10

Waste Management

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EXECUTIVE SUMMARY

Post-consumer waste is a small contributor to global greenhouse gas (GHG) emissions (<5%) with total emissions of approximately 1300 $MtCO_2$-eq in 2005. The largest source is landfill methane (CH_4), followed by wastewater CH_4 and nitrous oxide (N_2O); in addition, minor emissions of carbon dioxide (CO_2) result from incineration of waste containing fossil carbon (C) (plastics; synthetic textiles) *(high evidence, high agreement)*. There are large uncertainties with respect to direct emissions, indirect emissions and mitigation potentials for the waste sector. These uncertainties could be reduced by consistent national definitions, coordinated local and international data collection, standardized data analysis and field validation of models *(medium evidence, high agreement)*. With respect to annual emissions of fluorinated gases from post-consumer waste, there are no existing national inventory methods for the waste sector, so these emissions are not currently quantified. If quantified in the future, recent data indicating anaerobic biodegradation of chlorofluorocarbons (CFCs) and hydrochlorofluorocarbons (HCFCs) in landfill settings should be considered *(low evidence, high agreement)*.

Existing waste-management practices can provide effective mitigation of GHG emissions from this sector: a wide range of mature, environmentally-effective technologies are available to mitigate emissions and provide public health, environmental protection, and sustainable development co-benefits. Collectively, these technologies can directly reduce GHG emissions (through landfill gas recovery, improved landfill practices, engineered wastewater management) or avoid significant GHG generation (through controlled composting of organic waste, state-of-the-art incineration and expanded sanitation coverage) *(high evidence, high agreement)*. In addition, waste minimization, recycling and re-use represent an important and increasing potential for indirect reduction of GHG emissions through the conservation of raw materials, improved energy and resource efficiency and fossil fuel avoidance *(medium evidence, high agreement)*.

Because waste management decisions are often made locally without concurrent quantification of GHG mitigation, the importance of the waste sector for reducing global GHG emissions has been underestimated *(medium evidence, high agreement)*. Flexible strategies and financial incentives can expand waste management options to achieve GHG mitigation goals – in the context of integrated waste management, local technology decisions are a function of many competing variables, including waste quantity and characteristics, cost and financing issues, infrastructure requirements including available land area, collection and transport considerations, and regulatory constraints. Life cycle assessment (LCA) can provide decision-support tools *(high evidence, high agreement)*.

Commercial recovery of landfill CH_4 as a source of renewable energy has been practised at full scale since 1975 and currently exceeds 105 $MtCO_2$-eq, yr. Because of landfill gas recovery and complementary measures (increased recycling, decreased landfilling, use of alternative waste-management technologies), landfill CH_4 emissions from developed countries have been largely stabilized *(high evidence, high agreement)*. However, landfill CH_4 emissions from developing countries are increasing as more controlled (anaerobic) landfilling practices are implemented; these emissions could be reduced by both accelerating the introduction of engineered gas recovery and encouraging alternative waste management strategies *(medium evidence, medium agreement)*.

Incineration and industrial co-combustion for waste-to-energy provide significant renewable energy benefits and fossil fuel offsets. Currently, >130 million tonnes of waste per year are incinerated at over 600 plants *(high evidence, high agreement)*. Thermal processes with advanced emission controls are proven technology but more costly than controlled landfilling with landfill gas recovery; however, thermal processes may become more viable as energy prices increase. Because landfills produce CH_4 for decades, incineration, composting and other strategies that reduce landfilled waste are complementary mitigation measures to landfill gas recovery in the short- to medium-term *(medium evidence, medium agreement)*.

Aided by Kyoto mechanisms such as the Clean Development Mechanism (CDM) and Joint Implementation (JI), as well as other measures to increase worldwide rates of landfill CH_4 recovery, the total global economic mitigation potential for reducing landfill CH_4 emissions in 2030 is estimated to be >1000 $MtCO_2$-eq (or 70% of estimated emissions) at costs below 100 US$/$tCO_2$-eq/yr. Most of this potential is achievable at negative to low costs: 20–30% of projected emissions for 2030 can be reduced at negative cost and 30–50% at costs <20 US$/$tCO_2$-eq/yr. At higher costs, more significant emission reductions are achievable, with most of the additional mitigation potential coming from thermal processes for waste-to-energy *(medium evidence, medium agreement)*.

Increased infrastructure for wastewater management in developing countries can provide multiple benefits for GHG mitigation, improved public health, conservation of water resources, and reduction of untreated discharges to surface water, groundwater, soils and coastal zones. There are numerous mature technologies that can be implemented to improve wastewater collection, transport, re-use, recycling, treatment and residuals management *(high evidence, high agreement)*. With respect to both waste and wastewater management for developing countries, key constraints on sustainable development include the local availability of capital as well as the selection of appropriate and truly sustainable technology in a particular setting *(high evidence, high agreement)*.

10.1 Introduction

Waste generation is closely linked to population, urbanization and affluence. The archaeologist E.W. Haury wrote: 'Whichever way one views the mounds [of waste], as garbage piles to avoid, or as symbols of a way of life, they…are the features more productive of information than any others.' (1976, p.80). Archaeological excavations have yielded thicker cultural layers from periods of prosperity; correspondingly, modern waste-generation rates can be correlated to various indicators of affluence, including gross domestic product (GDP)/cap, energy consumption/cap, , and private final consumption/cap (Bingemer and Crutzen, 1987; Richards, 1989; Rathje *et al.*, 1992; Mertins *et al.*, 1999; US EPA, 1999; Nakicenovic *et al.*, 2000; Bogner and 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 and 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. A cornerstone of sustainable development is the establishment of affordable, effective and truly sustainable waste management practices in developing countries. It must be further emphasized that multiple public health, safety and environmental co-benefits accrue from effective waste management practices which concurrently reduce GHG emissions and improve the quality of life, promote public health, prevent water and soil contamination, conserve natural resources and provide renewable energy benefits.

The major GHG emissions from the waste sector are landfill CH_4 and, secondarily, wastewater CH_4 and N_2O. In addition, the incineration of fossil carbon results in minor emissions of CO_2. Chapter 10 focuses on mitigation of GHG emissions from *post-consumer* waste, as well as emissions from municipal wastewater and high biochemical oxygen demand (BOD) industrial wastewaters conveyed to public treatment facilities. Other chapters in this volume address *pre-consumer* GHG emissions from waste within the industrial (Chapter 7) and energy (Chapter 4) sectors which are managed within those respective sectors. Other chapters address agricultural wastes and manures (Chapter 8), forestry residues (Chapter 9) and related energy supply issues including district heating (Chapter 6) and transportation biofuels (Chapter 5). National data are not available to quantify GHG emissions associated with waste transport, including reductions that might be achieved through lower collection frequencies, higher routing efficiencies or substitution of renewable fuels; however, all of these measures can be locally beneficial to reduce emissions.

It should be noted that a separate chapter on post-consumer waste is new for the Fourth Assessment report; in the Third Assessment Report (TAR), GHG mitigation strategies for waste were discussed primarily within the industrial sector (Ackerman, 2000; IPCC, 2001a). It must also be stressed that there are high uncertainties regarding global GHG emissions from waste which result from national and regional differences in definitions, data collection and statistical analysis. Because of space constraints, this chapter does not include detailed discussion of waste management technologies, nor does this chapter prescribe to any one particular technology. Rather, this chapter focuses on the GHG mitigation aspects of the following strategies: landfill CH_4 recovery and utilization; optimizing methanotrophic CH_4 oxidation in landfill cover soils; alternative strategies to landfilling for GHG avoidance (composting; incineration and other thermal processes; mechanical and biological treatment (MBT)); waste reduction through recycling, and expanded wastewater management to minimize GHG generation and emissions. In addition, using available but very limited data, this chapter will discuss emissions of non-methane volatile organic compounds (NMVOCs) from waste and end-of-life issues associated with fluorinated gases.

The mitigation of GHG emissions from waste must be addressed in the context of integrated waste management. 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. LCA is an essential tool for consideration of both the direct and indirect impacts of waste management technologies and policies (Thorneloe *et al.*, 2002; 2005; WRAP, 2006). Because direct emissions represent only a portion of the life cycle impacts of various waste management strategies (Ackerman, 2000), this chapter includes complementary strategies for GHG avoidance, indirect GHG mitigation and use of waste as a source of renewable energy to provide fossil fuel offsets. Using LCA and other decision-support tools, there are many combined mitigation strategies that can be cost-effectively implemented by the public or private sector. 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 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.

Post-consumer waste is a significant renewable energy resource whose energy value can be exploited 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 to many biomass resources because it is regularly collected at public expense

(See also Section 11.3.1.4). 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 (Khan and 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). Using a conservative value of 900 Mt/yr for total waste generation in 2002 (discussed in Box 10.1 below), the energy potential of waste is approximately 5–13 EJ/yr. Assuming an average heating value of 9 GJ/t for mixed waste (Dornburg and 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 waste projections in Monni *et al.* (2006). Currently, more than 130 million tonnes per year of waste are combusted worldwide (Themelis, 2003), which is equivalent to >1 EJ/yr (assuming 9 GJ/t). The biogas fuels from waste – landfill gas and digester gas – typically have a heating value of 16–22 MJ/Nm^3, depending directly on 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/yr (using data from Willumsen, 2003).

An overview of carbon flows through waste management systems addresses the issue of carbon storage versus carbon turnover for major waste-management strategies including landfilling, incineration and composting (Figure 10.1). Because landfills function as relatively inefficient anaerobic digesters, significant long-term carbon storage occurs in landfills, 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; there are also minor emissions of CO_2 from incinerated fossil carbon (plastics). The CO_2 emissions from biomass sources – including the CO_2 in landfill gas, the CO_2 from composting, and CO_2 from incineration of waste biomass – are not taken into account in GHG inventories as these are covered by changes in biomass stocks in the land-use, land-use change and forestry sectors.

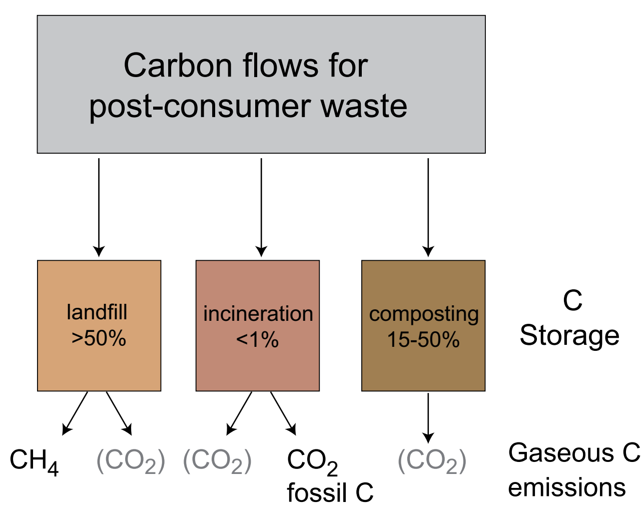
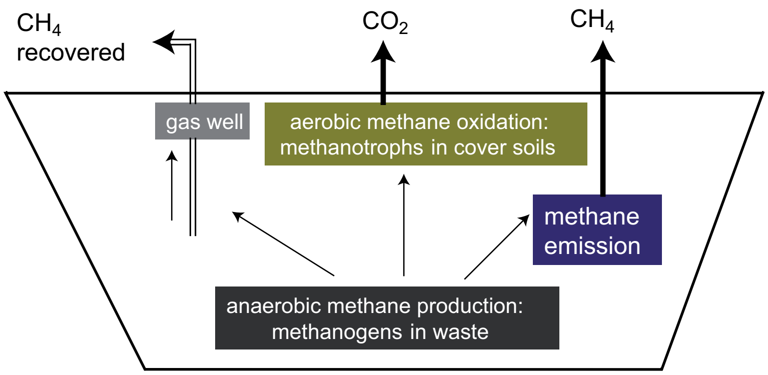


Figure 10.1: *Carbon flows through major waste management systems including C storage and gaseous C emissions. The CO_2 from biomass is not included in GHG inventories for waste.*
References for C storage are: Huber-Humer, 2004; Zinati et al., 2001; Barlaz, 1998; Bramryd, 1997; Bogner, 1992.

A process-oriented perspective on the major GHG emissions from the waste sector is provided in Figure 10.2. In the context of a landfill CH_4 mass balance (Figure 10.2a), emissions are one of several possible pathways for the CH_4 produced by anaerobic methanogenic microorganisms in landfills; other pathways include recovery, oxidation by aerobic methanotrophic microorganisms in cover soils, and two longer-term pathways: lateral migration and internal storage (Bogner and Spokas, 1993; Spokas *et al.*, 2006). With regard to emissions from wastewater transport and treatment (Figure 10.2b), the CH_4 is microbially produced under strict anaerobic conditions as in landfills, while the N_2O is an intermediate product of microbial nitrogen cycling promoted by conditions of reduced aeration, high moisture and abundant nitrogen. Both GHGs can be produced and emitted at many stages between wastewater sources and final disposal.

It is important to stress that both the CH_4 and N_2O from the waste sector are microbially produced and consumed with rates controlled by temperature, moisture, pH, available substrates, microbial competition and many other factors. As a result, CH_4 and N_2O generation, microbial consumption, and net emission rates routinely exhibit temporal and spatial variability over many orders of magnitude, exacerbating the problem of developing credible national estimates. The N_2O from landfills 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 and Svensson, 1997a) or aerobic/semi-aerobic landfilling practices are implemented (Tsujimoto *et al.*, 1994). Substantial emissions of CH_4 and N_2O can occur during wastewater transport in closed sewers and in conjunction with anaerobic or aerobic treatment. In many developing countries, in addition to GHG emissions, open sewers and uncontrolled solid waste disposal sites result in serious public health problems resulting from pathogenic microorganisms, toxic odours and disease vectors.

Major issues surrounding the costs and potentials for mitigating GHG emissions from waste include definition of system boundaries and selection of models with correct baseline assumptions and regionalized costs, as discussed in the TAR (IPCC, 2001a). Quantifying mitigation costs and potentials (Section 10.4.7) for the waste sector remains a challenge due to national and regional data uncertainties as well as the variety of mature technologies whose diffusion is limited by local costs, policies, regulations, available land area, public perceptions and other social development factors. Discussion of technologies



Simplified Landfill Methane Mass Balance

Methane (CH_4) produced (mass/time) = Σ(CH_4 recovered + CH_4 emitted + CH_4 oxidized)

Figure 10.2: *Pathways for GHG emissions from landfills and wastewater systems:*

Figure 10.2a: *Simplified landfill CH4 mass balance: pathways for CH_4 generated in landfilled waste, including CH_4 emitted, recovered and oxidized.*

Note: Not shown are two longer-term CH_4 pathways: lateral CH4 mitigation and internal changes in CH_4 storage (Bogner and Spokas, 1993; Spokas et al., 2006) Methane can be stored in shallow sediments for several thousand years (Coleman, 1979).

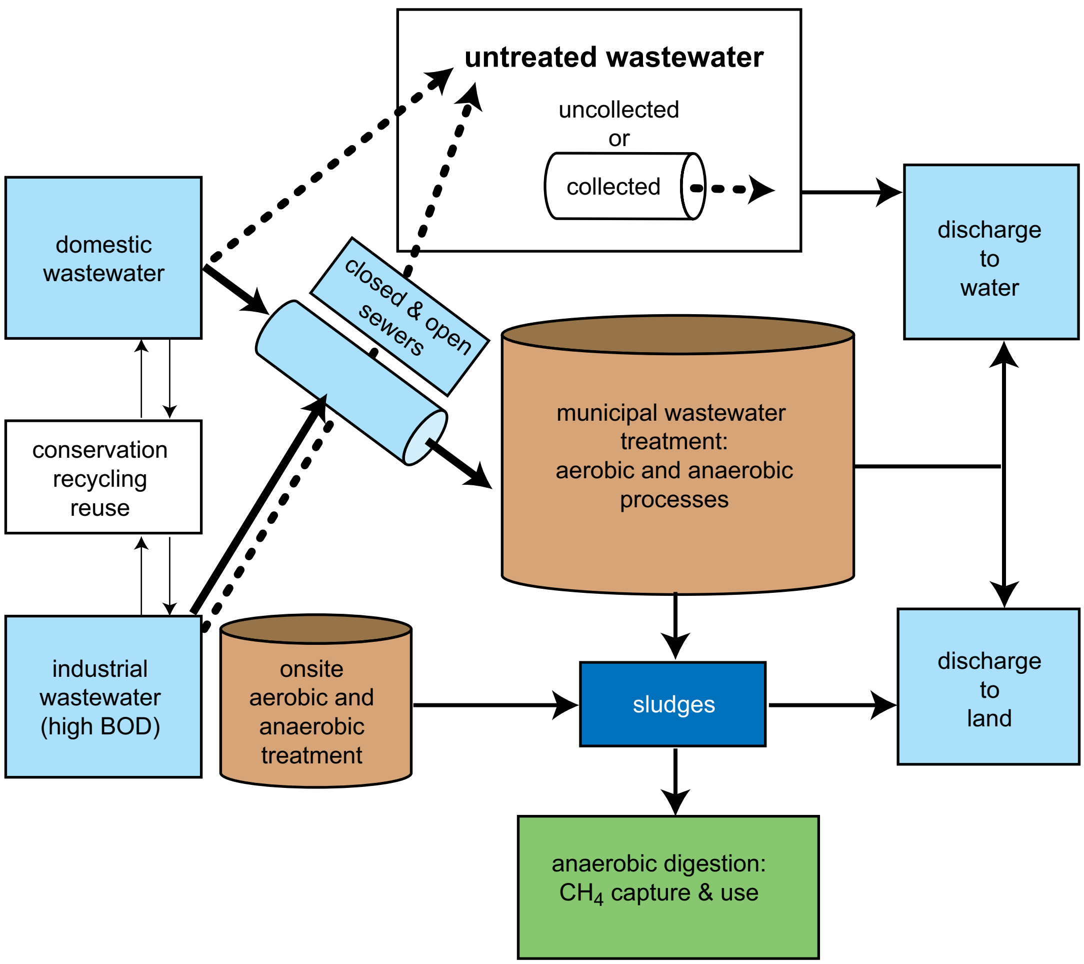


Figure 10.2b: *Overview of wastewater systems.*

Note: The major GHG emissions from wastewater – CH_4 and N_2O – can be emitted during all stages from sources to disposal, but especially when collection and treatment are lacking. N_2O results from microbial N cycling under reduced aeration; CH_4 results from anaerobic microbial decomposition of organic C substrates in soils, surface waters or coastal zones.

and mitigation strategies in this chapter (Section 10.4) includes a range of approaches from low-technology/low-cost to high-technology/high-cost measures. Often there is no single best option; rather, there are multiple measures available to decision-makers at the municipal level where several technologies may be collectively implemented to reduce GHG emissions and achieve public health, environmental protection and sustainable development objectives.

10.2 Status of the waste management sector

10.2.1 Waste generation

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, 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 or surveys, including IPCC methodologies (IPCC, 2006); 2) estimates based on population (e.g., SRES waste scenarios), 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, using population as the major driver, projected continuous 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 emissions significantly higher than those derived from IPCC inventory procedures (Nakicenovic *et al.*, 2000) (See also Section 10.3). 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, while less affluent societies generate less waste and practise 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 and 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 and Matthews, 2003). The box below illustrates 1971–2002 trends for regional solid-waste generation using the surrogate of energy consumption per capita. Using UNFCCC-reported values for percentage biodegradable organic carbon in waste for each country, this box also shows trends for landfill carbon storage based upon the reported data.

Solid waste generation rates range from <0.1 t/cap/yr in low-income countries to >0.8 t/cap/yr in high-income industrialized countries (Table 10.1). 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. It is important to recognize that the percentages of waste recycled, composted, incinerated or landfilled differ greatly amongst municipalities due to multiple factors, including local economics, national policies, regulatory restrictions, public perceptions and infrastructure requirements

Box 10.1: 1971–2002 Regional trends for solid waste generation and landfill carbon storage using a proxy variable.

Solid-waste generation rates are a function of both population and prosperity, but data are lacking or questionable for many countries. This results in high uncertainties for GHG emissions estimates, especially from developing countries. One strategy is to use a proxy variable for which national statistics are available on an annual basis for all countries. For example, using national solid-waste data from 1975–1995 that were reliably referenced to a given base year, Bogner and Matthews (2003) developed simple linear regression models for waste generation per capita for developed and developing countries. These empirical models were based on energy consumption per capita as an indicator of affluence and a proxy for waste generation per capita; the surrogate relationship was applied to annual national data using either total population (developed countries) or urban population (developing countries). The methodology was validated using post-1995 data which had not been used to develop the original model relationships. The results by region for 1971–2002 (Figure 10.3a) indicate that approximately 900 Mt of waste were generated in 2002. Unlike projections based on population alone, this figure also shows regional waste-generation trends that decrease and increase in tandem with major economic trends. For comparison, recent waste-generation estimates by Monni et al. (2006) using 2006 inventory guidelines, indicated about 1250 Mt of waste generated in 2000. Figure 10.3b showing annual carbon storage in landfills was developed using the same base data as Figure 10.3a with the percentage of landfilled waste for each country (reported to UNFCCC) and a conservative assumption of 50% carbon storage (Bogner, 1992; Barlaz, 1998). This storage is long-term: under the anaerobic conditions in landfills, lignin does not degrade significantly (Chen et al., 2004), while some cellulosic fractions are also non-degraded. The annual totals for the mid-1980s and later (>30 MtC/yr) exceed estimates in the literature for the annual quantity of organic carbon partitioned to long-term geologic storage in marine environments as a precursor to future fossil fuels (Bogner, 1992). It should be noted that the anaerobic burial of waste in landfills (with resulting carbon storage) has been widely implemented in developed countries only since the 1960s and 1970s.

Box 10.1 continued

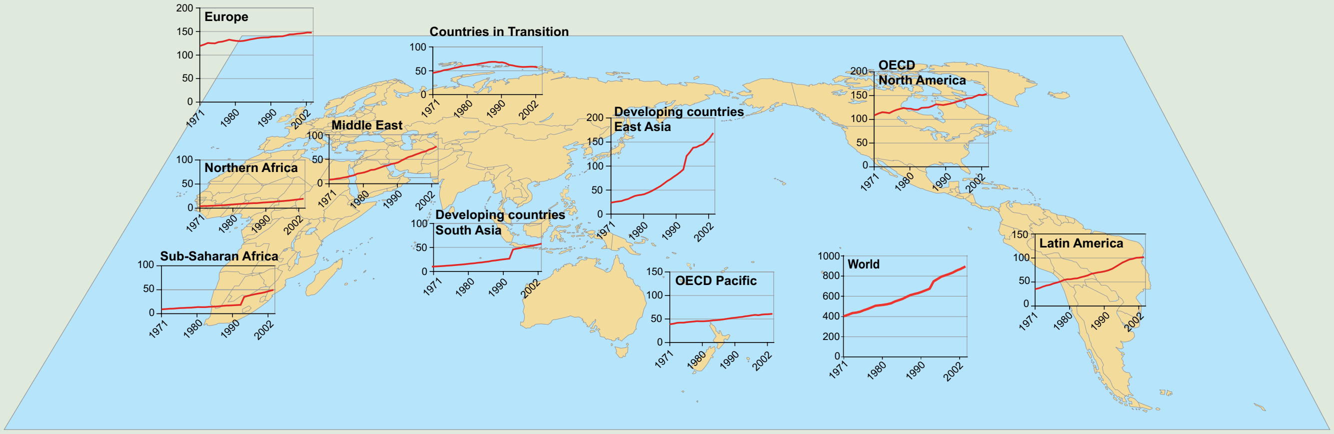


Figure 10.3a: *Annual rates of post-consumer waste generation 1971–2002 (Tg) using energy consumption surrogate.*

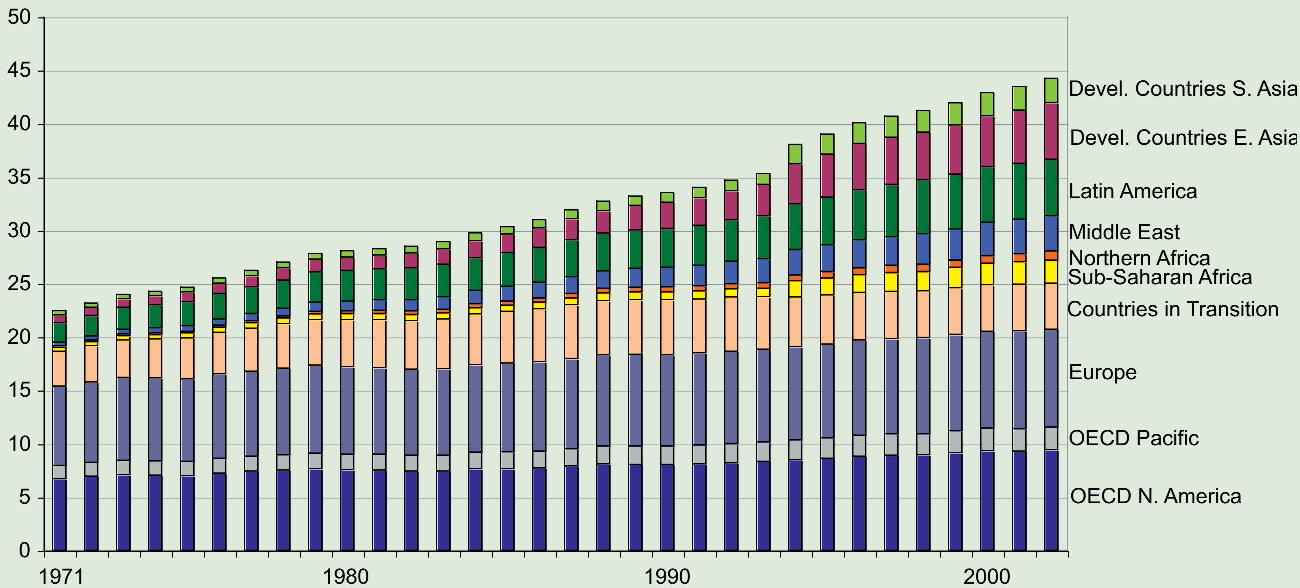


Figure 10.3b: *Minimum annual rates of carbon storage in landfills from 1971–2002 (Tg C).*

Table 10.1: *Municipal solid waste-generation rates and relative income levels*

Country	Low income	Middle income	High income
Annual income (US$/cap/yr)	825-3255	3256-10065	>10066
Municipal solid waste generation rate (t/cap/yr)	0.1-0.6	0.2-0.5	0.3 to >0.8

Note: Income levels as defined by World Bank (www.worldbank.org/data/wdi2005).

Sources: Bernache-Perez et al., 2001; CalRecovery, 2004, 2005; Diaz and Eggerth, 2002; Griffiths and Williams, 2005; Idris et al., 2003; Kaseva et al., 2002; Ojeda-Benitez and Beraud-Lozano, 2003; Huang et al., 2006; US EPA, 2003.

10.2.2 Wastewater generation

Most countries do not compile annual statistics on the total volume of municipal wastewater generated, transported and treated. 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, 2005; Jouravlev, 2004; PNUD, 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). Many countries in Eastern Europe and Central Asia lack reliable benchmarks for the early 1990s. Regional trends (Figure 10.4) indicate improved sanitation levels of <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. In both Southern and Eastern Asia, rapid urbanization is posing a challenge for the development of wastewater infrastructure. The highly urbanized region of Latin America and the Caribbean has also made slow progress in providing wastewater treatment. In the Middle East and North Africa, the countries of Egypt, Tunesia and Morocco have made significant progress in expanding wastewater-treatment infrastructure (World Bank and IMF, 2006). Nevertheless, globally, it has been estimated that 2.6 billion people lack improved sanitation (WHO-UNICEF, 2005).

Estimates for CH_4 and N_2O emissions from wastewater treatment require data on degradable organic matter (BOD; COD[1]) and nitrogen. Nitrogen content can be estimated using Food and Agriculture Organization (FAO) 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 addition, one must take into account the established infrastructure for wastewater treatment in developed countries and the lack of both infrastructure and financial resources in developing countries where open sewers or informally ponded wastewaters often result in uncontrolled discharges to surface water, soils, and coastal zones, as well as the generation of N_2O and CH_4. 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).

Most wastewaters within the industrial and agricultural sectors are discussed in Chapters 7 and 8, respectively. However, highly organic industrial wastewaters are addressed in this chapter, because they are frequently conveyed to municipal treatment facilities. Table 10.2 summarizes estimates for total and regional 1990 and 2001 generation in terms of kilograms of BOD per day or kilograms of BOD per worker per day, based on measurements of plant-level water quality (World Bank, 2005a). The table indicates that total global generation decreased >10% between 1990 and 2001; however, increases

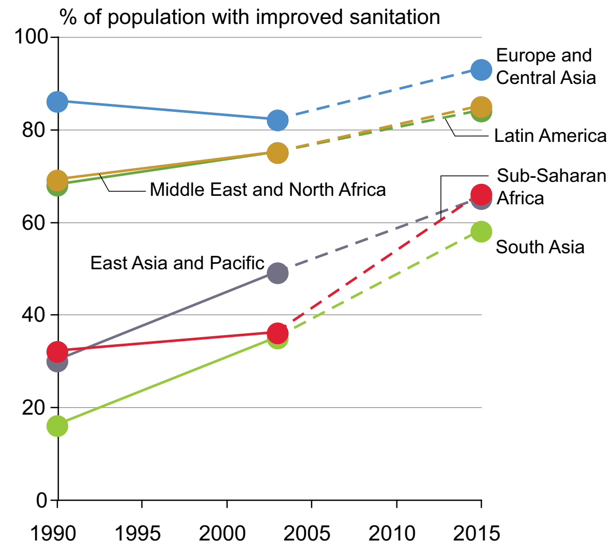


Figure 10.4: *Regional data for 1990 and 2003 with 2015 Millenium Development Goal (MDG) targets for the share of population with access to improved sanitation (sewerage + wastewater treatment, septic system, or latrine).*

Source: World Bank and IMF (2006)

of 15% or more were observed for the Middle East and the developing countries of South Asia.

10.2.3 Development trends for waste and wastewater

Waste and wastewater management are highly regulated within the municipal infrastructure under a wide range of existing regulatory goals 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. Thus, activities related to waste and wastewater management are, and will continue to be, controlled by national regulations, regional restrictions, and local planning guidelines that address waste and wastewater transport, recycling, treatment, disposal, utilization, and energy use. 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.

Depending on regulations, policies, economic priorities and practical local limits, developed countries will be characterized by increasingly higher rates of waste recycling and pre-treatment to conserve resources and avoid GHG generation. Recent studies have documented recycling levels of >50%

1 BOD (Biological or Biochemical Oxygen Demand) measures the quantity of oxygen consumed by aerobically biodegradable organic C in wastewater. COD (Chemical Oxygen Demand) measures the quantity of oxygen consumed by chemical oxidation of C in wastewater (including both aerobic/anaerobic biodegradable and non-biodegradable C).

Table 10.2: *Regional and global 1990 and 2001 generation of high BOD industrial wastewaters often treated by municipal wastewater systems.*

Regions	Kg BOD/day [Total, Rounded] (1000s)		Kg BOD/worker/ day		Primary metals (%)	Paper and pulp (%)	Chemicals (%)	Food and beverages (%)	Textiles (%)
Year	1990	2001	1990	2001	2001	2001	2001	2001	2001
1. OECD North America	3100	2600	0.20	0.17	9	15	11	44	7
2. OECD Pacific	2200	1700	0.15	0.18	8	20	6	46	7
3. Europe	5200	4800	0.18	0.17	9	22	9	40	7
4. Countries in transition	3400	2400	0.15	0.21	13	8	6	50	14
5. Sub-Saharan Africa	590	510	0.23	0.25	3	12	6	60	13
6. North Africa	410	390	0.20	0.18	10	4	6	50	25
7. Middle East	260	300	0.19	0.19	9	12	10	52	11
8. Caribbean, Central and South America	1500	1300	0.23	0.24	5	11	8	61	11
9. Developing countries, East Asia	8300	7700	0.14	0.16	11	14	10	36	15
10. Developing countries, South Asia	1700	2000	0.18	0.16	5	7	6	42	35
Total for 1-4 (developed)	**13900**	**11500**							
Total for 5-10 (developing)	**12800**	**12200**							

Note: Percentages are included for major industrial sectors (all other sectors <10% of total BOD).
Source: World Bank, 2005a.

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). In developing countries, a high level of labour-intensive informal recycling often occurs. Via various diversion and small-scale recycling activities, those who make their living from decentralized waste management can significantly reduce the mass of waste that requires more centralized solutions; however, the challenge for the future is to provide safer, healthier working conditions than currently experienced by scavengers on uncontrolled dumpsites. Available studies indicate that recycling activities by this sector can generate significant employment, especially for women, through creative microfinance and other small-scale investments. For example, in Cairo, available studies indicate that 7–8 daily jobs per ton of waste and recycling of >50% of collected waste can be attained (Iskandar, 2001).

Trends for sanitary landfilling and alternative waste-management technologies differ amongst countries. In the EU, the future landfilling of organic waste is being phased out via the landfill directive (Council Directive 1999/31/EC), while engineered gas recovery is required at existing sites (EU, 1999). This directive requires that, by 2016, the mass of biodegradable organic waste annually landfilled must be reduced 65% relative to landfilled waste in 1995. Several countries (Germany, Austria, Denmark, Netherlands, Sweden) have accelerated the EU schedule through more stringent bans on landfilling of organic waste. As a result, increasing quantities of post-consumer waste are now being diverted to incineration, as well as to MBT before landfilling to 1) recover recyclables and 2) reduce the organic carbon content by a partial aerobic composting or anaerobic digestion (Stegmann, 2005). The MBT residuals are often, but not always, landfilled after achieving organic carbon reductions to comply with the EU landfill directive. Depending on the types and quality control of various separation and treatment processes, a variety of useful recycled streams are also produced. Incineration for waste-to-energy has been widely implemented 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). Rates of incineration are expected to increase in parallel with implemention of the landfill directive, 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, both to comply with air-quality regulations and to provide energy, assisted by national tax credits and local renewable-energy/green power initiatives (see Section 10.5). The US, Canada, Australia and other countries are currently studying and considering the widespread implementation of 'bioreactor' landfills to compress the time period during which high rates of CH_4 generation occur (Reinhart and Townsend, 1998; Reinhart *et al.*, 2002; Berge *et al.*, 2005); bioreactors will also require the early implementation of engineered gas extraction. Incineration has not been widely implemented in these countries due to historically low landfill tipping fees in many regions, negative public perceptions and high capital costs. 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. Historically, there have also been 'semi-aerobic' Japanese landfills with potential for N_2O generation (Tsujimoto *et al.*, 1994). Similar aerobic (with air) landfill practices have also been studied or implemented in Europe and the US for reduced CH_4 generation rates as an alternative to, or in combination with, anaerobic (without air) practices (Ritzkowski and Stegmann, 2005).

In many developing countries, current trends suggest that increases in controlled landfilling resulting in anaerobic decomposition of organic waste will be implemented in parallel with increased urbanization. For rapidly growing 'mega cities', engineered landfills provide a waste disposal solution that is more environmentally acceptable than open dumpsites and uncontrolled burning of waste. There are also persuasive public health reasons for implementing controlled landfilling – urban residents produce more solid waste per capita than rural inhabitants, and large amounts of uncontrolled refuse accumulating in areas of high population density are linked to vermin and disease (Christensen, 1989). 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. This is largely the same transition that occurred in many developed countries in the 1950–1970 time frame. Paradoxically, this results in higher rates of CH_4 generation and emissions than previous open-dumping and burning practices. In addition, many developed and developing countries have historically implemented large-scale aerobic composting of waste. This has often been applied to mixed waste, which, in practice, is similar to implementing an initial 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 (Diaz *et al.*, 2002; Perla, 1997). 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 the least-cost environmentally acceptable solution – often this is landfilling or composting (Hoornweg, 1999; Hoornweg *et al.*, 1999; Johannessen and 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 proven useful in this regard, the post-2012 situation is unclear.

With regard to wastewater trends, a current priority in developing countries is to increase the historically low rates of wastewater collection and treatment. One of the Millennium Development Goals (MDGs) is to reduce by 50% the number of people without access to safe sanitation by 2015. One strategy may be to encourage more on-site sanitation rather than expensive transport of sewerage to centralized treatment plants: this strategy has been successful in Dakar, Senegal, at the cost of about 400 US$ per household. It has been estimated that, for sanitation, the annual investment must increase from 4 billion US$ to 18 billion US$ to achieve the MDG target, mostly in East Asia, South Asia and Sub-Saharan Africa (World Bank, 2005a).

10.3 Emission trends

10.3.1 Global overview

Quantifying global trends requires annual national data on waste production and management practices. Estimates for many countries are uncertain because data are lacking, inconsistent or incomplete; therefore, the standardization of terminology for national waste statistics would greatly improve data quality for this sector. Most developing countries use default data on waste generation per capita with inter-annual changes assumed to be proportional to total or urban population. Developed countries use more detailed methodologies, activity data and emission factors, as well as national statistics and surveys, and are sharing their methods through bilateral and multilateral initiatives.

For landfill CH_4, the largest GHG emission from the waste sector, emissions continue several decades after waste disposal; thus, the estimation of emission trends requires models that include temporal trends. Methane is also emitted during wastewater transport, sewage treatment processes and leakages from anaerobic digestion of waste or wastewater sludges. 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. Open burning of waste 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 10.3: *Trends for GHG emissions from waste using (a) 1996 and (b) 2006 IPCC inventory guidelines, extrapolations, and projections ($MtCO_2$-eq, rounded)*

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] and [b])	550	585	590	635	700	795	910		
Wastewater CH_4[a]	450	490	520	590	600	630	670		
Wastewater N_2O[a]	80	90	90	100	100	100	100		
Incineration CO_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.

Overall, the waste sector contributes <5% of global GHG emissions. Table 10.3 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 simple mass balance calculations using the 1996 IPCC Tier 1 methodology for landfill CH_4. Monni *et al.* (2006) calculated a time series for landfill CH_4 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. The estimates by Monni *et al.* (2006) are lower than US EPA (2006) for the period 1990–2005 because the former reflect slower growth 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 $MtCO_2$eq (UNFCCC, 2005). In general, data from Non-Annex I countries are limited and usually available only for 1994 (or 1990). In the TAR, annual global CH_4 and N_2O emissions from all sources were approximately 600 Tg CH_4/yr and 17.7 Tg N/yr as N_2O (IPCC, 2001b). The direct comparison of reported emissions in Table 10.3 with the SRES A1 and B2 scenarios (Nakicenovic *et al.*, 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 $MtCO_2$-eq/yr for 2050.

Table 10.3 indicates that total emissions have historically increased and will continue to increase (Monni *et al.*, 2006; US EPA, 2006; *see also* Scheehle and Kruger, 2006). However, between 1990 and 2003, the percentage of total global GHG emissions from the waste sector declined 14–19% for Annex I and 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 as a result of local waste management decisions including recycling, local economic conditions and policy initiatives. On the other hand, rapid increases in population and urbanization in developing countries are resulting in increases in GHG emissions from waste, especially CH_4 from landfills and both CH_4 and N_2O from wastewater. CH_4 emissions from wastewater alone are expected to increase almost 50% between 1990 and 2020, especially in the rapidly developing countries of Eastern and Southern Asia (US EPA, 2006; *Table 10.3*). Estimates of global N_2O emissions from wastewater are incomplete and based only on human sewage treatment, but these indicate an increase of 25% between 1990 and 2020 (*Table 10.3*). It is important to emphasize, however, that these are business-as-usual (BAU) scenarios, and actual emissions could be much lower if additional measures are in place. Future reductions in emissions from the waste sector will partially depend on the post-2012 availability of Kyoto mechanisms such the CDM and JI.

Uncertainties for the estimates in Table 10.3 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 twofold (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.

10.3.2 Landfill CH_4: regional trends

Landfill CH_4 has historically been the largest source of GHG emissions from the waste sector. The growth in landfill emissions has diminished during the last 20 years due to increased rates of landfill CH_4 recovery in many countries and decreased rates of landfilling in the EU. The recovery and utilization of landfill CH_4 as a source of renewable energy was first commercialized in 1975 and is now being implemented at >1150 plants worldwide with emission reductions of >105 $MtCO_2$-eq/yr (Willumsen, 2003; Bogner and Matthews, 2003). This number should be considered a minimum because there are also many sites that recover and flare landfill gas without energy recovery. Figure 10.5 compares regional emissions estimates for five-year intervals from 1990–2020 (US EPA, 2006) to annual historical estimates from 1971–2002 (Bogner and Matthews, 2003). The trends converge for Europe and the OECD Pacific, but there are differences for North America and Asia related to differences in methodologies and assumptions.

A comparison of the present rate of landfill CH_4 recovery to estimated global emissions (Table 10.3) indicates that the minimum recovery and utilization rates discussed above (>105 $MtCO_2$-eq yr) currently exceed the average projected increase from 2005 to 2010. Thus, it is reasonable to state that landfill CH_4 recovery is beginning to stabilize emissions from this source. A linear regression using historical data from the early 1980s to 2003 indicates a conservative growth rate for landfill CH_4 utilization of approximately 5% per year (Bogner and Matthews, 2003). For the EU-15, trends indicate that landfill CH_4 emissions are declining substantially. Between 1990 and 2002, landfill CH_4 emissions decreased by almost 30% (Deuber *et al.*, 2005) due to the early implementation of the landfill directive (1999/31/EC) and similar national legislation intended to both reduce the landfilling of biodegradable waste and increase landfill CH_4 recovery at existing sites. By 2010, GHG emissions from waste in the EU are projected to be more than 50% below 1990 levels due to these initiatives (EEA, 2004).

For developing countries, as discussed in the previous section (10.3.1), rates of landfill CH_4 emissions are expected to increase concurrently with increased landfilling. However, incentives such as the CDM can accelerate rates of landfill CH_4 recovery and use in parallel with improved landfilling practices. In addition, since substantial CH_4 can be emitted both before and after the period of active gas recovery, sites should be encouraged, where feasible, to install horizontal gas collection

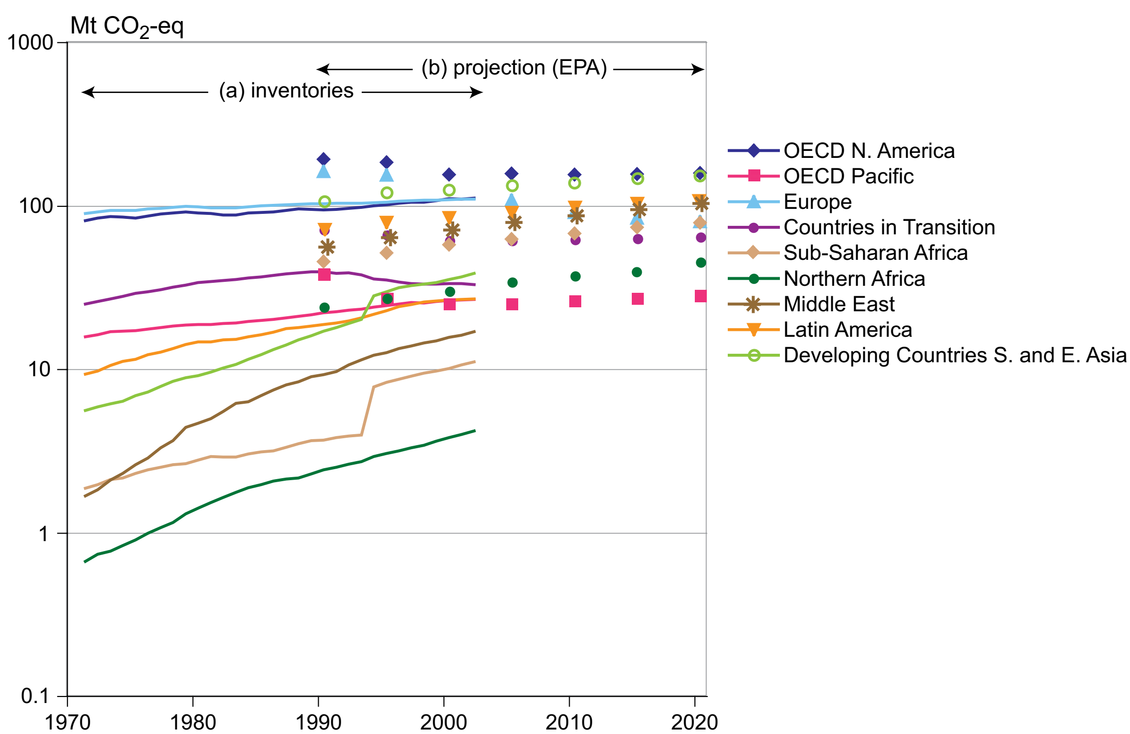


Figure 10.5: *Regional landfill CH_4 emission trends ($MtCO_2$-eq).*

Notes: Includes a) Annual historic emission trends from Bogner and Matthews (2003), extended through 2002; b) Emission estimates for five-year intervals from 1990–2020 using 1996 inventory procedures, extrapolations and projections (US EPA, 2006).

systems concurrent with filling and implement solutions to mitigate residual emissions after closure (such as landfill biocovers to microbially oxidize CH_4—see section 10.4.2).

10.3.3 Wastewater and human sewage CH_4 and N_2O: regional trends

CH_4 and N_2O can be produced and emitted during municipal and industrial wastewater collection and treatment, depending on transport, treatment and operating conditions. The resulting sludges 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 because of extensive infrastructure for wastewater treatment, usually relying on centralized treatment. With anaerobic processes, biogas is produced and CH_4 can be emitted if control measures are lacking; however, the biogas can also be used for process heating or onsite electrical generation.

In developing countries, due to rapid population growth and urbanization without concurrent development of wastewater infrastructure, CH_4 and N_2O emissions from wastewater are generally higher than in developed countries. This can be seen by examining the 1990 estimated CH_4 and N_2O emissions and projected trends to 2020 from wastewater and human sewage (UNFCCC/IPCC, 2004; US EPA, 2006). However, data reliability for many developing countries is uncertain. Decentralized 'natural' treatment processes and septic tanks in developing countries may also result in relatively large emissions of CH_4 and N_2O, particularly in China, India and Indonesia where wastewater volumes are increasing rapidly with economic development (Scheehle and Doorn, 2003).

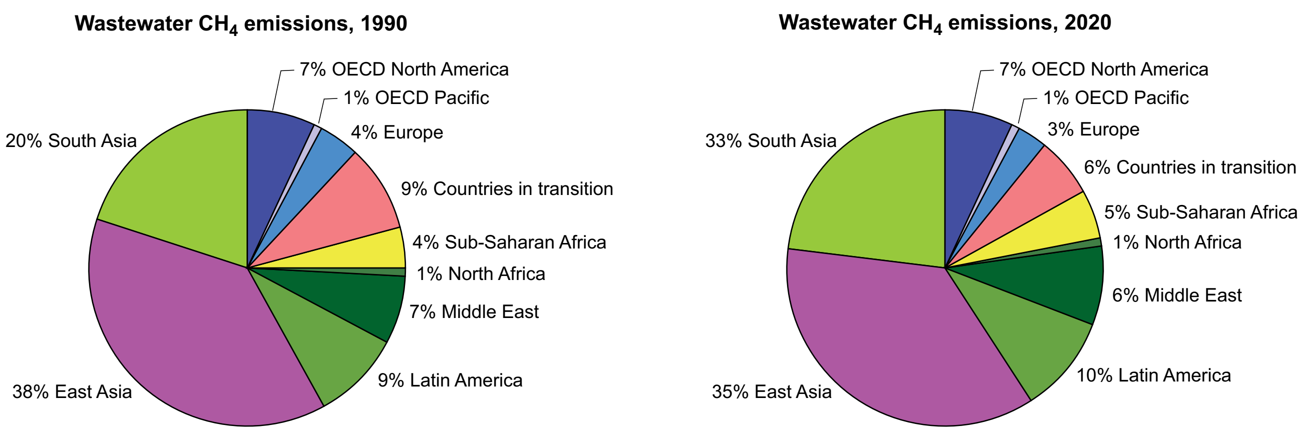


Figure 10.6a: *Regional distribution of CH_4 emissions from wastewater and human sewage in 1990 and 2020. See Table 10.3 for total emissions.*

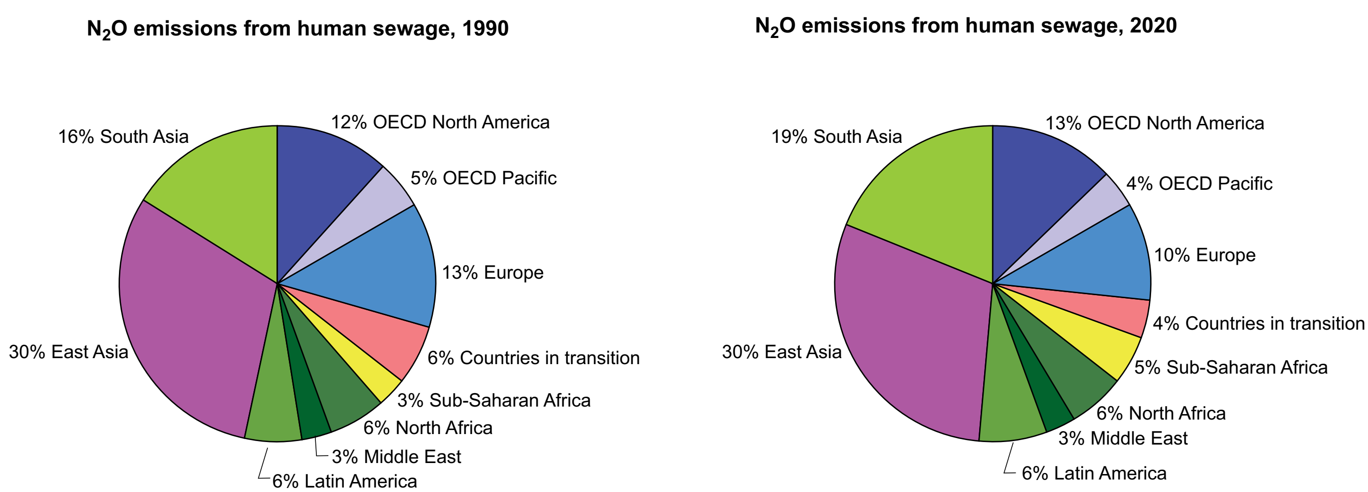


Figure 10.6b: *Regional distribution of N20 emissions from human sewage in 1990 and 2020. See Table 10.3 for total emissions.*

Notes: The US estimates include industrial wastewater and septic tanks, which are not reported by all developed countries.
Source: UNFCCC/IPCC (2004)

The highest regional percentages for CH_4 emissions from wastewater are from Asia (especially China, India). Other countries with high emissions in their respective regions 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 10.3) with much of the increase from the developing countries of East and South Asia, the middle East, the Caribbean, and Central and South America. 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 with emissions of 80–100 $MtCO_2$-eq/yr during the period 1990–2020 (Table 10.3) compared to current total global anthropogenic N_2O emissions of about 3500 $MtCO_2$-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 10.3). 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 (Figure 10.6b). 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 S and E Asia, the Caribbean, and Central and South America (US EPA, 2006). The only regions expected to have lower emissions in 2020 relative to 1990 are Europe and the EIT Countries.

10.3.4 CO_2 from waste incineration

Compared to landfilling, waste incineration and other thermal processes avoid most GHG generation, resulting only in minor emissions of CO_2 from fossil C sources, including plastics and synthetic textiles. Estimated current GHG emissions from waste incineration are small, around 40 $MtCO_2$-eq/yr, 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 $MtCO_2$-eq/yr (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 $MtCO_2$-eq/yr 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, 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/d (range of 170–1400 t/d). 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 $MtCO_2$-eq/yr when fossil-fuel offsets were taken into account.

In developing countries, controlled incineration of waste is infrequently practised because of high capital and operating costs, as well as a history of previous unsustainable projects. The uncontrolled burning of waste for volume reduction in these countries 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, 2006b; National Bureau of Statistics of China, 2006).

10.4 Mitigation of post-consumer emissions from waste

10.4.1 Waste management and GHG-mitigation technologies

A wide range of mature technologies is 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 compared to 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. There are many appropriate low- to high-technology strategies discussed in this section (see Figure 10.7 for a qualitative comparison of technologies). At the ‘high technology’ end, there are also advanced thermal processes for waste such as pyrolysis and gasification, which are beginning to be applied in the EU, Japan and elsewhere. Because of variable feedstocks and high unit costs, these processes have not been routinely applied to mixed municipal waste at large scale (thousands of tonnes per day). Costs and potentials are addressed in Section 10.4.7.

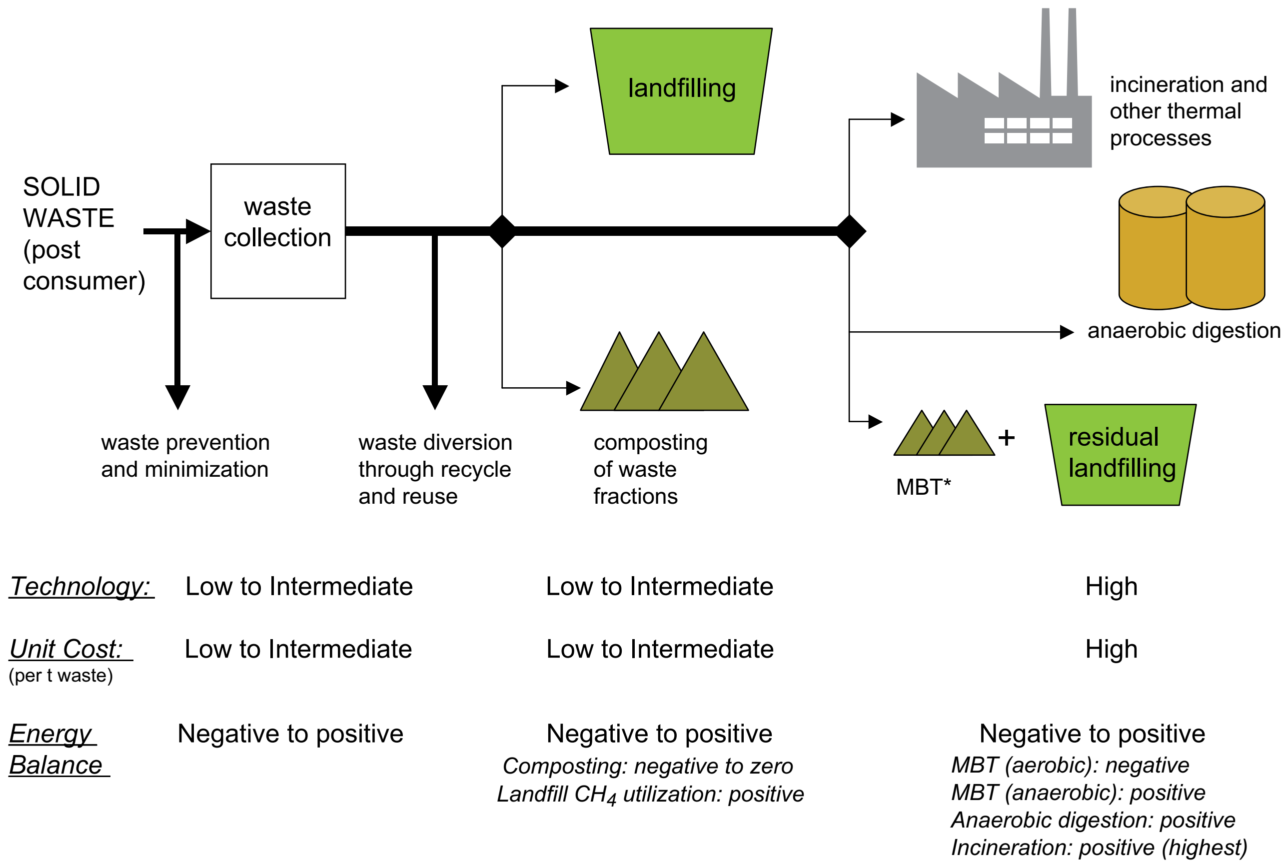


Figure 10.7: *Technology gradient for waste management: major low- to high-technology options applicable to large-scale urban waste management*

Note: MBT=Mechanical Biological Treatment.

10.4.2 CH_4 management at landfills

Global CH_4 emissions from landfills are estimated to be 500–800 $MtCO_2$-eq/yr (US EPA, 2006; Monni *et al.* 2006; Bogner and Matthews 2003). However, direct field measurements of landfill CH_4 emissions at small scale ($<1m^2$) can vary over seven orders of magnitude (0.0001–>1000 g CH_4 $/m^2/d$) depending on waste composition, cover materials, soil moisture, temperature and other variables (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 tCH_4/ha/d (Nozhevnikova *et al.*, 1993; Oonk and Boom, 1995; Borjesson, 1996; Czepiel *et al.*, 1996; Hovde *et al.*, 1995; Mosher *et al.*, 1999; Tregoures *et al.*, 1999; Galle *et al.*, 2001; Morris, 2001; Scharf *et al.*, 2002).

The implementation of an active landfill gas extraction system using vertical wells or horizontal collectors is the single most important mitigation measure to reduce emissions. Intensive field studies of the CH_4 mass balance at cells with a variety of design and management practices 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 and Boom, 1995), which argues for early implementation of gas recovery. Some measures that can be implemented to improve overall gas collection are 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, and frequent inspection and maintenance of cover materials. Currently, landfill CH_4 is being used to fuel industrial boilers; to generate electricity using internal combustion engines, gas turbines or steam turbines; 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. Significant barriers to increased diffusion of landfill gas utilization, especially where it has not been previously implemented, can be local reluctance from electrical utilities to include small power producers and from gas utilities/pipeline companies to transport small percentages of upgraded landfill gas in natural gas pipelines.

A secondary control on landfill CH_4 emissions is CH_4 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. 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; 1999; 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. Laboratory studies have shown that oxidation rates in landfill cover soils may be as high as 150–250 g $CH_4/m^2/d$ (Kightley *et al.*, 1995; de Visscher *et al.*, 1999). Recent field studies have demonstrated that oxidation rates can be greater than 200 g/m^2/d in thick, compost-amended 'biocovers' engineered to optimize oxidation (Bogner *et al.*, 2005; Huber-Humer, 2004). The prototype biocover design includes an underlying coarse-grained gas distribution layer to provide more uniform fluxes to the biocover above (Huber-Humer, 2004). Furthermore, engineered biocovers have been shown to effectively oxidize CH_4 over multiple annual cycles in northern temperate climates (Humer-Humer, 2004). In addition to biocovers, it is also possible to design passive or active methanotrophic biofilters to reduce landfill CH_4 emissions (Gebert and Gröngröft, 2006; Streese and Stegmann, 2005). In field settings, stable C isotopic techniques have proven extremely useful to quantify the fraction of CH_4 that is oxidized in landfill cover soils (Chanton and 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.*, 2003a).

Other measures to reduce landfill CH_4 emissions include installation of geomembrane composite covers (required in the US as final cover); design and installation of secondary perimeter gas extraction systems for additional gas recovery; and implementation of bioreactor landfill designs so that the period of active gas production is compressed while early gas extraction is implemented.

Landfills are a significant source of CH_4 emissions, but they are also a long-term sink for carbon (Bogner, 1992; Barlaz, 1998. See Figure 10.1 and Box 10.1). Since 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 10.1). Carbon storage makes landfilling a more competitive alternative from a climate change perspective, especially where landfill gas recovery is combined with energy use (Flugsrud *et al.* 2001; Micales and Skog, 1997; Pingoud *et al.* 1996; Pipatti and Savolainen, 1996; Pipatti and Wihersaari, 1998). 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 and Moriguchi, 2004 for a review addressing harvested wood products).

10.4.3 Incineration and other thermal processes for waste-to-energy

These processes 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 and Section 7.3.3). Incineration reduces the mass of waste and can offset fossil-fuel use; in addition, GHG emissions are avoided, except for the small contribution from fossil carbon (Consonni *et al.*, 2005). 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 million tonnes 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 waste (87–140 US$/t) (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/yr, 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 (Themelis, 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.

10.4.4 Biological treatment including composting, anaerobic digestion, and MBT (Mechanical Biological Treatment)

Many developed and developing countries practise 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: anaerobic digestion is particularly appropriate for wet wastes, while composting is often appropriate for drier feedstocks. Composting decomposes waste aerobically into CO_2, water and a humic fraction; some carbon storage also occurs in the residual compost (see references on Figure 10.1). Composting can be sustainable at reasonable cost in developing countries; however, choosing more labour-intensive processes over highly mechanized technology at 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 both 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, onsite 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 in comparison to uncontrolled CH_4 emissions from landfills without gas recovery (e.g. Petersen *et al.* 1998; Hellebrand 1998; Vesterinen 1996; 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 million tonnes with larger plants having a capacity of 600–1300 tonnes/day (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 for 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. Under landfill conditions, residual materials retain some potential for CH_4 generation (Bockreis and Steinberg, 2005). Reductions of as much as 40–60% of the original organic carbon are possible with MBT (Kaartinen, 2004). Compared with landfilling, MBT can theoretically reduce CH_4 generation by as much as 90% (Kuehle-Weidemeier and Doedens, 2003). In practice, reductions are smaller and dependent on the specific MBT processes employed (see Binner, 2002).

10.4.5 Waste reduction, re-use and recycling

Quantifying the GHG-reduction benefits of waste minimization, recycling and re-use requires the application of LCA tools (Smith *et al.*, 2001). Recycling reduces GHG emissions through lower energy demand for production (avoided fossil fuel) and by substitution of recycled feedstocks for virgin materials. Efficient use of materials also reduces waste. Material efficiency can be defined as a reduction in primary materials for a particular purpose, such as packaging or construction, with no negative impact on existing human activities. 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). Both material recycling and quality cascading occur in many countries at large scale for metals recovery (steel, aluminium) and recycling of paper, plastics and wood. All these measures lead to indirect energy savings, 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).

The magnitude of avoided GHG-emissions benefits from recycling 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.

10.4.6 Wastewater and sludge treatment

There are many available technologies for wastewater management, collection, treatment, re-use and disposal, ranging from natural purification processes to energy-intensive advanced technologies. Although decision-making tools are available that include environmental trade-offs and costs (Ho, 2000), systematic global studies of GHG-reduction potentials and costs 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 mean that smaller treatment plants with lower capital costs can be more extensively deployed. Wastewater collection and transport includes conventional (deep) sewerage and simplified (shallow) sewerage. Deep sewerage in developed countries has high capital and operational costs. Simplified (shallow) sewerage in both developing and developed countries uses smaller-diameter piping and shallower excavations, resulting in lower capital costs (30–50%) than deep systems.

Wastewater treatment removes pollutants using a variety of technologies. Small wastewater treatment systems include pit latrines, composting toilets and septic tanks. Septic tanks are inexpensive and widely used in both developed and developing countries. Improved on-site treatment systems used in developing countries include inverted trench systems and aerated treatment units. More advanced treatment systems include activated sludge treatment, trickling filters, anaerobic or facultative lagoons, anaerobic digestion and constructed wetlands. Depending on scale, many of these systems have been used in both developed and developing countries. Activated sludge treatment is considered the conventional method for large-scale treatment of sewage. In addition, separation of black water and grey water can reduce the overall energy requirements for treatment (UNEP/GPA-UNESCO/IHE, 2004). Pretreatment or limitation of industrial wastes is often necessary to limit excessive pollutant loads to municipal systems, especially when wastewaters are contaminated with heavy metals. Sludges (or biosolids) are the product of most wastewater treatment systems. Options for sludge treatment include stabilization, thickening, dewatering, anaerobic digestion, agricultural re-use, drying and incineration. The use of composted sludge as a soil conditioner in agriculture and horticulture recycles carbon, nitrogen and phosphorus (and other elements essential for plant growth). Heavy metals and some toxic chemicals are difficult to remove from sludge; either the limitation of industrial inputs or wastewater pretreatment is needed for agricultural use of sludges. Lower quality uses for sludge 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 either be re-used or discharged, but re-use is the most desirable option for agricultural and horticultural irrigation, fish aquaculture, artificial recharge of aquifers, or industrial applications.

10.4.7 Waste management and mitigation costs and potentials

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 and 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\$/tCO_2-eq for landfilling with gas recovery and on-site electrical generation to >370 US\$/tCO_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 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 waste (9–380 US$/t)) and, for composting, establishing local markets for the compost product. Costs for composting can range from 20–170 €/t waste (18–156 US$/t) and are typically 35 €/t waste (32 US$/t) for open-windrow operations and 50 €/t waste (46 US$/t) 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 waste (59 US$/t), with a range of 30–150 €/t (28–140 US$/t)). Landfill disposal is the most inexpensive waste management option in the EU (averaging 56 €/t waste (52 US$/t), ranging from 10–160 €/t waste (9–147 US$/t), 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 waste (83 US$/t, 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 (Bates and Haworth, 2001; Delhotal *et al.* 2006; Monni *et al.* 2006; Nakicenovic *et al.*, 2000; Pipatti and Wihersaari 1998). When reporting to the UNFCCC, most developed countries take the dynamics of landfill gas generation into account; however, most developing countries and non-reporting countries do 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$/ tCO_2-eq, with the lower value for direct use in industrial boilers and the higher value for on-site electrical generation. From the same study, break-even costs (all in US$/$tCO_2$-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 which 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. Most existing studies for the waste sector do not consider these temporal issues. 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. 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 and Matthews, 2003; Willumsen, 2003). Mitigation scenarios were developed for 2030 and 2050 which 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 10.8), emissions increase threefold during the period from 1990 to 2030 and more than fivefold 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 comes from non-OECD countries whose current emissions are smaller because of lower waste generation and a higher percentage of waste degrading aerobically. The mitigation scenarios show that reductions by individual measures in 2030 range from 5–20% of total emissions and increase proportionally with time. In 2050, the corresponding range is approximately 10–30%. As the measures in the scenarios are largely additive, total mitigation potentials of approximately 30% in 2030 and 50% in 2050 are projected relative to the baseline. Nevertheless, the estimated abatement potential is not capable of mitigating the growth in emissions.

The baseline emission estimates in the Delhotal *et al.* (2006) study are based on similar assumptions to the Monni *et al.* (2006) study: population and GDP growth with increasing amounts of landfilled waste in developing countries. Baselines also include documented or expected changes in disposal rates due to composting and recycling, as well as the effects of landfill-gas recovery. In Delhotal *et al.* (2006), emissions increase by about 30% between 2000 and 2020; therefore, the growth in emissions to 2020 is more moderate than in Monni *et al.* (2006). This more moderate growth can be attributed to the inclusion of current and planned policies and measures to reduce emissions, plus the fact that historical emissions from prior landfilled waste were only partially considered.

Scenario development in both studies was complemented with estimates on maximum mitigation potentials at given marginal cost levels using the baseline scenarios as the starting point. Monni *et al.* (2006) derived annual regional waste-generation estimates for the Global Times model by using static aggregate emission coefficients calibrated to regional FOD models. Some modifications to the assumptions used in the

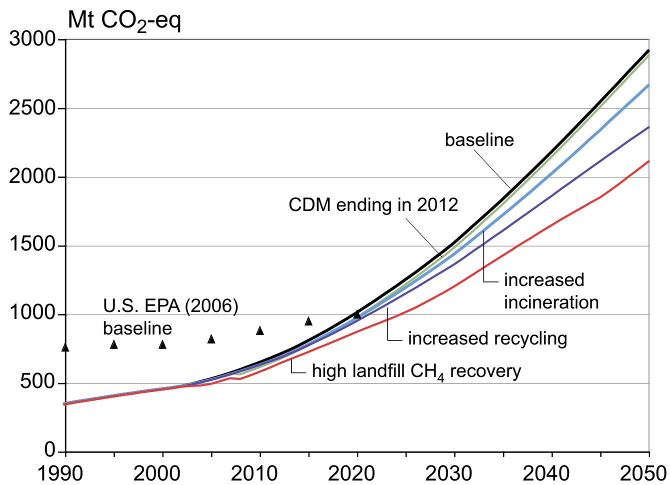


Figure 10.8: *Global CH_4 emissions from landfills in baseline scenario compared to the following mitigation scenarios: increased incineration, CDM ending by 2012 (end of the first Kyoto commitment period), increased recycling, and high landfill CH_4 recovery rates including continuation of CDM after 2012 (Monni et al., 2006). The emission reductions estimated in the mitigation scenarios are largely additional to 2050. This figure also includes the US EPA (2006) baseline scenario for landfill CH_4 emissions from Delhotal et al. (2006).*

scenario development were also made; for example, recycling was excluded due to its economic complexity, biological treatment was included and the technical efficiency of landfill-gas recovery was assumed the same in all regions (75%). Cost data were taken from various sources (de Feber & Gielen, 2000; OECD, 2004; Hoornweg, 1999).

As in the EMF-21 study (US EPA, 2003), both Delhotal *et al.* (2006) and Monni *et al.* (2006) assumed the same capital costs for all regions, but used regionalized labour costs for operations and maintenance.

Delhotal *et al.* (2006) and Monni *et al.* (2006) both conclude that substantial emission reductions can be achieved at low or negative costs (see Table 10.4). At higher costs, more significant reductions would be possible (more than 80% of baseline emissions) with most of the additional mitigation potential coming from thermal processes for waste-to-energy. Since combustion of waste results in minor fossil CO_2 emissions, these were considered in the calculations, but Table 10.4 only includes emissions reductions from landfill CH_4. In general, direct GHG emission reductions from implementation of thermal processes are much less than indirect reductions due to fossil fuel replacement, where that occurs. The emission reduction potentials for 2030 shown in Table 10.4 are assessed using a steady-state approach that can overestimate near-term annual reductions but gives more realistic values when integrated over time.

The economic mitigation potentials for the year 2030 in Table 10.5 take the dynamics of landfill gas generation into account. These estimates are derived from the static, long-term mitigation potentials previously shown in Table 10.4 (Monni *et al.* 2006). The upper limits of the ranges assume that landfill disposal is limited in the coming years so that only 15% of the waste generated globally is landfilled after 2010. This would mean that by 2030 the maximum economic potential would be almost 70% of the global emissions (see Table 10.5). The lower limits of the table have been scaled down to reflect a more realistic timing of implementation in accordance with emissions in the high landfill gas recovery (HR) and increased incineration (II) scenarios (Monni *et al.*, 2006).

It must be emphasized that there are large uncertainties in costs and potentials for mitigation of GHG emissions from waste due to the uncertainty of waste statistics for many countries and emissions methodologies that are relatively unsophisticated. It is also important to point out that the cost estimates are global

Table 10.4: *Economic reduction potential for CH_4 emissions from landfilled waste by level of marginal costs for 2020 and 2030 based on steady state models[a].*

	US$/t$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%
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%

[a] *The steady-state approach tends to overestimate the near-term annual reduction potential but gives more realistic results when integrated over time.*

Table 10.5: *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 ($MtCO_2$-eq) at <100 US$/$tCO_2$-eq	Economic mitigation potential ($MtCO_2$-eq) at various cost categories (US$/$tCO_2$-eq)			
			<0	0-20	20-50	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 high recycling and increased incineration scenarios in the Monni et al. 2006 study.

averages and therefore not necessarily applicable to local conditions.

10.4.8 Fluorinated gases: end-of-life issues, data and trends in the waste sector

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). The HFCs regulated under the Kyoto Protocol are promoted as substitutions for the ODS. High global-warming potential (GWP) fluorinated gases have been used for more than 70 years; the most important are the chlorofluorocarbons (CFCs), hydrochlorofluorocarbons (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). 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 and Jensen, 2001, Kjeldsen and Scheutz, 2003, Scheutz *et al*, 2003b), 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. 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 lowest 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 30-80 years, Gamlen *et al.*, 1986) and most of the fluorinated gases in building insulation have not yet reached the end of their useful life (TFFEoL, 2005). Daniel *et al.* (2007) discuss the uncertainties and some possible temporal trends for depletion of CFC-11 and CFC-12 banks.

Consumer products containing fluorinated gases are managed in different ways. After 2001, landfill disposal of appliances was prohibited in the EU (IPCC, 2005), resulting in appliance-recycling facilities. A similar system was established in Japan in 2001 (IPCC, 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 and Scheutz, 2003). A recent study estimating CFC-11 releases after shredding at three American facilities showed that 60–90% of the CFC 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 the CFC-11 is lost during segregation (Scheutz *et al.*, 2002; Fredenslund *et al.*, 2005). 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.*, 2003b). 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, recent laboratory experiments have indicated rapid CFC-11 degradation 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 of 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.

Table 10.6: *Examples of policies and measures for the waste management sector.*

Policies and measures	Activity affected	GHG affected	Type of instruments
Reducing landfill CH_4 emissions			
Standards for landfill performance to reduce landfill CH_4 emissions by capture and combustion of landfill gas with or without energy recovery	Management of landfill sites	CH_4	Regulation Economic Incentive
Reduction in biodegradable waste that is landfilled.	Disposal of biodegradable waste	CH_4	Regulation
Promoting incineration and other thermal processes for waste-to-energy			
Subsidies for construction of incinerator combined with standards for energy efficiency	Performance standards for incinerators	CO_2 CH_4	Regulation
Tax exemption for electricity generated by waste incineration with energy recovery	Energy recovery from incineration of waste	CO_2 CH_4	Economic incentive
Promoting waste minimization, re-use and recovery			
Extended Producer Responsibility (EPR)	Manufacture of products Recovery of used products Disposal of waste	CO_2 CH_4 Fluorinated gases	Regulation Voluntary
Unit pricing / Variable rate pricing / Pay-as-you-throw (PAYT)	Recovery of used products Disposal of waste	CO_2 CH_4	Economic incentive
Landfill tax	Recovery of used products Disposal of waste	CO_2 CH_4	Regulation
Separate collection and recovery of specific waste fractions	Recovery of used products Disposal of waste	CO_2 CH_4	Subsidy
Promotion of the use of recycled products	Manufacturing of products	CO_2 CH_4	Regulation Voluntary
Wastewater and sludge treatment			
Collection of CH_4 from wastewater treatment system	Management of wastewater treatment system	CH_4	Regulation Voluntary
Post-consumer management of fluorinated gases			
Substitutes for gases used commercially	Production of fluorinated gases	Fluorinated gases	Regulation Economic incentive Voluntary
Collection of fluorinated gases from end-of-life products	Management of end-of-life products	Fluorinated gases	Regulation Voluntary
JI and CDM in waste management sector			
JI and CDM	Landfill gas and biogas recovery	CO_2 CH_4	Kyoto mechanism

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 recent availability of the Clean Development Mechanism (CDM) in developing countries are providing strong economic incentives for improved landfilling practices (to permit gas extraction) and landfill CH_4 recovery. Box 10.2 summarizes the important role of landfill CH_4 recovery within CDM and gives an example of a successful project in Brazil.

10.5.2 Incineration and other thermal processes for waste-to-energy

Thermal processes can efficiently exploit the energy value of post-consumer waste, but the high cost of incineration with emission controls restricts its sustainable application in many developing countries. Subsidies for construction of incinerators have been implemented in several countries, usually combined with standards for energy efficiency (Austrian Federal Government, 2001; Government of Japan, 1997). 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 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 to 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 and Read (2001).

10.5.3 Waste minimization, re-use and recycling

Widely implemented policies 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 and 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). Initially, EPR programmes were reported to be expensive (Hanisch, 2000), but the EPR concept is very broad: a number of successful schemes have been implemented in various countries for diverse waste fractions such as packaging waste, old vehicles and electronic equipment. EPR programmes range in complexity and cost, but waste reductions have been reported in many countries and regions. 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 and Vogt, 2005).

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 ten-year sustainability of these programmes has been demonstrated (Yamakawa and Ueta, 2002).

Separate and efficient collection of recyclable materials is needed with both PAYT and landfill tax systems. For kerbside programmes, the percentage recycled is related to the efficiency of kerbside collection and the duration of the programme (Jenkins *et al.*, 2003). 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, 21 states have requirements for separate collection of garden (green) waste, which is diverted to composting or used as an alternative daily cover on landfills.

10.5.4 Policies and measures on fluorinated gases

The HFCs regulated under the Kyoto Protocol substitute for the ODS. 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).

10.5.5 Clean Development Mechanism/Joint Implementation

Because lack of financing is a major impediment to 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. As described in Section 10.3, open dumping and burning are common waste disposal methods in many developing countries, where GHG emissions occur concurrently with odours, public health and safety problems, and environmental degradation. In addition, developing countries often do not have existing infrastructure for collection and treatment of municipal wastewaters. Thus, the benefits from JI and CDM are twofold: improving waste management practices and reducing GHG emissions. To date, CDM has assisted many landfill gas recovery projects (see Box 10.2) while improving landfill operations, because adequate cover materials are required to minimize air intrusion during gas extraction (to prevent internal landfill fires). 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, certified emission reductions (CERs, with units of tCO2-eq) are determined directly from quantification of the CH_4 captured and combusted. In many countries, the anaerobic digestion of wastewaters and sludges could produce a useful biogas for heating use or onsite electrical generation (Government of Japan, 1997; Government of Republic of Poland, 2001); such projects could also be suitable for JI and CDM. In the future, waste sector projects involving municipal wastewater treatment, carbon storage in landfills or compost, and avoided GHG emissions due to recycling, composting, or incineration could potentially be implemented pending the development of approved methodologies.

10.5.6 Non-climate policies affecting GHG emissions from waste

The EIT and many developing countries have implemented market-oriented structural reforms that affect GHG emissions. As GDP is a key parameter to predict 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.

Box 10.2: Significant role of landfill gas recovery for CDM projects: overview and example

As of late October 2006, 376 CDM projects had achieved registration. These include 33 landfill gas projects, which collectively total 12% of the annual average CERs (12 million of approximately 91 million CERs per year). (http://cdm.unfccc.int/Projects/registered.html). The pie chart shows the distribution of landfill gas CERs by country. Most of these projects are located in Latin America and the Carribean region (72% of landfill gas CERs), dominated by Brazil (nine projects; 48% of CERs). Some projects are flaring gas, while others are using the gas for on-site electrical generation or direct-use projects (including leachate evaporation). 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.

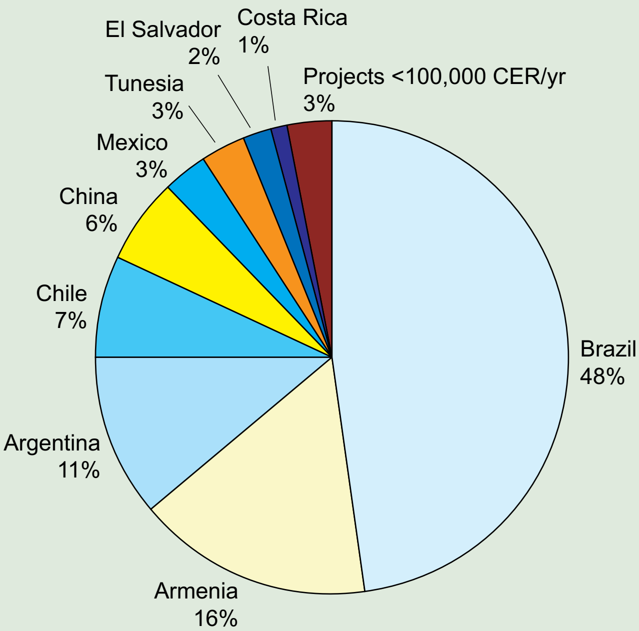


Figure 10.9: *Distribution of landfill gas CDM projects based on average annual CERs for registered projects late October 2006 (unfccc.org). Includes 10.9 Mt CERs for landfill CH_4 of 91 Mt total CERs. Projects <100,000 CERs/yr are located in Israel, Bolivia, Bangladesh and Malaysia*

An example of a successful Brazilian project is the ONYX SASA Landfill Gas Recovery Project at the VES landfill, Trémembé, Sao Paulo State (Figure 10.10). The recovered landfill gas is flared and used to evaporate leachate. As of December, 2005, approximately 93,600 CERs had been delivered (Veolia Environmental Services, 2005).

Figure 10.10: *ONYX SASA Landfill Gas Recovery Project .VES landfill, Trémembé, Sao Paulo State*

In 2002, the Johannesburg Summit adopted the Millennium Development Goals 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.

10.5.7 Co-benefits of GHG mitigation policies

Most policies and measures in the waste sector address broad environmental objectives, such as preventing pollution, mitigating odours, preserving open space and maintaining air, soil and water quality (Burnley, 2001). Thus, reductions in GHG emissions frequently occur as a co-benefit of regulations and policies not undertaken primarily for the purpose of climate-change mitigation (Austrian Federal Government, 2001). For example, the EU Landfill Directive is primarily concerned with preventing pollution of water, soil and air (Burnley, 2001).

10.6 Long-term considerations and sustainable development

10.6.1 Municipal solid waste management

GHG emissions from waste can be effectively mitigated by current technologies. 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, which is increasingly being implemented for landfill gas recovery projects, 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 a recycling rate of 50% was achievable, economically justified and environmentally preferable. This rate has already been achieved in many countries for the more valuable waste fractions such as metals and paper (OECD, 2002b).

Decisions for alternative waste management strategies are often made locally; however, there are also regional drivers based on national regulatory and policy decisions. Selected waste management options also determine GHG mitigation options. For the many countries which 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 biocover landfill cover designs 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., Lohiniva *et al.* 2002; Pipatti and Savolainen 1996; Consonni *et al.* 2005). High cost, however, is a major barrier to the increased implementation of waste-to-energy. Incineration has often proven to be unsustainable in developing countries – thus thermal processes are expected to be primarily (but not exclusively) 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 expected to continue in the future as part of the mix of mature waste management technologies. In general, anaerobic digestion technologies incur lower capital costs 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. Projects where 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.

Regarding the future of up-front recycling and separation technologies, 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. If analysed within an LCA perspective, waste can be considered a resource, and 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 be introduced and sustained to 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.

10.6.2 Wastewater management

Although current GHG emissions from wastewater are lower than emissions from waste, it is recognized that there are substantial emissions which are not quantified by current estimates, especially from septic tanks, latrines and uncontrolled discharges in developing countries. Nevertheless, the quantity of wastewater collected and treated is increasing in many countries in order to maintain and improve potable water quality, as well for other public health and environmental protection benefits. Concurrently, GHG emissions from wastewater will decrease relative to future increases in wastewater collection and treatment.

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 and Reingans, 2006). Historically, sanitation

Table 10.7: *Summary of adaptation, mitigation and sustainable development issues for the waste sector.*

Technologies and practices	Vulnerability to climate change	Adaptation implications & strategies to minimize emissions	Sustainable development dimensions			Comments
			Social	Economic	Environmental	
Recycling, reuse & waste minimization	Indirect low vulnerability or no vulnerability	Minimal implications	Usually positive Negative for waste scavenging without public health or safety controls	Positive Job creation	Positive Negative for waste scavenging from open dumpsites with air and water pollution	Indirect benefits for reducing GHG emissions from waste Reduces use of energy and raw materials. Requires implementation of health and safety provisions for workers
Controlled landfilling with landfill gas recovery and utilization	Indirect low vulnerability or positive effects: Higher temperatures increase rates of microbial methane oxidation rates in cover materials	Minimal implications May be regulatory mandates or economic incentives Replaces fossil fuels for process heat or electrical generation	Positive Odour reduction (non-CH_4 gases)	Positive Job creation Energy recovery potential	Positive Negative for improperly managed sites with air and water pollution	Primary control on landfill CH_4 emissions with >1200 commercial projects Important local source of renewable energy: replaces fossil fuels Landfill gas projects comprise 12% of annual registered CERs under CDM[a] Oxidation of CH_4 and NMVOCs in cover soils is a smaller secondary control on emissions
Controlled landfilling without landfill gas recovery	Indirect low vulnerability or positive effects: Higher temperatures increase rates of microbial methane oxidation rates in cover materials	Minimal implications Gas monitoring and control still required	Positive Odour reduction (non-CH_4 gases)	Positive Job creation	Positive Negative for improperly managed sites with air and water pollution	Use of cover soils and oxidation in cover soils reduce rate of CH_4 and NMVOC emissions
Optimizing microbial methane oxidation in landfill cover soils ('biocovers')	Indirect low vulnerability or positive effects: Increased rates at higher temperatures	Minimal implications or positive effects	Positive Odour reduction (non-CH_4 gases)	Positive Job creation	Positive Negative for improperly designed or managed biocovers with GHG emissions and NMVOC emissions	Important secondary control on landfill CH_4 emissions and emissions of NMVOCs Utilizes other secondary materials (compost, composted sludges) Low-cost low-technology strategy for developing countries
Uncontrolled disposal (open dumping & burning)	Highly vulnerable Detrimental effects: warmer temp. promote pathogen growth and disease vectors	Exacerbates adaptation problems Recommend implementation of more controlled disposal and recycling practices	Negative	Negative	Negative	Consider alternative lower-cost medium technology solutions (e.g., landfill with controlled waste placement, compaction, and daily cover materials)
Thermal processes including incineration, industrial co-combustion, and more advanced processes for waste-to-energy (e.g., fluidized bed technology with advanced flue gas cleaning)	Low vulnerability	Minimal implications Requires source control and emission controls to prevent emissions of heavy metals, acid gases, dioxins and other air toxics	Positive Odour reduction (non-CH_4 gases)	Positive Job creation Energy recovery potential	Positive Negative for improperly designed or managed facilities without air pollution controls	Reduces GHG emissions relative to landfilling Costly, but can provide significant mitigation potential for the waste sector, especially in the short term Replaces fossil fuels
Aerobic biological treatment (composting) Also a component of mechanical biological treatment (MBT)	Indirect low vulnerability or positive effects: Higher temperatures increase rates of biological processes (Q_{10})	Minimal implications or positive effects Produces CO_2 (biomass) and compost Reduces volume, stabilizes organic C, and destroys pathogens	Positive Odour reduction (non-CH_4 gases)	Positive Job creation Use of compost products	Positive Negative for improperly designed or managed facilities with odours, air and water pollution	Reduces GHG emissions Can produce useful secondary materials (compost) provided there is quality control on material inputs and operations Can emit N_2O and CH_4 under reduced aeration or anaerobic conditions
Anaerobic biological treatment (anaerobic digestion) Also a component of mechanical-biological treatment (MBT)	Indirect low vulnerability or positive effects: Higher temperatures increase rates of biological processes	Minimal implications Produces CH_4, CO_2, and biosolids under highly controlled conditions Biosolids require management	Positive Odour reduction (non-CH_4 gases)	Positive Job creation Energy recovery potential Use of residual biosolids	Positive Negative for improperly designed or managed facilities with, odours, air and water pollution	Reduces GHG emissions CH_4 in biogas can replace fossil fuels for process heat or electrical generation Can emit minor quantities of CH_4 during start-ups, shutdowns and malfunctions
Wastewater control and treatment (aerobic or anaerobic)	Highly vulnerable Detrimental effects in absence of wastewater control and treatment: Warmer temperatures promote pathogen growth and poor public health	Large adaptation implications High potential for reducing uncontrolled GHG emissions Residuals (biosolids) from aerobic treatment may be anaerobically digested	Positive Odour reduction (non-CH_4 gases)	Positive Job creation Energy recovery potential from anaerobic processes Use of sludges and other residual biosolids	Positive Negative for improperly designed or managed facilities with odours, air and water pollution and GHG emissions	Wide range of available technologies to collect, treat, recycle and re-use wastewater Wide range of costs CH_4 from anaerobic processes replaces fossil fuels for process heat or electrical generation Need to design and operate to minimize N_2O and CH_4 emissions during transport and treatment

a http://cdm.unfccc.int/Projects/registerd.html, October 2006 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 and 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 more than one 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 >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.

10.6.3 Adaptation, mitigation and sustainable development in the waste sector

In addition to providing mitigation of GHG emissions, improved public health, and environmental benefits, solid waste and wastewater technologies confer significant co-benefits for adaptation, mitigation and sustainable development (Table 10.7; see also Section 12.3.4). In developing countries, improved waste and wastewater management using low- or medium-technology strategies are recommended to provide significant GHG mitigation and public health benefits at lower cost. Some of these strategies include small-scale wastewater management such as septic tanks and recycling of grey water, construction of medium-technology landfills with controlled waste placement and use of daily cover (perhaps including a final biocover to optimize CH_4 oxidation), 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. Developing countries may also lack access to advanced technologies. However, technologies must be sustainable in the long term, and there are many examples of advanced, but unsustainable, technologies for waste management that have been implemented in developing countries. Therefore, the selection of truly sustainable waste and wastewater strategies is very important for both the mitigation of GHG emissions and for improved urban infrastructure.

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