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Mitigation of global greenhouse gas emissions from waste: conclusions and strategies from the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report. Working Group III (Mitigation)

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Intergovernmental Panel on Climate Change (IPCC): Working Group III (Mitigation)

Greenhouse gas (GHG) emissions from post-consumer waste and wastewater are a small contributor (about 3%) to total global anthropogenic GHG emissions. Emissions for 2004-2005 totalled 1.4 Gt CO_2-eq year^{-1} relative to total emissions from all sectors of 49 Gt CO_2-eq year^{-1} [including carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and F-gases normalized according to their 100-year global warming potentials (GWP)]. The CH_4 from landfills and wastewater collectively accounted for about 90% of waste sector emissions, or about 18% of global anthropogenic methane emissions (which were about 14% of the global total in 2004). Wastewater N_2O and CO_2 from the incineration of waste containing fossil carbon (plastics; synthetic textiles) are minor sources. Due to the wide range of mature technologies that can mitigate GHG emissions from waste and provide public health, environmental protection, and sustainable development co-benefits, existing waste management practices can provide effective mitigation of GHG emissions from this sector. Current mitigation technologies include landfill gas recovery, improved landfill practices, and engineered wastewater management. In addition, significant GHG generation is avoided through controlled composting, state-of-the-art incineration, and expanded sanitation coverage. Reduced waste generation and the exploitation of energy from waste (landfill gas, incineration, anaerobic digester biogas) produce an indirect reduction of GHG emissions through the conservation of raw materials, improved energy and resource efficiency, and fossil fuel avoidance. Flexible strategies and financial incentives can expand waste management options to achieve GHG mitigation goals; local technology decisions are influenced by a variety of factors such as waste quantity and characteristics, cost and financing issues, infrastructure requirements including available land area, collection and transport considerations, and regulatory constraints. Existing studies on mitigation potentials and costs for the waste sector tend to focus on landfill CH_4 as the baseline. The commercial recovery of landfill CH_4 as a source of renewable energy has been practised at full scale since 1975 and currently exceeds 105 Mt CO_2-eq year^{-1}. Although landfill CH_4 emissions from developed countries have been largely stabilized, emissions from developing countries are increasing as more controlled (anaerobic) landfilling practices are implemented; these emissions could be reduced by accelerating the introduction of engineered gas recovery, increasing rates of waste minimization and recycling, and implementing alternative waste management strategies provided they are affordable, effective, and sustainable. Aided by Kyoto mechanisms such as the Clean Development Mechanism (CDM) and Joint Implementation (JI), the total global economic mitigation potential for reducing waste sector emissions in 2030 is estimated to be > 1000 Mt CO_2-eq (or 70% of estimated emissions) at costs below 100 US$ t^{-1} CO_2-eq year^{-1}. An estimated 20–30% of projected emissions for 2030 can be

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reduced at negative cost and 30–50% at costs < 20 US$ t^{-1} CO_2-eq $year^{-1}$. As landfills produce CH_4 for several decades, incineration and composting are complementary mitigation measures to landfill gas recovery in the short- to medium-term – at the present time, there are > 130 Mt waste $year^{-1}$ incinerated at more than 600 plants. Current uncertainties with respect to emissions and mitigation potentials could be reduced by more consistent national definitions, coordinated international data collection, standardized data analysis, field validation of models, and consistent application of life-cycle assessment tools inclusive of fossil fuel offsets.

Keywords: methane, nitrous oxide, carbon dioxide, greenhouse gas, emissions, landfill, wastewater, incineration, global warming, IPCC, waste, wmr 1315–2

Introduction and background: the IPCC process and the fourth assessment report (AR4)

The Intergovernmental Panel on Climate Change (IPCC) was established in 1988 by the World Meteorological Organization (WMO) and the United Nations Environment Program (UNEP) to assess on a 'comprehensive, objective, open and transparent basis' … 'the scientific, technical and socio-economic information relevant to…human-induced climate change, its potential impacts and options for adaptation and mitigation' "[www.ipcc.ch]. The IPCC includes a Task Force on National GHG Inventories and three working groups (I–III), which provide regular assessment reports. The working groups address: the scientific basis of climate change (I); the consequences of climate change including vulnerability, adaptation strategies, and sustainable development implications (II); and climate change mitigation (III). In addition, the IPCC prepares periodic reports on specialized topics relating to climate change and, through the Task Force mentioned above, provides support to the United Nations Framework Convention on Climate Change (UNFCCC) on the methodologies for national GHG inventories.

The IPCC does not carry out independent research, nor does it monitor climate data. Each chapter in an assessment report is prepared by international groups of authors with expertise in the subject area who have been approved through their respective governments. The reports discuss policy options but are 'policy neutral' in their assessment of the existing peer-reviewed literature. The IPCC has issued four assessment reports to date. The fourth Assessment Report (AR4) was written during 2004–2007, approved by the UNFCCC in 2007, and published in four parts (separate reports for working groups I, II, and III plus a synthesis report). The working group III report for the AR4 conformed to organizational templates for the sectoral chapters (energy, industry, transport, buildings, agriculture, forestry, waste) and was consistent with a series of guidance documents for standardized units, page limits, avoidance of overlaps and double-counting, definitions of uncertainty, discussion of sustainable development, long-term trends, and other issues. For the first time, the AR4 included a separate chapter on post-consumer waste and wastewater. Other chapters address pre-consumer greenhouse gas (GHG) emissions from waste that is managed within the industrial, energy, agricultural, buildings, and forestry sectors. Global emissions from waste transport were not included because national data are not available on a global basis. This paper will address strategies, costs, and potentials for GHG mitigation from waste, as summarized in Chapter 10. Waste Management of the AR4 WGIII report (Bogner *et al.* 2007). The first author of this paper was the coordinating lead author for this chapter, while the other authors were lead authors, contributing authors, or review editors whose contributions are gratefully acknowledged.

Global trends for waste and wastewater generation, including carbon, nitrogen and energy balances

Waste generation is closely linked to population, urbanization and affluence. Waste-generation rates can be correlated to gross domestic product (GDP) $capita^{-1}$, energy consumption $capita^{-1}$, and private final consumption $capita^{-1}$ (Bingemer & Crutzen 1987, Richards 1989, Rathje *et al.* 1992, Mertins *et al.* 1999, US EPA 1999, IPCC 2000, Bogner & Matthews 2003, OECD 2004). In developed countries seeking to reduce waste generation, a current goal is to decouple waste generation from economic driving forces such as GDP (OECD 2003, Giegrich & Vogt 2005, EEA 2005). In most developed and developing countries with increasing population, prosperity and urbanization, it remains a major challenge for municipalities to collect, recycle, treat and dispose of increasing quantities of solid waste and wastewater.

Most technologies for waste management are mature and have been successfully implemented for decades in many countries. Nevertheless, there is significant potential for accelerating both the direct reduction of GHG emissions from waste as well as extended implications for indirect reductions within other sectors. Life-cycle assessment (LCA) can be an important tool for consideration of both the direct and indirect impacts of waste management technologies and policies (Thorneloe *et al.* 2002, 2005, WRAP 2006). Landfill CH_4 recovery and optimized wastewater treatment can directly reduce GHG emissions. GHG generation can be largely avoided through controlled aerobic composting and thermal processes such as incineration for waste-to-energy. Moreover, waste prevention, minimization, material recovery, recycling and re-use represent a growing potential for indirect reduction of GHG emissions through decreased waste generation, lower raw material consumption, reduced energy

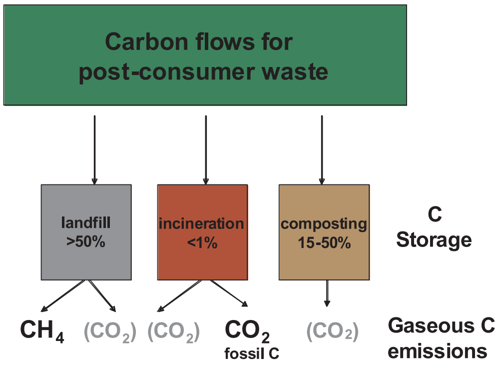


Fig. 1: Carbon flows through major waste management systems including C storage and gaseous C emissions. The CO_2 from biomass is not included in GHG inventories for the waste sector. (References for C storage are: Huber-Humer 2004. Zinati *et al.* 2001: Barlaz 1998: Bramryd 1997: Bogner 1992.)

demand and fossil fuel avoidance. Recent studies (e.g., Smith *et al.* 2001, WRAP 2006) have begun to comprehensively quantify the significant benefits of recycling for indirect reductions of GHG emissions from the waste sector.

An overview of carbon flows (Figure 1) addresses the issue of carbon storage versus carbon turnover for major waste management strategies including landfilling, incineration and composting. As landfills function as relatively inefficient anaerobic digesters, they provide significant long-term carbon storage, which is addressed in the 2006 IPCC *Guidelines for National Greenhouse Gas Inventories* (IPCC 2006). Landfill CH_4 is the major gaseous C emission from waste, with minor emissions of CO_2 from waste sector incinerated fossil carbon (plastics, synthetic textiles). It should be noted that the CO_2 emissions from biomass sources – including the CO_2 in landfill gas, from composting, and from incineration of waste biomass – are not taken into account in GHG inventories because they are attributed to the land use, land-use change and forestry sectors.

For quantifying the major GHG emissions from the waste sector, it is useful to consider major mass balance relationships. For landfill CH_4, emissions are one of several possible pathways for the CH_4 produced by anaerobic methanogenic micro-organisms; other pathways include recovery via active gas extraction systems; oxidation by aerobic methanotrophic micro-organisms in cover soils; and two longer-term pathways: lateral migration, and internal storage (Bogner & Spokas 1993, Spokas *et al.* 2006). Regarding CH_4 and nitrous oxide (N_2O) emissions from wastewater, the CH_4 is microbially produced under strict anaerobic conditions as in landfills, while the N_2O is an intermediate product of microbial N cycling promoted by conditions of reduced aeration, high moisture, and abundant N. Both GHGs can be produced and emitted at many stages between wastewater sources and final disposal.

Because both the CH_4 and N_2O are microbially produced and consumed, rates are controlled by temperature, moisture, pH, available substrates, microbial competition and many other factors. Emission rates can thus routinely exhibit temporal and spatial variability over many orders of magnitude, exacerbating the problem of developing credible national estimates. Landfill N_2O is considered an insignificant source globally (Bogner *et al.* 1999, Rinne *et al.* 2005), but may need to be considered locally where cover soils are amended with sewage sludge (Borjesson & Svensson 1997a) or where aerobic/semi-aerobic landfill designs are implemented. (Tsujimoto *et al.* 1994). Substantial emissions of CH_4 and N_2O can occur during wastewater transport in closed sewers.

The availability and quality of annual data are major problems for the waste sector. Solid waste and wastewater data are lacking for many countries; data quality is variable, national definitions are not uniform; and interannual variability is often not well quantified. There are three major approaches that have been used to estimate global waste generation: (1) data from national waste statistics, including IPCC methodologies (IPCC 2006), with extrapolations for non-reporting countries; (2) the assumptions and projections in the IPCC Special Report on *Emission Scenarios* (SRES) (IPCC 2000); and (3) the use of a proxy variable linked to demographic or economic indicators for which national data are annually collected. The SRES waste scenarios used population as the major driver and projected increases in waste and wastewater CH_4 emissions to 2030 (A1B-AIM), 2050 (B1-AIM), or 2100 (A2-ASF; B2-MESSAGE), resulting in current and future emission projections significantly higher than the other methods. A major reason is that waste generation rates are related to affluence as well as population – richer societies are characterized by higher rates of waste generation per capita, whereas less affluent societies generate less waste and practice informal recycling/re-use initiatives that reduce the waste per capita to be collected at the municipal level. The third strategy is to use proxy or surrogate variables based on statistically significant relationships between waste generation per capita and demographic variables, which encompass both population and affluence, including GDP per capita (Richards 1989, Mertins *et al.* 1999) and energy consumption per capita (Bogner & Matthews 2003). The use of proxy variables, validated using reliable datasets, can provide a cross-check on uncertain national data. Moreover, the use of a surrogate provides a reasonable methodology for a large number of countries where data do not exist, a consistent methodology for both developed and developing countries and a procedure that facilitates annual updates and trend analysis using readily available data (Bogner & Matthews 2003).

Current estimates indicate that global post-consumer waste generation totals approximately 900–1250 Mt yea^{-1} (Bogner & Matthews 2003, Monni *et al.* 2006). Per capita solid waste generation rates range from < 0.1 t cap^{-1} year^{-1} in low-income countries to > 0.8 t cap^{-1} year^{-1} in high-income industrialized countries (Bernache-Perez *et al*. 2001, Diaz & Eggerth 2002, Kaseva *et al*. 2002, Idris *et al*. 2003, Ojeda-Benitez & Beraud-Lozano 2003, US EPA, 2003, CalRecovery 2004, 2005, Griffiths & Williams 2005, Huang *et al*. 2006). Even though labour costs are lower in developing countries, waste management can constitute a larger percentage of municipal income because of higher equipment and fuel costs (Cointreau-Levine 1994). By 1990, many developed countries had initiated comprehensive recycling programmes. The percentages of waste recycled, composted, incinerated or landfilled can differ greatly among municipalities as a result of local economics, national policies, regulatory restrictions, public perceptions and infrastructure requirements.

It is important to emphasize that post-consumer waste is a significant renewable energy resource through thermal processes (incineration and industrial co-combustion), landfill gas utilization and the use of anaerobic digester biogas. Waste has an economic advantage in comparison with many biomass resources because it is regularly collected at public expense. The energy content of waste can be more efficiently exploited using thermal processes than with the production of biogas: during combustion, energy is directly derived both from biomass (paper products, wood, natural textiles, food) and fossil carbon sources (plastics, synthetic textiles). The heating value of mixed municipal waste ranges from < 6 to > 14 MJ kg^{-1} (Khan & Abu-Ghararath, 1991, EIPPC Bureau 2006). Thermal processes are most effective at the upper end of this range where high values approach low-grade coals (lignite). Based on a conservative value of 900 Mt $year^{-1}$ for total waste generation in 2002, the energy potential of waste could be approximately 5 to 13 EJ $year^{-1}$. Assuming an average heating value of 9 GJ t^{-1} for mixed waste (Dornburg & Faaij 2006) and converting to energy equivalents, global waste in 2002 contained about 8 EJ of available energy, which could increase to 13 EJ in 2030 using the waste projections in Monni *et al.* (2006). More than 130 Mt $year^{-1}$ of waste are combusted worldwide (Themelis 2003), which is equivalent to > 1 EJ $year^{-1}$ (assuming 9 GJ t^{-1}). The biogas fuels from waste – landfill gas and digester gas – typically have a heating value of 16–22 MJ Nm^3, depending upon the CH_4 content. Both are used extensively worldwide for process heating and on-site electrical generation; more rarely, landfill gas may be upgraded to a substitute natural gas product. Conservatively, the energy value of landfill gas currently being utilized is > 0.2 EJ $year^{-1}$ (using data from Willumsen 2003).

With regard to wastewater, most countries do not compile annual statistics on the total volume of municipal wastewater generated, transported, and treated. Estimates for CH_4 and N_2O emissions from wastewater treatment require data on degradable organic matter [biochemical oxygen demand (BOD); chemical oxygen demand (COD)] and nitrogen. Nitrogen content can be estimated using Food and Agriculture Organization data on protein consumption, and either the application of wastewater treatment, or its absence, determines the emissions. Aerobic treatment plants produce negligible or very small emissions, whereas in anaerobic lagoons or latrines 50–80% of the CH_4 potential can be produced and emitted.

In general, about 60% of the global population has sanitation coverage (sewerage) with very high levels (>90%) characteristic for the population of North America (including Mexico), Europe and Oceania, although in the last two regions rural areas decrease to approximately 75% and 80%, respectively (DESA/UN, 2005; Jouravlev, 2004; PNUD/UNEP, 2005; WHO/ UNICEF/WSSCC, 2000, WHO-UNICEF, 2005; World Bank, 2005a). In developing countries, rates of sewerage are very low for rural areas of Africa, Latin America and Asia, where septic tanks and latrines predominate. For 'improved sanitation' (including sewerage + wastewater treatment, septic tanks and latrines), almost 90% of the population in developed countries, but only about 30% of the population in developing countries, has access to improved sanitation (Jouravlev, 2004; World Bank, 2005a, b). Figure 2 indicates regional trends and Millenium Development Goals for improved sanitation levels. Note that current levels are <50% for eastern and southern Asia and sub-Saharan Africa (World Bank and IMF, 2006). In sub-Saharan Africa, at least 450 million people lack adequate sanitation whereas in both southern and eastern Asia, rapid urbanization is posing a challenge for the development of wastewater

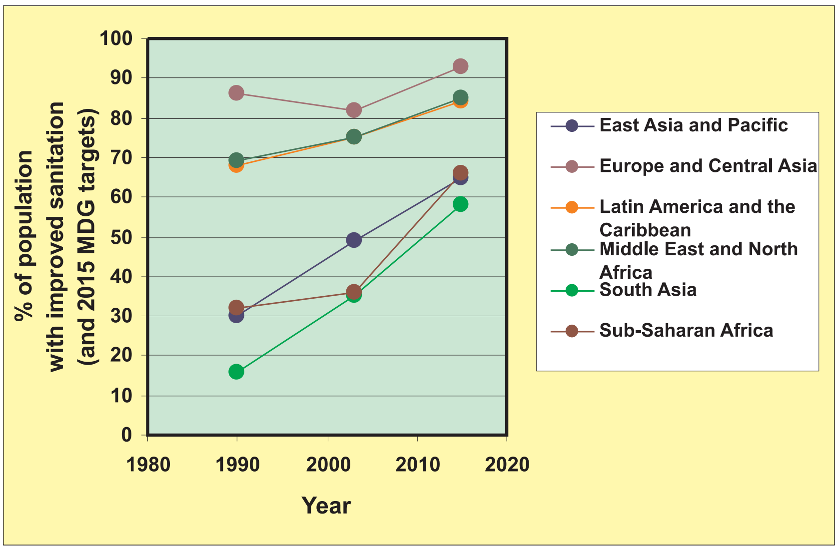


Fig. 2: Percentage of population with access to improved sanitation (sewerage + wastewater treatment, septic systems, or latrines). Regional data for 1990 and 2003 with Millenium Development Goals for 2015. Source: World Bank and IMF (2006).

infrastructure. The highly urbanized region of Latin America and the Caribbean (LAC) has also made slow progress in providing wastewater treatment. Globally, it has been estimated that 2.6 billion people lack improved sanitation (WHO-UNICEF, 2005). The majority of urban wastewater treatment facilities are publicly operated and only about 14% of the total private investment in water and sewerage in the late 1990s was applied to the financing of wastewater collection and treatment, mainly to protect drinking water supplies (Silva, 1998; World Bank 1997).

Both waste and wastewater are highly regulated within the municipal infrastructure in developed countries through a wide range of legislation to protect human health and the environment; promote waste minimization and recycling; restrict certain types of waste management activities; and reduce impacts to residents, surface water, groundwater and soils. National regulations, regional restrictions, and local planning guidelines address waste and wastewater transport, recycling, treatment, disposal, utilization, and energy use. Depending on regulations, policies, economic priorities and practical local limits, developed countries are characterized by increasingly higher rates of waste recycling and pre-treatment to conserve resources and avoid GHG generation. Studies have documented recycling levels of > 50% for specific waste fractions in some developed countries (i.e., Swedish Environmental Protection Agency, 2005). Recent US data indicate about 25% diversion, including more than 20 states that prohibit landfilling of garden waste (Simmons *et al.* 2006).

For developing countries, a wide range of waste management legislation and policies have been implemented with evolving structure and enforcement; it is expected that regulatory frameworks in developing countries will become more stringent in parallel with development trends. A high level of labour-intensive informal recycling often occurs in developing countries. Although this reduces the mass of waste requiring more centralized solutions, the challenge is to provide safer, healthier working conditions than currently experienced by scavengers on uncontrolled dumpsites. Ultimately, recycling activities by this sector can generate significant employment, especially for women, through creative microfinance and other small-scale investments. For example, in Cairo, 7–8 daily jobs per ton of waste and recycling of > 50% of collected waste can be attained (Iskandar 2001).

Regional trends for waste and wastewater management

In the EU, the landfill directive (Council Directive 1999/31/EC) requires that annual landfilling of biodegradable organic waste be reduced 65% by 2016 relative to 1995 levels, and that existing sites have engineered gas recovery (EU 1999). Several countries (Germany, Austria, Denmark, the Netherlands, Sweden) have adopted more stringent bans to accelerate the EU schedule. Thus larger quantities of post-consumer waste are now being diverted either to incineration or to mechanical-biological treatment (MBT) steps to recover recyclables and reduce the organic carbon content by partial aerobic composting or anaerobic digestion (Stegmann 2005). The MBT residuals are often, but not always, landfilled after achieving required organic carbon reductions. Depending on the types and quality control of various separation and treatment processes, a variety of useful recycled streams are also produced. Waste-to-energy (WTE) incinerators have been widely used in many European countries for decades. In 2002, EU WTE plants generated 41 million GJ of electrical energy and 110 million GJ of thermal energy (Themelis 2003). The landfill directive is expected to increase the use of incinerators, especially in countries such as the UK with historically lower rates of incineration compared to other European countries. In North America, Australia and New Zealand, controlled landfilling is continuing as a dominant method for large-scale waste disposal with mandated compliance to both landfilling and air-quality regulations. In parallel, larger quantities of landfill CH_4 are annually being recovered, driven both by air-quality regulations and expanding renewable energy markets, whose viability may be improved via tax credits or local renewable-energy/green power incentives. Incineration has not been widely adopted due to low landfill tipping fees in many regions and negative public perceptions regarding incineration. The US, Canada, Australia and other countries are currently assessing larger commitments to 'bioreactor' landfills to compress the time during which high CH_4 generation rates occur. (Reinhart & Townsend 1998, Reinhart *et al*. 2002, Berge *et al.* 2005); bioreactors also require early implementation of engineered gas extraction. In Japan, where open space is very limited for construction of waste management infrastructure, very high rates of both recycling and incineration are practised and are expected to continue into the future.

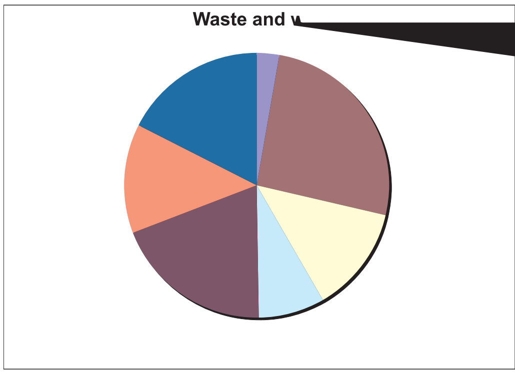
In many developing countries, current trends suggest that increases in controlled landfilling, resulting in increased anaerobic decomposition of organic waste, will be implemented in parallel with increased urbanization. The process of converting open dumping and burning to engineered landfills implies control of waste placement, compaction, the use of cover materials, implementation of surface water diversion and drainage, and management of leachate and gas, perhaps applying an intermediate level of technology consistent with limited financial resources (Savage *et al.* 1998). These practices shift the production of CO_2 (by burning and aerobic decomposition) to anaerobic production of CH_4; paradoxically, this leads to higher GHG generation and emissions than previous open-dumping and burning practices. This is largely the same transition that occurred in many developed countries in the 1950–1970 timeframe. In addition, many developing (and developed) countries have implemented large-scale aerobic composting of mixed waste, which, in practice, is similar to an aerobic MBT process. However, source-separated biodegradable waste streams are preferable to mixed waste in order to produce higher quality compost products for horticultural and other uses (Perla 1997, Diaz *et al.* 2002). In developing countries, composting can provide an affordable, sustainable alternative to controlled landfilling, especially where more labour-intensive lower technology strategies are applied to selected biodegradable wastes (Hoornweg *et al.* 1999). It remains to be seen if mechanized recycling and more costly alternatives such as incineration and MBT will be widely implemented in developing countries. Where decisions regarding waste management are made at the local level by communities with limited financial resources seeking an environmentally acceptable solution, often landfilling or composting are selected (Hoornweg 1999, Hoornweg *et al.* 1999, Johannessen & Boyer 1999). Accelerating the introduction of landfill gas extraction and utilization can mitigate the effect of increased CH_4 generation at engineered landfills. Although Kyoto mechanisms such as CDM and JI have already proved to be useful in this regard, the post-2012 situation is unclear [see section entitled 'Landfill methane']. These mechanisms permit the financing of GHG emission reduction projects in developing countries by entities in developed countries which have Kyoto compliance targets.

With regard to wastewater, a priority in developing countries is to increase the historically low rates of collection and treatment (Figure 2). One of the millennium development goals (MDGs) is to reduce by 50% the number of people without access to safe sanitation by 2015. More on-site sanitation rather than centralized collection and treatment may be warranted. In Dakar, Senegal, on-site systems were provided at a cost of about US$ 400 per household. It has been estimated that an annual investment increase from US$ 4 billion to US$ 18 billion will be needed to achieve the MDG target, mostly in east Asia, south Asia and sub-Saharan Africa (World Bank 2005a).

GHG emissions from waste: historical trends, projections, mitigation strategies and policies

As summarized in the AR4, global GHG emissions in 2004 totalled approximately 49 Gt CO_2 eq. This includes the anthropogenic emissions of CO_2, CH_4, N_2O, and F-gases weighted by their 100-year global warming potentials (GWPs) as reported in the IPCC Second Assessment Report (SAR). Total annual emissions of anthropogenic GHGs have increased by approximately 70% since 1970 (24% since 1990). Due primarily to increased fossil fuel combustion, CO_2 emissions have increased more rapidly than the other GHGs and accounted for about 77% of emissions in 2004; concurrently, CH_4 contributed about 14%, N_2O about 8%, and F-gases about 1%. (Rogner *et al*. 2007). Figure 3 compares sectoral emissions for 2004. Methane from landfills and wastewater are the major GHG emissions from the waste sector and accounted for approximately 18% of global anthropogenic CH_4 emissions in 2004.

As shown in Figure 3, the waste sector is responsible for approximately 3% of the annual total, including landfill CH_4, wastewater CH_4 and N_2O, and incinerator CO_2. In order to quantify global GHG emission trends for post-consumer waste, annual national data on waste production and management practices are needed. Annual estimates for many countries are uncertain because data are lacking, inconsistent, or incomplete. Most developing countries use default



data for per capita waste generation with inter-annual changes assumed to be proportional to total or urban population. Developed countries use more detailed methodologies, activity data and emission factors, national statistics and surveys. Standardization of terminology for national waste statistics would greatly improve waste sector data quality.

As landfill CH_4 emissions continue several decades after waste disposal, the estimation of emission trends requires models that include temporal trends. Methane is also emitted during wastewater transport, sewage treatment processes and leaks from anaerobic digesters. The major sources of N_2O are human sewage and wastewater treatment. The CO_2 from the non-biomass portion of incinerated waste is a small source of GHG emissions. The IPCC 2006 Guidelines also provide methodologies for CO_2, CH_4 and N_2O emissions from open burning of waste and for CH_4 and N_2O emissions from composting and anaerobic digestion of biowaste. Waste burning in developing countries is a significant local source of air pollution, constituting a health risk for nearby communities. Composting and other biological treatments emit very small quantities of GHGs but were included in 2006 IPCC Guidelines for completeness.

Table 1 compares estimated emissions and trends from two studies: US EPA (2006) and Monni *et al.* (2006). The US EPA (2006) study collected data from national inventories and projections reported to the United Nations Framework Convention on Climate Change (UNFCCC) and supplemented data gaps with estimates and extrapolations based on IPCC default data and mass balance calculations using the 1996 IPCC Tier 1 methodology for landfill CH_4, which assumes that CH_4 generation and emissions occur during the same year as disposal (IPCC 1996). For landfill CH_4, Monni *et al.* (2006) calculated a time series using the first-order decay (FOD) methodology and default data in the 2006 IPCC Guidelines, taking into account the time lag in landfill emissions compared to year of disposal. Thus the estimates by Monni *et al.* (2006) are lower than US EPA (2006) for the period 1990–2005 because the former reflect slower growth

Table 1: Trends for GHG emissions from waste using (a) 1996 and (b) 2006 IPCC inventory guidelines, extrapolations, and projections (Mt CO_2-eq, rounded). See notes.

Source	1990	1995	2000	2005	2010	2015	2020	2030	2050
Landfill CH_4[a]	760	770	730	750	760	790	820		
Landfill CH_4[b]	340	400	450	520	640	800	1000	1500	2900
Landfill CH_4 (average of a & b)	550	585	590	635	700	795	910		
Wastewater CH_4[a]	450	490	520	590	600	630	670		
WastewaterN_2O[a]	80	90	90	100	100	100	100		
IncinerationCO_2[b]	40	40	50	50	60	60	60	70	80
Total GHG emissions	1120	1205	1250	1345	1460	1585	1740		

Notes: Emissions estimates and projections as follows:
[a]Based on reported emissions from national inventories and national communications, and (for non-reporting countries) on 1996 inventory guidelines and extrapolations (US EPA 2006).
[b]Based on 2006 inventory guidelines and BAU projection (Monni *et al*. 2006).
Total includes landfill CH_4 (average), wastewater CH_4, wastewater N_2O and incineration CO_2.

in emissions relative to the growth in waste. However, the future projected growth in emissions by Monni *et al.* (2006) is higher, because recent European decreases in landfilling are reflected more slowly in the future projections. For comparison, the reported 1995 CH_4 emissions from landfills and wastewater from national inventories were approximately 1000 Mt CO_2-eq (UNFCCC 2005). In general, data from Non-Annex I countries are limited and usually available only for 1994 (or 1990). Direct comparison of emissions in this table with the SRES A1 and B2 scenarios (IPCC 2000) for GHG emissions from waste is problematical: the SRES do not include landfill-gas recovery (commercial since 1975) and project continuous increases in CH_4 emissions based only on population increases to 2030 (AIB-AIM) or 2100 (B2-MESSAGE), resulting in very high emission estimates of > 4000 Mt CO_2-eq year^{-1} for 2050.

Table 1 indicates that total emissions have historically increased and will continue to increase if business-as-usual (BAU) projections are assumed (Monni *et al.* 2006, US EPA, 2006, see also Scheehle & Kruger 2006). However, between 1990 and 2003, the percentage of total global GHG emissions from the waste sector declined by 14–19% for Annex I and Economies in Transition (EIT) countries (UNFCCC 2005). The waste sector contributed 2–3% of the global GHG total for Annex I and EIT countries for 2003, but a higher percentage (4.3%) for non-Annex I countries (various reporting years from 1990–2000) (UNFCCC 2005). In developed countries, landfill CH_4 emissions are stabilizing due to increased landfill CH_4 recovery, decreased landfilling, and decreased waste generation resulting from recycling, local economic conditions and policy initiatives. In developing countries, increased population and urbanization are increasing landfill CH_4 emissions. Likewise, wastewater CH_4 contributions are expected to increase almost 50% between 1990 and 2020, especially in the rapidly developing countries of eastern and southern Asia (US EPA 2006). Estimates of global wastewater N_2O emissions are incomplete and based only on human sewage treatment, but these indicate an increase of 25% between 1990 and 2020. It is important to emphasize, however, that these are BAU scenarios, and actual emissions could be much lower if additional measures are in place. Future reductions in emissions from the waste sector will also depend on the future (post-2012) availability of Kyoto enabling mechanisms such the CDM and JI. As discussed below, these mechanisms have already resulted in significant reductions of landfill CH_4 emissions; in addition, there are also CDM projects in the waste sector that involve avoided CH_4 emissions due to recycling, composting, or incineration.

Uncertainties in Table 1 values are difficult to assess and vary by source. According to 2006 IPCC Guidelines (IPCC 2006), uncertainties can range from 10–30% (for countries with good annual waste data) to more than two-fold (for countries without annual data). The use of default data and the Tier 1 mass balance method (from 1996 inventory guidelines) for many developing countries would be the major source of uncertainty in both the US EPA (2006) study and reported GHG emissions (IPCC 2006). Estimates by Monni *et al*. (2006) were sensitive to the relationship between waste generation and GDP, with an estimated range of uncertainty for the baseline for 2030 of –48% to +24%. Additional sources of uncertainty include the use of default data for waste generation, plus the suitability of parameters and chosen methods for individual countries. However, although country-specific uncertainties may be large, the uncertainties by region and over time are estimated to be smaller.

There are numerous mature technologies that are readily available to mitigate GHG emissions from waste. These technologies include landfilling with landfill gas recovery (reduces CH_4 emissions), post-consumer recycling (avoids waste generation), composting of selected waste fractions (avoids GHG generation), and processes that reduce GHG generation in comparison with landfilling (thermal processes including incineration and industrial co-combustion, MBT with landfilling of residuals, and anaerobic digestion). Therefore, the mitigation of GHG emissions from waste relies on multiple technologies whose application depends on local, regional and national drivers for both waste management and GHG mitigation. In many developing countries, the high

both before and after the period of active gas recovery, sites should be encouraged, where feasible, to install horizontal gas collection systems concurrent with filling and implement solutions to mitigate residual emissions after closure (such as landfill biocovers to microbially oxidize CH_4).

Global landfill CH_4 emissions are estimated to be 500–800 Mt CO_2-eq year^{-1} (Bogner & Matthews 2003, Monni *et al.* 2006, US EPA 2006). However, their direct measurement at small scale (< 1 m^2) can vary over seven orders of magnitude (0.0001 to > 1000 g CH_4 m^{-2} day^{-1}) (Bogner *et al.* 1997a). Results from a limited number of whole landfill CH_4 emissions measurements in Europe, the US and South Africa are in the range of about 0.1–1.0 t CH_4 ha^{-1} day^{-1} (Nozhevnikova *et al.* 1993, Oonk & Boom 1995, Hovde *et al.* 1995, Borjesson 1996, Czepiel *et al.* 1996, Mosher *et al.* 1999, Tregoures *et al.* 1999, Galle *et al.* 2001, Morris 2001, Scharf *et al.* 2002). Installing a vertical or horizontal landfill gas extraction system is the single most important measure to reduce emissions. Field studies have shown that > 90% recovery can be achieved at cells with final cover and an efficient gas extraction system (Spokas *et al.* 2006). Some sites may have less efficient or only partial gas extraction systems, and there are fugitive emissions from landfilled waste prior to and after the implementation of active gas extraction; thus estimates of 'lifetime' recovery efficiencies may be as low as 20% (Oonk & Boom 1995), which argues for early implementation of gas recovery. Other measures to improve gas collection include installation of horizontal gas collection systems concurrent with filling, frequent monitoring and remediation of edge and piping leakages, installation of secondary perimeter extraction systems for gas migration and emissions control, frequent inspection and maintenance of cover materials, installation of geomembrane composite covers (required in the US as final cover); and implementation of bioreactor landfill designs so that the period of active gas production is compressed while early gas extraction is implemented. Landfill CH_4 is being used to fuel industrial boilers; to generate electricity; and to produce a substitute natural gas after removal of CO_2 and trace components. Although electrical output ranges from small 30 kWe microturbines to 50 MWe steam turbine generators, most plants are in the 1–15 MWe range using internal combustion engines or gas turbines. Barriers to landfill gas utilization, especially where it has not been previously implemented, can be local reluctance from electrical utilities to enter into power purchase agreements, as well as reluctance from gas utilities/pipeline companies to transport small percentages of upgraded landfill gas in their pipelines.

A secondary control on landfill CH_4 emissions is microbial methane oxidation by indigenous methanotrophic microorganisms in cover soils. Landfill soils attain the highest rates of CH_4 oxidation recorded in the literature, with rates many times higher than in wetland settings. Recent field studies have demonstrated that oxidation rates can be greater than 200 g m^{-2} day^{-1} in thick, compost-amended 'biocovers' engineered to optimize oxidation (Huber-Humer 2004, Bogner *et al.* 2005). Passive or active methanotrophic biofilters are also effective (Streese & Stegmann 2005, Gebert & Gröngröft 2006). Methane-oxidizing biocovers, biofilters and related technologies are reviewed in this issue (Huber-Humer *et al.* 2008). The CH_4 oxidation rates at landfills can vary over several orders of magnitude and range from negligible to 100% of the CH_4 flux to the cover. Under circumstances of high oxidation potential and low flux of landfill CH_4 from the landfill, it has been demonstrated that atmospheric CH_4 may be oxidized at the landfill surface (Bogner *et al.*, 1995; 1997b; 2005; Borjesson and Svensson, 1997b). In such cases, the landfill cover soils function as a sink rather than a source of atmospheric CH_4. The thickness, physical properties, moisture content, and temperature of cover soils directly affect oxidation, because rates are limited by the transport of CH_4 upward from anaerobic zones and O_2 downward from the atmosphere. The prototype biocover design includes an underlying coarse-grained gas distribution layer to provide more uniform fluxes to the biocover above, and biocovers have been shown to effectively oxidize CH_4 over multiple annual cycles in northern temperate climates (Humer-Humer, 2004). In field settings, stable C isotopic techniques have proved extremely useful to quantify the fraction of CH_4 that is oxidized in landfill cover soils (Chanton & Liptay 2000, de Visscher *et al.* 2004, Powelson *et al.* 2007). A secondary benefit of CH_4 oxidation in cover soils is the co-oxidation of many non-CH_4 organic compounds, especially aromatic and lower chlorinated compounds, thereby reducing their emissions to the atmosphere (Scheutz *et al.* 2003).

Landfills are a significant source of CH_4 emissions, but they are also a long-term sink for carbon (Bogner 1992, Barlaz 1998). As lignin is recalcitrant and cellulosic fractions decompose slowly, a minimum of 50% of the organic carbon landfilled is not typically converted to biogas carbon but remains in the landfill (See references cited on Figure 1). Carbon storage makes landfilling a more competitive alternative from a climate change perspective, especially where landfill gas recovery is combined with energy use (Pingoud *et al.* 1996, Pipatti & Savolainen 1996, Micales & Skog 1997, Pipatti and Wihersaari 1998, Flugsrud *et al.* 2001). The fraction of carbon storage in landfills can vary over a wide range, depending on original waste composition and landfill conditions (for example, see Hashimoto & Moriguchi 2004 for a review addressing harvested wood products).

The reduction of landfill CH_4 emissions in developed countries has been encouraged or mandated by regulatory and policy strategies to increase landfill CH_4 recovery, reduce the landfilling of biodegradable waste, and encourage renewable energy use. In the US, landfill CH_4 emissions are regulated indirectly under the Clean Air Act (CAA) Amendments/New Source Performance Standards (NSPS) by applying a landfill-gas generation model, either measured or default mixing ratios for total non-methane organic compounds (NMOCs) and restricting the emissions of NMOCs. Larger quantities of landfill CH_4 are also being recovered each year to comply with air-quality regulations and provide energy, assisted by national tax credits and local renewable-energy/ green-power initiatives. As discussed above, the EU landfill directive (1999/31/EC) requires a phased reduction in landfilled biodegradable waste to 50% of 1995 levels by 2009 and 35% by 2016, as well as the collection and flaring of landfill gas at existing sites (Commission of the European Community 2001). However, increases in the availability of landfill alternatives (recycling, composting, incineration, anaerobic digestion and MBT) are required to achieve these regulatory goals (Price 2001). Landfill CH_4 recovery has also been encouraged by economic incentives. In the UK, the Non-Fossil Fuel Obligation, which required that a portion of electrical generation capacity be from non-fossil sources, was an incentive for landfill gas-to-electricity projects during the 1980s and 1990s. It has now been replaced by the Renewables Obligation. In the USA, along with CAA regulations, the US EPA Landfill Methane Outreach Program provides technical support, tools and resources to facilitate landfill gas utilization projects in the US and abroad. Periodic US state or federal tax credits also promote landfill gas utilization; almost 50 of the 400+ commercial US projects started up in 1998, just before the expiration of federal tax credits. A small US tax credit has again become available for landfill gas and other renewable energy sources; in addition, some states also provide economic incentives through tax structures or renewable energy credits and bonds. Other US drivers include state requirements that a portion of electrical energy be derived from renewables, green-power programmes (which allow consumers to select renewable providers), regional programmes to reduce GHG emissions (the RGGI/ Regional GHG Initiative in the northeastern states; a state programme in California) and voluntary markets (such as the Chicago Climate Exchange with binding commitments by members to reduce GHG emissions).

In non-Annex I countries, it is anticipated that landfill CH_4 recovery will increase significantly in the developing countries of Asia, South America and Africa during the next two decades as controlled landfilling is phased in as a major waste-disposal strategy. Where this occurs in parallel with deregulated electrical markets and more decentralized electrical generation, it can provide a strong driver for increased landfill CH_4 recovery with energy use. Significantly, both JI in the EIT countries and the CDM in developing countries are providing strong economic incentives for improved landfilling practices (to permit gas extraction) and landfill CH_4 recovery. As of September 2007, 52 landfill gas CDM projects had achieved registration that collectively totalled 16 Mt CO_2-eq [16 million certified emission reductions (CERs)] per year, mostly in Brazil, Argentina, Korea, Chile, and China (http://cdm.unfccc.int/Projects/registered.html) (see Figure 5).

Although eventual landfill gas utilization is desirable, an initial flaring project under CDM can simplify the CDM process (fewer participants, lower capital cost) and permit definition of composite gas quantity and quality prior to capital investment in engines or other utilization hardware. As financial constraints hinder improved waste and wastewater management in EIT and developing countries, the JI and CDM have been useful mechanisms for obtaining external investment from industrialized countries. Further, their benefits are two-fold: improving waste management practices and reducing GHG emissions. The validation of CDM projects requires attention to baselines, additionality and other criteria contained in approved methodologies (Hiramatsu *et al.* 2003); however, for landfill gas CDM projects, CERs (units of t CO_2-eq) are determined directly from quantification of the CH_4 captured and combusted.

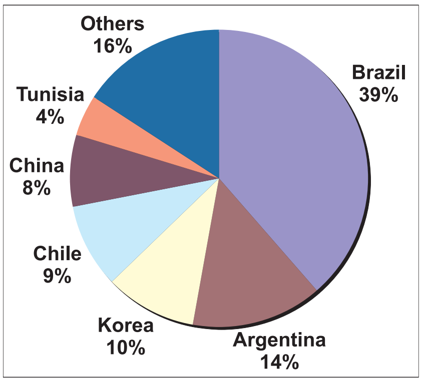


Fig. 5: Distribution of landfill gas CDM projects based on average annual CERs for registered projects as of September 2007 (unfccc.org). Includes 16 million CERs year^{-1} for landfill CH_4 'Others' includes Mexico, Egypt, Peru, South Africa, Tanzania, El Salvador, Israel, Costa Rica, Armenia, Malaysia, Bolivia, Bangladesh, Ecuador, Georgia (each less than 2% of total).

Wastewater methane and nitrous oxide

Both CH_4 and N_2O can be emitted during municipal and industrial wastewater collection and treatment, depending on transport, treatment and operating conditions. Biogas is also produced from anaerobic treatment processes. Residual solids (sludges and biosolids) can also microbially generate CH_4 and N_2O, which may be emitted without gas capture. In developed countries, these emissions are typically small and incidental due to centralized infrastructure and biogas use.

Small wastewater treatment systems include pit latrines, composting toilets and septic tanks. Other systems used in developing countries include inverted trench systems and aerated treatment units. More advanced systems include activated sludge treatment, trickling filters, anaerobic or facultative lagoons, anaerobic digestion and constructed wetlands. Activated sludge treatment is the conventional method for large-scale sewage treatment, and separation of black and grey water can reduce the energy requirements for treatment (UNEP/GPA-UNESCO/IHE 2004). Where feasible, residual solids can be composted and used as a soil conditioner in agriculture and horticulture to recycle carbon, nitrogen and phosphorus (and other elements essential for plant growth); however, high metals content may preclude this option. Lower quality uses for solids may include mine site rehabilitation, highway landscaping, or landfill cover (including biocovers). Some sludges are landfilled, but this practice may result in increased volatile siloxanes and H_2S in the landfill gas. Treated wastewater can be re-used or discharged, but re-use is the most desirable option for agricultural and horticultural irrigation, artificial recharge of aquifers, or industrial applications.

In many developing countries, the absence of a highly developed wastewater infrastructure results in higher wastewater CH_4 and N_2O emissions than in developed countries (see UNFCCC/IPCC, 2004; US EPA, 2006). Decentralized 'natural' treatment processes and septic tanks in developing countries may also result in relatively large CH_4 and N_2O emissions, particularly in China, India and Indonesia where wastewater volumes are increasing rapidly with economic development (Scheehle & Doorn, 2002). The highest regional percentages for CH_4 emissions from wastewater are from Asia (especially China, India), and others with high levels include Turkey, Bulgaria, Iran, Brazil, Nigeria and Egypt. Total global emissions of CH_4 from wastewater handling are expected to rise by more than 45% from 1990 to 2020 (Table 1) with much of the increase from the developing countries of East and South Asia, the middle East, the Caribbean, and Central and South America. In contrast, the EU has projected lower emissions in 2020 relative to 1990 (US EPA 2006).

The contribution of human sewage to atmospheric N_2O is very low and is expected to fluctuate from 80–100 Mt CO_2-eq year^{-1} during the period 1990–2020 (Table 1) in comparison with current total global anthropogenic N_2O emissions of about 3500 Mt CO_2-eq (US EPA 2006). Emission estimates for N_2O from sewage for Asia, Africa, South America and the Caribbean are significantly underestimated since limited data are available, but it is estimated that these countries accounted for > 70% of global emissions in 1990 (UNFCCC/IPCC 2004). Compared with 1990, it is expected that global emissions will rise by about 20% by 2020 (Table 1). The regions with the highest relative N_2O emissions are the developing countries of East Asia, the developing countries of South Asia, Europe and the OECD North America. Regions whose emissions are expected to increase the most by 2020 (with regional increases of 40 to 95%) are Africa, the Middle East, the developing countries of south and east Asia, the Caribbean, and Central and South America (US EPA 2006). Europe and the EIT countries are projected to have lower emissions in 2020 relative to 1990.

Although decision-making tools are available that include environmental trade-offs and costs (Ho 2000), systematic global studies of GHG-reduction costs and potentials for wastewater are still needed. When efficiently applied, wastewater transport and treatment technologies reduce or eliminate GHG generation and emissions; in addition, wastewater management promotes water conservation by preventing pollution, reducing the volume of pollutants, and requiring a smaller volume of water to be treated. Because the size of treatment systems is primarily governed by the volume of water to be treated rather than the mass loading of nitrogen and other pollutants, smaller volumes require smaller treatment plants with lower capital costs. In 2002, the Johannesburg Summit adopted the Millennium Development Goal to reduce the number of people without access to sanitation services by 50% via the financial, technical and capacity-building expertise of the international community. If achieved, the Johannesburg Summit goals would significantly reduce GHG emissions from wastewater. In many countries, wastewater and sludge anaerobic digestion also produce useful biogas for process heating or onsite electricity generation (Government of Japan 1997, Government of Republic of Poland 2001). Anaerobic digestion and composting are also beginning to be implemented or considered in countries eligible for JI and CDM projects.

Carbon dioxide from waste incineration

Waste incineration and other thermal processes reduce the mass of waste and can offset fossil-fuel use; in addition, GHG emissions are largely avoided, except for small emissions of CO_2 from fossil carbon sources (Consonni *et al.* 2005), including plastics and synthetic textiles. Estimated current GHG emissions from waste incineration are small, around 40 Mt CO_2-eq year^{-1}, or less than one-tenth of landfill CH_4 emissions. Recent data for the EU-15 indicate CO_2 emissions from incineration of about 9 Mt CO_2-eq year^{-1} (EIPPC Bureau 2006). Future trends will depend on energy price fluctuations, as well as incentives and costs for GHG mitigation. Monni *et al.* (2006) estimated that incinerator emissions would grow to 80–230 Mt CO_2-eq year^{-1} by 2050 (not including fossil fuel offsets due to energy recovery). Major contributors to this minor source would be the developed countries with high rates of incineration, including Japan (> 70% of waste incinerated), Denmark and Luxembourg (> 50% of waste), as well as France, Sweden, the Netherlands and Switzerland. Incineration rates are increasing in most European countries as a result of the EU Landfill Directive. In 2003, about 17% of municipal solid waste was incinerated with energy recovery in the EU-25 (Eurostat 2003, Statistics Finland 2005). More recent data for the EU-15 (EIPCC Bureau 2006) indicate that 20–25% of the total municipal solid waste is incinerated at over 400 plants with an average capacity of about 500 t day^{-1} (range of 170–1400 t day^{-1}). In the US, only about 14% of waste is incinerated (US EPA 2005), primarily in the more densely populated eastern states. Thorneloe *et al.* (2002), using a life-cycle approach, estimated that US plants reduced GHG emissions by 11 Mt CO_2-eq year^{-1}, when fossil-fuel offsets were taken into account.

Thermal processes for waste include incineration with and without energy recovery, production of refuse-derived fuel (RDF), and industrial co-combustion (including cement kilns: see Onuma *et al.* 2004). Incineration has been widely applied in many developed countries, especially those with limited space for landfilling such as Japan and many European countries. Globally, about 130 Mt of waste are annually combusted in > 600 plants in 35 countries (Themelis 2003). Waste incinerators have been extensively used for more than

20 years with increasingly stringent emission standards in Japan, the EU, the US and other countries. Mass burning is relatively expensive and, depending on plant scale and flue-gas treatment, currently ranges from about 95–150 € t^{-1} waste (87–140 US\$ t^{-1}) (Faaij *et al.* 1998, EIPPC Bureau 2006). Waste-to-energy plants can also produce useful heat or electricity, which improves process economics. Japanese incinerators have routinely implemented energy recovery or power generation (Japan Ministry of the Environment 2006). In northern Europe, urban incinerators have historically supplied fuel for district heating of residential and commercial buildings. Starting in the 1980s, large waste incinerators with stringent emission standards have been widely deployed in Germany, the Netherlands and other European countries. Typically such plants have a capacity of about 1 Mt waste $year^{-1}$, moving grate boilers (which allow mass burning of waste with diverse properties), low steam pressures and temperatures (to avoid corrosion) and extensive flue gas cleaning to conform with EU Directive 2000/76/EC. In 2002, European incinerators for waste-to-energy generated 41 million GJ electrical energy and 110 million GJ thermal energy (Themeli, 2003). Typical electrical efficiencies are 15% to > 20% with more efficient designs becoming available. In recent years, more advanced combustion concepts have penetrated the market, including fluidized bed technology (Dornburg *et al.* 2006).

In developing countries, controlled incineration of waste is infrequently practised because of high capital and operating costs, as well as a history of previously unsustainable projects; however, the uncontrolled burning of waste for volume reduction is still a common practice that contributes to urban air pollution (Hoornweg 1999). Incineration is also not the technology of choice for wet waste, and municipal waste in many developing countries contains a high percentage of food waste with high moisture contents. In some developing countries, however, the rate of waste incineration is increasing. In China, for example, waste incineration has increased rapidly from 1.7% of municipal waste in 2000 to 5% in 2005 (including 67 plants). (Du *et al.* 2006a, b, National Bureau of Statistics of China 2006).

Policies that have encouraged incineration include construction subsidies for incinerators in several countries, usually combined with standards for energy efficiency (Government of Japan 1997, Austrian Federal Government 2001). Tax exemptions for electricity generated by waste incinerators (Government of the Netherlands 2001) and for waste disposal with energy recovery (Government of Norway 2002) have also been adopted. In Sweden, it has been illegal to landfill pre-sorted combustible waste since 2002 (Swedish Environmental Protection Agency 2005). Landfill taxes have also been implemented in a number of EU countries to elevate the cost of landfilling and encourage more costly alternatives (incineration, industrial co-combustion, MBT). In the UK, the landfill tax has also been used as a funding mechanism for environmental and community projects, as discussed by Morris *et al*. (2000) and Grigg & Read (2001).

GHG emissions from biological processes including composting, anaerobic digestion, and mechanical biological treatment

Many developed and developing countries practice composting and anaerobic digestion of mixed waste or biodegradable waste fractions (kitchen or restaurant wastes, garden waste, sewage sludge). Both processes are best applied to source-separated waste fractions. When composting decomposes waste aerobically into CO_2, water and a humic fraction; some carbon storage occurs in the residual compost (see references on Figure 1). Composting can be sustainable at reasonable cost in developing countries; however, choosing more labour-intensive processes over highly mechanized technology on a large scale is typically more appropriate and sustainable; Hoornweg *et al.* (1999) give examples from India and other countries. Depending on compost quality, there are many potential applications for compost in agriculture, horticulture, soil stabilization and soil improvement (increased organic matter, higher water-holding capacity) (Cointreau 2001). However, CH_4 and N_2O can be formed during composting by poor management and the initiation of semi-aerobic (N_2O) or anaerobic (CH_4) conditions; recent studies also indicate potential production of CH_4 and N_2O in well-managed systems (Hobson *et al.* 2005).

Anaerobic digestion produces biogas (CH_4 + CO_2) and biosolids. In particular, Denmark, Germany, Belgium and France have implemented anaerobic digestion systems for waste processing, with the resulting biogas used for process heating, on-site electrical generation and other uses. Minor quantities of CH_4 can be vented from digesters during start-ups, shutdowns and malfunctions. However, the GHG emissions from controlled biological treatment are small compared to CH_4 emissions from landfills without gas recovery (e.g. Vesterinen 1996, Petersen *et al.* 1998, Hellebrand 1998, Beck-Friis, 2001, Detzel *et al.* 2003). The advantages of biological treatment over landfilling are reduced volume and more rapid waste stabilization. Depending on quality, the residual solids can be recycled as fertilizer or soil amendments, used as a CH_4-oxidizing biocovers on landfills (Barlaz *et al.* 2004, Huber-Humer 2004), or landfilled at reduced volumes with lower CH_4 emissions.

Mechanical biological treatment (MBT) of waste is now being widely implemented in Germany, Austria, Italy and other EU countries. In 2004, there were 15 facilities in Austria, 60 in Germany and more than 90 in Italy; the total throughput was approximately 13 Mt with larger plants having a capacity of 600–1300 t day^{-1} (Diaz *et al.* 2006). Mixed waste is subjected to a series of mechanical and biological operations to reduce volume and achieve partial stabilization of the organic carbon. Typically, mechanical operations (sorting, shredding, crushing) first produce a series of waste fractions for recycling or subsequent treatment (including combustion or secondary biological processes). The biological steps consist of either aerobic composting or anaerobic digestion. Composting can occur either in open windrows or in closed buildings with gas collection and treatment. In-vessel anaerobic digestion of selected organic fractions produces biogas for energy use. Compost products and digestion residuals can have potential horticultural or agricultural applications; some MBT residuals are landfilled, or soil-like residuals can be used as landfill cover. Although reductions of as much as 40–60% of the original organic carbon are possible with MBT (Kaartinen 2004), residual materials still retain some potential for CH_4 generation (Bockreis & Steinberg 2005). Compared with landfilling, MBT can theoretically reduce CH_4 generation by as much as 90% (Kuehle-Weidemeier & Doedens 2003). In practice, reductions are smaller and dependent on the specific MBT processes employed (see Binner 2002).

GHG implications of waste reduction, re-use and recycling

Quantifying the GHG-reduction benefits of waste minimization, recycling and re-use requires the application of life-cycle analysis (LCA) tools (Smith *et al.* 2001). At several stages in the life cycle of a product, material efficiency can be increased by more efficient design, material substitution, product recycling, material recycling, and quality cascading (use of recycled material for a secondary product with lower quality demands). All of these measures lead to indirect energy savings (Worrell *et al*. 1995, Dornberg & Faaij 2006), reductions in GHG emissions, and avoidance of GHG generation. This is especially true for products resulting from energy-intensive production processes such as metals, glass, plastic and paper (Tuhkanen *et al.* 2001). In general, the magnitude of avoided GHG-emissions is highly dependent on the specific materials involved, the recovery rates for those materials, the local options for managing materials, and (for energy offsets) the specific fossil fuel avoided (Smith *et al.* 2001). Therefore, existing studies are often not comparable with respect to the assumptions and calculations employed. Nevertheless, virtually all developed countries have implemented comprehensive national, regional or local recycling programmes. For example, Smith *et al.* (2001) thoroughly addressed the GHG-emission benefits from recycling across the EU, and Pimenteira *et al.* (2004) quantified GHG emission reductions from recycling in Brazil.

Widely-implemented policies that encourage waste minimization and recycling include extended producer responsibility (EPR), unit pricing (or PAYT/pay as you throw) and landfill taxes. Waste reduction can also be promoted by recycling programmes, waste minimization and other measures (Miranda *et al.* 1994, Fullerton & Kinnaman 1996). The EPR regulations extend producer responsibility to the post-consumer period, thus providing a strong incentive to redesign products using fewer materials as well as those with increased recycling potential (OECD 2001). A number of successful schemes have been implemented for diverse waste fractions such as packaging waste, old vehicles and electronic equipment. In Germany, the 1994 Closed Substance Cycle and Waste Management Act, other laws and voluntary agreements have restructured waste management over the past 15 years (Giegrich & Vogt 2005). In addition, unit pricing has been widely adopted to decrease landfilled waste and increase recycling (Miranda *et al.* 1996). Some municipalities have reported a secondary increase in waste generation after an initial decrease following implementation of unit pricing, but the 10-year sustainability of these programmes has been demonstrated (Yamakawa & Ueta 2002). Other policies and measures include local subsidies and educational programmes for collection of recyclables, domestic composting of biodegradable waste and procurement of recycled products (green procurement). In the US, for example, states with requirements for separate collection of garden (green) waste, this waste is diverted to composting or used as an alternative daily cover on landfills.

As GDP is correlated to waste generation (Daskalopoulos *et al.* 1998), economic growth affects the consumption of materials, the production of waste, and hence GHG emissions from the waste sector. Decoupling waste generation from economic and demographic drivers, or dematerialization, is often discussed in the context of sustainable development. Many developed countries have reported recent decoupling trends (OECD 2002a), but the literature shows no absolute decline in material consumption in developed countries (Bringezu *et al.* 2004). In other words, solid waste generation does not support an environmental Kuznets curve (Dinda 2004), because environmental problems related to waste are not fully internalized. In Asia, Japan and China are both encouraging ‘circular economy’ or ‘sound material-cycle society’ as a new development strategy, whose core concept is the circular (closed) flow of materials and the use of raw materials and energy through multiple phases (Japan Ministry of the Environment 2003, Yuan *et al.* 2006). This approach is expected to achieve efficient economic growth while discharging fewer pollutants.

Costs and potentials for GHG mitigation from waste

In the waste sector, it is often not possible to clearly separate costs for GHG mitigation from costs for waste management. In addition, waste management costs can exhibit high variability depending on local conditions. Therefore the baseline and cost assumptions, local availability of technologies, and economic and social development issues for alternative waste management strategies need to be carefully defined. An older study by de Jager & Blok (1996) assumed a 20-year project life to compare the cost-effectiveness of various options for mitigating CH_4 emissions from waste in the Netherlands, with costs ranging from –2 US\$ t^{-1} CO_2-eq for landfilling with gas recovery and on-site electrical generation to > 370 US\$ t^{-1} CO_2-eq for incineration. In general, for landfill CH_4 recovery and utilization, project economics are highly site-specific and dependent on the financial arrangements as well as the distribution of benefits, risks and responsibilities among multiple partners. Some representative unit costs for landfill gas recovery and utilization (all in 2003 US\$ kW^{-1} installed power) are: 200–400 for gas collection; 200–300 for gas conditioning (blower/compressor, dehydration, flare); 850–1200 for internal combustion engine/generator; and 250–350 for planning and design (Willumsen 2003).

Smith *et al.* (2001) highlighted major cost differences between EU member states for mitigating GHG emissions from waste. Based on fees (including taxes) for countries with data, this study compared emissions and costs for various waste management practices with respect to direct GHG emissions, carbon sequestration, transport emissions, avoided emissions from recycling due to material and energy savings, and avoided emissions from fossil-fuel substitution via thermal processes and biogas (including landfill gas). Recycling costs are highly dependent on the waste material recycled. Overall, the financial success of any recycling venture is dependent on the current market value of the recycled products. The price obtained for recovered materials is typically lower than separation/reprocessing costs, which can be, in turn, higher than the cost of virgin materials – thus recycling activities usually require subsidies (except for aluminium and paper recycling). Recycling, composting and anaerobic digestion can provide large potential emission reductions, but further implementation is dependent on reducing the cost of separate collection [10–400 € t^{-1} waste (9–380 US\$ t^{-1})] and, for composting, establishing local markets for the compost product. Costs for composting can range from 20–170 € t^{-1} waste (18–156 US\$ t^{-1}) and are typically about 35 € t^{-1} waste (32 US\$ t^{-1}) for open-windrow operations and 50 € t^{-1} waste (46 US\$ t^{-1}) for in-vessel processes. When the replaced fossil fuel is coal, both mass incineration and co-combustion offer comparable and less expensive GHG emission reductions compared to recycling [averaging 64 € t^{-1} waste (59 US\$ t^{-1}), with a range of 30–150 € t^{-1} (28–140 US\$ t^{-1})]. Landfill disposal is the most inexpensive waste management option in the EU [averaging 56 € t^{-1} waste (52 US\$ t^{-1}), ranging from 10–160 € t^{-1} waste (9–147 US\$ t^{-1}), including taxes], but it is also the largest source of GHG emissions. With improved gas management, landfill emissions can be significantly reduced at low cost. However, landfilling costs in the EU are increasing due to increasingly stringent regulations, taxes, and declining capacity. Although there is only sparse information regarding MBT costs, German costs are about 90 € t^{-1} waste (83 US\$ t^{-1}, including landfill disposal fees); recent data suggest that, in the future, MBT may become more cost-competitive with landfilling and incineration.

Costs and potentials for reducing GHG emissions from waste are usually based on landfill CH_4 as the baseline (Pipatti and Wihersaari 1998, IPCC 2000, Bates & Haworth 2001, Delhotal *et al.* 2006, Monni *et al.* 2006). When reporting to the UNFCCC, most developed countries have historically taken the dynamics of landfill gas generation into account whereas most developing countries have not. Basing their study on reported emissions and projections, Delhotal *et al.* (2006) estimated break-even costs for GHG abatement from landfill gas utilization that ranged from about –20 to +70 US\$ $t^{-1}CO_2$-eq, with the lower value for direct use in industrial boilers and the higher value for on-site electrical generation. From the same study, break-even costs (all in US\$ $t^{-1}CO_2$-eq) were approximately 25 for landfill-gas flaring; 240–270 for composting; 40–430 for anaerobic digestion; 360 for MBT and 270 for incineration. These costs were based on the EMF-21 study (US EPA 2003), which assumed a 15-year technology lifetime, 10% discount rate and 40% tax rate.

Compared to thermal and biological processes that only affect future emissions, landfill CH_4 is generated from waste landfilled in previous decades, and gas recovery, in turn, reduces emissions from waste landfilled in previous years. Monni *et al.* (2006) developed baseline and mitigation scenarios for solid waste management using the first order decay (FOD) methodology in the 2006 IPCC Guidelines, which takes into account the timing of emissions. Figure 6 compares the baselines and mitigation scenarios from Monni *et al*. (2006) to the baseline scenario from Delhotal *et al.* (2006). The baseline scenario by Monni *et al.* (2006) assumed that: (1) waste generation will increase with growing population and GDP (using the same population and GDP data as SRES scenario A1b); (2) waste management strategies will not change significantly; and (3) landfill gas recovery and utilization will continue to increase at the historical rate of 5% per year in developed countries (Bogner & Matthews 2003, Willumsen 2003). Mitigation scenarios were developed for 2030 and 2050 that focus on increased landfill gas recovery, increased recycling, and increased incineration. In the increased landfill gas recovery scenario, recovery was estimated to increase 15% per year, with most of the increase in developing countries because of CDM or similar incentives (above baseline of current CDM projects). This growth rate is about triple the current rate and corresponds to a reasonable upper limit, taking into account the fact that recovery in developed countries has already reached high levels, so that increases would come mainly from developing countries, where current lack of funding is a barrier to deployment. Landfill gas recovery was capped at 75% of estimated annual CH_4 generation for developed countries and 50% for developing countries in both the baseline and increased landfill gas recovery scenarios. In the increased incineration scenario, incineration grew 5% each year in the countries where waste incineration occurred in 2000. For OECD countries where no incineration took place in 2000, 1% of the waste generated was assumed to be incinerated in 2012. In non-OECD countries, 1% waste incineration was assumed to be reached only in 2030. The maximum rate of incineration that could be implemented was 85% of the waste generated. The increased recycling scenario assumed a growth in paper and cardboard recycling in all parts of the world using a technical maximum of 60% recycling (CEPI 2003). This maximum was assumed to be reached in 2050. In the mitigation scenarios, only direct emission reductions compared to the baseline CH_4 emissions from landfills were estimated – thus avoided emissions from recycled materials, reduced energy use, or fossil fuel offsets were not included. In the baseline scenario (Figure 6), emissions increase three-fold during the period from 1990 to 2030 and more than five-fold by 2050. These growth rates do not include current or planned legislation relating to either waste minimization or landfilling – thus future emissions may be overestimated. Most of the increase

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Table 2: Economic reduction potential for CH_4 emissions from landfilled waste by level of marginal costs for 2020 and 2030 based on steady state models.

	US$ t^{-1} CO_2 equivalent				
2020 (Delhotal *et al.* 2006)	0	15	30	45	60
OECD	12%	40%	46%	67%	92%
EIT	NA	NA	NA	NA	NA
Non-OECD	NA	NA	NA	NA	NA
Global	12%	41%	50%	57%	88%
	US$ t^{-1} CO_2 equivalent				
2030 (Monni *et al.* 2006)	0	10	20	50	100
OECD	48%	86%	89%	94%	95%
EIT	31%	80%	93%	99%	100%
Non-OECD	32%	38%	50%	77%	88%
Global	35%	53%	63%	83%	91%

Table 3: Economic potential for mitigation of regional landfill CH_4 emissions at various cost categories in 2030 (from estimates by Monni *et al.* 2006). See notes.

Region	Projected emissions for 2030	Total economic mitigation potential (Mt CO_2-eq) at < 100 US$ t^{-1} CO_2-eq	Economic mitigation potential (Mt CO^2-eq) at various cost categories (US$ $t^{-1}CO_2$-eq)			
			< 0	0–US$ 20	US$ 20–50	US$ 50–100
OECD	360	100–200	100–120	20–100	0–7	1
EIT	180	100	30–60	20–80	5	1–10
Non-OECD	960	200–700	200–300	30–100	0–200	0–70
Global	1500	400–1000	300–500	70–300	5–200	10–70

Notes:
1. Costs and potentials for wastewater mitigation are not available.
2. Regional numbers are rounded to reflect the uncertainty in the estimates and may not equal global totals.
3. Landfill carbon sequestration is not considered.
4. The timing of measures limiting landfill disposal affect the annual mitigation potential in 2030. The upper limits of the ranges given assume that landfill disposal is limited in the coming years to 15% of the waste generated globally. The lower limits correspond to the sum of the mitigation potential in the HR and II scenarios in the Monni *et al*. 2006 study.

Other gases in the waste sector: fluorinated gases, non-methane volatile organic compounds and combustion emissions

High global-warming potential (GWP) fluorinated gases have been used for more than 70 years; the most important are the chlorofluorocarbons (CFCs), hydrochloro-fluorocarbons (HCFCs) and the hydrofluorocarbons (HFCs) with the existing bank of CFCs and HCFCs estimated to be > 1.5 Mt and 0.75 Mt, respectively (TFFEoL 2005, IPCC 2005). The CFCs and HCFCs regulated as ozone-depleting substances (ODS) under the Montreal Protocol can persist for many decades in post-consumer waste and occur as trace components in landfill gas (Scheutz *et al.* 2003). These gases have been used as refrigerants, solvents, blowing agents for foams and as chemical intermediates. End-of-life issues in the waste sector are mainly relevant for the foams; for other products, release will occur during use or just after end-of-life. For the rigid foams, releases during use are small (Kjeldsen & Jensen 2001, Kjeldsen & Scheutz 2003, Scheutz *et al.* 2005a), so most of the original content is still present at the end of their useful life. The rigid foams include polyurethane and polystyrene used as insulation in appliances and buildings; in these, CFC-11 and CFC-12 were the main blowing agents until the mid-1990s. The HFCs regulated under the Kyoto Protocol are promoted as substitutions for the ODS. After the mid-1990s, HCFC-22, HCFC-141b and HCFC-142b with HFC-134a have been used (CALEB 2000). Considering that home appliances are the foam-containing product with the shortest lifetime (average maximum lifetime 15 years; TFFEoL 2005), a significant fraction of the CFC-11 in appliances has already entered waste management systems. Building insulation has a much longer lifetime (estimated to be 30–80 years, Gamlen *et al.* 1986), and most of the fluorinated gases from it have not reached the end of their useful life (TFFEoL 2005). Daniel *et al.* (2007) discuss some uncertainties and possible temporal trends for depletion of CFC-11 and CFC-12 banks. A number of countries have adopted collection systems for products still in use based on voluntary agreements (Austrian Federal Government 2001) or EPR regulations for appliances (Government of Japan 2002). Both the EU and Japan have successfully prohibited landfill disposal of appliances containing ODS foams after 2001

(TFFEoL 2005). For other developed countries, appliance foams are often buried in landfills, either directly or following shredding and metals recycling. For rigid foams, shredding results in an instantaneous release with the fraction released related to the final particle size (Kjeldsen & Scheutz 2003). A recent study estimating CFC-11 releases after shredding at three American facilities showed that 60–90% remains and is slowly released following landfill disposal (Scheutz *et al.* 2005a). In the US and other countries, appliances typically undergo mechanical recovery of ferrous metals with landfill disposal of residuals. A study has shown that 8–40% of CFC-11 is lost during segregation (Scheutz *et al.* 2005a); then, during landfilling, the compactors shred residual foam materials and further enhance instantaneous gaseous releases.

In the anaerobic landfill environment, some fluorinated gases may be biodegraded because CFCs and, to some extent, HCFCs can undergo dechlorination (Scheutz *et al.* 2005b). Potentially this may result in the production of more toxic intermediate degradation products (e.g., for CFC-11, the degradation products can be HCFC-21 and HCFC-31); however, experiments have indicated that rapid CFC-11 degradation occurs with only minor production of toxic intermediates (Scheutz *et al.* 2005b). HFCs have not been shown to undergo either anaerobic or aerobic degradation. Thus, landfill attenuation processes may decrease emissions of some fluorinated gases, but not others; however, data are entirely lacking for PFCs, and field studies are needed to verify that CFCs and HCFCs are being attenuated *in situ* in order to guide future policy decisions.

Landfill gas also contains trace concentrations of aromatic, chlorinated and fluorinated hydrocarbons, reduced sulphur gases and other species. High hydrocarbon destruction efficiencies are typically achieved in enclosed flares (> 99%), which are recommended over lower efficiency open flares. Hydrogen sulfide is mainly a problem at landfills which co-dispose large quantities of construction and demolition debris containing gypsum board. At landfill sites, field studies show that NMVOC fluxes through final cover materials are very small with both positive and negative fluxes ranging from approximately 10^{-8} to 10^{-4} g m^{-2} day^{-1} for individual species (Scheutz *et al.* 2003, Bogner *et al.* 2003, Barlaz *et al.* 2004). In general, the emitted compounds consist of species recalcitrant to aerobic degradation (especially higher chlorinated compounds), while negative emissions (uptake from the atmosphere) are observed for species which are readily degradable in aerobic cover soils, such as the aromatics and vinyl chloride (Scheutz *et al.* 2003). Emissions of NO_x resulting from landfill gas combustion can sometimes be a problem for permitting landfill gas engines in strict air quality regions.

For waste incinerators, advanced emission controls are mandated in developed countries (EIPPC Bureau 2006). For reducing emissions of volatile heavy metals and dioxins/dibenzofurans during incineration, the removal of batteries, other electronic waste and polyvinyl chloride (PVC) plastics is recommended prior to combustion (EIPPC Bureau 2006).

Conclusions and long-term considerations for sustainable development

GHG emissions from waste can be effectively mitigated by current technologies. Decisions for alternative waste management strategies are often made locally; however, there are also regional drivers based on national regulatory and policy decisions. Many existing technologies are also cost effective; for example, landfill gas recovery for energy use can be profitable in many developed countries. However, in developing countries, a major barrier to the diffusion of technologies is lack of capital – thus the CDM can provides a major incentive for both improved waste management and GHG emission reductions. For the long term, more profound changes in waste management strategy are expected in both developed and developing countries, including more emphasis on waste minimization, recycling, re-use and energy recovery. Huhtala (1997) studied optimal recycling rates for municipal solid waste using a model that included recycling costs and consumer preferences; results suggested that an overall recycling rate of 50% was achievable, economically justified and environmentally preferable. This rate has already been achieved in many countries with respect to the more valuable waste fractions such as metals and paper (OECD 2002b).

For the many countries that continue to rely on landfilling, increased utilization of landfill CH_4 can provide a cost-effective mitigation strategy. The combination of gas utilization for energy with biocovers to increase CH_4 oxidation can largely mitigate site-specific CH_4 emissions (Huber-Humer 2004, Barlaz *et al.* 2004). These technologies are simple ('low technology') and can be readily deployed at any site. Moreover, R&D to improve gas-collection efficiency, design biogas engines and turbines with higher efficiency, and develop more cost-effective gas purification technologies are underway. These improvements will be largely incremental but will increase options, decrease costs, and remove existing barriers for expanded applications of these technologies.

Advances in waste-to-energy have benefited from general advances in biomass combustion; thus the more advanced technologies such as fluidized bed combustion with emissions control can provide significant future mitigation potential for the waste sector. When the fossil fuel offset is also taken into account, the positive impact on GHG reduction can be even greater (e.g., Pipatti and Savolainen 1996, Lohiniva *et al.* 2002, Consonni *et al.* 2005). Because of the high capital costs, thermal processes are expected to be primarily deployed in developed countries. Advanced combustion technologies are expected to become more competitive as energy prices increase and renewable energy sources gain larger market share.

Anaerobic digestion as part of MBT, or as a stand-alone process for either wastewater or selected wastes (high moisture), is another waste management option. In general, anaerobic digestion technologies incur lower capital costs per unit mass of waste input than incineration; however, in terms of national GHG mitigation potential and energy offsets, their potential is more limited than landfill CH_4 recovery and incineration. When compared to composting, anaerobic digestion has advantages with respect to energy benefits (biogas), reduced process times and reduced volume of residuals; however, as applied in developed countries, it typically incurs higher capital costs per unit mass waste than composting. Projects in which mixed municipal waste was anaerobically digested (e.g., the Valorga project) have been largely discontinued in favour of projects using specific biodegradable fractions such as food waste. In some developing countries such as China and India, small-scale digestion of biowaste streams with CH_4 recovery and use has been successfully deployed for decades as an inexpensive local waste-to-energy strategy – many other countries could also benefit from similar small-scale projects. For both as a primary wastewater treatment process or for secondary treatment of sludges from aerobic processes, anaerobic digestion under higher temperature using thermophilic regimes or two-stage processes can provide shorter retention times with higher rates of biogas production than single-stage mesophilic processes.

With respect to up-front recycling and separation, it is expected that wider implementation of incrementally improving technologies will provide more rigorous process control for recycled waste streams transported to secondary markets or secondary processes, including paper and aluminium recycling, composting and incineration. These improvements should result in more advantageous material and energy balances for both individual components and urban waste streams as a whole. For developing countries, provided sufficient measures are in place to protect workers and the local environment, more labour-intensive recycling practices can conserve materials, gain energy benefits and reduce GHG emissions. In general, existing studies on the mitigation potential for recycling yield variable results because of the differing assumptions and methodologies applied; however, recent studies (i.e., Myllymaa *et al.* 2005) are beginning to quantitatively examine the environmental benefits of alternative waste strategies, including recycling.

With regard to wastewater, it should be recognized that there are substantial emissions from septic tanks, latrines and uncontrolled discharges in developing countries that are not accounted for in current estimates. For developing countries, it is a significant challenge to develop and implement innovative, low-cost but effective and sustainable measures to achieve a basic level of improved sanitation (Moe & Reingans 2006). Historically, sanitation in developed countries has included costly centralized sewerage and wastewater treatment plants, which do not offer appropriate sustainable solutions for either rural areas in developing countries with low population density or unplanned, rapidly growing, peri-urban areas with high population density (Montgomery & Elimelech 2007). It has been demonstrated that a combination of low-cost technology with concentrated efforts for community acceptance, participation and management can successfully expand sanitation coverage; for example, in India where more than 1 million pit latrines have been built and maintained since 1970 (Lenton *et al.* 2005). The combination of household water treatment and 'point-of-use' low-technology improved sanitation in the form of pit latrines or septic systems has been shown to lower diarrhoeal diseases by more than 30% (Fewtrell *et al.* 2005). Wastewater is also a secondary water resource in countries with water shortages. Future trends in wastewater technology include buildings where black water and grey water are separated, recycling the former for fertilizer and the latter for toilets. In addition, low-water-use toilets (3–5 L) and ecological sanitation approaches (including ecological toilets), where nutrients are safely recycled into productive agriculture and the environment, are being used in Mexico, Zimbabwe, China, and Sweden (Esrey *et al*. 2003). These could also be applied in many developing and developed countries, especially where there are water shortages, irregular water supplies, or where additional measures for conservation of water resources are needed. All of these measures also encourage smaller wastewater treatment plants with reduced nutrient loads and proportionally lower GHG emissions.

Through the mitigation of GHG emissions, improved public health, and environmental protection, solid waste and wastewater technologies confer significant co-benefits for sustainable development. In developing countries, improved waste and wastewater management using low- to medium-technology strategies can provide GHG mitigation and public health benefits at low cost through small-scale wastewater management, construction of engineered medium-technology landfills, and controlled composting of organic waste. The major impediment in developing countries is the lack of capital, which jeopardizes improvements in waste and wastewater management. Selected technologies need to be affordable and sustainable in the long term for the combined benefits of improved urban infrastructure and mitigation of GHG emissions.

References

Austrian Federal Government (2001) *Third National Climate Report of the Austrian Federal Government*. Vienna, Austria.

Barlaz, M. (1998) Carbon storage during biodegradation of municipal solid waste components in laboratory-scale landfills. *Global Biogeochemical Cycles*, **12**, 373–380.

Barlaz, M., Green, R., Chanton, J., Goldsmith, R.D. & Hater, G. (2004) Evaluation of a biologically-active cover for mitigation of landfill gas emissions. *Environmental Science and Technology*, **38**, 4891–4899.

Bates, J. & Haworth, A. (2001) *Economic Evaluation of Emission Reductions of Methane in the Waste Sector in the EU: Bottom-up Analysis*. Final Report to DG Environment, European Commission by Ecofys Energy and Environment, by AEA Technology Environment and National Technical University of Athens as part of Economic Evaluation of Sectoral Emission Reduction Objective for Climate Change. AEA Technology, Oxfordshire.

Beck-Friis, B.G. (2001) *Emissions of Ammonia, N_2O, and CH_4 during Composting of Organic Household Waste.* PhD Thesis, Swedish University of Agricultural Sciences, Uppsala.

Berge, N., Reinhart, D., & Townsend, T. (2005) A review of the fate of nitrogen in bioreactor landfills. *Critical Reviews in Environmental Science and Technology*, **35**, 365–399.

Bernache-Perez, G., Sánchez-Colón, S., Garmendia, A.M., Dávila-Villarreal, A., & Sánchez-Salazar, M.E. (2001) Solid waste characterization study in Guadalajara Metropolitan Zone, Mexico. *Waste Management & Research*, **19**, 413–424.

Bingemer, H.G. & Crutzen, P.J. (1987) The production of CH_4 from solid wastes. *Journal of Geophysical Research*, **92**, 2182–2187.

Binner, E. (2002) The impact of mechanical-biological pretreatment on the landfill behaviour of solid wastes. In: Proc. of the Workshop on Biowaste, Brussels, 8–10 April 2002. EU, Brussels.

Bockreis, B. & Steinberg, I. (2005) Influence of mechanical-biological waste pre-treatment methods on gas formation in landfills. *Waste Management*, **25**, 337–343.

Bogner, J. (1992) Anaerobic burial of refuse in landfills: increased atmospheric methane and implications for increased carbon storage. *Ecological Bulletin*, **42**, 98–108.

Bogner, J. & Spokas, K. (1993) Landfill CH_4: rates, fates, and role in global carbon cycle. *Chemosphere,* **26***,* 366–386.

Bogner, J. & Matthews, E. (2003) Global methane emissions from landfills: New methodology and annual estimates 1980–1996. *Global Biogeochemical Cycles,* **17**, 34-1–34-18.

Bogner, J., Spokas, K., Burton, E., Sweeney, R., & Corona, V. (1995) Landfills as atmospheric methane sources and sinks. *Chemosphere,* **31**, 4119–4130.

Bogner, J., Meadows, M., & Czepiel, P. (1997a) Fluxes of methane between landfills and the atmosphere: natural and engineered controls. *Soil Use and Management*, **13**, 268–277.

Bogner, J., Spokas, K., & Burton, E. (1997b) Kinetics of methane oxidation in landfill cover materials: major controls, a whole-landfill oxidation experiment, and modeling of net methane emissions. *Environmental Science and Technology*, **31**, 2504–2614.

Bogner, J., Spokas, K., & Burton, E. (1999) Temporal variations in greenhouse gas emissions at a midlatitude landfill. *Journal of Environmental Quality*, **28**, 277–288.

Bogner, J., Scheutz, C., Chanton, J., Blake, D., Morcet, M., Aran, C., & Kjeldsen, P. (2003) Field measurement of non-methane organic compound emissions from landfill cover soils. In: Proc. of the Sardinia '03, International Solid and Hazardous Waste Symposium. CISA, University of Cagliari, Sardinia.

Bogner, J., Spokas, K., Chanton, J., Powelson, D., & Abichou, T. (2005) Modeling landfill methane emissions from biocovers: a combined theoretical-empirical approach. In: Proc. of the Sardinia '05, International Solid and Hazardous Waste Symposium. CISA, University of Cagliari, Sardinia.

Bogner, J., Abdelrafie Ahmed, M., Diaz, C., Faaij, A., Gao, Q., Hashimoto, S., Mareckova, K., Pipatti, R., & Zhang, T. (2007) Chapter 10. Waste Management. In:. Metz, B., Davidson, O.R., Bosch, P.R., Dave, R. & Meyer, L.A. (eds): *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridaijokd

Bojessson,G.(1997a)(L)-5.6(a)-028(n-5.8dr)0.8f(il-9(l04(oU)3.8v)-4.9ea)-0.1r).4(SU)3.8oU)3.8(i)-(il-9so)-4.1.e
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EEA (2004) *Greenhouse Gas Emission Trends and Projections in Europe 2004*. European Environment Agency (EEA) Report no5/2004, Progress EU and its member states towards achieving their Kyoto targets. EEA, Luxembourg.

EEA (2005) *European Environment Outlook*. EU Report 4/2005, ISSN 1725-9177, Luxembourg, published by the European Environment Agency (EEA), Copenhagen.

EIPPC Bureau (2006) *Reference Document on the Best Available Techniques for Waste Incineration (BREF), Integrated Pollution Prevention and Control*. European IPPC, Seville, Spain, <http://eippcb.jrc.es>

Esrey, S., Andersson, I., Hillers, A. & Sawyer, R. (2003): Closing the loop – ecological sanitation for food security. Swedish International Development Agency (SIDA), *Publications on Water Resources,* 18, (2nd edition) Mexico City, Mexico.

EU (1999) Official Journal of the European Communities 16.7.1999. Council Directive 1999/31/3C of 26 April 1999 on the landfill of waste. European Commission, Brussels.

Eurostat (2003) *Waste Generated and Treated in Europe. Data 1990–2001*. Office for Official Publications of the European Communities, Luxemburg.

Faaij, A., Hekkert, M., Worrell, E., & van Wijk, A. (1998) Optimization of the final waste treatment system in the Netherlands. *Resources, Conservation, and Recycling*, **22**, 47–82.

Fewtrell, L., Kaufmann, R., Kay, D., Enanoria, W., Haller, L., & Colford, J. (2005) Water, sanitation, and hygiene interventions to reduce diarrhoea in less developed countries: a systematic review and meta-analysis. *Lancet*, **5**, 42–52.

Flugsrud, K., Hoem, B., Kvingedal, E., & Rypdal, K. (2001) *Estimating Net Emissions of CO_2 from Harvested Wood Products*. SFT report 1831/200, Norwegian Pollution Control Authority, Oslo. <http://www.sft.no/publikasjoner/luft/1831/ta1831.pdf> (accessed 29 June 2007).

Fullerton, D. & Kinnaman, T.C. (1996) Household responses to pricing garbage by the bag. *American Economic Review*, **86**, 971–984.

Galle, B., Samuelsson, J., Svensson, B., & Borjesson, G. (2001) Measurements of methane emissions from landfills using a time correlation tracer method based on FTIR absorption spectroscopy. *Environmental Science and Technology*, **35**, 21–25.

Gamlen, P.H., Lane, B.C., & Midgley, P.M. (1986) The production and release to the atmosphere of CCl_3F and CCl_2F_2 (Chlorofluorocarbons CFC-11 and CFC-12). *Atmospheric Environment*, **20**, 1077–1085.

Gebert, J. & Gröngröft, A. (2006) Passive landfill gas emission – influence of atmospheric pressure and implications for the operation of methane-oxidising biofilters. *Waste Management*, **26**, 245–251.

Giegrich, J. & Vogt, R. (2005) *The Contribution of Waste Management to Sustainable Development in Germany*. Umweltbundesamt Report FKZ 203 92 309, Berlin.

Government of Japan (1997) *Japan's Second National Communication to the UNFCCC.*, Tokyo.

Government of Japan (2002) *Japan's Third National Communication to the UNFCCC*. Tokyo.

Government of the Netherlands (2001) *Third Netherlands National Communication on Climate Change Policies.* The Hague.

Government of Norway (2002) *Norway's Third National Communication to the UNFCC*. Oslo.

Government of Republic of Poland (2001) *Third National Communication to the COP of the UNFCCC*. Warsaw.

Griffiths, A.J. & Williams, K.P. (2005) Thermal treatment options. *Waste Management World*, July-August 2005.

Grigg, S.V.L. & Read, A.D. (2001) A discussion on the various methods of application for landfill tax credit funding for environmental and community projects. *Resources, Conservation and Recycling,* **32**, 389–409.

Hashimoto, S. and Moriguchi, Y. (2004) *Data book: Material and Carbon Flow of Harvested Wood in Japan*. CGER Report D034, National Institute for Environmental Studies, Japan, Tsukuba. 40 pp. <http://www-cger.nies.go.jp/> (accessed 29 June 2007).

Hellebrand, H.J. (1998) Emissions of N_2O and other trace gases during composting of grass and green waste. *Journal of Agricultural Engineering Research*, **69**, 365–375.

Hiramatsu, A., Hanaki, K., & Aramaki, T. (2003) Baseline options and greenhouse gas emission reduction of clean development mechanism project in urban solid waste management. *Mitigation and Adaptation Strategies for Global Change*, **8**, 293–310.

Ho, G. (2000) Proceedings of the Regional Workshop on Sustainable Wastewater and Stormwater Management in Latin America and the Caribbean. 27–31 March 2000, International Environmental Technology Centre, Report Series – Issue 10, pp. 115–156. United Nations, Osaka, Shiga.

Hobson, A., Frederickson, J., & Dise, N. (2005) CH_4 and N_2O from mechanically turned windrow and vermicomposting systems following in-vessel pre-treatment. *Waste Management*, **25**, 345–352.

Hoornweg, D. (1999) *What a Waste: Solid Waste Management in A*sia. Report of Urban Development Sector Unit, East Asia and Pacific Region, World Bank, Washington, DC.

Hoornweg, D., Thomas, L., & Otten, L. (1999) *Composting and its Applicability in Developing Countries*. Urban Waste Management Working Paper 8, Urban Development Division, World Bank, Washington, DC.

Hovde, D.C., Stanton, A.C., Meyers, T.P., & Matt, D.R. (1995) Methane emissions from a landfill measured by eddy correlation using a fast-response diode laser sensor. *Journal of Atmospheric Chemistry,* **20**, 141–162.

Huang, Q., WM6(its (s)0.Q(C.Dnto(M6(itsn,)9(L)-7.2(.)2ts (s)0.Xtri)2ts B.sn,) Khan, M.Z.A. & Abu-Ghararath, A.H. (1991) New approach for estimating energy content of municipal solid waste. *Journal of Environmental Engineering*, **117**, 376–380.

Kjeldsen, P. & Jensen, M.H. (2001) Release of CFC-11 from disposal of polyurethane from waste. *Environmental Science and Technology*, **35**, 3055–3063.

Kjeldsen, P. & Scheutz, C. (2003) Short and long term releases of fluorocarbons from disposal of polyurethane foam waste. *Environmental Science and Technology*, **37**, 5071–5079.

Kuehle-Weidemeier, M. & Doedens, H. (2003) Landfilling and properties of mechanical-biological treated municipal waste. In: Proc. of the Sardinia '03, International Solid and Hazardous Waste Symposium, October 2005. CISA, University of Cagliari, Sardinia.

Lenton, R., Wright, A.M., & Lewis, K. (2005) *Health, Dignity, and Development: What Will it Take? UN Millennium Project Task Force on Water and Sanitation*. Earthscan, London.

Lohiniva, E., Sipilä, K., Mäkinen, T., & Hietanen, L. (2002) *Waste-to-energy and Greenhouse Gas Emissions*. Espoo, VTT – Tiedotteita – Research Notes 2139, 119 pp. (in Finnish) (published in English in 2006).

Mertins, L., Vinolas, C., Bargallo, A., Sommer, G., and Renau, J. (1999) *Development and Application of Waste Factors – An Overview*. Technical Report No. 37, European Environment Agency, Copenhagen.

Micales, J.A. & Skog, K.E. (1997) The decomposition of forest products in landfills. *International Biodeterioration and Biodegradation*, **39**, 145–158.

Miranda, M.L., Everett, J.W., Blume, D., & Roy, B.A. (1994) Market-based incentives and residential municipal solid waste. *Journal of Policy Analysis and Management*, **13**, 681–698.

Miranda, M.L., Bauer, S.D., & Aldy, J.E. (1996) *Unit Pricing Programs for Residential Municipal Solid Waste: an Assessment of the Literatures*. Report prepared for U.S. Environmental Protection Agency, Washington DC.

Moe, C.L. & Reingans, R.D. (2006) Global challenges in water, sanitation, and health. *Journal of Water Health*, **4**, 41–57.

Monni, S., Pipatti, R., Lehtilä, A., Savolainen, I., & Syri, S. (2006) *Global Climate Change Mitigation Scenarios for Solid Waste Management*. Technical Research Centre of Finland. VTT Publications, Espoo.

Montgomery, M.A. & Elimelech, M. (2007) Water and sanitation in developing countries: including health in the equation. *Environmental Science and Technology*, **41**, 16–24.

Morris, J.R. (2001) *Effects of Waste Composition on Landfill Processes under Semi-arid Conditions*. PhD Thesis, Faculty of Engineering, University of the Witwatersrand, Johannesburg, S. Africa.

Morris, J.R., Phillips, P.S., & Read, A.D. (2000) The UK landfill tax: financial implications for local authorities. *Public Money and Management*, **20**, 51–54.

Mosher, B., Czepiel, P., Harriss, R., Shorter, J., Kolb, C., McManus, J.B., Allwine, E., & Lamb, B. (1999) Methane emissions at nine landfill sites in the northeastern United States. *Environmental Science and Technology*, **33**, 2088–2094.

Myllymaa, T., Dahlbo, H., Ollikainen, M., Peltola, S., & Melanen, M. (2005) *A Method for Implementing Life cycle Surveys of Waste Management Alternatives' Environmental and Cost Effects,* vol. 750. Suomen Ympäristö – Finnish Environment Helsinki.

National Bureau of Statistics of China (2006) *China Statistical Yearbook 2005/2006*. China Statistics Press, Beijing.

Nozhevnikova, A.N., Lifshitz, A.B., Lebedev, V.S., & Zavarin, G.A. (1993) Emissions of methane into the atmosphere from landfills in the former USSR. *Chemosphere*, **26**, 401–417.

OECD (2001) *Extended Producer Responsibility: A Guidance Manual for Governments*. OECD Publishers, Paris.

OECD (2002a): *Indicators to Measure Decoupling of Environmental Pressure from Economic Growth*. OECD Environment Directorate. SG/SD(2002)1/FINAL.

OECD (2002b) *Environmental Data Waste Compendium 2002.* Environmental Performance and Information Division, OECD Environment Directorate. Working Group on Environmental Information and Outlooks. 27 pp.

OECD (2003): *OECD Environmental Data Compendium 2002*. Paris. <www.oecd.orgn> (accessed 25 June 2007).

OECD (2004) *Towards Waste Prevention Performance Indicators*. OECD Environment Directorate. Working Group on Waste Prevention and Recycling and Working Group on Environmental Information and Outlooks. 197 pp.

Ojeda-Benitez, S. & Beraud-Lozano, J.L. (2003) Characterization and quantification of household solid wastes in a Mexican city. *Resources, Conserva.1(rc-7-4. ((ervan.9((an (R)13. (e)c.5(m)-y. (oi)5lirc-7-g.4(.) T*

Scharf, H., Oonk, H., Hensen, A., & van Rijn, D.M.M. (2002) Emission measurements as a tool to improve methane emission estimates. In: Proc. of the Intercontinental Landfill Research Symposium, Asheville, NC. North Carolina State University, Raleigh, North Carolina.

Scheehle, E. & Doorn, M. (2002) Improvements to the U.S. Wastewater CH_4 and N_2O Emissions Estimates. US EPA report, UE EPA, Washington, DC.

Scheehle, E. & Kruger, D. (2006) Global anthropogenic methane and nitrous oxide emissions. In: Yatchew, A. (ed.): *Multi-Greenhouse Gas Mitigation and Climate Policy, Special Issue #3 Energy Journal*. International Association for Energy Economics, Cleveland, Ohio.

Scheutz, C., Bogner, J., Chanton, J., Blake, D., Morcet, M., & Kjeldsen, P. (2003) Comparative oxidation and net emissions of CH_4 and selected non-methane organic compounds in landfill cover soils. *Environmental Science and Technology,* **37**, 5143–5149.

Scheutz, C., Dote, Y., Fredenslund, A.M., Mosbæk, H., & Kjeldsen, P. (2005a) Attenuation of insulation foam-released fluorocarbons in landfills. *Environmental Science and Technology* (submitted).

Scheutz, C., Fredenslund, A.M., Tant, M., & Kjeldsen, P. (2005b) Release of fluorocarbons from insulation foam in home appliances during shredding. *Environmental Science and Technology* (submitted).

Silva, G. (1998) Private participation in the water and sewerage sector – recent trends. In: *Public Policy for the Private Sector (Viewpoint): Water*, pp. 12–20. World Bank, Washington, DC.

Simmons, P., Goldstein, N., Kaufman, S., Themelis, N., & Thompson, Jr., J. (2006) The state of garbage in America. *Biocycle*, **47**, 26–35.

Smith, A., Brown, K., Ogilvie, S., Rushton, K., & Bates, J. (2001) *Waste Management Options and Climate Change*. Final Report ED21158R4.1 to the European Commission, DG Environment, AEA Technology, Oxfordshire.

Spokas, K., Bogner, J., Chanton, J., Morcet, M., Aran, C., Graff, C., Moreau-le-Golvan, Y., Bureau, N., & Hebe, I. (2006) Methane mass balance at three landfill sites: what is the efficiency of capture by gas collection systems? *Waste Management*, **26**, 516–525.

Statistics Finland (2005) Environment Statistics. *Environment and Natural Resources*, 2, Helsinki, 208 pp.

Stegmann, R. (2005) Mechanical biological pretreatment of municipal solid waste. In: Proc. of the Sardinia ’05, International Waste Management and Landfill Symposium, October, 2005. CISA, University of Cagliari, Sardinia.

Streese, J. & Stegmann, R. (2005) Potentials and limitations of biofilters for methane oxidation. In: Proc. of the Sardinia ’05, International Waste Management and Landfill Symposium, October, 2005. CISA, University of Cagliari, Sardinia.

Swedish Environmental Protection Agency (2005) *A Strategy for Sustainable Waste Management: Sweden’s Waste Plan*. Swedish Environmental Protection Agency.

TFFEoL (2005) *Report of the UNEP Task Force on Foam End-of-life Issues. Task Force on Foam End-of-Life Issues*, May 2005, Montreal Protocol on Substances that Deplete the Ozone Layer. UNEP Technology and Economic Assessment Panel. UNEP, Nairobi, Kenya.

Themelis, N. (2003) An overview of the global waste-to-energy industry. *Waste Management World*, 2003–2004 Review Issue July–August 2003, 40–47.

Thorneloe, S., Weitz, K., Nishtala, S., Yarkosky, S., & Zannes, M. (2002) The impact of municipal solid waste management on greenhouse gas emissions in the United States. *Journal of the Air & Waste Management Association*, **52**, 1000–1011.

Thorneloe, S., Weitz, K., & Jambeck, J. (2005) Moving from solid waste disposal to materials management in the United States. In: Proc. of the Sardinia ’05, International Solid and Hazardous Waste Symposium. CISA, University of Cagliari, Sardinia.

Tregoures, A., Beneito, A., Berne, P., Gonze, M.A., Sabroux, J.C., Savanne, D., Pokryszka, Z., Tauziede, C., Cellier, P., Laville, P., Milward, R., Arnaud, A., & Burkhalter, R. (1999) Comparison of seven methods for measuring methane flux at a municipal solid waste landfill site. *Waste Management and Research*, **17**, 453–458.

Tsujimoto, Y., Masuda, J., Fukuyama, J., & Ito, H. (1994) N_2O emissions at solid waste disposal sites in Osaka City. *Air Waste*, **44**, 1313–1314.

Tuhkanen, S., Pipatti, R., Sipilä, K., & Mäkinen, T. (2001) The effect of new solid waste treatment systems on greenhouse gas emissions. In: Williams, D.J., Durie, R.A., Mcmullan, P., Paulson, C.A.J., & Smith, A.Y. (eds): Greenhouse Gas Control Technologies. Proc. of the Fifth International Conference on Greenhouse Gas Control Technologies (GHGT-5), pp. 1236-1241. CSIRO Publishing, Collingwood.

UNEP/GPA-UNESCO/IHE (2004) *Improving Municipal Wastewater Management in Coastal Cities.* Training Manual, version 1, February 2004, pp. 49-81 and 103-117, UNEP/GPA Coordination Office, The Hague, Netherlands.

UNFCCC (2005) Key GHG Data. United Nations Framework Convention on Climate Change, <http://www.unfccc.int/resource/docs/publications/keyghg.pdf> (accessed 25 June 2007).

UNFCCC/IPCC (2004) Historical data and projections of greenhouse gas emissions – CH_4 and N_2O. IPCC Secretariat, Geneva, Switzerland.

US EPA (1999) *National Source Reduction Characterization Report for Municipal Solid Waste in the United States*. EPA 530R-99-034, Office of Solid Waste and Emergency Response, Washington, DC.

US EPA (2003) *International Analysis of Methane and Nitrous Oxide Abatement Opportunities*. Report to Energy Modeling Forum, Working Group 21. US Environmental Protection Agency, Washington, DC. <http://www.epa.gov/methane/intlanalyses.html> (accessed 5 January 2008).

US EPA (2005) *Municipal Solid Waste Generation, Recycling and Disposal in the United States: Facts and Figures for 2003*. US Environmental Protection Agency, Washington, DC. www.epa.gov./epaoswer/nonhw/muncpl/pubs/msw05rpt.pdf (accessed 25 June 2007).

US EPA (2006) *Global Anthropogenic Non-CO_2 Greenhouse Gas Emissions: 1990–2020*. Office of Atmospheric Programs, Climate Change Division. <http://www.epa.gov/nonco2/econ-inv/pdfs/global_emissions.pdf> (accessed 25 June 2007).

Vesterinen, R. (1996) Greenhouse gas emissions from composting. SIHTI-Research Programme, Seminar 13–14 March 1996, Hanasaari, Espoo, Finland. (in Finnish)

WHO-UNICEF (2005) *Joint Monitoring Programme: Worldwide Sanitation*. <http://www.who.int/mdg/publications/MDG_Report_08_2005.pdf/mdg_report/en/index> (accessed 25 June 2007).

WHO/UNICEF/WSSCC (2000) Global water supply and sanitation assessment: 2000 Report. WHO/UNICEF Joint Monitoring Programme for Water Supply and Sanitation. <http://www.who.int> (accessed 25 June 2007).

Willumsen, H.C. (2003) Landfill gas plants: number and type worldwide. In: Proc. of the Sardinia ’05, International Solid and Hazardous Waste Symposium, October 2005. CISA, University of Cagliari, Sardinia.

World Bank and IMF (2006) *Global Monitoring Report 2006: Ensuring Environmental Sustainability Target 10*. World Bank and International Monetary Fund, Washington, DC.

World Bank (1997) *Toolkits for Private Sector Participation in Water and Sanitation*. World Bank, Washington DC.

World Bank (2005a) World Development Indicators 2005: Environment, Table 2.15 Disease Prevention: Coverage and Quality. Table 3.6: Water pollution, Table 3.11: Urban Environment, <http://www.worldbank.org/data/wdi2005/wditext> (accessed 25 June 2007).

World Bank (2005b) Global Monitoring Report 2005, Chapter 3: Scaling up service delivery. WHO and UNICEF joint monitoring program, Washington, DC.

Worrell, E., Faaij, A., Phylipsen, G.J.M., & Blok, K (1995) An approach for analysing the potential for material efficiency improvement. *Resources, Conservation, and Recycling*, **13**, 215–232.

WRAP (2006) Environmental benefits of recycling, an international review of life cycle comparisons for key materials in the UK recycling sector. Waste and Resources Action Program, Peer reviewed report prepared by H. Wenzel *et al.*, Danish Technical University, published by WRAP, Banbury, Oxfordshire, UK.

Yamakawa, H. & Ueta, K. (2002) Waste reduction through variable charging programs: its sustainability and contributing factors. *Journal of Material Cycles and Waste Management*, **4**, 77–86.

Yuan, Z., Bi, J. & Moriguchi, Y. (2006) The circular economy-a new development strategy in China. *Journal of Industrial Ecology,* **10**, 4–8.

Zinati, G.M., Li, Y.C., & Bryan, H.H. (2001) Utilization of compost increases organic carbon and its humin, humic, and fulvic acid fractions in calcareous soil. *Compost Science & Utilization,* **9**, 156–162.