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Carbon – Making the right choice for waste management in developing countries

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Abstract

Due to initiatives such as the clean development mechanism (CDM), reducing greenhouse gas emissions for a developing country can offer an important route to attracting investment in a variety of qualifying project areas, including waste management. To date CDM projects have been largely confined to schemes that control emission from landfill, but projects that avoid landfilling are beginning to be submitted.

In considering the waste options which might be suitable for developing countries certain ones, such as energy from waste, have been discounted for a range of reasons related primarily to the lack of technical and other support services required for these more sophisticated process trains. The paper focuses on six options: the base case of open dumping; three options for landfill (passive venting, gas capture with flaring, and gas capture with energy production), composting and anaerobic digestion with electricity production and composting of the digestate. A range of assumptions were necessary for making the comparisons based on the effective carbon emissions, and these assumptions will change from project to project.

The highest impact in terms of carbon emissions was from using a sanitary landfill without either gas flaring or electricity production; this was worse than the baseline case using open dumpsites. Landfills with either flaring or energy production from the collected gas both produced similar positive carbon emissions, but these were substantially lower than both open dumping and sanitary landfill without flaring or energy production. Composting or anaerobic digestion with energy production and composting of the digestate were the two best options with composting being neutral in terms of carbon emissions and anaerobic digestion being carbon negative. These generic conclusions were tested for sensitivity by modifying the input waste composition and were found to be robust, suggesting that subject to local study to confirm assumptions made, the opportunity for developing CDM projects to attract investment to improved waste management infrastructure is significant. Kyoto credits in excess of 1 tCO_2e/t of waste could be realised.

1. Introduction

As with most other global activities, waste management has its rich and poor relations. In this paper, we propose to focus on the poor relations, the majority of whom live in the developing economies. However, it will be necessary to consider the rich relatives in order to put any waste management option in the appropriate context.

Under the United Nations Framework Convention on Climate Change (UNFCCC), three flexible mechanisms have been developed to assist countries with binding targets (Annex B countries of the Kyoto Protocol) to reduce their greenhouse gas (GHG) emissions. Furthermore, within the Annex B countries a range of measures have been implemented in national regulatory frameworks to ensure that the agreed targets are met. For example, in Europe, one of the key issues of the landfill directive (LD) was to substantially reduce the biodegradable municipal waste (BMW) going to landfill and thus reduce uncontrolled emissions of methane, a major greenhouse gas (EC, 1999). Those countries in Europe taking the LD

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seriously have been exploring a range of options for BMW, which at one extreme includes the relatively simple windrow composting and at the other sophisticated systems for gasification, pyrolysis and energy from waste. As well as direct measures such as the LD aimed at reducing greenhouse gas emissions from waste management activities, mechanisms have been introduced to subsidise non-fossil energy generation which can also support the waste management system, for example, renewable obligation certificates (ROCs) for electricity generated from landfill gas, advanced thermal systems and combined heat and power (CHP) facilities (DTI, 2005). Furthermore, the European Union Emission Trading Scheme (EU-ETS) (EC, 2003) allows companies to sell carbon credits to the energy intensive industrial sectors that need to offset their own emissions to meet statutory targets and hence national targets of the Kyoto Protocol.

In considering the options available for waste management in developing countries (DCs) there are certain routes which can be eliminated fairly early on in the process of selection for a range of reasons which include:

- the characteristics of the waste (often high in organics and not suitable for waste to energy plants);
- the lack of technical support which would be required for the more sophisticated options;
- the lack of suitable technical, managerial and logistical infrastructure; and
- a lower financial base to operate and maintain expensive waste management systems.

To counteract what might appear to be these somewhat negative factors, there are certain general statements of a more positive nature which can be made that are applicable to many of the developing economies:

- the waste is high in organic matter and low in contaminants – highly suitable for composting and anaerobic digestion;
- waste densities are high – reducing the need for high cost, sophisticated compactor trucks for collection;
- the production rates per household are less than in developing countries – reducing the transport needs; and
- labour costs are low – increasing the incentives for recycling and scavenging.

Despite the above potential advantages, finance remains a major barrier. However, just as the Kyoto targets have led to measures that can support waste management in developed countries, developing countries can benefit through the clean development mechanism (Article 12 of the Kyoto Protocol), which permits projects that reduce greenhouse gas emissions to allocate the carbon reductions to companies in developed countries. The Kyoto credit scheme developed by the UNFCCC and the more mature EU ETS provides the mechanisms for exchange and, in principle, provides a more economically efficient way to meeting targets: reducing costs of obligated parties and transfer of funds to developing countries. In Europe, obligated companies face penalty charges of €40/t of CO_2 (€100/t in 2008) and EU ETS trading values have varied between €8 and €30/t over the last 2 yr. Kyoto credit prices have been lower than in Europe and both markets are currently trading at the bottom end of the range but, in the past and looking ahead, this offers a significant incentive for attracting investment. A baseline methodology has been developed for waste projects (AM0025, Avoided emissions from organic waste through alternative treatment processes (UNFCCC, 2006a)), and numerous projects related to landfill gas have already achieved success under the scheme (e.g., Nova Gerar, a Brazilian landfill gas to energy project won PointCarbon's best CDM project 2005 (Point Carbon, 2005)). In 2006, the first municipal solid waste (MSW) composting project was accepted under the CDM banner (composting of organic waste in Dhaka, Project Ref 0169, (UNFCCC, 2006b)). This paper reviews some of the problems faced by DCs in terms of waste management, assesses the net impact of various waste options on GHG emissions and confirms the potential for DCs to use CDM projects to attract the investment needed to improve waste management practices.

2. Urbanisation, poverty and waste management

Urbanisation induces a consumer based society; an increased concentration of people and industrial/commercial development implies an accumulation of waste, which for public health and environmental reasons needs to be properly managed and safely disposed of. The onus of managing and regulating this waste lies with local authorities. In developing countries 620,000 t/d of solid waste (approximately 226 million t/yr) will be produced from the one billion people living in slums alone (on average 0.6 kg per capita per day) (Shimura et al., 2001). These slum dwellers have no access to adequate water supply, sanitation or solid waste collection/disposal services. Even though per capita waste generation rates in developing countries is less than in higher-income countries, the capacity of the responsible local authorities to manage waste, from collection, to recycling or reuse and disposal, is limited.

It is acknowledged that greenhouse gas (GHG) content in the atmosphere has been increasing in the last decades due to anthropogenic activities. The rapid rates of urbanisation and growing volumes requiring collection and treatment are responsible for 18% of the total anthropogenic methane emissions globally (Table 1). In Sub-Saharan Africa, the World Bank (2006) is advocating that annual economic growth rates of 7% up to the year 2015 be reached and maintained, as well as an increase in infrastructure investments from 4.7% to 9% of GDP, in order to raise people out of poverty. For the 4.9 billion people living in less developed countries (UN-HABITAT, 2003), increased economic activity to meet these GDP targets will

Table 1
Global anthropogenic methane budget by source (EPA website)

Source of anthropogenic emissions (%)	
Rice	11
Natural gas	15
Coal	8
Oil	1
Solid waste	13
Manure	4
Wastewater	10
Fuel, stationary and mobile	1
Biofuel combustion	4
Biomass burning	5
Enteric fermentation	28

http://www.epa.gov/nonco2/econ-inv/international.html. Accessed 11th February 2007.

inevitably result in a considerable increase in municipal and hazardous waste; as increased GDP has been positively correlated to increasing per capita waste generation (ISWA, 2002; Bogner and Mathews, 2003). Therefore, pursuing economic growth in the interest of poverty alleviation also has significant repercussions on municipalities to provide basic services, of which waste management is only one of them.

2.1. Waste management in developing countries

In most developing countries, solid waste management is undertaken by the local authority, and the service includes waste collection (either from households or communal collection points) to final disposal. However, the low financial base and human resource capacity of these local authorities means that in most cases these authorities are only able to provide a limited service. For example in Nairobi 25% of waste is collected by the city council (UNEP, 2005), and in cities in China 36% is collected (ISWA, 2002). A study of five major urban centres (Accra, Kumasi, Tema, Tamale and Sekondi-Takoradi) in Ghana by UN-HABITAT (2001) revealed that on average, the local authorities are only able to manage 40% of the waste generated and that the poor performance was attributed to: accelerated urban growth; economic problems and inadequate funding of waste management by national government; inadequacy of available infrastructure and high service cost. In some of the larger cities, the private sector supplements the service, but in the small to middle-size towns where the profit margins for waste collection are smaller, the local authorities remain the only providers.

In general, solid waste management costs are covered indirectly through taxes, permits and rates. The lack of capacity within local authorities for billing and revenue generally results in a very low portion of revenue being collected and thus a low financial base to cover salaries and running costs associated with SWM (Ogawa, 1996). It is common to find old and broken down refuse collection vehicles and related equipment because the local authorities are unable to pay for the repair; this is not only as a result of lack of finances but also a poor choice of equipment in the first place, often by development agencies and national governments (UN-HABITAT, 2004; Coad, 2000) The poor operation and maintenance therefore leads to local authorities only being able to service a small area of the urban centres, in most cases on the central business districts. Urban residents who do not receive a waste collection service are forced to either bury their waste, burn it or dump it in open spaces.

Landfills are the main disposal option for most countries in the world, a World Bank study in 1997–98 (Johannessen and Boyer, 1999) on waste disposal methods in developing countries found that in Africa, Asia and in Latin America only four countries had engineered landfills with active pumping and flaring of landfill gas. In other cities and large towns in developing countries, most of the disposal sites are nothing more than open dumpsites. Open dumpsites are a resource for the urban poor; the scavenging and informal recycling of waste means that waste is being turned and thus aerated, and due to space limitation of the dumpsite waste it is often burnt thus emitting carbon dioxide, dioxins, furans etc. The depth of waste on the dumpsites is not generally very great and not only does this reduce the potential for CH_4 generation but also increases the opportunity for the methane to be oxidised through the relatively thick aerobic cover layer giving a lower CH_4 to CO_2 ratio.

2.2. Characterisation of urban waste in developing countries

Waste is biodegraded aerobically or anaerobically. Depending on the availability of oxygen during the degradation process and whether nitrogen is present in the appropriate form, the principal gaseous end products are carbon dioxide (CO_2) and/or methane (CH_4) and water, and to a lesser extent nitrous oxide (N_2O).

The growing volume of waste is contributing to the anthropogenic sources of GHG, which ultimately has an influence on climate change. Ten countries alone represent about 50% of global methane emissions from solid waste disposal (US EPA, 2006) – Fig. 1 represents the share of global methane emissions from solid waste. With continuing trends in population growth and urbanisation, developing countries accounted for 30–40% of methane emissions of this source in 2000 (US EPA, 2006). With developed countries rapidly adopting LFG utilisation and upstream recovery and treatment systems, the proportional contribution of DCs can only increase.

The quantities of GHGs emitted depend on the volumes of waste disposed and fraction of degradable organic carbon (DOC). A feasibility study on solid waste incineration for the largest cities in Kenya, Malawi and Zimbabwe (DFID, 1999) found that 75–80% of municipal solid waste was organic (putrescibles and paper). In India, 70% was organic (Yedla and Parikh, 2002), and reported values for Dhaka city (Bangladesh) from domestic properties were between 85% and 95% (JICA, 2005; BCAS, 1998). In developed

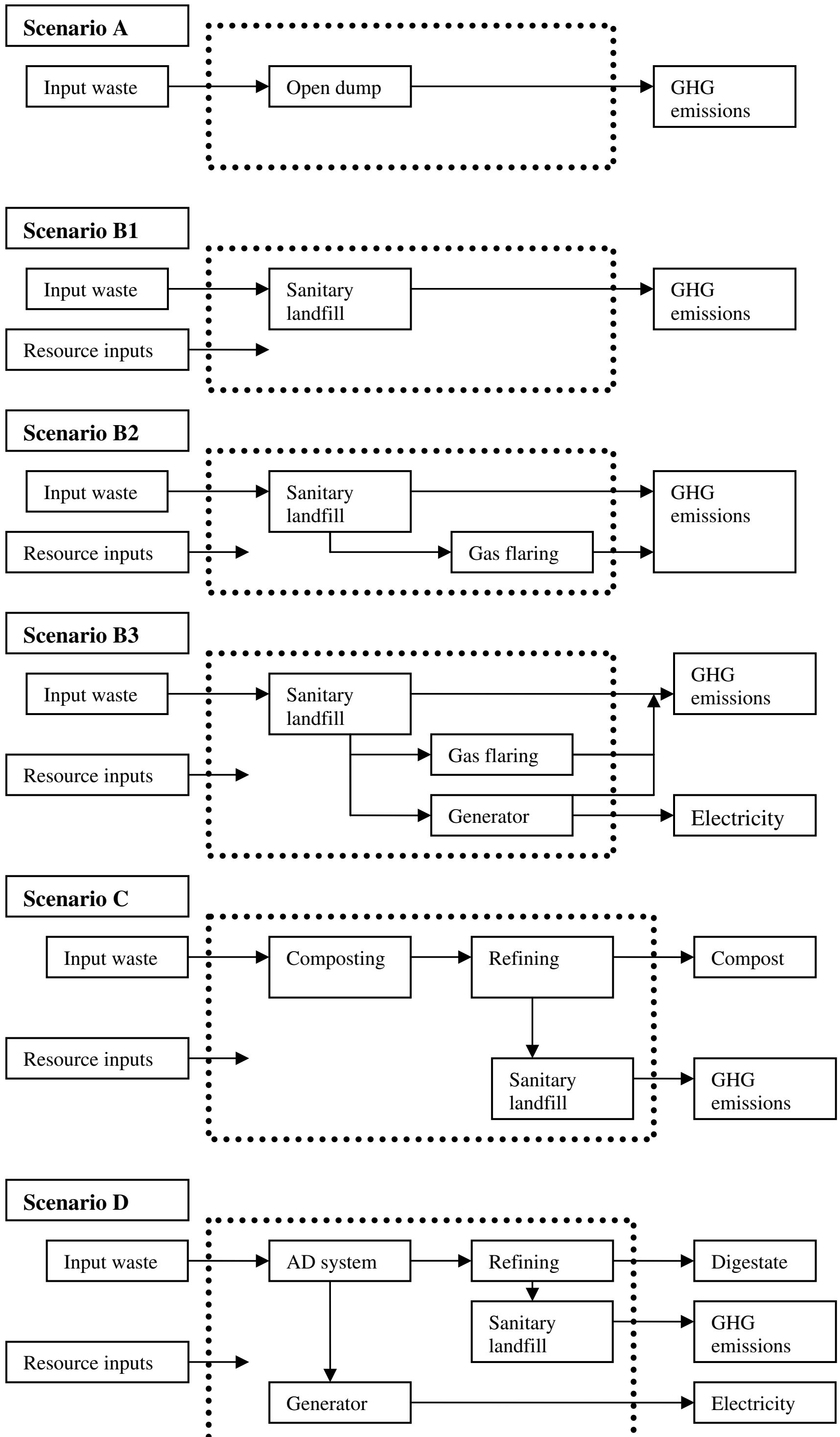

Fig. 1. Schematic system flow diagrams for the different scenarios.

countries, total organic content is lower, typically 60% of which in some cases only 10% is food waste and the rest is paper and cardboard (Tchobanoglous and Kreith, 2002). However, as paper and cardboard have much lower moisture contents, the DOC content is usually higher. Per capita waste generation for developing countries is lower than average, but the high rates of urbanisation and increasing poverty may have a considerable influence on inter country per capita waste generation. In Egypt for example, per capita waste generation from urban areas is 0.8 kg per capita per day while in rural areas it is 0.3 kg per capita per day (Badran and El-Haggar, 2006). In addition, the waste from DCs (especially lower income groups) has a higher density of around 400 kg/m^3 (Cointreau, 1982). Not only does the quantity and characteristic of waste have implications for the GHG emissions, it also requires different SWM options.

Clearly the observations made above indicate that waste management priorities vary both within and between developing countries and resources are needed from initial collection through to disposal. However, it is at the treatment and disposal end of the chain that the opportunity exists for attracting investment if verifiable GHG reduction schemes can be developed. Hence this paper concentrates on the GHG performance of residual waste treatment and disposal options.

3. 'Life-cycle analysis of different waste management options for developing countries'

In comparing the performance of options, a life-cycle approach needs to be taken. Life cycle analysis (LCA) is an established environmental assessment tool (ISO, 1997) for looking at the cradle-to-grave impacts of products and services. LCA stages include goal and scoping definitions and inventory analysis (ISO, 1998), impact assessment (ISO, 1999a) and interpretation (ISO, 1999b). This study is primarily concerned with the steps leading to generating the inventory.

3.1. Goal and scoping

Defining the purpose or goal of a study dictates the appropriate scope and level of detail. In this case the goal was to assess, in terms of carbon efficiency, a range of simple waste management that might be applied in DCs. Use of LCA in waste management is commonplace in developed countries and would routinely include a wide range of impact categories (e.g., land use, global warming, eutrophication, acidification, human health) and require detailed knowledge of resource inputs, waste flows and compositions, operational characteristics of facilities and the fate of recovered materials, energy and residues. The waste scenarios considered tend to be complex and cover all flows from the household (source separated, residuals, collected and delivered). Overall, such a detailed approach is not warranted or necessary in order to make an initial assessment of ranking of options in terms of GHG emissions, particularly for developing countries where the data necessary for a full study are unlikely to be readily available. Thus, while a life-cycle approach is still appropriate, the limited and comparative nature of the goal is compatible with a simple and pragmatic approach to defining the scope, and decisions were made to restrict the level of detail in which activities were disaggregated, eliminate activities common to all scenarios from the system boundary and use readily available inventory data/emission factors. The approach taken for assessing the direct emissions for the waste management systems is compatible with CDM methodologies developed to date (e.g., AM0025 and approved small-scale methodologies, Type III F and G (UNFCCC, 2006a) but it should be recognised that CDM projects must calculate credits (certified emission reductions (CERs) or voluntary emission reductions (VERs)) for each year. This requires specific project information and use of suitable decomposition rates for the biodegradable categories. The method adopted in this report predicts the relative GHG performance per tonne of waste disregarding the precise time period of the emissions. Over the project life, this does not change the relative GHG performance of options considered, but clearly prospective project developers need to refer to the CDM methodology tool to determine the annual avoided emissions from dumping waste (UNFCCC, 2006c).

3.1.1. System definition and system boundary

Bearing in mind the constraints which are likely to exist in developing countries, a number of options (Fig. 1) were considered for the management of urban waste. The dotted lines in the figure encompass the "foreground" waste management activities considered in the various scenarios, and their direct contributions to GHG emissions are assessed. The area outside the dotted line shows resource and emission flows that may generate or offset GHG emissions in the full system boundary. The description of the various scenarios clarifies what has been included in calculating the total GHG emissions for each scenario to enable equitable comparison.

Base case A assumes scavenging activities (at source/in transit and temporary storage) for materials recycling and open dumping of the residual waste. Dumpsites were assumed to be within or close to the urban centre and a 50:50 mix (in terms of waste weight) of unmanaged deep (>5 m) and shallow (<5 m) sites.

Scenarios B1, B2 and B3 consider moving to large-scale sanitary (engineered) landfills (depth $\geqslant 10$ m) within direct transport distance of the conurbation; B1 with passive venting of landfill gas (LFG), B2 with capture and flaring and B3 capture and electricity production by gas-engine/generator units.

Scenario C assumed composting and scenario D assumed anaerobic digestion with electricity production. Both of the latter options assumed compost/digestate were used after simple refining (e.g., screening) with any process residues going to landfill. The inputs are the collected non-segregated wastes and therefore both of these options assume the use of a central facility. It was also assumed that the introduction of new facilities would leave the upstream scavenging operations largely unaffected.

As noted earlier, to assess impact on the carbon balance, a life-cycle approach is needed and a systems boundary adopted to permit equitable comparison. Many uncertainties are associated with estimating greenhouse emissions from waste management systems, even for countries such as Holland and the UK (IPCC, 2000; Golder Associates, 2005) that have had years of monitoring data suited to the task. Thus, it is not appropriate to go to a high level of detail or sophistication for a non-specific overview of options for DCs, and the analysis was restricted to considering the major elements contributing to estimating the net difference in GHG for the options. Where appropriate, activities common to all systems were ignored, e.g., although offsetting GHG emission benefits of materials reuse/recycling are highly positive, it was assumed that any minor changes (e.g., location) of materials recycling activities had minimal impact on the carbon balance and these activities were ignored. Further it was assumed that the precise location of facilities (and hence transport distances and modes) had minor impact on the carbon balance compared to the management practices adopted. Capital resource inputs were ignored in this study and consumable inputs were only considered where data were readily available and they were considered to be potentially significant in terms of the carbon balance.

3.2. Inventory analysis

To calculate the carbon balance in terms of CO_2 equivalent emissions, the biogenic CO_2 component generated by the various systems (whether through bio-degradation or combustion) was taken to be carbon neutral and greenhouse gas contributions from biogenic materials were confined to emissions of methane (CH_4) and nitrous oxide (N_2O) assuming 21 and 310 times the impact of CO_2, respectively, for global warming potential. CO_2/N_2O emissions associated with resources used to operate systems were accounted for, but other GHGs, e.g., PFCs HFCs and SF6, were not considered given these emissions are not relevant to the MSW treatment and disposal processes considered in this paper.

The first task was to allocate some specific characteristics to the waste input in order to produce a mass balance for the fate of carbon at the various stages based on expected performance of the various processes. This required making a judgement about such issues as design and operational standards of both the waste management systems and background systems that supply or accept resources. For this study, Table 2 provides the assumed baseline (Case 1) composition of the waste based on typical reported compositions. For consistency and convenience, the composition has been expressed in terms of the categorisation used in IPCC guidance along with degradable organic carbon contents (DOC) (IPCC, 1997). More recent (IPCC, 2006) guidance makes minor changes to categorisation but these are not significant in terms of this assessment.

3.2.1. Landfill inventory analysis

For the landfill options, 0.5 was taken as the fraction degraded (assumes lignin included in DOC) and a 50:50 mix of CH_4 to CO_2 as the initial breakdown products of anaerobic degradation. Emissions from open dumping for a mix of unmanaged sites (less and more than 5 m) were based on IPCC guidance to use a methane correction factor (MCF) of 0.6 (IPCC, 2006) which assumed significant aerobic activity. For the sanitary landfill options, an MCF of 1 was assumed. Methane oxidation (MO) was assumed to be zero for open dumps (oxidation accounted for in allocated MCF) and 0.1 for landfill gas from the engineered but passively vented site B1 (IPCC, 2006). For actively pumped sites, reduced gas flow-rates in the cover permit higher MO rates and recent UK research has led to adopting a consensus value of 0.25 (Golder Associates, 2005) which combined a 50% loss via fissures and a 50% MO for the residual gas passing through the full cover layer. Thus a value of 0.25 was used for B2 and B3 scenarios (note IPCC (2006) uses 0.1 for all managed sites). High standards were adopted in terms of landfill gas capture, flaring and electricity production (ERM, 2006). For both B2 and B3, 80% capture of landfill gas was assumed, all was flared in B2, 50% and was utilised for electricity generation in B3 (at 35% efficiency). It is pertinent to note that in reports for UK government agencies, values for factors such as methane oxidation rates, inclusion of fissures, gas collection efficiencies etc. are reviewed/refined on a regular basis and most authors use of a range of values to accommodate uncertainty when predicting inventories. Consideration of operational resource inputs to landfill were confined to diesel use (~1 kg/t of waste (ERM, 2006)).

3.2.2. Bio-treatment inventory analysis

For composting, a simple windrow system was assumed operating to UK standards (ERM, 2006) and GHG contributions covered N_2O emissions and methane associated with unintentional anaerobic activity in the piles and GHG emissions associated with operational demands (~0.5 kWh/t and 3 kg/t of diesel); 416 kg of compost per

Table 2
Assumed category composition and degradable carbon contents

Waste category	DOC for category	Waste compositions (wt%)		
		Case 1 (baseline)	Case 2 (more developed)	Case 3 (less developed)
Paper and textiles	0.4	10	30	5
Garden/non-food putrescibles	0.17	10	10	15
Food waste	0.15	60	30	70
Wood/straw	0.3	10	10	5
Non-putrescibles	0	10	20	5
Calculated DOC		0.177	0.212	0.166

http://www.ipcc-nggip.igas.or.jp./public/gl/invs6.htm.

t of input was predicted. The residues to landfill were assumed to be inert.

For anaerobic digestion (AD), a high solids system was selected and, for the waste composition given and the degradation factors selected, a yield of 77 m^3 of methane per t of input was predicted, which was used to generate electricity at the same conversion efficiency as landfill gas engines. The assumptions led to an electricity generation rate of 267 kWh/t, reduced to a net figure of 247 kWh/t for inventory purposes given reported in-house usage of 20 kWh/t (ERM, 2006). The methane yield was approximated to 75% of the predicted methane generation for the landfill scenarios and was in line with reported performance of such plants (MCDougal et al., 2001; Defra, 2005). The digestate underwent aerobic stabilisation prior to use and, based on expected volatile and moisture losses and screened rejects, a yield of 380 kg of compost was predicted.

3.2.3. Resource input and product output contributions to the inventory

For each system, resource use and product outputs have associated GHG impacts that are not directly emitted by the "foreground" waste management systems and assumptions need to be made to estimate levels.

Operational resource demands for running the various systems included diesel and, based on UK data, the GHG emissions associated with diesel production and use was taken as 3.66 kg CO_2e/kg fuel (ERM, 2006), but consumption per tonne of waste treated rarely exceeded 3 kg for any of the treatment systems and had minimal impact on the inventory.

For all systems, the GHG offset benefits for electricity recovered and exported were based on substitution for oil-fired plant. A value based on UK oil fired power stations of 0.95 kg CO_2e/kWh was used and, given this is an important offset, country and case specific information would be needed to assess the true potential for a given scheme.

For composting and AD systems, the GHG benefits of the compost/digestate were confined to considering offsetting emissions of production of peat (restricting offset resource benefit to soil conditioning). These were estimated to be 16.2 kgCO_2e/t (ERM, 2006) and had a very limited impact on the overall inventory. In many cases, peat substitution is not a factor that would be relevant for a developing country and other offsets from use of the compost may need consideration; however, inclusion does highlight the limited impact it had on the overall inventory.

4. Results and discussion

Fig. 2 illustrates the results for the various scenarios. Moving from open dumpsites (A) to sanitary landfill with no provision for landfill gas capture (B1) may have many environmental and public health benefits but reduced GHG emission is not one of them. Emissions increased

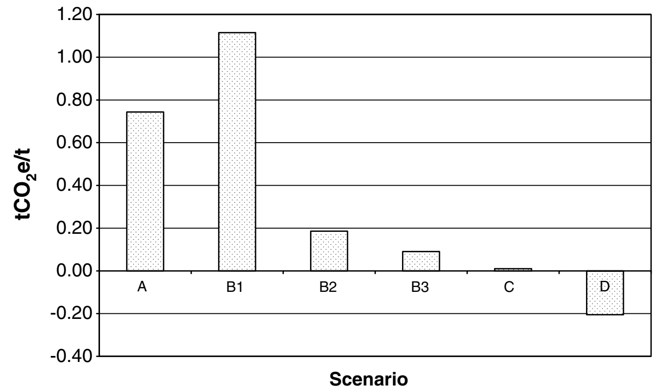

Fig. 2. Effect of changing the waste management systems on GHG emissions.

from 0.74 t/CO_2e/t to 1.2 t/CO_2e/t of waste. Ensuring effective gas collection and flaring (B2) dramatically improved the emission factor to around 0.19 tCO_2e/t of waste which can be reduced further if the biogas is utilised for electricity production to 0.09 tCO_2e/t of waste. The composting option was essentially carbon neutral and the AD plant provided a net benefit in terms of GHG at −0.21 tCO_2e/t of waste. The graph illustrates that significant benefits relative to the baseline case of open dumping are attributable to all scenarios apart from sanitary landfill with passive venting. Furthermore, realising the potential for electricity production from an engineered landfill site is an added bonus as far as greenhouse emission reductions are concerned, effective landfill gas collection and flaring should be the priority. The offset in terms of greenhouse gas reductions from the compost/digestate produced was very small, although it only accounted for "soil conditioning" benefit by considering the energy used in peat extraction. Considering offsetting fertilizer benefits would improve the position marginally but requires more specific knowledge of the waste inputs and end-use applications. Extending the system boundary to encompass schemes where the use of the compost was specifically designed to sequester carbon/grow energy crops on previously infertile land/desert would be necessary to significantly enhance the potential GHG benefits. Again, scheme specific assessment is required. The AD scenario provided the best result (due to the added benefit of electricity production) but, in practical terms, this option may well not be a feasible choice. Large-scale plants have a limited commercial track record for MSW; it represents the highest capital cost option and the most demanding in terms of operations. This paper has considered large central facilities processing the residual wastes but there are many successful small-scale 'back garden' AD plants around the world, particularly in China and India, which operate on source separated feedstocks, primarily animal wastes, but often including food wastes. These have proved very successful from the homeowner's point of view, in terms of providing free gas for cooking and heating, but data on the impact of such schemes on the residual waste and stray methane emissions from such plants need to be included. However, it is highly unlikely that fugitive emissions would offset the carbon benefits of this approach and it is worth noting that Nepal has successfully registered a small-scale CDM project via this route (Project 0136: Biogas Support Program – Nepal (BSP – Nepal) Activity-1, UNFCCC (URL 1)). This project was submitted on the basis of use of animal dung in the digesters, and while accounting for substitution of kerosene and unsustainable firewood in assessing carbon credits, made no claim for avoided methane emissions from prior disposal practices of the dung. Interestingly, the potential to use of food wastes in these digesters was not included in the submission but a visit by one of the authors confirmed food waste is also used as a feedstock. Where food waste is an input, the potential offset to GHG emissions from avoided dumping could provide significant additional credits, but demonstrating/validating the amounts diverted from the MSW stream is fraught with problems.

4.1. Sensitivity of results to waste composition

The assumed composition of the waste in this study, while not untypical, is unlikely to reflect the specific values for a given scheme. Waste compositions change over time and analysis is a costly activity for an authority to undertake. However, by looking at the sensitivity of the findings to compositional change, the importance of this factor in terms of impact on GHG emissions can be assessed.

Two additional compositions are given in Table 2 alongside the base case. Composition 2 assumed lower food waste content (30%) with increased paper/textile (30%) and non-compostables (20%) fractions, i.e., a waste composition reflecting a more affluent and developed urban economy. Composition 3, representing a more undeveloped situation, had very high putrescible food (70%) and non-food putrescibles (15%) with the other categories being reduced to 5% each. Fig. 3 illustrates that the predicted emissions and savings for the scenarios were affected by changes in composition. The main effect relates to the landfill scenarios (A, B1, B2 and B3) and indicates that where urbanisation moves the waste towards a more developed country composition, GHG emissions rise significantly (e.g., 20% for sanitary landfill B1) mainly due to increase in paper/textile content. The low moisture contents of these categories relative to food wastes result in a higher overall DOC value (0.212 compared with 0.177 for case1). For the very high putrescible content composition, the emissions fall under the landfill options. As the move to a larger, deeper "sanitary" landfill is most pressing and prevalent for economically developing urban centres, the case for ensuring such sites are engineered to collect and flare/utilise the gas is likely to strengthen over time. The impact of the waste changes tested on the GHG emissions from composting and AD scenarios is minimal. Thus, the changes did not affect the broad conclusions regarding the impact of the scenarios tested, and the relative benefits are similar. However, they do confirm that GHG emissions will increase significantly from developing countries as economic development and urbanisation continues apace unless the non-gas collection "sanitary" landfill phase of development of their waste management systems is bypassed.

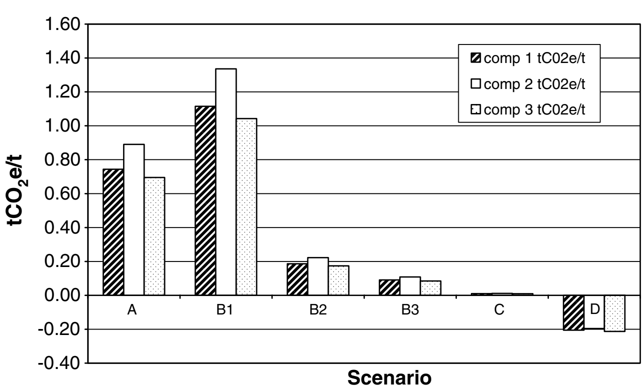

Fig. 3. Impact of waste composition on predicted GHG emissions for the selected scenarios.

5. Conclusions

This paper has adopted a simple and generalised approach to identifying the GHG emissions for options considered likely to represent the baseline conditions and prospective improvements in the management of MSW in developing countries in order to indicate the potential for carbon reduction schemes to both developing countries and potential sponsors from Annex B countries. A number of conclusions and observations emerge.

Developing sanitary landfills has often been seen as "the solution" for countries where the priority is avoidance of the severe health and safety problems associated with poor collection coverage and open dumping practices in rapidly developing urban conurbations. In the past, few such developments have included systems for actively managing the landfill gas emissions. In terms of carbon impact, it is important when considering sanitary landfill as the final deposit point for MSW that it is designed and operated either with gas flaring or to produce energy. Without these systems being installed, then not only is the landfill without proper gas control worse than those with gas control, it is also worse than having open dumping. The CDM offers a route to attracting investment to rehabilitate existing sites and new facilities. Examples of support for LFG projects through this mechanism are available and it is relatively easy to demonstrate that the collected and utilised landfill gas qualifies.

Anaerobic digestion with energy production and composting of the digestate was the only option which was considered to be carbon negative. Composting did not come out quite as well as anaerobic digestion, being essentially carbon neutral. However, in terms of its relative simplicity compared to large-scale AD plants, composting is seen as probably the first process to consider when replacing open dumping. Constructing a case, demonstrating and validating GHG savings in the context of adopting either of these latter treatment options for qualifying as a CDM project is more demanding. A baseline of uncaptured GHG emission from a landfill, now captured by the project and hence providing easily validated savings, no longer pertains. In developed countries, the baseline landfill offset would have to be, at minimum, based on scenario B2 (gas collection and flaring) as such standards are mandated by legislation and hence additional offsets are limited to less than 0.4 tCO_2e/t. For developing countries the CDM methodology permits the baseline to be A or B1 type scenarios provided legislation/safety considerations do not require gas collection. In urban conurbations, an argument that B1 or deepsite A type landfill (with an MCF of 0.8) should be taken as the baseline is justifiable. Thus potential offsets in excess of 1 tCO_2e/t of waste treated could be realised. At current EU ETS values, this represents up to €10/t of waste treated.

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