.108×10^3 MJ/kg of waste recovered over the lifespan of the waste. Although aerobic landfills provide the opportunity for increase energy recovery, there comes the associated fuel consumption needed to introduce air and/or leachate into the landfill, which is absent in a sanitary landfill or a conventional WtE landfill. Energy consumption for aerobic landfill operations is quoted by White et al. (1995) to be 0.198 MJ/kg of waste, while that for conventional landfill was reported to be 3.99×10^{-3} MJ/kg of waste (Baky and Eriksson, 2003). This shows that the fuel consumption and hence GHG emissions from aerobic landfills is likely to be considerably higher than the operational requirement of conventional landfills. Whether this elevated GHG emissions from increased operational requirements can be offset by energy recovery should be quantified with further studies.

Depending on the landfill site, aeration of the landfill may be feasible at different stages of landfill operation. Early aeration means that WtE is forfeited, but may be suitable for landfills where WtE is unfeasible. Late aeration is more common as it allows for energy recovery and continues to mitigate CH_4 emissions when the production of CH_4 has plateau and is no longer cost-effective to continue operation. Should aerobic landfilling be carried out, consideration should be made to the type of energy, fossil fuel or alternatives, needed to operate the facility. As WtE is no longer an option once air is introduced, it may be that aerobic landfilling emits more GHG than mitigated in the grand scheme, should fossil fuel be used as the energy source. Hence, it would also be valuable to evaluate the net GHG mitigation potential of aerobic landfilling, taking into account external factors such as operational energy emissions, to determine if aerobic landfilling is truly sustainable.

In semi-arid climates, natural decomposition is slow, resulting in large C storage and little GHG emissions in the short-medium term (Bogner et al., 2007; Humer and Lechner, 1999b). Aerobic landfilling will thus cause more GHG emissions in the short-term as degradation is accelerated. Conversely, GWP will be reduced significantly in the tropics as degradation is rapid anyway (Kumar et al., 2004). Several European and Asian countries have deployed their own aerobic landfills studies (Hudgins, 2000). It would be useful to consolidate these findings to evaluate the effectiveness of aerobic landfilling on CH_4 emission reduction under different climates.

2.3. Landfill cover – compost application

Following closure, a landfill continues to emit LFG, which poses odour problems and increases the GWP of the landfill (Tchobanoglous and O'Leary, 1994). As a result, landfill covers are sought to resolve this. Conventional methods of degasifying closed landfills rely on non-organic materials, such as clay, which is designed to limit infiltration into the waste, thereby retarding decomposition in an attempt to stifle gaseous production and emissions into the atmosphere (Sadek et al., 2007; Chanton and Liptay, 2000). This has inherent problems, as these inert materials are bound to break down eventually, especially in semi-arid climates (Sadek et al., 2007), reducing the cover's efficiency. Hence, biological covers such as compost are turned to as an alternative landfill cover.

As oppose to clay covers, compost covers aim to improve water filtration into the landfill. This infiltration aids in the decomposition of waste and establishment of an optimal functional methanotrophic community in the compost that oxidises CH_4 into CO_2 (Stern et al., 2007). Although CH_4 is generated within the anaerobic regions, the majority of its uptake is performed by aerobes which are most densely located throughout the upper layer of the compost (Whalen et al., 1990). The first major field trial assessment of the CH_4 removal effectiveness of compost as a landfill cover is ongoing at an Austrian landfill (Humer and Lechner, 1999b, 2001). Results showed 100% CH_4 capture when compost is used in conjunction with gravel under bed, and 10% to 50% CH_4 capture when only compost was used (Humer and Lechner, 2001). Subsequent various studies have supported the concept of biological CH_4 removal via oxidation as an effective GHG reduction tool in landfill emissions (Stern et al., 2007; Hilger and Humer, 2003; Barlaz et al., 2004; Brown and Subler, 2007) and certainly an improvement from the clay cover with respect to CH_4 mitigation (Chanton and Liptay, 2000). Literature shows CH_4 removal rates ranges from 10% to 100% (Humer and Lechner, 2001; Stern et al., 2007; Zeiss, 2006; Chanton and Liptay, 2000; Whalen et al., 1990). This wide variation can be attributed to the various factors, such as temperature, nutrient level and moisture, controlling CH_4 oxidation (Borjesson et al., 2004). Hence, manipulation of these factors can contribute to increase in CH_4 oxidation, resulting in reduced GHG emissions.

In addition, negative CH_4 fluxes have also been reported in compost landfill cover indicating that the landfill surface is consuming, rather than emitting CH_4 (Bogner and Spokas, 1995; Bogner et al., 1997). Field verification of landfill cover soils functioning as CH_4 sink has profound implication for revision of landfill contribution to global CH_4 budget (Bogner and Spokas, 1995) and climate prediction models (Hilger and Humer, 2003).

Microbial oxidation of CH_4 in compost cover is a low-cost alternative to the conventional synthetic capping method in mitigating GHG from closed landfills (Humer and Lechner, 1999a). However, they are not suitable in all situations. For example, compost functionality may be inhibited in cold climates. Experiments by Zeiss (2006) observed a halt in microbial oxidation during the winter period, and did not resume oxidation until a month later. Zeiss (2006) suggested a method of filter bed heating, which resulted in double the emissions reduction in cold climate. However, considerations should be made as to whether operation of this heating device will offset the benefits achieved from the CH_4 reduction through the compost cover, with respect to GWP.

Further research of biotic cover is needed to address its durability and operational lifespan would directly influence the time span of which GHG is being mitigated in the landfill. It would also be of interest to investigate different strands of methanotrophs that may be inoculated into the compost to enhance CH_4 uptake. With the increased interest in phytoremediation, it would also be valuable to investigate the impact of compost cover on the ability of the vegetation in reducing landfill emissions. Would the compost cover enhance the growth of the vegetation? Or would the vegetation compete with the C consuming microbes for nutrients and water, so that the overall CH_4 consumption is decreased? Irregardless, compost cover remains as a competitive option in mitigating landfill CH_4 emissions under certain conditions.

2.4. Pre-composting prior to landfilling

There are two primary objectives for pre-treatment of waste before landfill disposal: (1) rapid stabilization of in order to reduce the landfilled waste volume, thereby prolonging landfill lifespan (Slater and Frederickson, 2001), and (2) to enhance initial methanogenic conditions in landfills (Komilis et al., 1999b; Leikam and Stegmann, 1999). In the course of attaining these objectives, overall CH_4 reduction is achieved (Leikam and Stegmann, 1999; Rieger and Bidlingmaier, 1995). This is achieved via 2 processes:

1. Pre-treatment induces aerobic environment which decomposes the organic fraction of MSW rapidly, releasing CO_2 gas and H_2O instead of CH_4. Binner (1995) found that pre-treating MSW halves the organic C of untreated waste. Thus, when this is later landfilled, CH_4 production is lessened.
2. As pre-treatment attains a thermophilic state, which favours rapid establishment of methanogenesis, moving this waste to an anaerobic condition (landfill) will temporary enhance CH_4 production (Brummeler and Koster, 1990; Brinkmann et al., 1995). This observation is supported by studies by Stegmann (1983), Stegmann and Spendlin (1987) and Norbu et al (2005), where results correlated on an increase in CH_4 production in landfills within the first few years with pre-treated waste as compared to one without. However, as composting has removed a portion of DOC, the total CH_4 production from pre-composted waste will be lessened as compared to a landfill that receives fresh waste.

With regards to overall GHG balance, it is generally agreed that a reduction of total gas production up to 90% can be achieved through pre-composting (Stegemann et al., 1995; Zach et al., 2000; Leikam and Stegmann, 1999). According to Tränkler and Ranaweera (2002), overall gaseous formation could be reduced by more than 35% and GWP would be abated by more than 63%, within 20 years of using pre-treatment technology in a landfill. However, further considerations have to be taken before implementation. With respect to process (1), although methanogenesis is enhanced in the landfill following pre-treatment, owing to the mineralization of some organic matter during composting, CH_4 yield are lower as compared to untreated MSW (Stegmann, 1983; Stegmann and Spendlin, 1987). Reduction in CH_4 production when MSW are later landfilled implies that gas recovery may not be economical if pre-treatment proceeds. Therefore, the duration of the pre-composting step is crucial to the overall GHG mitigation potential.

A short pre-treatment will allow some methanogenesis to occur in the landfill. Whether or not this remains economical is unknown. A longer pre-treatment will stabilize the waste to such a large extent that CH_4 potential is reduced significantly. A value of up to 90% reduction in CH_4 production was reported by Zach et al. (2000) and Stegemann et al. (1995). The time period to establish waste stabilization and hence reduced CH_4 production, varies widely between countries due to different climate influenced factors. For example, Komilis et al. (1999b) in Germany suggested a pre-treatment of one to two months to allow some generation of CH_4 while Norbu et al. (2005) in Thailand suggested a pre-treatment period of approximately one week, due to more rapid decomposition in tropical climate.

It can be seen that the suitability and duration of anaerobic pre-treatment of waste prior to landfilling are dependent on the LFG recovery objectives. This technology may be appropriate for landfills unsuitable for LFG recovery; in which MSW can be pre-treated till highly stabilized. This allows significant reduction in gaseous emissions and alternative strategies (such as landfill cover) can be sought to further mitigate any residual CH_4 emissions.

3. Composting

Composting has been concerned with the efficient and economic production of a product, while functioning within a larger process oriented approach to solid WM (Zeman et al., 2002; Murphy and Power, 2006). One of the conundrums which has emerged due to this product versus process, is that much of the composting research has been focused with the goal of improving production and the quality of the finished product, while the issue of composting process's contribution to GHG emissions is often neglected.

Aerobic decomposition from well managed composting results in the emission of CO_2 and H_2O. Due to the heterogeneous nature of a compost pile, some CH_4 may form in anaerobic pockets within the pile (Bogner et al., 2007; Brown and Subler, 2007). However, studies have shown that the majority of this CH_4 emission oxidizes to CO_2 in aerobic pockets and near the surface of the compost pile, making CH_4 emission negligible (Zeman et al., 2002; Brown and Leonard, 2004). This claim is refuted by Edelmann and Josss (1999), Amlinger et al. (2008) and Rynk and Richards (2001), who reported considerable CH_4 emission even in well managed systems. This is probably due to various variables controlling the nature of the compost piles.

GHG emission from the composting process is highly dependent on the feedstock. Feedstock for composting ranges from greenwaste to MSW to manure. Waste stream with a higher DOC will result in higher GHG emissions from aerobic decomposition, while waste with lower rate decay constant will require a longer period of composting, which may cause a difference in the operational requirements to compost that waste stream. Theoretical calculations by White et al. (1995) estimated approximately 0.323 ton CO_{2-e}/ton of mixed waste, while that from Lou (2008) estimated

a value of 0.284 ton CO_{2-e}/ton of mixed waste, taking into account the emissions from operational activities. Practical GHG measurements from alternatives studies have produced emissions from composting operations ranging from 0.183 ton CO_{2-e}/ton of mixed waste to 193.2 ton CO_{2-e}/ton of mixed waste (Jackson and Line, 1997; Jakobsen, 1994) which are lower than those predicted from theoretical calculations. This suggests theoretical estimates may tend to overestimate what is actually emitted. These emission estimates suggests the importance of operational activities on the impact of composting on GHG emissions. Composting tends to occur via windrows composting or aerobic in-vessel composting, the former tending to have lower associated operating emissions (White et al., 1995; Lou, 2008). For example, aerobic in-vessel composting has been reported to consume 0.198 MJ/kg of waste, while windrows composting consumed a reasonable reduced 0.071 MJ/kg of waste (White et al., 1995). Emissions from operational activities vary with reports due to the type of composting operations, the amount of waste treated and other variations, an average emission from operation of composting if thus difficult to quantify. As a brief indication of the emissions from windrow operations, White et al. (1995) reported 0.071 MJ/kg of waste, Baky and Eriksson (2003) estimated 0.00151 MJ/kg of waste, and Franklin Associates (1994) estimated 0.134 MJ/kg of waste. In order to obtain an accurate estimation of a composting facility GHG emission, it would be best to undertake a site specific GHG inventory instead of taking an average from the literature.

Recently, there has been a renewed interest in the emission of N_2O from composting (Beck-Friis et al., 2000; He et al., 2000). N_2O can be produced by either incomplete ammonium oxidation or via incomplete denitrification (Beck-Friis et al., 2000). Results of N_2O generation have been mixed. He et al., (2000) reported high emissions at the beginning of composting, and negligible amount after the initial 2 days, while Beck-Friis et al., (2000) reported higher emissions with time. Little is known regarding N_2O emission, however the fact that it has a GWP 310 times that of CO_2, makes its study justifiable (IPCC, 2006).

Despite potential CH_4 and N_2O emissions, these are usually not taken into account in C budgeting. In fact in the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) and the Australian Methodology for the Estimation of Greenhouse Gas Emissions and Sinks (DCC 2007), only CO_2 is assumed to be emitted during composting. Even so, although this CO_2 is climate active when released, it is generally not considered when it comes to GHG accounting. This is due to the general consensus that CO_2 from composting is of biogenic origin and hence does not add to the overall GHG emissions that contribute to global warming (IPCC, 2006). Therefore, when C accounting is performed by composting facilities, their GHG contribution is a result of operational activities, as opposed to the actual decomposition process. Nevertheless, there are some that advocate the inclusion of all GHG emissions derived from non-fossil C (Hogg et al., 2008). The rationale behind this inclusion is that the climate responds the same whether the source is no differently to fossil or non-fossil C, and thus all emissions should be accounted for on a like-for-like basis.

Due to the exclusion of CO_2 from composting as a GHG contribution, there have been relatively fewer studies devoted to composting contribution to GHG emissions than for landfills; and the need for this is often questioned by the composting waste sectors. This is because, unlike landfilling, where there is increasing pressure to quantify their GHG contribution, there is little legislative pressure for composting facilities to quantify their GHG contribution (Wong, 2008). However, from an environmental perspective, the negligence of composting contribution to GHG emissions is questionable, especially from a life cycle analysis approach. This is especially relevant to centralized composting systems where operational activities are energy consuming (ROU, 2007). The

question is how can composting contribution to GHG emissions be objectively quantified using a life cycle approach that will be acceptable to the scientific and managerial community? Through logical reasoning, most waste managers and literature agree that with regards to landfilling, the majority of their GHG impact lies with the degradation of waste so much that, emissions from operational activities are negligible (Chen and Lin, 2007; Wilson, 2002). The converse may be said for composting. However, there has been few comparative studies between landfilling and composting, utilizing a life cycle approach to justify this statement.

Some centralized composting facilities are driven to produce high quality compost that can be used for land application (Wei et al., 2000; Murphy and Power, 2006). The production of compost, especially high quality compost, requires energy. Source separation of MSW received at the composting facility is needed prior to landfilling. This activity is carried out using heavy machineries which makes emissions of GHG inevitable. Machines are essential to grind and mix the waste material, set up compost piles, aeration *inter alia.* All these processes require energy which will have an associated equivalent GHG cost (Brown and Subler, 2007). An alternative would be to initial source separation at household level, therefore reducing the need for heavy machinery centralized separation and the associated GHG emissions (ROU, 2007). It should also be noted that, no matter how much waste is composted and recycled, a portion of waste will still remain that requires landfilling.

Nonetheless, the production of compost can help mitigate GHG by:

1. Decreasing the need of chemical fertilisers and pesticides; thereby reducing GHG emissions from the use of fossil fuel associated with their production and application (Cogger, 2005; Favoino and Hogg, 2008).
2. Allowing for more rapid growth in plants, thereby increasing C uptake and storage within the plant. Studies have shown the increase biomass of plants growth with compost application (Gonzalez and Cooperband, 2002; Lynch et al., 2005; Wei et al., 2000). This is a form of C sequestration which removes CO_2 from the atmosphere.
3. Sequestering C in soil that has received compost application (Favoino and Hogg, 2008; Mondini et al., 2007). The USEPA (2002) estimated approximately 50 kg C (183 kg CO_2) sequestered per ton of wet compost. On a large scale, this may be significant.
4. Improving tillage and workability of soil (thereby reducing emissions from fossil fuel that would otherwise be used to work the soil) (Favoino and Hogg 2008).

With the exception of C sequestration (point 3), these offsets have been recognized but actual quantification of them remains incomplete. The USEPA has derived an estimated figure of approximately 50 kg C (183 kg CO_2) sequestered per ton of wet compost (USEPA, 2002). This figure is however specific to the US, and to a particular soil type.

4. Carbon sequestration

Although C sequestration is not within the scope of this review, mention should be made as its impact on GHG mitigation is relevant to the landfilling strategies and land application of compost mentioned above. Landfills can be great C sinks owing to slow degradation of waste (Bogner et al., 2007; Pipatti and Wihersaari, 1998). The amount of C stored in a landfill is dependent on the deposited waste stream's DOC and their rate constant, both of which can be obtain from the IPCC (2006). Waste material that

tend to degrade rapidly such as food waste will not remain stable as stored C, while other materials such as wood finds difficulty in decomposing under anaerobic conditions in landfill but is easily compostable under aerobic composting (US Composting Council, 2008). This suggests the need to direct certain waste to appropriate waste management facilities in order to degrade waste efficiently. Compost application to land also allows for C sequestration. Although this provides a short-medium term solution to climate change mitigation, it may be perceived as simply a manner of buying time (Kirschbaum, 2006). The importance of C sequestration is represented by Weitz et al. (2002) calculations which demonstrated that when C storage is considered in determining the net GHG emitted, it dramatically offsets the emissions from landfill operations. This is supported by many who suggest the probability of landfilling as a more competitive option with respect to climate change when C storage is included; especially when coupled with WtE (Pipatti and Wihersaari, 1998; Bogner et al., 2007). This would greatly influence how landfilling strategies are perceived with regards to climate change mitigation.

5. Conclusion

Despite the various waste management strategies ability to reduce GHG emissions from conventional landfills, a certain amount of GHG will inevitably be produced. This literature review suggests GHG emissions from waste decomposition are greatly higher for landfills than for composting system. This is because anaerobic decomposition facilitates the production of CH_4 which has a GWP 25 times that of CO_2, Energy recovery and appropriate landfill capping strategies can help to reduce this impact, but composting still remains a simpler and effective mean of reducing GHG emissions. This brings about the fundamental difference between landfilling and composting – landfilling, together with its mitigation strategies are often reactive measures in environmental protection, seeking to remediate harmful effects it has caused. Conversely, composting adopts a more proactive approached, with an objective to prevent or minimise such negative impacts in the first place.

With regards to emissions from operational activities, results are mixed. Depending on the landfill operations and composting operations, composting may emit more GHG emissions than landfill activities. For example, aerobic in-vessel composting has approximately doubled the operational fuel consumption than aerobic landfills. In general, when the sum of GHG emissions is considered, factoring decomposition and operational emissions, landfills appears to have a heavier impact on GHG emissions than composting. However, the existence and continual usage of landfills now and in the future cannot be denied. Not all waste can be composted or recycled, and a certain portion of waste will inevitable be landfilled. Nonetheless, countries should always aim to reduce the amount of waste produced and divert waste to appropriate waste management facilities so as to ensure overall operational and environmental sustainability.

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