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Contents

EXECUTIVE SUMMARY

The importance of the waste sector for reducing global GHG emissions has been underestimated because most waste management decisions are made locally without concurrent consideration or quantification of GHG mitigation effects. Collectively, a wide range of environmentally-effective waste management technologies are reducing emissions globally through landfill CH_4 recovery and utilization, improved landfill management practices, engineered wastewater management, and waste-to-energy. Moreover, waste minimization, recycling, and re-use represent a growing but largely undefined potential for indirect reduction of GHG emissions through improved energy efficiency and fossil fuel avoidance.

There are a range of mature low- to high-technology strategies that can be implemented to mitigate GHG emissions from waste and enhance sustainable development. In the context of integrated waste management, the choice of a particular technology is a function of many competing variables, including cost, available land area, waste quantity and characteristics, regulatory constraints, collection and transport issues, and policy considerations. Flexible national policies and regulations, especially, can expand waste management options to achieve GHG mitigation goals. For developing countries, environmentally-responsible waste management at an appropriate level of technology promotes sustainable development.

Consistent and coordinated data collection and analysis at the national level could greatly improve the quantification of both direct and indirect GHG mitigation for the waste sector. Currently, the draft 2006 UNFCCC national inventory guidelines provide more advanced default methods, recognize the need to improve field measurement strategies, and allow greater flexibility for quantifying national emissions depending on data quality and quantity. Within this framework, if implemented, the quantification of global GHG emissions from waste would greatly benefit from increased coordination of international data collection and statistical analysis.

Landfill CH_4 recovery for energy use has been fully commercial since 1975, currently exceeds 105 Mt CO_2e/yr, and is beginning to stabilize landfill CH_4 emissions globally via decreased landfill CH_4 emissions from developed countries. However, landfill CH_4 emissions from developing countries are increasing as more controlled (anaerobic) landfilling practices are implemented. The availability of the Clean Development Mechanism under the Kyoto Protocol has the potential to significantly reduce landfill CH_4 emissions from developing countries by accelerating the introduction of existing technologies for CH_4 recovery and utilization.

Increased infrastructure for wastewater management in developing countries could provide multiple benefits for GHG mitigation, improved public health, conservation of water resources, and the reduction of untreated discharges to surface water, groundwater, and coastal zones. There are numerous existing technologies that can be implemented for improved wastewater collection, transport, recycling, treatment, and use of byproducts (sludges). A key aspect of sustainable development is the selection of appropriate and sustainable wastewater methodologies that are consistent with the development status of a particular country.

10.1 Introduction

Waste generation is closely linked to population, affluence, and industrial structure. 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...” (Haury, 1976, p. 80). Archaeological excavations yield thicker cultural layers from periods of prosperity; correspondingly, modern waste generation rates can be correlated to various indicators of affluence normalized by population, including energy consumption/cap and GDP/cap (Bingemer and Crutzen, 1986; Richards, 1989; Rathje *et al.*, 1992; Mertins *et al.*, 1999; Nakicenovic *et al.*, 2000; Bogner and Matthews, 2003). Within the EU, a current goal is to "decouple" waste generation and GHG emissions from economic driving forces such as GDP (OECD, 2003; EEA, 2005). However, in most developed countries, as well as 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.

This chapter focuses on GHG mitigation in the context of integrated waste management. Landfill CH_4 and wastewater CH_4 and N_2O are the major GHG emissions from the waste sector. Thus, in contrast to most other sectors but similar to agriculture, the waste sector is dominated by non-CO_2 GHGs with high GWPs. In this chapter, regional and source-related emissions from the TAR and the SRES will be updated, focusing on mitigation of greenhouse gas emissions from *post-consumer* waste management practices, as well as emissions from municipal and industrial wastewaters conveyed to public treatment facilities. Other chapters in this volume address *pre-consumer* GHG emissions from waste in the industry (see 7.9.8), energy, forestry, agriculture, transportation, and construction sectors which are managed within their respective sectors. This chapter will also address emissions of volatile organic compounds (VOCs) from waste and post-consumer, end-of-life issues associated with F-gases (CFC’s, HCFC’s).

There is significant potential for accelerating the direct reduction of GHG emissions from waste as well as extended implications for indirect reduction within the industrial, agriculture, forestry, and energy sectors. Appropriate waste and wastewater management, including landfill gas recovery, directly reduce emissions from the waste sector. Waste prevention, reuse, and material recovery indirectly reduce GHG emissions by reducing waste generation, energy demand, and raw material consumption. The TAR (IPCC, 2001) included the following waste management practices:

- mitigation of landfill CH_4 emissions via landfill gas recovery,
- mitigation of wastewater and human sewage emissions of CH_4 and N_2O through improved management practices,
- reductions in fossil fuel use through waste-to-energy (incineration),
- recycling with decreased demand for virgin materials and decreased energy demand during production,
- reduction of emissions during waste transport.

Although new concepts and methodologies are discussed in this chapter, it must be stressed that the major technologies for GHG mitigation from waste are mature and include multiple technologies implemented at the municipal level. Using a life cycle approach, there are thus many choices and creative approaches which can be implemented by the public or private sector to mitigate GHG emissions from waste.

It must be emphasized that multiple benefits accrue from cost-effective waste management practices which improve the quality of life, promote public health, conserve natural resources, and concurrently reduce GHG emissions. Thus the mitigation of GHG emissions from waste and wastewater also bestows a wide range of public safety, health, and auxiliary environmental benefits through improved waste and wastewater management. Moreover, waste minimization, recycling, and re-use represent a growing but largely undefined potential for indirect reduction of GHG emissions through improved energy efficiency and fossil fuel avoidance.

Figure 10.1 provides an overview of carbon flows through waste management systems; this figure also indicates approximate carbon storage vs. carbon turnover for the major waste management strategies. Landfills function as relatively inefficient anaerobic digesters, and, for national inventories, the substantial carbon sequestration that occurs in landfills is currently not included in the methodological guidance—this will, however, be addressed by the IPCC 2006 Guidelines. Note in Figure 10.1 that the major gaseous C emission from waste is landfill CH_4 with minor CO_2 from incinerated fossil C (plastics). The CO_2 from composting or incineration of waste biomass is not considered in national GHG inventories under the UNFCCC.

[INSERT **Figure 10.1** here]

Figure 10.2 provides a process-oriented perspective on the major GHG emissions from landfills and wastewater. Figure 10.2a indicates CH_4 pathways in the context of a landfill CH_4 mass balance—emissions are one of several possible pathways for the CH_4 produced by anaerobic methanogenic microorganisms; others include recovery, oxidation, lateral migration, and internal storage. Figure 10.2b shows CH_4 and N_2O emissions from wastewater transport and treatment—as in landfills, the CH_4 is biogenically produced under strict anaerobic conditions, while the N_2O is an intermediate gaseous product of microbial N cycling under conditions of reduced aeration (not strictly anaerobic), high moisture, and available N.

[INSERT **Figure 10.2** here]

Therefore both the CH_4 and N_20 from the waste sector are microbially produced and consumed with rates controlled by temperature, pH, available substrates, microbial competition, and other factors. As a result, CH_4 and N_20 generation and consumption rates can 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 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, open sewers and uncontrolled solid waste disposal sites lead to serious public health problems resulting from pathogenic microorganisms, toxic odors, and disease vectors, in addition to GHG emissions.
Some major mitigation measures for the waste sector were previously addressed in the TAR (IPCC, 2001; Ackerman, 2000). These are updated and expanded in this chapter to encompass landfill CH_4 recovery for flaring or energy use; optimizing methanotrophic CH_4 oxidation in landfill cover soils; reduction at source through waste minimization, recycling, and re-use; alternative strategies to landfilling (incineration; mechanical and biological pretreatment/MBP); offsetting fossil fuel use via waste-to-energy, and wastewater management to minimize emissions via closed sewers, efficient wastewater treatment, at-source reduction in wastewater quantities, and wastewater recycling/reuse. In particular, it is important to emphasize that landfill CH_4 recovery as an alternative source of renewable energy has been fully commercial since 1975 and is now being implemented at >1150 plants worldwide with emission reductions of >105 Mt CO_2e/yr (Willumsen, 2003; Bogner and Matthews, 2003). This number should be considered a minimum because there are also many sites which recover and flare landfill gas without energy recovery.

With respect to the economic aspects of mitigating GHG emissions from waste and wastewater, the TAR (IPCC, 2001) previously indicated some important issues for developing comparative costs for GHG mitigation. These include a judicious choice of system boundaries, assessment approaches or models, baseline assumptions, and regionalized costs; also, one must consider local economic and social development factors. Because the waste sector is characterized by mature technologies whose diffusion limited by local costs, policies, available land area, and public perceptions, the discussion of mitigation costs and policies in this chapter will be organized according to a "technology gradient" approach: from low-technology/low-cost measures to high-technology/high-cost options. There is no single best option, but rather there are multiple commercially-available technologies. Because waste management technology decisions are often made locally, the goal of this analysis is to suggest strategies that can be collectively implemented to reduce GHG emissions and achieve sustainable development and public health goals.

10.2 Status of the waste management sector

10.2.1 Waste Generation Data

The availability and quality of annual data are a major problem for the waste sector. Solid waste and wastewater data are lacking for many countries, the reliability of existing data for many countries is questionable, definitions of waste and waste fractions are not uniform, and, because steady state assumptions are typically applied to solid waste and wastewater generation, interannual variability is often not well quantified (Bogner and Matthews, 2003). In this section, we focus on global waste and wastewater generation data. There are three major approaches that have been used to estimate global waste generation: country-specific data from a variety of methods for the IPCC national inventory process, population-based estimates used for the SRES waste scenarios, and the use of a proxy variable linked to demographic or economic indicators for which national data are annually collected. For the IPCC national inventories, most countries use population-based estimates based on regional or national statistics, data from one or more urban areas, or data from an adjoining country with similar demographics. This variety of methods results in data uncertainties both within and between countries. The SRES scenarios (Nakicenovic *et al.*, 2000) project continuous increases in waste and wastewater CH_4 emissions based upon population as the driver for increased waste generation to 2030 (A1B-AIM), 2050 (B1-AIM), or 2100 (A2-ASF; B2-MESSAGE).

In addition to population, waste generation rates are related to affluence - richer societies are characterized by higher rates of waste generation/cap, while less affluent societies generate less waste as well as practice informal recycling/reuse initiatives that decrease the waste/cap to be collected at the municipal level. It is thus possible to develop statistically significant relationships between waste generation/cap and certain proxy or surrogate variables which encompass both population and affluence, including GDP/cap (Richards, 1989; Mertins *et al.*, 1999) and energy consumption/cap (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 which 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 energy consumption/cap. Using UNFCCC-reported values for % biodegradable organic C in waste for each country, this BOX also shows trends for landfill C storage based upon the reported data.

Box 10.1. *1971-2002 Trends for regional solid waste*

THE PROBLEM OF WASTE GENERATION DATA*:* *Use of a Surrogate or Proxy Variable for Waste Generation/cap* (Bogner and Matthews, 2003). Solid waste data are lacking for many countries, the reliability of existing data for many countries is questionable, definitions differ among countries, and, because steady state assumptions are typically applied to solid waste generation, interannual variability is often not well quantified. In general, waste generation rates are a function of both population and prosperity. 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/cap for developed and developing countries. These empirical models were based on energy consumption/cap as an indicator of affluence and a proxy for waste generation/cap; the surrogate relationship was applied to annual national data using either total population (developed countries) or urban population (developing countries). Figure 10.3 plots the results for each country for 1971-2002 grouped by region, indicating almost 900 Tg waste in 2002. A conversion to energy equivalents (assuming 12,000 MJ/kg) indicates that waste in 2002 contained a substantial 1.1 X 10^{10} TJ of available energy. Unlike projections based on population alone, this figure shows waste generation from individual regions decreasing as well as increasing in tandem with major economic trends.

[INSERT **Figure 10.3a.** here]

[INSERT **Figure 10.3b.** here]

Figure 10.3b, using the same base data, illustrates annual C storage in landfills. This figure was developed from the percentages of landfilled waste for each country (reported to UNFCCC) and a conservative assumption of 50% C storage (Bogner, 1992; Barlaz, 1998). In landfills, lignin is recalcitrant to anaerobic decomposition while some fraction of the cellulose is also non-degraded. The annual totals for the mid-1980's and later (>30 Tg carbon/year) exceed estimates in the literature for the annual quantity of organic C partitioned to long-term geologic storage as a precursor to future fossil fuels (Bogner, 1992). In developed countries, anaerobic burial of waste in landfills has been widely implemented only since the 1960's and 1970's; since then, there have been large quantities of organic C annually sequestered in landfills; moreover, this is longer-term storage than typical for forest biomass C and many types of soil C.

Table 10.1 compares average income, solid waste generation, and % recyclables for low, middle, and high income countries (Cointreau-Levine, 1994). In general, waste generation rates range from <0.2 t/cap/yr in low income countries to >0.6 t/cap/yr in high income industrialized countries. Even though labor costs are lower in developing countries, solid waste management constitutes a higher share of income because both equipment and fuel costs are typically higher (Cointreau-Levine, 1994); also, waste collection in developing countries is less mechanized and more labor-intensive. By 1990, many developed countries had initiated comprehensive recycling programs. It is important to recognize that the % recycled or composted, incinerated, or landfilled differs greatly among municipalities as a result of multiple factors, including local economics, national policies, regulatory restrictions, public perceptions, and availability of open-space for landfill siting.

[INSERT **Table 10.1.** here]

10.2.2 Wastewater Generation Data

Wastewater data, like waste data, are not collected on an annual basis for many countries. For IPCC inventories, wastewater generation is estimated from protein consumption, using FAO data. Many countries, including most developed countries, do not collect annual statistics on the volume of wastewater generated, transported, and treated. In general, 58% of the global population has sanitation coverage (sewerage) with very high levels characteristic for the population of North America (100 %), Europe (92%), and Oceania (93%), although in the last two regions the percentage for rural areas reduces to 74% and 81%, respectively (Jouravlev, 2004; WHO/UNICEF/WSSCC, 2000, DESA, 2005; WHO-UNICEF, 2005; World Bank, 2005; PNUD, 2005). In developing countries, rates of sewerage are very low for rural areas of Africa, Latin America, and Asia, where septic tanks and latrines predominate. Moreover, for wastewater treatment, 88% of the population in developed countries but only 29% of the population in developing countries is served by wastewater treatment. North America has high levels of coverage (90%), followed by Europe (66%). There are no available data for Oceania. Other regions have, in general, low levels of waste treatment, with Asia at 35%, Latin America and the Caribbean at 14%, and much of Africa lacking wastewater treatment (Jouravlev, 2004; World Bank, 2005).

When considering CH_4 and N_2O emissions from wastewater treatment, one must take into account the existence of an established infrastructure in developed countries and the lack of infrastructure in developing countries resulting in very low treatment percentages (typically <25%). In developing countries, open sewers or informally-ponded wastewaters often result in generation of N_2O and CH_4 as well as uncontrolled discharges to rivers, lakes, and small streams. The majority of urban wastewater treatment facilities are publicly-operated, and only about 14% of the present total private investment in water and sewerage is financing wastewater treatment, mainly directed toward protection of drinking water supplies (Silva,1998; World Bank 1997).

Industrial wastewaters were previously discussed in Chapter 7 with the exception of highly organic wastewaters. The high BOD (Biochemical Oxygen Demand) industrial wastewaters will be included here as these are often transported to municipal treatment facilities. Table 10.2 summarizes regional and total 1990 and 2001 generation in terms of kg BOD/day or kg BOD/worker/day. [BOD measures the strength of an organic waste relative to the mass of oxygen required for aerobic degradation]. Unlike estimates from earlier studies based on engineering or economic models, these estimates are based on actual measurements of plant-level water quality (World Bank, 2005). The table indicates that total global generation decreased >7% between 1990 and 2001; however, increases were observed for the Middle East (+17%) and the developing countries of South Asia (+24%) with overall 2001 generation 35% higher from developing countries than for developed countries. The regions with the highest 2001 percentages relative to the global total are the developing countries of East Asia (43%), Europe (22%), and OECD North America (12%). Among developed countries, the highest percentages of the 2001 total are for Europe (52%), OECD North America (28%), and OECD Pacific (19%). For developing countries in 2001, the highest percentages are for the developing countries of South Asia (17%) and East Asia (63%), including China with an estimated 25% of the world total.

[INSERT **Table 10.2.** here]

10.2.3 Development Trends for Waste: Public Health, Regulatory, Policy, and Economic Trends for Waste Generation and Treatment

Developed countries are characterized by higher rates of waste recyling and waste pre-treatment. Economics largely dictate that the bulk of the residual solid waste is either landfilled or incinerated, although more costly practices such as anaerobic digestion have been locally implemented. In addition, many countries practice solid waste composting; however, because of compost quality issues, this is best applied to specific biodegradable waste streams that are source-separated. In countries with available open space (North America, Australia, New Zealand, Korea), landfilling is expected to continue as the dominant method for large scale waste disposal. In parallel, larger quantities of landfill CH_4 are being recovered for energy use. North America and Australia are actively implementing “bioreactor” landfill designs to compress the time period during which high rates of CH_4 generation occur. Decisions regarding waste management are made at the local level by communities with limited financial resources seeking the least-cost environmentally-acceptable solution. In most cases, this means landfilling, except where adequate open space is not available. In the EU, the landfill directive (Council Directive 1999/31/EC) mandates that by 2010, there must be a 75%

reduction in the mass of biodegradable organic waste that is annually landfilled. As a result, post-consumer waste is now being diverted to increased incineration, as well as the use of mechanical/biological treatment (MBT) before landfilling to 1) recover recyclables; and 2) reduce the organic C content by a partial aerobic composting. Current landfill cover designs in the EU tend to retard infiltration to limit leachate generation, resulting in lower rates of CH_4 generation; nevertheless, CH_4 production will continue at existing landfills with declining rates for the next 2-3 decades. In the U.S., incineration capacity has not increased significantly because the Supreme Court mandated that the interstate movement of waste to large regional landfills cannot be curtailed, denying flow control of waste to incinerators. In Japan, where there is little open space for landfill construction, high rates of incineration are practiced with utilization or landfilling of residuals. Historically, there have also been "semi-aerobic" Japanese landfills with potential for generation of N_2O (Tsujimoto *et al.*, 1994). Aerobic landfill practices have also been studied or implemented in Europe and the U.S. as an alternative, or in combination with, anaerobic practices (Ritzkowski and Stegman, 2005).

In developing countries, more controlled landfilling with anerobic decomposition of organic waste and increased CH_4 emissions 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. There are also persuasive public health reasons for implementing controlled landfilling - urban dwellers produce more solid waste per capita than rural inhabitants, and large amounts of refuse accumulating in areas of high population density are linked to vermin and disease (Christensen *et al.*, 1999). 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. To a large extent, this is the same transition that occurred in many developed countries in the 1950's and 1960's.

One must not neglect the role of informal waste recycling in developing countries. 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 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 jobs/t waste and recycling of >50% of collected waste can be attained (Iskandar, 2005).

10.3 Emission trends

10.3.1 Global Overview

Historically, for UNFCCC national inventories, a simplified C or N mass balance has been applied to waste and wastewater with some portion of the C or N annually partitioned to gaseous emissions as CH_4 or N_2O. 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; thus the standardization of terminology and 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.

Landfill CH_4 is the major GHG emission from the waste sector. In addition, CH_4 is emitted from wastewater, 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 also a small source of GHG emissions from this sector.

For national inventories, the IPCC Guidelines and Good Practice Reports (Houghton *et al.* 1997; Penman *et al.* 2000; Penman *et al.* 2003) provide guidance. Currently, two methods are available for landfill CH_4 emissions: (1) a simple mass balance method where all the potential CH_4 from a specified fraction of the degradable organic C is assumed to be released into the atmosphere during the year of disposal; and (2) a first order decay method using a first order kinetic equation to proportion the CH_4 emissions over a period of years, thus producing more accurate annual estimates. At specific landfill sites, it is standard practice to apply a first order kinetic equation, based on the annual quantities of waste disposal, to predict annual methane production and interannual variability for commercial gas recovery projects. The first order decay method has now been proposed as the default methodology for landfill CH_4 for the FOD 2006 UNFCCC Inventory Guidelines (Pipatti and Vieira, 2005); these guidelines discourage the use of the mass balance method. Nevertheless, the mass balance method provides a good estimate of emissions over the full life-cycle of landfill disposal and is suitable for studies comparing the long term GHG impact of different waste management strategies (Pipatti and Wihersaari 1998).

Non-CO_2 greenhouse gas emissions from wastewater treatment (CH_4, N_2O), and waste incineration (N_2O) are estimated using emission factors based on empirical data from a limited number of studies. The CO_2 from waste incineration is estimated from the fossil C content. 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 takes place mostly in developing countries, can be a significant local source of GHG emissions, and also results in emissions of air pollutants which are a health risk for nearby communities. Composting and other biological treatments emit very small quantities of GHGs compared to other treatments and have only been included in the 2006 IPCC Guidelines for completeness. [Note: to be updated when 2006 IPCC Guidelines approved by the IPCC Panel – scheduled for April 06]

Five-year global emission estimates and trends are given in Table 10.3 from UNFCCC inventories and projections. The table compares two recent compilations: (a) estimated CH_4 and N_2O emission trends from landfills and wastewater from Scheehle and Kruger (2005), including N_2O trends from human sewage only; and (b) reported 1990 and 1995 GHG emissions for waste management from Konte (2005), including landfill CH_4, wastewater CH_4 and N_2O, and CO_2 from incineration of fossil C. Totals for Annex I countries only are shown in brackets. Data from non-Annex 1 countries are limited and usually available only for 1994 (or 1990). Information on N_2O emissions from waste water is minimal, and therefore global estimates are based on human sewage treatment only. A comparison of (a) and (b) with the A1B and B2 SRES scenarios is also shown in Table 10.3 and will be discussed below.

Based on the UNFCCC estimates, landfills and waste water annually contributed about 5%–10% of global CH_4 and 1 % of global N_2O emissions, respectively, in the 1990-2005 period, based on (a) and (b) from Table 10.3 compared to global emissions from the TAR (IPCC, 2001b) of 600 Tg CH_4/yr and 17.7 Tg N as N_2O. Note that most of the increase from 1990 to 1995 reported by Konte

(1995) is from non-Annex I countries. Scheehle and Kruger (2005) project increases in annual CH_4 and N_2O emissions of 33-36% from 1990 to 2020, with the non-Annex 1 countries contributing over half of the total CH_4 and N_2O emissions over this time period.

[INSERT **Table 10.3.** here]

A direct comparison with the A1 and B2 SRES scenarios (N. Nakicenovic *et al.*, 2000) is problematical because these do not include landfill gas recovery and project continuous increases in CH_4 emissions from the waste sector to 2030 or later (Table 10.3). For example, based on assumed population growth and increased use of landfills, the A1B marker scenario (A1B-AIM) projects increased CH_4 emissions from waste and wastewater through 2030 followed by a decline to about 1990 levels. The B2-MESSAGE scenario projects that CH_4 emissions continually increase from 1990 to 2100. These two scenarios estimated 1990 CH_4 emissions of approximately 1300 Tg CO_2e/yr and very high 2050 emissions of >4000 Mt CO_2e per year.

10.3.2 Regional Trends: Landfill CH_4

Although landfill CH_4 has historically been the largest source in the waste sector, the combination of improved estimation methodologies, "bottom up" field measurements constraining rates of CH_4 generation and emissions, and increased rates of landfill CH_4 recovery have resulted in decreasing global estimates during the last 20 years. Given the uncertainty associated with waste generation and GHG emissions estimates, we will compare two regional approaches for landfill CH_4: the Scheehle and Kruger (2005) inventory estimates with projections for selected years and annual estimates using a proxy variable (energy consumption/cap) for waste generation/cap (methodology of Bogner and Matthews, 2003). Figure 10.4 compares the regional estimates for the years 1990, 1995, and 2000 plus projections for 2005-2020 from Scheehle and Kruger (2005) to historical estimates for 1971-2002 using the energy consumption surrogate. Note that the Scheehle and Kruger (2005) estimates for OECD North America and the developing countries of S and E. Asia are typically higher than those referenced to energy consumption; however, the European and OECD Pacific trends converge at the present time.

[INSERT **Figure 10.4.** here]

Landfill CH_4 recovery has been fully commercial since 1975 and has already achieved significant emissions reductions in many countries. Currently, >105 Mt CO_2e /yr are recovered globally (Willumsen, 2003). Moreover, a linear regression using historical data indicates a current growth rate of approximately 5% per year (Bogner and Matthews, 2003). A comparison of the present rate of landfill CH_4 recovery to estimated global emissions from Scheehle and Kruger (2005) in Table 10.3 indicates that annual recovery and utilization currently exceed the projected 5 year emissions increase from 2005 to 2010. Thus, it is reasonable to state that landfill CH_4 recovery is beginning to stabilize emissions from this source. Moreover, because there are many projects which recover and flare landfill gas without energy use, it is likely that the present rate of landfill CH_4 recovery may realistically be much higher than 105 Mt CO_2e per year. If recovery continues to increase by 5% annually, more than 1000 Mt CO_2e per year could be recovered by 2050. It is anticipated that, as developing countries implement more controlled landfilling practices, incentives such as the CDM will increasingly promote landfill CH_4 recovery and use.

For the EU15, trends indicate that landfill CH_4 emissions are declining. Between 1990 and 2002, landfill CH_4 emissions decreased by almost 30% due to the early implementation of the landfill directive (199/31/EC) and similar national legislation intended to reduce biodegradable waste going to landfills and increase landfill CH_4 recovery at existing sites. Emissions from the waste sector are projected to be more than 50% below 1990 levels in 2010 largely due to existing policy measures, including the EU landfill directive and mandated recovery of landfill CH_4. The largest reductions are projected for Portugal, Sweden and the United Kingdom with more than a 60% decline, and for Finland and Portugal (more than 75 %) in the additional measures scenario (EEA, 2004).

10.3.3 Regional Trends: Wastewater and Human Sewage CH_4 and N_2O

Wastewater emissions are significantly correlated to population trends (US EPA, 2001). CH_4 and N_2O are produced and emitted during municipal and industrial wastewater collection and treatment, depending on transport, treatment, and operating conditions. The resulting sludge may also generate CH_4 and N_2O if it is further biodegraded without gas capture.

Figure 10.5 summarizes the 1990 estimated CH_4 and N_2O emissions and projected trends to 2020 from wastewater and human sewage (UNFCCC/IPCC, 2004). Although there are few studies on controlling emissions from wastewater, there is general consensus that emissions of CH_4 and N_2O from wastewater are relatively small compared to other sources. In developed countries there is an extensive infrastructure for wastewater treatment, typically relying on centralized aerobic treatment; thus CH_4 emissions are small and incidental. Wastewater can also be treated anaerobically, with significant CH_4 being produced and emitted if control measures are lacking; usually the resulting biogas is used for heat or onsite electrical generation. In general, due to rapid population growth, urbanization, and industrialization, the wastewater CH_4 and N_2O emissions from developing countries are higher than from developed countries. However, data reliability for developing countries is a major issue. Decentralized "natural" treatment processes in developing countries may produce 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, 2002).

[INSERT **Figure 10.5.** here]

Figure 10.5a summarizes estimated regional CH_4 emissions from wastewater. The highest emissions are from China, India, and the United States. Other countries with high emissions in their respective regions include Korea, Turkey, Bulgaria, Iran, Brazil, Nigeria, and Egypt. Global emissions of CH_4 from wastewater handling are expected to rise more than 40% from 1990 to 2020. The only regions with decreased emissions in 2020 relative to 1990 are Europe and Countries in Transition. Comparing CH_4 emissions in 1990 and 2020 indicates that OECD Annex 1 countries emitted 59 Mt CO_2e in 1990 and are expected to rise about 12% by 2020; the Non-OECD Annex 1 countries emitted 12 Mt CO_2e and are expected to decrease 25% by 2020. The Non-Annex 1 countries emitted 302 Mt CO_2e and are expected to increase 47% by 2020 with the highest emissions from the developing countries of East and South Asia and the Caribbean, Central and South America. The highest emissions are from Asia, Africa, and Central and South America. Regions with the highest projected growth in emissions by 2020 are: the Middle East with a projected increase of 168%; the Caribbean, Central America and South America with 144%; the developing countries of East Asia with 134%, and Africa with 119% .

Figure 10.5b summarizes estimated N_2O emissions from human sewage. (UNFCCC/IPCC, 2004). The contribution of human sewage to atmospheric N_2O is very low and is expected to fluctuate from 70-90 Mt CO_2e/yr during the period 1990-2020. 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. It is expected that global emissions will rise 32% by 2020 in comparison with 1990. The regions with the highest emissions are the developing countries of East Asia, the developing countries of South Asia, Europe, and the OECD North America. The continents with the highest emissions are (in decreasing order): Asia, America, Africa, and Europe. The highest emissions are from China, India, and United States; other countries with high emissions are Japan, Germany, France, Italy, Russia, Sub-Saharan Africa, Iran, and Brazil. Regions whose emissions are expected to increase the most are: Sub – Saharan Africa 88%, N Africa 75%, the Middle East 60%, the developing countries of S Asia 58%, and the OECD North America 43%. The only regions that are expected to decrease their emissions in 2020 relative to 1990 are Europe and the Countries in Transition.

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10.3.4 Carbon Dioxide: Fossil C Incineration

The major GHG emission from waste incineration is CO_2 from fossil C sources such as plastics. Detailed data on waste incineration are difficult to obtain for most countries. Japan incinerates > 70% of the waste generated, while in the EU25 about 17% of municipal solid waste was incinerated with energy recovery in 2003 (Eurostat 2003; Statistics Finland 2005); for the U.S., approximately 15% of waste is incinerated. Incineration is especially important in Denmark and Luxembourg, at 52% and 59% of waste, respectively, as well as in France, Sweden, the Netherlands, and Switzerland. Incineration is increasing in most EU countries as a result of the EU Landfill Directive.

In developing countries, incineration of waste is less common than in developed countries because of high capital and operating costs. Incineration is also not the technology of choice for wet waste, and many developing countries have a high % of putresibles with high moisture content in their mixed waste. Uncontrolled burning of waste is common is developing countries to reduce volume and contributes significantly to urban air pollution (World Bank, 1999).

10.4 Mitigation of post-consumer emissions from waste

10.4.1 Waste management and GHG mitigation technologies

Mitigation of GHG emissions from waste includes a range of mature 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 10.6). These technologies include post-consumer recycling, landfilling, composting of selected waste fractions, MBT with landfilling of residuals, anaerobic digestion, and incineration. At the "high technology" end, there are also advanced processes for waste management such as pyrolysis and gasification, which have not been applied at large scale (thousands of t/d) and are largely inappropriate for mixed waste. There are also numerous low- to high- technology methods for quantification of GHG emissions from waste management - from bottom up measurements to top down estimation methods for constraining GHG emissions at landscape scales.

[INSERT **Figure 10.6.** here]

10.4.2 CH_4 management at landfills

CH_4 emissions from landfills are estimated to range from approximately 400 to 800 Mt CO2e/yr (Bogner and Matthews 2003; Scheehle and Kruger, 2005). Landfill CH_4 emissions from small scale surface measurements (area <1m^2) can vary over 7 orders of magnitude (0.0001 - >1000 g CH_4 m^{-2} day^{-1}) 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 U.S., and South Africa exhibit about one order of magnitude variation--from 0.1 to 1.0 t CH_4 $ha^{-1}d^{-1}$ (equivalent to 0.03 to 0.3 g m^{-2} d^{-1}) (Nozhevnikova *et al.*, 1993; Borjesson, 1996; Czepiel *et al.*, 1996; Hovde *et al.*, 1995; Mosher *et al.*, 1999; Tregoures *et al.*, 1999; Galle *et al.*, 2001; Morris, 2001).

The implementation of an active gas extraction system using vertical wells or horizontal collectors is the single most important mitigation measure to reduce CH_4 emissions. This technology has been fully commercial since 1975 and has been implemented at more than 1150 facilities worldwide (Willumsen, 2003), primarily in the U.S., Europe, and Australia. 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.*, 2005). Landfill CH_4 is currently 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 carbon dioxide 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.

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). The thickness, physical properties, and moisture content 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. Oxidation rates in conventional landfill cover soils may be as high as 166-240 g CH_4 m^{-2} d^{-1} (Kightley *et al.*, 1995; de Visscher *et al.*, 1999) and greater than 300 g m^{-2} d^{-1} in thick, compost-amended "biocovers" engineered to optimize oxidation (Huber-Humer, 2004; Bogner *et al.*, 2005). Thus the combination of engineered gas extraction and natural CH_4 oxidation can be extremely effective to reduce emissions. 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 biofilters which utilize methanotrophic microorganisms to reduce emissions (Gebert and Gröngröft, 2005; Streese and Stegman, 2005). Stable C isotopic techniques are useful to quantify the fraction of CH_4 that is oxidized in landfill cover materials (Chanton and Liptay, 2000; de Visscher and Chanton, 2004) A secondary benefit of CH_4 oxidation in cover soils is the aerobic oxidation of many non-CH_4 organic compounds, especially aromatic and lower chlorinated compounds, thereby reducing their emissions to the atmosphere (Scheutz *et al.*, 2003).

Landfills are a significant source of CH_4 emissions, but they are also a long-term sink for carbon (Bogner, 1992; Barlaz, 1998; 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 converted to biogas carbon but remains in the landfill. Carbon storage makes landfills 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).

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

Incineration reduces the mass of waste and substitutes for fossil fuels; in addition, GHG emissions are avoided except for a small contribution from fossil C. Incineration has been widely applied, especially in countries with limited space for landfilling such as Japan and the Netherlands. Waste-to-energy plants produce heat or electricity for productive use, which improves process economics. In northern Europe, urban incinerators have historically supplied fuel for district heating of residential and commercial buildings. In developing countries, rural areas, and also historically in developed countries, waste has often been inefficiently burned to reduce volume and recover noncombustible recyclables, especially metals.

Waste incinerators have been extensively used for more than 20 years, primarily in Europe and Japan, with increasingly stringent emission standards. Mass burning is relatively expensive (range of 50-150 €/tonne) [Faaij *et al.*, 1998]. Typical electrical efficiencies are 15% to >20% with more efficient designs (>30%) now available. Starting in the 1980s, large waste incinerators with stringent emission standards were widely deployed in Germany and the Netherlands. Typically such plants have a capacity of about 1 Mt waste/yr, moving grate boilers (which allow mass burning of very diverse waste properties), low steam pressures and temperatures (to avoid corrosion) and extensive flue gas cleaning. In recent years advanced combustion concepts have penetrated the market, including fluidized bed technology and advanced flue gas cleaning.

10.4.4 Biological treatment including composting, anaerobic digestion, and MBP (Mechanical Biological Pretreatment)

Many developed and developing countries practice composting and anaerobic digestion of mixed waste or biodegradable waste fractions (kitchen or restaurant wastes, garden waste, manures, sewage sludge). Both processes are best applied to source separated waste fractions: anaerobic digestion is particularly appropriate for wet wastes while composting is more appropriate for drier feedstocks. Composting decomposes waste aerobically into CO_2, water and a humic fraction. Anaerobic digestion produces CH_4, CO_2 and biosolids; the CH_4 can be used for process heating or onsite electrical generation. In particular, Denmark, Germany, Belgium, and France have implemented anaerobic digestion systems for waste processing, including the large Valorga plant in France.
The CO_2 emissions from composting and anaerobic digestion are biogenic and therefore not included in UNFCCC inventories. 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. Minor quantities of CH_4 can also be vented from digesters during start-ups, shut-downs, and malfunctions. However, the GHG emissions from controlled biological treatment are small in comparison to uncontrolled CH_4 emissions from landfills (e.g. Petersen *et al.* 1998; Hellebrand 1998; Vesterinen 1996; Beck-Friis 2001; Detzel *et al.* 2003). The advantages of biological treatment are reduced volume, waste stabilisation, and pathogen destruction. Depending on quality, the residual solids can be recycled as fertiliser or soil amendments, used as a CH_4-oxidizing biocovers on landfills (Huber-Humer, 2004), or be landfilled at reduced volumes, reducing CH_4 emissions.

Mechanical-biological treatment (MBT) of waste is now being widely implemented in Germany, Austria and other EU countries. Mixed waste is subjected to a series of mechanical and biological operations which reduce volume and achieve partial stabilisation of the organic C. Typically, mechanical operations such as shredding and crushing produce waste fractions for further treatment (composting, anaerobic digestion, combustion, recycling); then the subsequent biological processes may include composting or anaerobic digestion. Composting may occur either in open windrows or in closed buildings with gas collection and treatment. Reductions of as much as 40 - 60% of the organic C are possible with MBT (Kaartinen 2004). Due to reductions in mass, organic C, and biological activity, MBT-treated waste theoretically produces up to 95% less CH_4 when landfilled. In practice, reductions have been smaller, depending on the type and duration of various treatments (see Binner, 2002).

10.4.5 Waste reduction, reuse and recycling of secondary materials

Material efficiency can be defined as a reduction in the use of primary materials for a particular purpose such as packaging or construction, without negatively impacting existing human activities. Efficient use of materials also reduces waste. At several stages in the life cycle of a product, material efficiency can be increased by efficient design, material substitution, product recycling, material recycling, and quality cascading (use of recycled material for a secondary product with lower quality demands). All these measures lead to indirect energy savings and reductions in GHG emissions. In this chapter, we address material recycling, and quality cascading whereas other chapters discuss efficient design, material substitution, and product recycling in various industrial contexts. Recycling reduces GHG emissions through lower energy demand for production (avoided fossil fuel) and by substitution of recycled feedstocks for virgin materials. This is especially true for products resulting from energy-intensive production processes such as metals (steel and Al), glass, plastic, and paper (Tuhkanen *et al.*, 2001).

Both material recycling and quality cascading are deployed in many countries at large scale for re-use of metals (steel, aluminium) and recycling of paper, plastics, and wood. The reductions in energy use and GHG emissions are quantified using the GER (Gross Energy Requirement, or GJ primary fuel/t) and other methods. When lower grade materials are displaced, the GER values need to be adjusted.

10.4.6 Wastewater and sludge treatment

There are many available technologies for wastewater management, collection, treatment, reuse and disposal, ranging from energy-intensive advanced technologies to natural purification processes. Systematic decision-making tools are now available which include both environmental tradeoffs and costs (Ho, 2000). When efficiently applied, wastewater transport and treatment technologies reduce or eliminate CH_4 and N_2O generation and emissions, as well as promote water conservation by preventing pollutants from entering the water or requiring a smaller volume of water to be treated. Since the size of treatment systems is primarily governed by the volume of water to be treated rather than the mass loading of pollutants, smaller volume implies smaller treatment plants and lower capital costs.

Low-water use toilets are also desirable at the household level to limit water use (Ho, 2000).

Wastewater collection and transport includes conventional (deep) sewerage, simplified (shallow) sewerage, and settled 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 capital costs reduced by 30-50% compared to deep systems. Settled sewerage, or the transport of wastewater following solids settlement (septic tanks), is inexpensive and widely used in both developed and developing countries.

Wastewater treatment removes pollutants using a variety of technologies. Small wastewater treatment systems include pit latrines, composting toilets, pour-flush toilets, and septic tanks. Improved on-site treatment systems used in developing countries include inverted trench systems and aerated treatment units in urban areas. More advanced treatment systems include activated sludge treatment, trickling filters, anaerobic or facultative lagoons, anaerobic digestion, and constructed wetlands and other land-based treatment systems. 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 Trickling filters have lower energy requirements than activated sludge systems and are also widely used in developed countries. Anaerobic lagoons produce CH_4 while facultative lagoons are aerobic during the day and anaerobic at night. Frequently, there can be a series of lagoons with anaerobic, facultative, and aerobic stages. Reduction or pretreatment of industrial wastes is often necessary to limit excessive pollutant loads on municipal systems., especially when contaminated with heavy metals. Also, separation of black water and grey water can reduce the overall energy requirements for treatment (UNEP/GPA-UNESCO/IHE, 2004).

Sludges (or biosolids) are the product of most wastewater treatment systems. Options for sludge treatment include stabilization, thickening, dewatering, anaerobic digestion, agricultural reuse, drying and incineration. The agricultural application of raw sludge from activated sludge plants is unsatisfactory because of the presence of pathogens and, in the case of industrial inputs, heavy metals. However, pathogens can be reduced to non-hazardous levels by composting sludge at 55 ^{0}C for two weeks. The use of composted sludge as a soil conditioner in agriculture and horticulture returns C, N, P and other elements essential for plant growth back to the soil. Heavy metals and some toxic chemicals are difficult to remove from sludge; either 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. Sludge incineration is expensive due to the necessity of costly air pollution control measures but can be used when the sludge is too heavily contaminated for alternative uses (UNEP/GPA-UNESCO/IHE, 2004). Some sludges are landfilled, but this practice may result in volatile siloxanes (from silicones retained through the wastewater treatment process) or H_2S (from sulfate reduction) in the landfill gas. Siloxanes result in siliceous deposits while H_2S results in acid gas corrosion in engines utilizing the landfill gas for onsite electrical generation.

Treated wastewater can either be reused or discharged; where possible, reuse is the most desirable option. Uses include agricultural and horticultural irrigation, fish aquaculture, artificial recharge of aquifers, and industrial applications. Disposal options include percolation through the ground, in many cases to recharge shallow aquifers, or discharge to waterways. When discharged, the capacity of the receiving water to assimilate wastewater nutrients (N and P) must be considered to prevent excessive algal growth.

10.4.7 Waste management and mitigation costs

All climate change policies and measures necessitate some costs. In the waste sector, it is often not possible to clearly separate costs for direct or indirect GHG mitigation from costs for waste management. Thus one must be particularly careful about baseline assumptions, assumed costs, local availability of technologies, and economic and social development issues when costing alternative waste management strategies. Some common metrics applied to large CO_2 sources, such as marginal abatement curves (Delhotel, 2005), should be applied with caution within the waste sector where facilities are smaller, dispersed, and influenced by site-specific economics based on many factors in addition to GHG abatement. It is important to emphasize that waste management costs can exhibit high variability depending on local conditions.

Table 10.4 qualitatively assesses applicability and costs for a range of waste management technologies for GHG mitigation, concluding that low to moderate costs were associated with most technology options except incineration. Thus there are multiple mature, cost-effective technologies. The dominant GHG emission from the waste sector is landfill CH_4, which is readily controlled using existing landfill gas recovery, combustion, and utilization technologies. Calculations for the EU indicate that, in order to reach a 28% overall reduction by 2010 from the 1990 total GHG emissions from waste and wastewater (CH_4, N_2O, and CO_2), a reduction of 30% in CH_4 emissions would be required (EEA, 2004).

[INSERT **Table 10.4.** here]

Table 10.5 summarizes cost estimates for GHG mitigation from waste. This table provides an overview of a bottom-up cost analysis for GHG mitigation from waste management strategies compared to landfilling, based primarily on data from the U.K. and the Netherlands (Bates and Haworth, 2001). For the landfill comparison, the 2001 rate of landfill gas recovery for the EU as a whole was estimated to be 20% while 70% was assumed to be the maximum % CH_4 recovery over the lifetime of an individual site. For the 70% scenario, all the alternative technologies showed modest gains of approximately 0.5 t CO_2 e/t waste.

[INSERT **Table 10.5** here]

Table 10.6 gives additional cost information. In a study for the Netherlands, de Jager and Block (1996) estimated the cost-effectiveness for mitigating CH_4 emissions from waste using alternative technologies (Table 10.6a). Table 10.6b gives the direct costs associated with landfill CH_4 recovery and utilization for onsite generation of electricity (Willumsen, 2003).

[INSERT **Table 10.6.** here]

In general, and this is especially true for emissions from the waste sector which is dominated by non-CO_2 gases, national multi-gas abatement strategies can significantly reduce the cost of mitigation as compared to achieving the same level of GHG reduction via CO_2 alone.

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

High GWP F-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). 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 only relevant for the foams; for other products, release will occur during use or just after end-of-life. For the rigid foams, releases of F-gases during use are small (Kjeldsen and Jensen, 2001, Kjeldsen and Scheutz, 2003, Scheutz et al, 2005a), so most of the original content is still present at the end of their useful life. The rigid foams include polyurethane and polystyrene; these have been used as insulation in appliances and buildings; CFC-11 and CFC-12 were the main blowing agents until the mid-1990s. After the mid-1990's, 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 F-gases in building insulation have not yet reached the end of their useful life (TFFEoL, 2005).

Products containing F-gases are managed in different ways. After 2001, disposal of appliances containing F-gas foams was prohibited in EU landfills (TFFEoL, 2005), resulting in appliance-recycling facilities where F-gases were destroyed. A similar system has been established in Japan since 2001 (TFFEoL, 2005). For other developed countries, appliance foams are buried in landfills, either directly or following shredding and 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 U.S. and some 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, 2005a). Then, when landfilled, compactors reduce the size of residual foam materials and enhance subsequent release of the F-gases.

In the anaerobic landfill environment, however, released F-gases may be biodegraded because CFCs and, to some extent, HCFCs undergo dechlorination (Scheutz *et al.*, 2005b). Potentially this may result in the production of more toxic intermediate degradation products (e.g., for CFC-11 the degradation products can be HCFC-21 and HCFC-31); however, 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; however, field studies are needed to verify that CFCs and HCFCs are being attenuated in full-scale landfills.

10.4.9 Air quality issues associated with waste management activities: VOCs and combustion emissions

This section will address landfill gas and incinerator emissions of VOCs, metals, and other air pollutants. Uncontrolled emissions of collected landfill gas and incinerated waste are not permitted in developed countries; landfill gas must be flared and incinerators must be equipped with advanced emission controls.

Landfill gas contains trace concentrations of aromatics, chlorinated and fluorinated hydrocarbons, reduced S gases, and other species; however, high hydrocarbon destruction efficiencies are typically achieved in enclosed flares (>99.99%), which are recommended over lower efficiency open flares. In some cases for landfill gas utilization projects (engines, turbines, or gas upgrading), landfill gas must be pre-treated to remove VOCs and H_2S to comply with manufacturer's warranty specifications. Hydrogen sulfide is mainly a problem at sites which co-disposed large quantities of construction and demolition debris containing gypsum board. At a landfill, the type and level of gas treatment depend on site-specific factors (gas flow rate, trace gas concentrations, proposed use, hardware vendor specifications). Treatment techniques can include solid adsorption, chemical oxidation, liquid absorption, and membrane processes. Emissions of NO_x can sometimes be a problem for permitting biogas engines as new sources in strict air quality regions.

At landfill sites, recent field studies have indicated that VOC fluxes through final cover materials are very small with both positive and negative fluxes on the order of 10^{-8} to 10^{-5} $g{\cdot}m^{-2}{\cdot}d^{-1}$ for individual species (Scheutz *et al.*, 2003; Bogner *et al.*, 2003). In general the emitted compounds consist of species recalcitrant to aerobic degradation (especially higher chlorinated compounds), while negative emissions (uptake from the atmosphere) are observed for species which are readily degradable in aerobic cover soils, such as the aromatics and vinyl chloride. Uptake (negative emission) occurs when air in urban areas and above landfill sites contains elevated VOCs (especially aromatics from mobile sources), and soil gas profiles indicate that the direction of diffusive flux is from the atmosphere into the soil. In contrast, emissions from the temporary cover area are mainly positive and higher, on the order of 10^{-5} to 10^{-4} $g{\cdot}m^{-2}{\cdot}d^{-1}$ for individual species.

For reducing incinerator emissions of volatile heavy metals and dioxins/dibenzofurans, the pre-combustion removal is recommended for batteries, plastics, and other waste materials containing heavy metals (Pb.Cd) and chlorinated compounds. Modern incinerators must meet stringent emission control standards in Japan, Germany and the rest of the EU, the U.S., and other developed countries.

10.5 Policies and measures: waste management and climate

GHG emissions from waste are influenced by numerous policy and regulatory strategies which encourage energy recovery from waste, restrict choices for ultimate waste disposal, promote waste recycling and reuse, and encourage waste minimization. In many developed countries, especially Japan and the EU, waste management policies are closely related to and integrated with climate policies. Although policy instruments within the waste sector consist mainly of regulations, there are also economic measures in a number of countries to encourage particular waste management technologies, recycling, and waste minimization. In industrialized countries, waste minimization and recycling are encouraged through policy and regulatory drivers. In developing countries, major policies are aimed at restricting the uncontrolled dumping of waste. Table 10.7 provides an overview of policies and regulations.

[INSERT **Table 10.7.** here]

10.5.1 Reducing landfill CH_4 emissions

Landfill CH_4 is the dominant GHG emission from the waste sector. There are two major strategies to reduce landfill CH_4 emissions: implementation of standards that require or encourage landfill CH_4 recovery and a reduction in the quantity of biodegradable waste that is landfilled. The U.S. has implemented regulations under the Clean Air Act (CAA) which require large landfills to capture and combust landfill gas (U.S. Department of State, 2002). The EU requires member states to insure that landfill gas will be captured and flared at all landfills receiving biodegradable waste (Commission of the European Community, 2001). More broadly, the EU Landfill Directive (1999/31/EC) requires a phased reduction in the quantity of biodegradable waste landfilled to 50% of 1995 levels by 2009 and 35% by 2016 (Commission of the European Community, 2001). An increase in the availability of landfill alternatives is required (Price, 2001) to achieve these regulatory goals; alternatives include increased recycling, composting, incineration, and combined strategies such as mechanical biological treatment (MBT) prior to landfilling.

Landfill CH_4 recovery has also been encouraged by several country-specific economic and regulatory incentives. In the U.K., for example, the Non Fossil Fuel Obligation (NFFO), requiring a portion of electrical generation capacity from non-fossil sources, provided a major incentive for landfill-CH_4-to-electricity projects during the 1980's and 1990's. In other European countries, the decentralization of electrical generation capacity via renewable sources provides greater incentives for the development of on-site electrical generation from landfill CH_4. In the U.S., as mentioned above, the implementation of Clean Air Act (CAA) regulations in the early 1990's provided a regulatory driver for gas recovery at large landfills; in parallel, the U.S. EPA Landfill Methane Outreach Program has provided technical support to project developers. Also, periodic tax credits in the U.S. have provided an economic incentive for landfill gas utilization—for example, almost 50 of the 400 commercial projects in the U.S. came on line in 1998, just before the expiration of Section 29 (IRS code) tax credits. A small U.S. tax credit has again become available for landfill gas and other renewable energy sources; in addition, some states also provide economic incentives through tax structures or renewable energy credits. It is anticipated that landfill CH_4 recovery will increase significantly in the developing countries of Asia, South America, and Africa during the next two decades as controlled landfilling is phased in as a major waste disposal strategy. Where this occurs in parallel with deregulated electrical markets and more decentralized electrical generation, it provides a strong driver for increased landfill CH_4 recovery for energy use. Significantly, the recent availability of the Clean Development Mechanism (CDM) for Kyoto signatory countries is providing a strong economic incentive for both improved landfilling practices (to permit gas extraction) and landfill CH_4 recovery.

10.5.2 Promoting waste minimization, reuse, and recycling

Widely implemented policies include Extended Producer Responsibility (EPR), unit pricing (or PAYT/"Pay As You Throw"), and landfill taxes. Extended Producer Responsibility (EPR) regulations extend the producer's responsibility to the post-consumer period, thus providing a strong incentive to redesign products with fewer materials in smaller quantities more amenable to recycling (OECD, 2001). In general, EPR programs are expensive (Hanisch, 2000), and their economic and environmental benefits are still under debate. On the other hand, unit pricing (PAYT) has been widely adopted to decrease landfilled waste and increase recycling (Miranda *et al.*, 1996). In general, decreases can be partly explained by increased recycling, waste minimization, and other measures (Miranda *et al.*, 1994; Fullerton and Kinnaman, 1996). Although a large number of municipalities have instituted PAYT, there are remaining questions related to its long term sustainability (Yamakawa and Ueta, 2002).

Another economic instrument to reduce waste is the landfill tax, an environmental tax added to tipping fees for waste disposal by landfill. Its purpose is to reduce landfilled waste by artificially increasing the cost to levels commensurate with competing technologies such as incineration, thus encouraging the alternatives. In the U.K., the landfill tax has been used as a funding mechanism for environmental and community projects as discussed by Moriis *et al.* (2000) and Grigg and Read (2001).
Separate and efficient collection of recyclables is needed with both PAYT and landfill tax systems. For curbside programs, the percentage recycled is related to the efficiency of curbside collection and the duration of the program (Jenkins *et al.*, 2003). Other policies and measures include local subsidies and educational programs for collection of recyclables, domestic composting of biodegradable waste, and promoting recycled products (green procurement). In the U.S., 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.

10.5.3 Promoting incineration (waste-to-energy)

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 also been adopted. As discussed above, landfill taxes have been implemented in a number of EU countries to elevate the cost of landfilling to be equivalent to incineration.

10.5.4 Miscellaneous policies and measures: F gases and JI/CDM

Refrigerants and blowing agents for foam insulation are sources of F gases. Promoting substitutions for F gases is an effective countermeasure, and several countries have promoted substitutions and gradually phased-out the use of the long-lived high GWP fluorinated gases based on voluntary agreements, taxes, or direct regulation. A number of countries have also 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). As discussed above, neither the EU nor Japan have permitted landfill disposal of appliances containing F-gas foams after 2001 (TFFEoL, 2005).

As described in Section 10.3, open dumping is the common disposal method for solid waste in many developing countries where pollutant and GHG emissions occur concurrently with odor, public safety, and health problems. Because lack of financing is a major impediment to improved solid waste management in EIT and developing countries, the JI and CDM have already proven to be useful mechanisms for external investment from industrialized countries. The benefits are twofold: improving waste management practices and reducing GHG emissions. This is especially true for landfill gas recovery projects because an engineered landfill with cover materials is required to minimize air intrusion during gas extraction (to prevent internal landfill fires). Thus landfilling practices in many developing countries will require upgrading so that sites are suitable for gas recovery. To validate CDM projects, it is important to set proper baselines, as these can cause significant differences in the resulting emission reductions (Hiramatsu *et al.*, 2003). However, at all landfill gas CDM projects, the GHG credits are determined directly from quantification of the CH_4 captured and combusted.

10.5.5 Non-climate policies affecting GHG emissions

The EIT and many developing countries have implemented market-oriented structural reforms which 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. To date, solid waste generation does not support an environmental Kuznets curve (Dinda, 2004) because environmental problems related to waste can be externalized. Decoupling waste generation from economic and demographic drives, or dematerialization, is often discussed in the context of sustainable development, but the literature shows no absolute decline in material consumption in developed countries (Bringezu *et al.*, 2004). Currently, China is encouraging “circular economy”, which includes a closed loop for material flows associated with new economic development activities.

In many countries in Europe, Asia, and N America, the anaerobic digestion of wastewater and sludges to produce CH_4 creates a useful biofuel for heat or onsite electrical generation (Government of Japan, 1997; Government of Republic of Poland, 2001). In the Millennium Summit held in September 2000, it was agreed to reduce by 50 % the number of people without access to potable water in 2015. In 2002, the Johannesburg Summit reaffirmed this commitment, adding another goal to reduce the number of people without access to sanitation services by 50 %. It is very difficult to reach this goal in the majority of developing countries from Africa, Asia, and Latin America without the financial, technical, and capacity-building expertise of the international community.

Most policies and measures in the waste sector address broad environmental objectives, such as preventing pollution, avoiding odors, 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, EU Landfill Directive is primarily concerned with preventing pollution of water, soil, and air (Burnley, 2001).

10.6 Long-term considerations and mitigation options

Projections of GHG emissions through 2020 were previously discussed (see Table 10.3). This section will discuss long-term trends and options.

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, e.g. 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 Clean Development Mechanism (CDM), which is being increasingly 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, reuse, and energy recovery. Huhtala (1997) studied optimal recycling rates for municipal solid waste using a model which included recycling costs (including social 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, paper/cardboard, and glass (OECD, 2002).

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 landfill cover designs to increase CH_4 oxidation ("biocovers") can completely mitigate site-specific CH_4 emissions. These technologies are simple (“low tech “) 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 mitigation potential for the waste sector. When the fossil fuel offset is also taken into account, the GHG impact can even be negative (e.g., Lohiniva *et al.* 2002; Pipatti and Savolainen 1996). High cost, however, is a major barrier to the increased implementation of waste-to-energy, but advanced technologies are expected to become more competitive as both energy prices and emissions trading increase.

In some developing countries, small-scale anaerobic digestion with CH_4 recovery and use is locally deployed as a simpler waste-to-energy strategy. These 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.

The mitigation potential of recycling technologies is still largely unexplored with existing studies yielding variable results, in part because of the differing assumptions and methodologies applied. A recent study (Myllymaa *et al.*, 2005) examined the environmental benefits of alternative waste management strategies.

10.6.2 Wastewater Management

GHG emissions from wastewater are lower than emissions from solid waste management. In addition, the quantity of wastewater collected and treated is increasing in many parts of the world to maintain and improve potable water quality. This will decrease GHG emissions because well-managed wastewater treatment plants result in lower emissions than septic tanks, latrines, or uncontrolled discharges into waterways. Wastewater can also become a secondary resource in countries with water shortages. Future trends in wastewater technology will include buildings where black water and gray water are separated, recycling the former for fertilizer and the latter for toilets. This will permit smaller treatment plants with reduced nutrient loads and concurrently lower emissions of CH_4 and N_2O. Other trends include the increased use of ozone and UV light for wastewater purification.

REFERENCES

Ackerman, F., 2000: Waste Management and Climate Change, Local Environment **5**(2), 223-229.

Austrian Federal Government, 2001:Third National Climate Report of the Austrian Federal Government, Vienna, Austria.

Barlaz, M., 1998: Carbon storage during biodegradation of municipal solid waste components in laboratory-scale landfills, Global Biogeochemical Cycles, **12**(2), 373-380.

Bates, J., and A. Haworth, 2001: *Economic Evaluation of Emission Reductions of Methane in the Waste Sector in the EU: Bottom-up Analysis,* Final Report to DG Environment, European Commission by Ecofys Energy and Environment, by AEA Technology Environment and National Technical University of Athens as part of Economic Evaluation of Sectoral Emission Reduction Objective for Climate Change, 73 pp.

Beck-Friis, B.G. 2001: *Emissions of ammonia, N_2O, and CH_4 during composting of organic household waste.* PhD Thesis, Swedish University of Agricultural Sciences, Uppsala, 331 pp.

Bingemer, H.G., and P.J. Crutzen, 1987: The production of CH_4 from solid wastes, J. Geophysical Research., **92**(D2), 2182-2187.

Binner, E. 2002: The impact of mechanical-biological pretreatment on the landfill behaviour of solid wastes. Workshop on Biowaste. Brussels, April 8-10, 2002. 16 pp.

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

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

Bogner, J. and E. Matthews, 2003: Global Methane Emissions from Landfills: New Methodology and Annual Estimates 1980-1996, Global Biogeochemical Cycles **17**, 34-1 to 34-18.

Bogner, J., C. Scheutz, J. Chanton, D. Blake, M. Morcet, C. Aran, and P. Kjeldsen, 2003: Field measurement of non-methane organic compound emissions from landfill cover soils, Proceedings Sardinia '03 International Solid and Hazardous Waste Symposium, published by CISA, Univ. of Cagliari, Sardinia.

Bogner, J., and K. Spokas, 1993: Landfill CH_4: rates, fates, and role in global carbon cycle, Chemosphere, **26**(1-4), 366-386.

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

Bogner, J., K. Spokas and E. Burton E., 1999: Temporal variations in greenhouse gas emissions at a midlatitude landfill. Journal of Environmental Quality, **28**, 278-288.

Bogner, J., K. Spokas, E. Burton, R. Sweeney, and V Corona, 1995: Landfills as atmospheric methane sources and sinks, Chemosphere, **31**(9), 4119-4130.

Bogner, J., K. Spokas, J. Chanton, D. Powelson, and T. Abichou, 2005: Modeling landfill methane emissions from biocovers: a combined theoretical-empirical approach, Proc Sardinia '05 International Solid and Hazardous Waste Symposium, October, 2005, published by CISA, Univ. of Cagliari, Sardinia.

Borjesson, G., 1996: Methane oxidation in landfill cover soils, Doctoral Thesis, Dept. of Microbiology, Swedish University of Agricultural Sciences, Uppsala, Sweden,

Borjesson G. and B. Svensson 1997a: Nitrous oxide release from covering soil layers of landfills in Sweden, Tellus B, **49**, 357-363, 1997.

Borjesson, G., and B. Svensson, 1997b: Seasonal and diurnal methane emissions from a landfill and their regulation by methane oxidation, Waste Management and Research, **15**(1), 33-54.

Burnley, S., 2001: The impact of the European landfill directive on waste management in the United Kingdom, Resources, Conservation and Recycling, **32**, 349-358.

Bringezu, S., H. Schutz, S.Steger, and J. Baudisch, 2004: International comparison of resource use and its relation to economic growth: the development of total material requirement, direct material inputs and hidden flows and the structure of TMR, Ecological Economics, **51**, 97-124.

CALEB, 2000: Development of a global emission function for blowing agents used in closed cell foam. Final report prepared for AFEAS. Caleb Management Services, Bristol, United Kingdom.

Chanton J. , and K. Liptay, 2000: Seasonal variation in methane oxidation in a landfill cover soil as determined by an i*n situ* stable isotope technique, Global Biogeochemical Cycles, **14**, 51-60.

Christiansen, T., 1989: Environmental aspects of sanitary landfilling, in Sanitary Landfilling, Christensen, T., *et al.* (ed.), Academic Press, San Diego, pp.19-29.

Cointreau-Levine, S., 1994: Private Sector Participation in Municipal Solid Waste Services in Developing Countries, Vol. 1 The Formal Sector. Urban Management and the Environment, 13, UNDP/ UNCHS(United Nations Centre for Human Settlements), World Bank, Washington, DC, 52 pp.

Commission of the European Community, 2001: Third Communication from the European Community under the UN Framework Convention on Climate Change, Brussels.

Czepiel, P., B. Mosher, R. Harriss, J.H. Shorter, J.B. McManus, C.E. Kolb, E. Allwine, and B. Lamb, 1996; Landfill methane emissions measured by enclosure and atmospheric tracer methods, J. Geophysical Research, **D101,** 16711-16719.

Daskalopoulos E., O. Badr, and S.D. Probert, 1998: Municipal solid waste: a prediction methodology for the generation rate and composition in the European Union countries and the United States of America, Resources, Conservation and Recycling, **24**, 155-166.

de Jager, D. and K. Blok, 1996: Cost-effectiveness of emission-reduction measures for CH_4 in the Netherlands, Energy Conserv. Management, **37**, ?pp.

Delhotel, C., 2005: Methane and Nitrous Oxide from Non-Agricultural Sources, Chapter 3, International Non-CO_2 Greenhouse Gas Marginal Abatement Report, U.S. EPA Office of Air and Radiation (Draft).

DESA (Dept. of Economic and Social Affairs)/UN, 2005: Country, Regional and Global Estimates on Water and Sanitation, Statistics Division, MDG Report - WHO/UNICEF WES estimate, http://www.unicef.org/wes/mdgreport/who_unicef_WESestimate. pdf (updated 10Nov04).

Detzel, A., R. Vogt, H. Fehrenbach, F. Knappe and U. Gromke, 2003. Anpassung der deutschen Methodik zur rechnerischen Emissionsermittlung and internationale Richtlinien: Teilberich Abfall/Abwasser. IFEU Institut - Öko-Institut e.V. 77 p.

de Visscher, A., D. Thomas, P. Boeckx, and O. Van Cleemput, 1999: Methane oxidation in simulated landfill cover soil environments, Environmental Science and Technology, **33**(11), 1854-1859.

de Visscher A., I. De Pourcq, and J. Chanton , 2004: Isotope fractionation effects by diffusion and methane oxidation in landfill cover soils, Journal of Geophysical Research, **109** (D18), paper D18111.

Dinda, S., 2004: Environmental Kuznets curve hypothesis: a survey, Ecological Economics, **49**, 431-455.

EEA(European Environment Agency), 2004: Greenhouse gas emission trends and projections in Europe 2004. EEA Report no5/2004, Progress EU and its member states towards achieving their Kyoto targets, Luxembourg, pp., ISSN 1725 -9177, 40 pp.

EEA(European Environment Agency), 2005: European Environment Outlook, EU Report 4/2005, ISSN 1725-9177, Luxembourg, published by EEA, Copenhagen, 92 pp.

Eurostat, 2003**:** Waste generated and treated in Europe. Data 1990 - 2001. Office for Official Publications of the European Communities, Luxemburg, 140 pp.

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

Flugsrud, K., B. Hoem, E. Kvingedal and K. Rypdal. 2001: Estimating net emissions of CO_2 from harvested wood products. SFT report 1831/200. Norwegian Pollution Control Authority, Oslo, 47 pp. http://www.sft.no/publikasjoner/luft/1831/ta1831.pdf

Fullerton, D. and T.C. Kinnaman, 1996: Household responses to pricing garbage by the bag, American Economic Review, **86**(4), 971-984.

Galle, B., J. Samuelsson, B. Svensson, and G. Borjesson, 2001: Measurements of methane emissions from landfills using a time correlation tracer method based on FTIR absorption spectroscopy, Environmental Science and Technology, **35**(1), 21-25.

Gamlen, P.H., B.C. Lane, and P.M. Midgley, 1986: The production and release to the atmosphere of CCl_3F and CCl_2F_2 (Chlorofluorocarbons CFC-11 and CFC-12), Atmospheric Environment, **20**(6), 1077-1085.

Gebert, J., and A. Gröngröft, 2005: Microbial Oxidation of Landfill Methane in a Biofilter System - Passive Ventilation and Oxygen Supply, Proceedings Sardinia '05 International Solid and Hazardous Waste Symposium, October, 2005, CISA, Univ. of Cagliari, Sardinia.

Government of Japan, 1997: Japan’s Second National Communication to the UNFCCC, Tokyo.

Government of Japan, 2002: Japan’s Third National Communication to the UNFCCC, Tokyo.

Government of Norway, 2002: Norway’s Third National Communication to the UNFCCC, Oslo.

Government of the Netherlands, 2001: Third Netherlands National Communication on Climate Change Policies, The Hague.

Government of Republic of Poland, 2001: Third National Communication to the COP of the UNFCCC, Warsaw.

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

Hanisch, C., 2000: Is extended producer responsibility effective? Environmental Science & Technology, **34**(7), 170A-175A.

Haury, E.W., 1976: The Hohokam: Desert Farmers and Craftsmen, University of Arizona Press, Tucson, Arizona.

Hellebrand, H.J. 1998: Emissions of N_2O and other trace gases during composting of grass and green waste. J. Agric. Engineering Research, **69**, 365-375.

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

Houghton, J.T., Filho. L., Griggs, D., and K. Maskell, 1997: An Introduction to Simple Climate Models used in the IPCC Second Assessment Report, IPCC Technical Paper, UNEP/WMO.

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

Huber-Humer M. , 2004: Abatement of Landfill Methane Emissions by Microbial Oxidation in Biocovers made of Compost, PhD Thesis, University of Natural Resources and Applied Life Sciences (BOKU), Vienna, 279 pp.

Huhtala, A. 1997: A post-consumer waste management model for determining optimal levels of recycling and landfilling. Environmental and Resource Economics, 10, 301-314.

IPCC, 2001a: Climate Change 2001: Mitigation (Contribution of Working Group III to the third assessment report of the Intergovernmental Panel on Climate Change). Cambridge: Cambridge University Press.

IPCC, 2001b: Climate Change 2001: The Scientific Basis (Contribution of Working Group I to the third assessment report of the Intergovernmental Panel on Climate Change). Cambridge: Cambridge University Press.

Iskandar, L., 2001, The Informal Solid Waste Sector in Egypt: Prospects for Formalization, published by Community and Institutional Development, Cairo, Egypt, 65 pp.

Jenkins, R.R., S.A. Martinez, K. Palmer, and M.J. Podolsky, 2003: The determinants of household recycling: a material-specific analysis of recycling program features and unit pricing, Journal of Environmental Economics and Management, 45(2), 294-318.

Jouravlev, A., 2004: Los servicios de agua potable y saneamiento en el umbral del siglo XXI, Santiago de Chile, julio 2004, CEPAL – Serie Recursos Naturales e Infraestructura No 74.

Kaartinen, T. 2004: Sustainable disposal of residual fractions of MSW to future landfills. M.S.Thesis, Technical University of Helsinki, Espoo, Finland. In Finnish.

Kightley, D., D. Nedwell, and M. Cooper, 1995: Capacity for methane oxidation in landfill cover soils measured in laboratory-scale microcosms, Applied and Environ. Micro**., 61**, 592-601.

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

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

Konte, 2005: UNFCCC Metadata, UNEP/DEWA/GRID, Geneva, Switzerland. http://ghg.unfccc.int.

Lohiniva, E., K. Sipilä, T. Mäkinen and L. Hietanen, 2002: Waste-to-energy and greenhouse gas emissions. Espoo, VTT - Tiedotteita - Research Notes 2139, 119p. In Finnish (to be published in English in 2006)

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

Micales, J.A. and K.E. Skog, 1997: The decomposition of forest products in landfills, International Biodeterioration and Biodegradation, **39**(2-3), 145-158

Miranda, M.L., J.W. Everett, D. Blume, and B.A. Roy, 1994: Market-based incentives and residential municipal solid waste, Journal of Policy Analysis and Management, **13**(4), 681-698.

Miranda, M.L., S.D. Bauer, and J.E. Aldy, 1996: Unit pricing programs for residential municipal solid waste: an assessment of the literatures, Report prepared for U.S. Environmental Protection Agency, Washington D.C.

Morris, J., 2001: Effects of waste composition on landfill processes under semi-arid conditions, PhD Thesis, Faculty.of Engin.,Univ. of the Witwatersrand, Johannesburg, S. Africa. 1052 pp.

Moriis, J.R., P.S. Phillips, and A.D. Read, 2000: The UK landfill tax: financial implications for local authorities, Public Money and Management, **20**(3), 51-54.

Mosher, B., P. Czepiel, R. Harriss, J. Shorter, C. Kolb, J.B. McManus, E. Allwine, and B. Lamb, 1999: Methane emissions at nine landfill sites in the northeastern United States, Environmental Science and Technology, **33**(12), 2088-2094.

Myllymaa, T., H. Dahlbo, M. Ollikainen, S. Peltola and M. Melanen, 2005: A method for

implementing life cycle surveys of waste management alternatives' environmental and cost effects. Helsinki, Suomen Ympäristö - Finnish Environment 750, 108 pp.
Nakicenovic, N., J. Alcamo, G. Davis, B. de Vries, J. Fenhann, S. Gaffin, K Gregory, A. Grübler, T-Y. Jung, T. Kram, E.L. La Rovere, L. Michaelis, S. Mori, T. Morita, W. Pepper, H. Pitcher, L. Price, K. Riahi, A. Roehrl, H.-H. Rogner, A. Sankovski, M. Schlesinger, P. Shukla, S. Smith, R. Swart, S. van Rooijen, N. Victor, and Z. Dadi, 2000: Special Report on Emissions Scenarios, Intergovernmental Panel on Climate Change (IPCC), http://www.grida.no/climate/ipcc/emission/ index.html.
Nozhevnikova, A.N., A.B. Lifshitz, V.S. Lebedev, and G.A. Zavarin, 1993: Emissions of methane into the atmosphere from landfills in the former USSR, Chemosphere, 26(1-4), 401-417.
OECD, 2001: Extended Producer Responsibility: A Guidance Manual for Governments, OECD publishers, Paris.
OECD, 2002: Environmental Data. Waste. Compendium 2002. Environmental Performance and Information Division, OECD Environment Directorate. Working Group on Environmental Information and Outlooks. 27 pp.
OECD, 2003: OECD Environmental Data Compendium 2002, Paris, www.oecd.orgn
Penman, J., D. Kruger. I. Galbally T. Hiraishi, B. Nyenzi, S. Enmanuel, L. Buendia, R. Hoppaus, T. Martinsen , J. Meijer, K. Miwa, and K Tanabe (eds), 2000: Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories. IPCC/OECD/IEA/IGES, Hayama, Japan.
Penman, J., M. Gytarsky, T. Krug, D. Kruger. T. Hiraishi, R. Pipatti, L. Buendia, K. Miwa, T. Ngara, K. Tanabe and F. Wagner (Eds), 2003: Good Practice Guidance for Land Use, Land-Use Change and Forestry, Institute for Global Environmental Strategies (IGES), Hayama, Japan.
Petersen, S.O., A.M. Lind and S. G. Sommer, 1998: Nitrogen and organic matter losses during storage of cattle and pig manure. J. Agric. Sci., **130**, 69-79.
Pingoud, K., A-L. Perälä, S. Soimakallio and A. Pussinen, 1996: Greenhouse impact of the Finnish forest sector including forest products and waste management. Ambio, **25**, 318-326.
Pipatti, R., and M. Wihersaari, 1998: Cost-effectiveness of alternative strategies in mitigating the greenhouse impact of waste management in three communities of different size. Mitigation and Adaption Strategies for Global Change **2**, 337 -358.
Pipatti, R., and I. Savolainen, 1996, Role of energy production in the control of greenhouse gas emissions from waste management, Energy Conservation Management, **37**(6-8), 1105-1110.
Pipatti, R., and Vieira, S., 2005: Waste and Wastewater, First Order Draft (FOD) 2006 IPCC Guidelines for Greenhouse Gas Inventories.
PNUD(Programa de las Naciones Unidas)/UNDP, 2005: Informe sobre Desarrollo Humano 2005: la cooperación internacional ante una encrucijada, Cuadro 7 – Agua, Saneamiento y Nutrición, publicado para el Programa de las Naciones Unidas para el Desarrollo, Ediciones Mundi – Prensa 2005, Barcelona, Spain. http://hdr.undp.org/statistics
Price, J.L., 2001: The landfill directive and the challenge ahead: demands and pressures on the UK householder, Resources, Conservation and Recycling, **32**, 333-348.
Rathje, W.L., W.W. Hughes, D.C. Wilson, M.K. Tani, G.H. Archer, R.G. Hunt, and T.W. Jones, 1992: The archaeology of contemporary landfills, American Antiquity, **57**(3): 437-447.
Richards, K., 1989: Landfill gas: working with Gaia, Biodeterioration Abstracts, **3**(4), 317-331.
Rinne, J., M. Pihlatie, A. Lohila, T. Thum, M. Aurela, J-P Tuovinen, T. Laurila, and R.Vesala, 2005: N_2O emissions from a municipal landfill, Environmental Science and Technology, **39**, 7790-7793.
Ritzkowski, M., and Stegman, R., 2005: Reduction of GHG Emissions by Landfill In Situ Aeration, Proceedings Sardinia '05 International Solid and Hazardous Waste Symposium, October, 2005, published by CISA, Univ. of Cagliari, Sardinia.

Savage, G., L. Diaz, C. Golueke, C. Martone, and R.Ham, 1998: Guidance for Landfilling Waste in Economically Developing Countries, ISWA Working Group on Sanitary Landfilling, published under U.S. EPA Contract 68-C4-0022. ISBN No. 87-90402-07-3, 300 pp.

Scheehle, E. and M. Doorn, 2002, Improvements to the U.S. Wastewater CH_4 and N_2O emissions estimates, U.S. EPA report June 2002.

Scheehle, E. and D. Kruger, D, 2005: Global anthropogenic methane and nitrous oxide emissions, Energy Journal, (in review).

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

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

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

Silva, G., 1998: Private Participation in the Water and Sewerage Sector – Recent Trends, in Public Policy for the Private Sector(Viewpoint): Water, World Bank, Washington D.C. pp.12-20.

Spokas, K., J. Bogner, J. Chanton, M. Morcet, C. Aran, C. Graff, Y. Moreau-le-Golvan, N. Bureau, and I. Hebe, 2005: Methane mass balance at three landfill sites: what is the efficiency of capture by gas collection systems? Waste Management, (in press).

Statistics Finland, 2005: Environment Statistics. Environment and Natural Resources 2005:2, Helsinki, 208 pp.

Streese, J., and Stegman, R., 2005: Potentials and Limitations of Biofilters for Methane Oxidation, Proceedings Sardinia '05 International Solid and Hazardous Waste Symposium, October, 2005, CISA, Univ. of Cagliari, Sardinia.

TFFEoL (Task force on Foam End-of-Life Issues), 2005: Report of the UNEP task force on foam end-of-life issues. Task force on Foam End-of-Life Issues, May 2005, Montreal Protocol on Substances that Deplete the Ozone Layer. UNEP Technology and Economic Assessment Panel, 113 pp.

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

Tsujimoto Y., J. Masuda, J. Fukuyama, and H. Ito, 1994: N_2O emissions at solid waste disposal sites in Osaka City, Air Waste, **44**, 1313-14..

Tuhkanen, S., A. Lehtilae, and I. Savolainen, 1999: The role of CH_4 and N_2O emission reductions in the cost-effective control of the greenhouse gas emissions from Finland, Mitigation and Adaptation Strategies for Global Change, **4** (2), 91-111

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

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

UNFCCC/IPCC, 2004: Historical Data and Projections of Greenhouse Gas Emissions – CH_4 and N_2O.

US EPA, 2001: Non-CO_2 Greenhouse Gas Emissions from Developed Countries: 1990-2010, EPA-430-R-01-007, Office of Air and Radiation, Washington, DC.

U.S. Department of State, 2002: U.S. Climate Action Report 2002, Washington, D.C.

Vesterinen, R. 1996: Greenhouse gas emissions from composting. SIHTI-Research Programme, Seminar 13-14-3-1996. Hanasaari, Espoo, Finland. 3 pp. In Finnish.

WHO/UNICEF, 2005: Joint Monitoring Programme: Worldwide Sanitation, http://www.who.int/mdg/publications/MDG_Report_08_2005.pdf/mdg_report/en/index

WHO/UNICEF/WSSCC, 2000: Global Water Supply and Sanitation Assessment: 2000 Report, WHO/UNICEF Join Monitoring Programme for Water Supply and Sanitation, ISBN924156202 1, 2000. http://www.who.int

Willumsen, H.C., 2003, Landfill gas plants: number and type worldwide, Proc. Sardinia '05 International Solid and Hazardous Waste Symposium, October, 2005, published by CISA, Univ. of Cagliari, Sardinia.

World Bank, 1997: Toolkits for private sector participation in water and sanitation, World Bank, Washington D.C.

World Bank, 1999: What a Waste: Solid Waste Mangement in Asia. World Bank, Washington, DC. 43 pp.

World Bank, 2005: World Development Indicators 2005:Environment, Table 2.15 Disease Prevention: Coverage and Quality, Table 3.11: Urban Environment; Table 3.6 Water pollution, http://www.worldbank.org/data/wdi2005/wditext

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