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The Climate Change Mitigation Potential of the Waste Sector

Illustration of the potential for mitigation of greenhouse gas emissions from the waste sector in OECD countries and selected emerging economies; Utilisation of the findings in waste technology transfer

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The Climate Change Mitigation Potential of the Waste Sector

Illustration of the potential for mitigation of greenhouse gas emissions from the waste sector in OECD countries and selected emerging economies; Utilisation of the findings in waste technology transfer

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Abstract

This study presents the greenhouse gas (GHG) mitigation potential of municipal solid waste (MSW) management in OECD countries as well as India and Egypt. Three detailed GHG balances for the USA, India, Egypt and one balance for the OECD countries are elaborated applying the life cycle assessment (LCA) method according to ISO 14040/14044 for waste management. For each balance the respective status quo is determined and compared with two scenarios to 2030. The methodology as well as the underlying data and assumptions were profoundly discussed at workshops with LCA experts and local stakeholders. A GHG calculation approach was developed, which uses harmonised emission factors to credit avoided emissions from material recycling. With regard to the status quo, the net results for the OECD countries, the USA, India and Egypt show that methane emissions from landfilling are the main contributor to the GHG burdens. Only OECD countries with little or no landfill of (organic) waste achieve a net credit (e.g. Japan). These credits are the more evident the higher recycling rates are and the more efficient energy recovery is. The findings of this study were presented in May 2014 at the environmental fair IFAT in Munich.

The study's most important conclusion is that the potential for GHG mitigation in waste management is significant. However, further incentives are necessary to support developing countries as well as some OECD and/or EU countries to develop an integrated closed-cycle waste management system. With regard to the EU, targets to promote diversion of biodegradable waste from landfill and for further development of recycling are important steps in the right direction. For emerging and developing countries the integration of the informal sector in future MSW concepts should be taken into account.

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Abbreviations

CDM	Clean Development Mechanism
CHP	combined heat and power, cogeneration
DOC	degradable organic carbon
DOCf	fraction of DOC dissimilated
EEAA	Egyptian Environmental Affairs Agency
ERC	Energy Recovery Council
EU-OECD countries	The 21 members of the EU28 that are also members of the OECD
EU ETS	EU Emissions Trading System
Fe metals	ferrous metals
GCA	Greater Cairo Area (governorates of Cairo, Giza and Qalyubia)
GDP	gross domestic product
GHG	greenhouse gas
GIZ	Gesellschaft für internationale Zusammenarbeit
IPCC	Intergovernmental Panel on Climate Change
JPN	Japan
KOR	South Korea
MBS	mechanical-biological stabilisation
MBT	mechanical-biological treatment
MCF	methane correction factor
Mg	megagram, equivalent to one metric ton
MSW	municipal solid waste
MSW-DST	MSW – Decision Support Tool (of USEPA ORD)
MSWI	municipal solid waste incinerator, municipal solid incineration
MWC	municipal waste combustion
NAMA	Nationally Appropriate Mitigation Actions
NIR	National Inventory Report
non-Fe metals	non-ferrous metals
OECD	Organisation for Economic Co-operation and Development
OX	oxidation factor
PoA	Programmes of Activities
RDF	refuse-derived fuel
RTO	regenerative thermal oxidation

SOG survey	State of Garbage in America (EEC 2014)
TMP	thermochemical pulp
UNFCCC	United Nations Framework Convention on Climate Change
USEPA	United States Environmental Protection Agency
USEPA ORD	US EPA, Office of Research and Development
USEPA OSWER	US EPA, Office of Solid Waste and Emergency Response
WARM	Waste Reduction Model (of USEPA OSWER)
YWCF	yard waste composting facilities

1 Summary

The potential for climate change mitigation in the waste management sector is substantial. It can contribute appreciably to national greenhouse gas (GHG) mitigation goals. This has already been examined and shown in previous studies for Germany, the EU27 and in a first approach for selected developing countries and emerging economies using the life cycle assessment (LCA) method (Öko-Institut/IFEU 2005, 2010). The present study determines the mitigation potential for the OECD countries using the LCA method, with an in-depth analysis of the USA. In addition, the study provides a more detailed analysis of the GHG mitigation potentials of India and Egypt as selected developing countries and emerging economies.

In the context of national reporting commitments under the Kyoto Protocol, only direct GHG emissions are considered in the waste sector, and only those from treatment options without energy recovery. In the National Inventory Reports (NIR), effects from energy use or recycling activities are reported in the energy or industry sector. In contrast to this, the LCA method makes it possible to assess the full effects of waste management activities. The LCA method considers both the direct emissions (debits) from waste treatment and the avoided emissions (benefits, credits) resulting from secondary products or energy generation. The system boundary starts with the waste generated. It thus considers the fate of non-collected waste, if present to a relevant extent, as is the case e.g. in India and Egypt. For collected waste, collection, treatment and recycling to produce secondary products or recover energy are included. The benefit from secondary products in the form of the potential to substitute primary products and conventional energy in other sectors is taken into account via credits in the GHG balance (offsets). In contrast to National Inventory Reports, the LCA method thus allows analysis and assessment of optimisation potentials in the waste sector, providing orientation for decision-makers. The results represent statements of potentials.

The methodological approach follows ISO 14040/14044. For waste management there are some specifics. For example, the system boundaries start with the waste generated and end with final disposal or production of secondary products instead of “cradle-to-grave”. In addition, the waste sector is typically a multifunctional system. Apart from the main function of disposal of a particular quantity of waste, there are usually additional benefits from the production of secondary products and energy. The ISO standard offers general guidelines for multifunctional systems without further concretisation. In system comparisons, additional benefits from secondary products or recovered energy are usually considered by taking into account substitution processes, accounted for as credits. If avoided emissions through substitution are higher than the direct emissions from waste treatment the net results are negative values (“savings or mitigation potential”). This is to be understood as GHG mitigation potentially taking place in other sectors, namely the energy or industry sector.

The offsetting of additional system benefits is required for system comparison, e.g. between status quo and future scenarios, in order to establish equal benefits among the systems. However, no specifications as to how substitution processes should be selected are available. This and further methodological questions were discussed with international experts at a methodology workshop in Berlin on 18 June 2012. The goal of that exchange was to improve the comparability and transparency of LCA studies in order to strengthen their suitability as decision-support tools for politics and for planning. The experts agreed that the choice of substitution processes should be limited as it has a major influence on the final results. It was agreed that harmonised emission factors should be used for studies designed to identify mitigation potentials. For material recycling the technical substitution potential is applicable.

Taking into account the market-related substitution potential would be contradictory because this would mean "the more you substitute the less credit you get".

The methodological remarks underscore the fact that mitigation potentials in waste management should not be misunderstood as exact GHG reductions but rather as potentials revealing important methods, options and actions which can significantly contribute to GHG mitigation.

As an outcome of the workshop, consistent emission factors for material recycling were established (see Sections 4.2.4 and 11.1) and used in this study. For energy recovery the marginal substitution approach (substitution of fossil fuels) was used in the detailed country balances (USA, India, Egypt). In the OECD balance emission factors for the national electricity grid were used instead, as valid data were available for these.

Data situation for the OECD countries, EU28 and USA

The presentation of waste management in the OECD countries and the EU28 is based on statistical data of the OECD and Eurostat. These data, which vary in quality, could not be verified and scrutinised for all of the 34 OECD countries and/or 28 EU countries. Some data important for the balance that are not contained in the statistics were taken instead from national publications. Often, though, these gaps had to be closed by plausible assumptions.

For the separate USA balance, publications by USEPA were analysed comprehensively. This enabled a relatively high degree of accuracy to be achieved. Nevertheless, uncertainties also exist with the data given by USEPA. These were identified and examined in sensitivity analyses. Data from the USA balance were used in the OECD balance, but for symmetry reasons not in the same degree of detail as in the USA balance.

OECD balance

The municipal solid waste (MSW) streams in the 34 OECD member states are captured by statistical data from Eurostat and the OECD for the time period 2008 to 2010. In the OECD region it is assumed that the waste generated is equal to the waste treated; i.e. that there are no noteworthy amounts of non-collected or non-treated MSW in this region. Eurostat and OECD do not provide recycling rates per waste fraction. These had to be determined from national information and other assumptions. Information on the waste composition or the state of technology of waste treatment options was also extracted from national data wherever possible, although these partly refer to more distant time horizons.

In many cases, plausible assumptions had to be made. Thus the waste characteristics (calorific value, carbon content) and the waste composition for Germany were used for the EU28, as in the previous study (Öko-Institut/IFEU 2010). For waste incineration the efficiency ratios of thermal recovery for the EU from (CEWEP 2012) were used for all OECD countries with the exception of the USA. For collected landfill gas, 50% use in combined heat and power (CHP) plants and 50% flaring was assumed, following the findings for the USA.

The analysis distinguishes the three regions of "America", "Europe, Turkey and Israel", and "Japan, South Korea and Pacific", in the style of (OECD 2012). In total 649 Mt MSW (521 kg/(cap*a)) were treated in the 34 OECD countries. The amounts in each of the three analysed regions and the percentages of different waste management methods are shown in Table 1.

Table 1: Waste amounts and management methods in OECD countries

	America	Europe, Turkey and Israel	Japan, South Korea and Pacific	OECD total
Waste amount treated in 1,000 tonnes	291,508	263,893	85,339	**640,740**
Waste amount treated in kg/(cap*a)	607	469	421	**514**
Recycling in %	24%	25%	31%	**25%**
Composting in %	8%	13%	0.3%	**9%**
Incineration (no energy) in %	0.1%	3%	4%	**2%**
Incineration (with energy) in %	9%	20%	48%	**18%**
Landfill in %	60%	38%	17%	**45%**

Deviation from 100% total waste treatment results from mixed waste composting not listed in the table

Overall in the OECD countries the largest proportion of MSW is landfilled. In the “America” region landfilling also predominates, while in “Europe, Turkey and Israel” about the same amount of MSW is landfilled as is recycled/composted. In “Japan, South Korea and Pacific” waste incineration with energy recovery (“with energy”) is the dominant waste management method due to the conditions in Japan and in South Korea. Waste incineration without energy recovery (“no energy”) only takes place to a limited extent, in some EU countries and in Japan and South Korea.

The GHG balance of waste treatment in the OECD countries leads to a net GHG debit of about 66 Mt CO_2-eq (Table 2). This is due above all to landfilling with the associated methane emissions from biological degradation of the organic fraction in the waste. This includes the savings from landfill gas collection and 50% use in small-scale CHP units for energy recovery. Based on the National Inventory Report of the EU28, the average gas collection efficiency for the EU countries was calculated to be 34.6%, assuming that the maximum national collection efficiency is 50%. This cap on the maximum gas collection efficiency was set uniformly (also for the USA) as it is assumed to be the maximum technically possible gas collection for the overall landfill period (100 year time horizon). The gas collection efficiency for non-EU-OECD countries was extracted from national data wherever possible. Otherwise the calculated average value for the EU-OECD countries was used. The outcome of this is the following gas collection efficiency by region:

“America”:	43.7%
“Europe, Turkey and Israel”:	31.0%
“Japan, South Korea and Pacific”:	16.0%[1]
OECD total:	37.9%

Because only 17% MSW is landfilled in the “Japan, South Korea and Pacific” region, this is the only region to show a net credit (negative value) in the GHG balance. The contribution of waste incineration with energy recovery varies by country depending on the national electricity grid

[1] In place of gas collection Japan relies on ventilation to reduce methane emissions; this is taken into account via the methane correction factor in the GHG balance.

used to calculate the avoided emissions. In countries with a high share of renewable energy or nuclear power plants the net result can be a net debit even for incineration with energy recovery, as less fossil fuel is substituted (e.g. Switzerland, Norway).

Table 2: Absolute net global warming potential (GWP) results in the OECD countries

in 1,000 t CO_2-eq	America	Europe, Turkey and Israel	Japan, South Korea and Pacific	OECD total
Collection, sorting, transport*	6,041	5,094	2,271	**13,407**
Landfill	122,336	81,904	13,122	**217,362**
Incineration (no energy)	77	3,226	1,183	**4,486**
Incineration (with energy)	-574	-3,234	-2,241	**-6,049**
Recycling	-67,764	-74,107	-21,642	**-163,514**
Composting	174	291	50	**466**
Total**	**60,323**	**13,339**	**-7,305**	**66,358**

* Collection, sorting and transport were calculated identically for all OECD countries
**Results for residual-waste composting are included in the net result but are not listed separately, due to small quantities

From a climate protection point of view, there is relevant optimisation potential for waste management in OECD countries. This was analysed in two scenarios to 2030 – a medium and an ideal one. The optimisation mainly addresses the steering of waste streams, as well as some technical optimisations such as the assumed increase of energy recovery efficiency for waste incineration.

In the medium scenario it is assumed that the waste amount landfilled can be cut by half. All landfills are equipped with a gas collection system; this increases the weighted average gas collection efficiency in OECD countries from 37.9% to 50% (maximum value, see above). Some of the amount no longer landfilled is treated via material recycling. 80% of the remaining residual waste is treated in municipal solid waste incinerators (MSWI) with energy recovery and 20% in anaerobic mechanical-biological treatment plants (MBT). Treatment via MBT results in a refuse-derived fuel (RDF) fraction, one half of which is used in RDF plants and one half in coal power plants or cement kilns. The residual waste remaining at that stage is treated biologically, generating a stabilised MBT residue which is landfilled with substantially reduced methane generation. The assumption of treatment via MBT simplifies the phasing out of direct landfilling of MSW because existing landfill capacities can continue to be used.

In the ideal scenario, as for all balances, complete diversion from direct landfilling is assumed. The no longer landfilled amount is partly treated via material recycling and the remaining waste is treated as described for the medium scenario. However, in the ideal scenario it is assumed that the 20% of residual waste is not consigned to MBT as in the medium scenario but to mechanical-biological stabilisation (MBS). The main purpose of MBS is to produce RDF, which again is assumed to be used one half each in RDF cogeneration plants and in coal power plants or cement kilns. Only an inert fraction separated from the stabilised material is still landfilled.

The result for the scenarios compared to the status quo (business as usual, BAU) is shown in Table 3. With reduction of or complete diversion from direct landfilling, and material recycling and use with energy recovery as alternative waste treatment methods, significant GHG mitigation can be achieved. Even the medium scenario achieves a net credit in the OECD balance. For “Japan, South Korea and Pacific” the net credit increases; in the other two regions the net debit of the BAU scenario changes into a net credit, with the highest mitigation effect being achieved in the “America” region. The ideal scenario leads to a further significant increase in net credits. Overall, under the assumed general conditions and optimisations, a net credit of some -155 Mt CO_2-eq in the medium scenario and of some -287 Mt CO_2-eq in the ideal scenario is achieved in the OECD.

Table 3: Absolute net GWP results, status quo and scenarios to 2030, in the OECD countries

in 1,000 t CO_2-eq	America	Europe, Turkey and Israel	Japan, South Korea and Pacific	OECD total
Status quo (BAU)	60,323	13,339	-7,305	**66,358**
Medium scenario	-57,195	-69,696	-27,572	**-154,646**
Ideal scenario	-144,143	-108,475	-34,387	**-286,906**

EU28 balance

Not all European countries are OECD member states. Therefore an additional calculation was performed for the EU28. For this the statistical data for the seven non-OECD-EU countries were collected additionally. According to Eurostat, waste generation in the EU28 totalled about 240 Mt (476 kg/(cap*a)). The total waste was treated as follows:

34% landfilling
20% incineration with energy recovery
27% recycling
15% composting
4% incineration without energy recovery

For the status-quo of waste management in the EU28 the GHG balance shows a net credit of about -8 Mt CO_2-eq. Although the result is not directly comparable to the previous study (Öko-Institut/IFEU 2010) due to differing amounts of total MSW (260 Mt in 2007), there are signs of a qualitative improvement, because less MSW is landfilled, gas collection efficiency is higher and higher savings are achieved through materials recycling.

A relevant GHG mitigation can be obtained in the two scenarios. The medium scenario results in a net credit of about -65 Mt CO_2-eq, the ideal scenario in a net credit of about -100 Mt CO_2-eq. Overall, under the assumed general conditions and optimisations, the ideal scenario leads to a GHG mitigation potential of 92 Mt CO_2-eq.

USA balance

Data on waste generation and treatment were taken from statistical information from USEPA for the year 2011, given in US short tons (USEPA 2013a, b). To allow comprehensive traceability of the data, the values have not been converted into metric tons. All waste amounts in this section are given in short tons; for precise differentiation metric tons are given in megagrams (Mg).

In addition, it should be noted that the results here are not directly comparable with those for the USA in the OECD balance as it was possible to calculate the USA balance to a higher degree of accuracy. For example, waste amounts which are usually not considered as MSW (old tyres, lead from lead-acid batteries) were excluded from the mass balance and/or the inventory. On account of national conditions in the USA, the oxidation rate for landfill gas generated was set to 10% , while it was symmetrically and conservatively set to 0% (IPCC default value) for all countries in the OECD balance because the actual situation could not be determined for all 34 member states. Another important difference is that in the USA balance energy generated from waste treatment is offset using the marginal approach instead of the country-specific electricity mix as in the OECD balance. The marginal electricity for the USA is electricity from coal.

In total about 250 million short tons of waste were treated in the USA in 2011. This is equivalent to about 225 million Mg (721 kg/(cap*a)). The total waste was treated as follows:

54% landfilling
11% incineration with energy recovery
27% recycling
8% composting

The vast majority of the waste was landfilled. US landfills are widely equipped with gas collection systems. The landfill gas collection efficiency for the overall landfilling period (100 year time horizon) was set to 50% in the calculations (general cap, see above), even though US landfill operators quote significantly higher landfill gas collection efficiencies and higher values and measurements are also given in the literature and in USEPA publications. These gas collection efficiency data relate to the gas collection phase and cannot be claimed to be valid for the overall landfill period. According to the National Inventory of GHG Emissions (USEPA 2012b), approximately half of the landfill gas collected is used with energy recovery. In the balance, use in a CHP plant was assumed.

Most waste-to-energy (WtE) plants in the USA produce electricity only. The average net efficiency of electricity generation was determined to be 19%. Key data for waste incineration (calorific value and fossil carbon content) were derived from measurement data provided by the Covanta Energy Cooperation, a relevant operator and contractor of WtE-plants in the USA. The degradable organic carbon content for landfilling was calculated as the difference between the values for incineration and the values determined from the USEPA data on waste composition.

Composting in the USA is mainly yard waste composting in simple open facilities. Mixed-waste composting also takes place but involves only about 0.2% of the total waste; this was disregarded in the balance.

For material recycling the shares of source-segregated collection (pre-sorted by residents), single-stream recyclables collection (treated in materials recovery facilities, MRF) and mixed waste collection (treated in mixed waste processing facilities) were each identified and calculated (electricity demand). Waste paper at 70% has by far the largest share in the separated recyclables. The recycling rate for waste paper is 66%. The recycling rate for the other recyclable fractions is between 8% for plastics and 33%/38% (ferrous/non-ferrous metals)[2].

The GHG balance of waste treatment in the USA in 2011 shows a net debit of about 18 million Mg CO_2-eq. This is made up of

net debit collection:	+2.2 million Mg CO_2-eq
net debit landfilling:	+64.7 million Mg CO_2-eq

[2] The recycling rate for non-ferrous metals breaks down into 21% for aluminum and 68% for other non-ferrous metals; the latter is lead from lead-acid batteries which is not considered in the GHG balance.

net credit recycling:	-44.7 million Mg CO_2-eq
net credit incineration:	-3.5 million Mg CO_2-eq
net credit composting:	-0.6 million Mg CO_2-eq

The influence of a higher gas collection efficiency and the effect of allowing for a carbon sink (landfilling, compost use) were examined in sensitivity analyses. In both cases the result changes from a net debit to a net credit, but neither aspect can be proved securely. The carbon sink is also excluded from the national inventory according to the IPCC guidelines (IPCC 2006). For incineration the difference that arises when the marginal approach rather than the country-specific electricity mix is used to offset energy generation was analysed, as were the effect of a higher fossil carbon content and the effect of cogeneration of heat and power from WtE. In all three cases the result is still a net debit; this is increased in the first two cases and decreased in the last.

USEPA collects data using a top-down approach: data on waste generation and management method are gathered by analysing production and trade statistics. In contrast to this, a regular bottom-up evaluation is undertaken by the Earth Engineering Center (ECC) of the University of Columbia and the journal *BioCycle* ("State of Garbage in America", SOG survey). The SOG survey is based on data provided by waste management agencies in the fifty states. Nevertheless, uncertainties also exist here because only landfills and WtE facilities are required to report the waste amounts treated. In addition, the reported MSW tonnages sometimes include non-MSW; this was excluded wherever possible.

The SOG survey results in considerably larger waste amounts than the USEPA data; in particular, it shows larger amounts landfilled. According to the survey, the MSW generated in 2011 was about 389 million short tons, of which 64% was landfilled. On the basis of the volumes in the SOG survey the net debit in the GHG balance for the USA is 3.6 times higher at 64.5 million Mg CO_2-eq. The GHG emissions from landfilling are nearly twice as high.

For the USA a medium and an ideal future scenario were analysed with the following conditions:

2030 medium:	45% recycling, 25% incineration, 30% landfill
2030 ideal:	60% recycling, 40% incineration, 0% landfill

As in the OECD balance, it was assumed that 80% of the waste amount "incinerated" is delivered directly to WtE facilities while 20% goes to an anaerobic MBT plant in the medium scenario and to an MBS plant in the ideal scenario. The increase in recycling was arrived at by adapting the recycling rates per fraction. Additional source-segregated food waste in the ideal scenario is assumed to be treated via anaerobic digestion. The characteristics of incineration and landfilling were recalculated to take account of the revised composition of the remaining waste. In contrast to the status quo, the same calculated characteristics were applied equally for incineration and landfilling. Differentiation of quality cannot be retained with an increasing share of incineration in the future scenarios. In addition to the redirection of waste streams in the future scenarios, some technical optimisations were assumed, such as cogeneration of heat and power from waste incineration in line with the assumption in the OECD balance.

The net results of the future scenarios compared to the status quo are shown in Table 4. As in the OECD balance, the results show that significant GHG mitigation can be achieved by reducing or halting direct landfilling and instead promoting material recycling and energy recovery from mixed waste. The increased credits from incineration are caused mainly by the shift from electricity generation only to cogeneration of heat and power. The medium scenario achieves a GHG reduction of about 72 million Mg CO_2-eq. In the ideal scenario the phasing out of landfilling and the correspondingly increased recycling lead to a net credit that is a further 2.6 times higher than in the medium scenario.

Table 4: Absolute net results – global warming potential, status quo and future scenarios to 2030 in the USA

in 1,000 Mg CO_2-eq	status quo	2030 medium	2030 ideal
Collection	2,151	2,151	2,151
Landfill	64,689	39,591	0
Incineration (with energy)	-3,454	-28,840	-50,840
Recycling	-44,688	-65,906	-89,850
Composting/anaerobic digestion	-595	-712	-2,863
Total	**18,104**	**-53,717**	**-141,402**

India balance

Data on waste generation and management method were extracted from official sources and other relevant publications. As is often the case for developing countries and emerging economies, the quality of the data is limited. In many cases the actual waste streams are not known. In India this is especially the case in rural areas; some studies of urban areas do exist. The described situation leads for India with its 1.2 billion inhabitants to possible waste generation of between 42 million tons (147 kg/(cap*a)) in rural areas and 243 million tons (201 kg/(cap*a)) for India as a whole. For the GHG balance values from a World Bank study were used (WBI 2008). Although these refer to rural areas the study offers a cohesive overall picture.

According to (WBI 2008) the 42 million tonnes MSW generated in rural areas were treated as follows:

- 9.5% uncollected MSW (98% unmanaged landfill, 2% open burning)
- 9.5% informal doorstep collection of recyclables
- 81% collected MSW:
  - 94% unmanaged landfill (of which 10% landfill fires)
  - 5% composting (mixed waste)
  - 1% recyclables sorted out by informal sector

GIZ India assumes that relevant amounts (up to 60%) of organic waste on the streets and at transfer stations are eaten by cows and other animals. This aspect was not taken into account in the balance, partly because the amounts cannot be quantified and partly because the practice is questionable on hygienic grounds and does not comply with legal requirements.

The informal sector in India is active on two levels. On the one hand there is doorstep collection where recyclables (newspapers, cans, glass, plastic bags, textiles) are bought from residents; on the other hand there are waste pickers who sort out recyclables from the mixed waste on the streets and at landfills. The quantities of recyclables collected by the informal sector quoted in (WBI 2008) are estimates which probably significantly underrate the activities of the waste pickers. Nevertheless, in relation to the waste composition for India these estimated values already lead to a recycling rate of 45% for the recyclables fractions in the calculations. Recycling for India was again calculated using the standard emission factors established in this study. Plastics recycling was assumed to have a low substitution effect because secondary granulate tends to be used in thick-walled products like flower tubs and planters. This means that in line with experience from Europe the secondary granulate will usually substitute wood and concrete; only a limited amount of primary plastic is substituted.

Landfills in India are unmanaged, without basal liners and without landfill gas or leachate collection. At best the deposited waste is levelled by bulldozers. Thus there is practically no difference between “official” landfills and illegal dumps. Both are assessed in the same way in the GHG balance. According to India’s Second National Communication to the UNFCCC (MoEF 2012) landfills are assumed to be shallow (< 5 m waste). Under the IPCC guidelines such landfills are assessed with a methane correction factor (40% of the methane formation potential), since it is assumed that on account of the limited anaerobic conditions methane formation is low. This value has been adopted in the GHG balance for India.

With composting (“simple MBT”) MSW is first separated mechanically; the sieved fraction <100 mm is composted. The product is a mixed-waste compost which, because of its high heavy metal content and low nutrient content, is not assigned any credit in the GHG balance. The separated inert fraction is usually landfilled. The sieve overflow can be used as RDF. However, the possibilities for RDF use in India are limited. There are probably only two RDF power plants, and cement works can purchase subsidised coal. In the GHG balance it was assumed that 70% of the RDF fraction is landfilled.

The GHG balance for management of the 42 million tonnes of Indian MSW considered in the baseline case shows a net debit of about 9.4 Mt CO_2-eq. With high waste generation of 243 million tonnes the net debit would be nearly six times higher at 54.5 million CO_2-eq. The net results are made up as follows (values in brackets for high waste generation):

net debit landfilling:	12.3 Mt CO_2-eq	(71 Mt CO_2-eq)
net debit open burning:	0.6 Mt CO_2-eq	(3 Mt CO_2-eq)
net debit simple MBT:	0.2 Mt CO_2-eq	(1 Mt CO_2-eq)
net credit recycling:	-3.6 Mt CO_2-eq	(-21 Mt CO_2-eq)

Taking the carbon sink into account does not reverse the net result. If a lower national collection rate of 60% is applied, the result hardly changes, because in India this situation chiefly means that there is a shift from unmanaged landfilling to illegal dumping, both of which are assessed in the same way in the GHG balance.

For India two future scenarios – a medium and an ideal one – were again analysed. In general, no change in the informal collection of recyclables was assumed in either scenario. It was assumed that all the remaining waste was collected. In the medium scenario measures and technologies were considered that have at least some likelihood of being implemented. Diversion from landfill is desirable in India, although more for reasons of space than for climate change mitigation reasons. Space is scarce and relatively expensive. Waste incineration is highly controversial in India. This is partly because of negative experiences with plants that have failed because of bad planning (heating value too low, organic fraction lower than expected, too expensive, maintenance problems) and partly because of health concerns. Sample measurements taken at a plant in Okhla, New Delhi, that has been in operation since the middle of 2012 have revealed high levels of dioxin and furan emissions.

Against this background the following assumptions were made with regard to the treatment of residual waste in the medium scenario:

50% managed landfill, 20% gas collection efficiency, flaring

50% simple MBT as in status quo, but 100% RDF use in cement works

For the ideal scenario it was again assumed for India that direct landfilling is phased out. The following management methods were assumed for mixed waste:

50% MSWI, pre-sorting and recycling of plastics
50% MBS with RTO, RDF use in cement works

The net results of the scenarios are shown in Table 5. Landfilling shows no improvement in the medium scenario by comparison with the status quo; there is a slight increase in the net debit. This is due to the fact that managed landfill involves anaerobic conditions and hence has 100% methane formation potential. Thus in the medium scenario, despite less landfilling of MSW and some landfill gas collection, somewhat more methane is released than in the status-quo with 40% methane formation potential (see above). This must if necessary be accepted as part of the transition to a managed, integrated waste management system. For health reasons this approach is most definitely to be commended if no other opportunities for achieving a rapid phasing out of landfilling are available. Because of the continuation of landfilling, the overall net result in the medium scenario remains a net debit. Only in the ideal scenario, with the cessation of landfilling, is the result converted into a net credit.

Table 5: Absolute net results – global warming potential, status quo and future scenarios to 2030 in India

in 1,000 t CO_2-eq	status quo	2030 medium	2030 ideal
Landfill	12,293	12,792	-
Incineration (open burning / MSWI)	573	-	-1,363
Simple MBT / MBS	201	-1,269	-7,560
Recycling	-3,636	-3,636	-3,636
Total	**9,430**	**7,886**	**-12,559**

Egypt balance

As in India, collection of data on waste generation and treatment in Egypt is difficult. The data available consists of estimated values from a regional waste management network (Sweep-Net) and relatively old (2006) official figures from the Ministry of Trade; both sources were used for the calculations. The total waste amount for the year 2010 was determined to be about 21 million tonnes (260 kg/(cap*a)). On the basis of information for the 27 governorates of Egypt, the national collection rate was calculated at 54%. 11% of this relates to waste collection in Greater Cairo, which is traditionally informal. There the "Zabbaleen" collect MSW from households for a fee; they purchase a licence from the local authorities that permits them to do this. In the "garbage cities" where the Zabbaleen live the collected waste is sorted manually and in some cases processed into secondary raw materials or products. According to (CID/GTZ 2008), 28% of the collected waste was recycled. Organic waste was used as animal feed (52%, predominantly for pigs), and 20% remaining waste was dumped. It is known that further informal activities take place, involving for example the purchase and exchange of household articles or recyclable waste and the activities of waste pickers, but these were not considered in the GHG balance because the amounts involved are small or unquantifiable.

It is assumed that most uncollected waste is dumped; in rural areas some is also burned in the open. However, it is also reported that in rural areas MSW is also used as fuel for cooking or as animal feed (sensitivity). Most of the formally collected MSW is dumped in unmanaged landfills; a smaller amount is composted or recycled. Overall the following picture emerges for Egypt:

- 46% uncollected MSW (88% unmanaged landfill, 12% open burning)
- 11% informal doorstep collection (Zabbaleen, Greater Cairo)
- 43% formally collected MSW:
  - 83.5% unmanaged landfill
  - 5% managed landfill
  - 9% composting (mixed waste)
  - 2.5% recycling

For Egypt a particular difficulty lies in the available figures on waste composition, as these differ markedly. Estimated values from Sweep-Net for national waste generation in Egypt contrast with measurements of formally collected MSW in two of the 27 governorates. According to the former of the waste composition is 56% organic waste and 29% dry recyclables; according to the latter it is 88% organic waste and 7% recyclables (the remainder in both cases is “other”). The discrepancy could not be resolved by assuming that the majority of recyclables have already been removed from the formally collected MSW. The resulting figures differ widely from other data sources (e.g. national study on plastic recycling). In the GHG balance the problem was addressed by means of assumptions and sensitivity analysis. For example, the Sweep-Net composition was used for the 11% informal doorstep collection because this is more plausible in the light of the type of waste treatment (large proportion of recyclables). For the formally collected and uncollected waste the analysed waste composition was used (large organic waste fraction).

For Egypt, recycling was again assessed using the standard emission factors established in this study. As for India, plastics recycling was assumed to have “low” substitution potential.

The vast majority of landfills in Egypt are unmanaged. As in India, shallow landfilling is assumed (< 5 m), with reduced methane formation potential. However, in many cases waste is dumped on canal banks in Egypt. It was therefore assumed that half of the unmanaged dumping involves “wet” dumping (high water level; according to IPCC 80% methane formation potential). Overall, though, it is possible that because of the very dry climate in Egypt the calculation of methane emissions based on the IPCC default values is an over-estimate. Managed landfills in Egypt are not fitted with gas collection systems.

Composting in Egypt is mixed-waste composting. In contrast to India, recyclables are rarely or never pre-sorted and no RDF fraction is produced. This seems plausible in the light of the apparently very large share of organic waste in formally collected MSW. This would also explain why compost of acceptable quality is produced despite the fact that mixed waste is composted. To reflect this, a credit for 10% of the compost was included in the GHG balance. The feeding to animals of some of the organic matter separated from mixed waste was assigned neither debits nor credits in the GHG balance because this practice must be regarded as hygienically questionable.

For the 21 million tonnes of MSW considered in the baseline case for Egypt, the GHG balance shows a net debit of 14.5 Mt CO_2-eq. This is made up of

net debit collection (100 km):	0.3 Mt CO_2-eq
net debit landfilling:	14.5 Mt CO_2-eq
net debit open burning:	0.1 Mt CO_2-eq

net debit composting (MSW): 0.1 Mt CO_2-eq
net credit recycling: -0.4 Mt CO_2-eq

Taking the carbon sink into account does not reverse the net result. If some home composting, feeding to animals and use of organic waste for fuel is assumed in rural areas (nearly 20% reduction in landfilled waste) the net debit is reduced to 12.7 Mt CO_2-eq. If the calculation is based on the Sweep-Net waste composition (large share of recyclables), the net debit is reduced to 12.1 Mt CO_2-eq. The small reduction in spite of the larger recyclables share is due to the relatively low recycling rates in the baseline case, which were not changed in the sensitivity analysis.

As in India, the two future scenarios for Egypt were determined by the given situation and the possibilities available. According to the national objectives, incineration of waste is not an option. The country is aiming for source-segregated collection of organic and dry waste. In general, complete collection was assumed in the future scenarios: in Scenario 1 (medium) with the assumption that previously uncollected waste is distributed equally between formal and informal collection, in Scenario 2 (ideal) with complete formal collection. For Egypt, too, phasing out of direct landfilling was assumed in the ideal scenario. Based on this the following further assumptions were made:

Scenario 1 ("medium"):

- informal collection: MSW treatment as in status quo (but larger amounts)
- formal collection: 70% source-segregation of organic waste (composting), remaining MSW to managed landfill (20% effective gas collection efficiency)

Scenario 2 ("ideal"):

- 70% source-segregation of organic waste (share of feeding as in Scenario 1, remaining share 50% composting, 50% anaerobic digestion)
- residual waste to MBT with output: 12% recycling, 11% RDF (cement works), otherwise MBT residue and inert fraction

Due to the source-segregated collection of organic waste in the future scenarios, the feeding of organic waste to animals was assigned a credit for the substitution of animal feed. However, steps should be taken to investigate whether sanitisation of the organic waste should be required because of the possible risks of epidemics.

The net results of the scenarios are shown in Table 6. The net debits in the status quo are already halved in the medium scenario. This is mainly due to the high separate collection rate for organic waste and the corresponding prevention of methane emissions from landfill. The alternative composting results in a net credit, albeit a small one ("recycling organic waste").

Table 6: Absolute net results – global warming potential, status quo and future scenarios to 2030 in Egypt

in 1,000 t CO_2-eq	status quo	2030 medium Scenario 1	2030 ideal Scenario 2
Collection	261	485	485
Landfill	14,528	8,242	-
Open burning	117	-	-
Recycling organic waste	54	-65	-560
Simple MBT	-	-	1,225
Recycling	-408	-738	-429
Total	**14,552**	**7,924**	**722**

The complete cessation of direct landfilling in the ideal scenario leads to a further GHG mitigation but not yet to a reversal of the net result. The reasons are the low recyclables share in the waste composition and the fact that simple MBT leads to net debits which are higher than the credits from composting and anaerobic digestion. This becomes obvious in a sensitivity analysis for Scenario 2 in which the waste composition according to Sweep-Net was used. Here a net credit of -0.9 Mt CO_2-eq is achieved. This is attributable to the larger shares of recyclables, which result in larger credits for recycling, and a net credit for simple MBT, because more recyclables and more RDF are sorted out, yielding corresponding credits.

Conclusion

The analysis of the OECD countries and the EU28 and the detailed studies of the USA, India and Egypt confirm the findings of the previous studies:

- Diversion from landfill is the main contributor to GHG mitigation in the waste management sector.
- Even if waste is deposited in managed landfills with gas collection and landfill gas use, there is still significant GHG mitigation potential if waste is materially recycled wherever possible and otherwise used for energy recovery.
- An integrated waste management system that prioritises reuse, material recycling and otherwise energy recovery instead of landfilling can contribute significantly to national GHG mitigation goals. Synergy effects are more efficient resource use and the reduction of environmental impacts on human health and ecosystems.

For the development of an integrated waste management system the following recommendations – some of them country-specific and some more general – can be made:

- In countries in which the majority of MSW is still landfilled, action plans for the progressive reduction of landfilling should be introduced and systematically implemented. In industrialised countries anaerobic MBT and MBS can be used as bridging technologies; in developing countries and emerging economies simple MBT with due regard to good composting practice is appropriate. A market for the RDF fraction produced must exist or be created (in compliance with emission standards).
- The superordinate goal of closed-cycle materials management is comprehensive high-value material recycling. Source-segregated collection is recommended because, provided that

the population is well-educated on the issue, it can ensure high quality standards. Alternatively, sophisticated sorting facilities are a possibility, but they cannot be readily established in all countries and they need a market for secondary products to justify the cost and labour involved.

- Systematically implemented political and legal conditions are essential for the development of action plans. The formulation of policy objectives for the waste management sector is recommended for the USA and India, and in Egypt a waste management law should be enacted. Of special importance for the EU28 is full implementation of the Landfill Directive: despite existing derogations, eight EU countries are having difficulty complying with the landfill reduction targets. The recycling targets and rules on comprehensive source-segregated collection for the Waste Framework Directive proposed on 2 July 2014 by the EU Commission of the time are an important step in the right direction; they should not be withdrawn as planned by the current Commission.
- Information about waste streams and waste characteristics is indispensable for proper steering of waste streams and for planning. For the USA it is therefore recommended that, instead of using the top-down approach, actual waste streams are evaluated. This requires compulsory reporting of the waste delivered to composting, sorting and recycling facilities. In addition, MSW should be weighed separately from other waste. The waste composition should be analysed, at least on a sample basis. The recommendations also apply to OECD countries in which they are not (yet) being implemented. For India and Egypt random samples are recommended as an initial step, especially in rural areas.

In general, a closed-cycle management system requires creation of the infrastructure needed to facilitate high-quality recycling. A market for secondary raw materials or products is also needed. If the corresponding demand can be created, closed-cycle management can be established.

Countries like Germany can support the process of implementing closed-cycle management – for example through transfer of know-how either at either technology or government agency level. They must also ensure, however, that they consolidate and further refine the standards achieved in their own waste management systems.

For developing countries and emerging economies, access to funding is also important. Most of the current funding is directed at the private sector as it usually targets single projects such as CDM projects. However, in countries such as India funds are urgently needed for further capacity development in the public sector. There may be scope here for pursuing an integrated funding approach via NAMAs. For single projects it is important that regional conditions are taken into account in order to avoid technological failures. In addition, sophisticated technical installations should as a matter of urgency be required to meet the emission standards that are standard practice in industrialised countries (negative example: dioxin and furan emissions from the MSWI in New Delhi). Finally, assistance for developing countries and emerging economies should be planned and implemented in a manner that takes account of the activities of the informal sector.

3 Introduction

The present study explores the performance and potential of the waste management sector with regard to climate change mitigation in the OECD countries, India and Egypt. As a special emphasis, it was agreed that the USA would be included not only as a member of the OECD states but also as a separate focus of analysis. In all, therefore, four inventories were drawn up for the study.

A greater depth of detail is possible for the GHG inventories of the individual countries than for that of the OECD. Nevertheless, some country-specific data were obtained for the OECD balance, while some conclusions were also reached by analogy and assumptions were made. A key source of data on waste streams in the OECD countries was statistics from Eurostat and the OECD. For the inventories of individual countries, the relevant official sources were analysed and supplemented by information from the literature. For India, selection of the underlying data and in particular the assumptions for the future scenarios were based on the outcomes of discussion and debate at a workshop in New Delhi on 7 November 2012 organised by the Gesellschaft für Internationale Zusammenarbeit (GIZ). In Egypt the difficult political situation rendered such an approach impossible and the exchange of information with local experts was therefore confined to email correspondence and telephone calls.

In each case the greenhouse gas (GHG) balance of the status quo was calculated and analysed. The results were used to identify possible improvements. Possible obstacles were taken into account and suitable strategies for greenhouse gas mitigation by the waste sector were drawn up with due regard to the political objectives of the individual countries, wherever such objectives have been adopted. In connection with the inventories for the individual countries, legal and economic instruments were also considered. In the case of the developing countries and emerging economies, the aim was to devise feasible scenarios that have a realistic prospect of being implemented. Thanks are due in this connection to the Gesellschaft für Internationale Zusammenarbeit (GIZ), which made the exchange of information in India possible. A successful workshop there yielded important information; in particular, the discussion among the panel members and the comments from the audience highlighted the areas of waste management in India in which action is needed. Key information and findings from the workshop are given in the Annex (Section 11.3).

The year 2030 was selected as the time horizon for the future scenarios. It was agreed that in addition to the realistic or "medium" scenario, an ideal scenario would also be considered. The four country inventories that were drawn up are documented in detail in this report. The inventory for the USA should be regarded as a special aspect of the inventory for the OECD countries. The methodology is identical for all inventories, as are some other aspects of the analysis. For example, in the course of the project it was agreed that harmonised emission factors should be used for substitution processes, irrespective of the country being considered. This decision is in line with the recommendations made by experts following the methodology workshop for this project held in Berlin on 18 June 2012.[3]

[3] The workshop protocol and the presentations can be downloaded at http://www.umweltbundesamt.de/themen/abfall-ressourcen/abfallwirtschaft/klimaschutz-in-der-abfallwirtschaft.

The following key findings of the project:

- the clear confirmation that ending landfilling of untreated waste makes a significant contribution to climate change mitigation,
- the fact that even at landfills with state-of-the-art gas collection there is still significant potential for GHG mitigation through materials recovery and subsequently through the use of waste for energy generation, and
- the fact that the waste sector can make a relevant contribution to climate change mitigation in the national context

were presented and discussed at the final workshop at IFAT in Munich on 8 May 2014.[4] It was very clear from the presentation of results and the subsequent discussion that there are obstacles to the implementation of measures in the waste sector. These obstacles are not necessarily financial in nature. They may arise from national priorities, or from inadequate information. In connection with the latter aspect, participants at the closing meeting urged that the present study be made available in English.

Another aspect of the present project was an evaluation of waste measures that could contribute to the generation of emission certificates, or in the case of India and Egypt to nationally appropriate mitigation actions (NAMAs), where these are defined or approved. Consideration was also given to the strategic goals of the two countries in relation to the use of the above-mentioned mechanisms and to whether and under what conditions funding for waste management measures involving reduction of GHG emissions is available from technology cooperation funds. The project also included a rough calculation of the CO_2 abatement costs of various waste management measures.

[4] Details and presentations can be downloaded at http://www.umweltbundesamt.de/service/termine/konferenz-abfall-klimaschutz-auf-der-ifat-2014.

4 Principles of the emissions accounting method used

The study uses the method of life cycle assessment in waste management based on ISO 14040/14044. Special considerations apply to waste management: these are described in Section 4.1, which also details other methodological agreements that apply to this study. Section 4.2 describes the scope of the study and the procedure for calculating the emissions of the various management methods.

4.1 Method

4.1.1 Life cycle assessment in waste management – system comparisons

As in previous studies, assessment of the global warming effect is based on the method of life cycle assessment in waste management (IFEU 1998). In the context of the present project, which is concerned exclusively with the potential for greenhouse gas mitigation, the following specific issues are relevant:

1. Departing from the "cradle-to-grave" approach, the study considers the life cycle of waste management services. The inventory commences with the generation of the waste. The previous history of the waste is irrelevant to the issue of disposal – that is, it is normally the same for all disposal options and can be omitted from the assessment if the quantities of waste being considered are constant. If waste avoidance or temporal variation in the generation of waste is an issue, the production of the waste must be considered if, as in the present study, system comparisons are to be performed (status quo, 2030 scenarios). In consequence, this study uses constant total waste generation for system comparisons and does not consider waste prevention.[5]
2. In the inventory, all current and future debits and credits arising from the disposal of one tonne of waste are allocated to the waste quantity considered in the relevant base year. This is particularly relevant to landfilling, where methane emissions from biodegradation of the organic waste in landfill are released over decades. The calculations comply with the requirements of (IPCC 2006) and normally involve a 100-year time horizon.
3. In another departure from the classic product LCA, in which a product is considered, if appropriate over a number of recycling loops, until fully disposed of through incineration or landfilling, analysis in waste management – apart from considering direct incineration or landfilling – usually tracks materials only as far as the production of a secondary product. Equivalence of the utility of different disposal systems is achieved by crediting the utility produced in each case. Thus for each system or scenario the same utility is considered in the inventory: "disposal of the same quantity of waste".
4. Because of the major importance of this issue for our society, the study deliberately focuses on the potential and possible contribution of waste management in relation to greenhouse gas mitigation. On account of this restriction to one impact category, the study does not meet the criteria of the LCA method described in ISO 14040 and 14044, which requires all

[5] This means that the source reduction considered in (OECD 2012) is not pursued, since in this approach waste prevention is assessed without considering the generation of the waste.

other relevant environmental impacts such as acidification, eutrophication, human toxicity, etc. to be considered.

To ensure equality of utility between the systems being compared, the additional benefits arising from the recovery of waste – such as secondary products or energy produced – are compared with the primary products or conventionally produced energy that they replace in equivalence or substitution processes. This applies equally to all the scenarios considered in the four inventories – OECD, USA, India, Egypt. In addition, in all the scenarios evaluated in the various inventories, the quantity of waste considered is that identified for the status quo. In each case this quantity is the functional unit of the comparative analysis.

However, in order to take account of quantity trends where necessary, the extent to which significant future changes can be expected was estimated. As in the OECD study (2012a), this can be done by modelling increased waste amounts using information on management methods from the status quo and comparing this business-as-usual scenario with future scenarios involving similarly increased quantities of waste. This ensures that the condition of equal total waste generation for the LCA comparison is met. In the event, though, this option was not pursued in the final assessment. For the USA the available data on the trend over time indicates stabilisation of waste generation rather than an increase. In India, because per-capita waste generation is currently relatively low, waste volumes are likely to rise. Here, though, data on the quantity of current waste generation are beset by such high levels of uncertainty that – with the agreement of the Indian stakeholders – the decision was taken not to engage in further speculative forecasting of quantity increases. Instead, a sensitivity analysis of the waste volume in the status quo situation was carried out. This was also done for Egypt. For the OECD inventory, an assessment at the level of the individual member states would be required, but it was not possible to conduct this as part of the project.[6]

4.1.2 Other methodological agreements

Other methodological aspects and definitions involve in particular the procedure for particular types of waste and the method of offsetting the secondary products and energy produced. In general, the substituted primary processes taken into account via credits have a major influence on the results of any life cycle assessment of the waste management sector. To understand the significance of the results, it is important to be aware of the conditions that have been specified:

Crediting of energy produced is often performed using the marginal approach. This assumes that all the fuel substituted by “additionally” produced energy is fossil fuel. This approach was also used in the previous study (Öko-Institut/IFEU 2010). In the present study, the marginal approach was used in the assessment of the individual countries of India, Egypt and the USA. However, for the assessment of the OECD countries there was no reasonably straightforward and reliable way of calculating the marginal electricity for all member states. For simplification, average electricity generation in each country was used as a substitute. The relevant country-specific emission factors were taken from the Ecoinvent database. This method

[6] The scenario for the OECD termed “business as usual” in the comparison of results represents a completely unchanged waste management system in the year 2030 – i.e. one in which the volume of waste and all other boundary conditions remain unchanged.

was applied consistently in the OECD inventory, including for the USA. This means that the results of the USA inventory cannot be directly compared with the results of the OECD inventory: the USA inventory is not a subset of the OECD inventory. To demonstrate the influence of the relevant electricity credit, a sensitivity analysis was performed for the USA inventory. For the purpose of crediting generated heat, a simplified general formula for average heat composition was used according to which 50% of heat is assumed to be generated from heating oil and 50% from natural gas. An emission factor of 334 g CO_2-eq/kWh heat was used; the same factor was used in the previous study (Öko-Institut/IFEU 2010). This simplification was regarded as permissible because heat generation plays a subsidiary role in the overall waste management system and because other types of heat provision are not expected to result in any significant differences. For the co-incineration in power plants or cement works that is relevant in the future scenarios, crediting takes place at the level of the fossil fuel that is usually used. In this study, substitution of coal of equivalent calorific value is assumed for co-incineration.

In the future scenarios to 2030, no changes were made to the emission factors for energy supply, either for energy demand or for the substitution processes. This is in line with the method used in the previous studies (Öko-Institut/IFEU 2010 and 2005; IFEU 2006) and ensures that differences by comparison with the status quo are the result of changes in waste management and not of energy management measures. However, it should be borne in mind that for the recovery of energy from waste the substituted electricity production is extremely important. In countries in which the share of fossil fuels in electricity generation is high, the recovery of energy from waste is of necessity more beneficial in the average analysis than in countries with higher shares of renewables. If the shares of fossil fuels and renewables change over time, this affects the outcome. In the marginal approach this is not directly given, but here the question arises of how small it is possible for the fossil share in the energy mix to be while still assuming marginal substitution of this share. However, set against the possible "error" in the result on account of the retention of constant emission factors are the uncertainties with regard to the forecasting of energy supply in the year 2030 in the various countries. A possible solution for the future may involve using global emission factors – one for electricity with high shares of fossil fuels and one for electricity with low shares – and drawing up two inventories in parallel. A similar idea was put forward at the methodology workshop on 18 June 2012 (see footnote 5).

With regard to **possible carbon sinks (C sinks)**, these – as in the previous study (Öko-Institut/IFEU 2010) – are included only in the context of a sensitivity analysis. A C sink can arise if biogenic waste is withdrawn from the carbon cycle (the atmosphere) long-term (100-year time horizon). This can occur when organic waste is landfilled, when compost is applied, or when organic materials are used in long-life products (such as a wooden table or even books). However, considerable uncertainties attach to the actual long-term storage of biogenic carbon. For example, it has not yet been possible to arrive at an even approximately reliable assessment of this option in relation to the material recycling of wood or paper. In the case of compost use, it was shown in (IFEU 2012) that a C sink only arises if compost is used to enrich humus (humus carbon storage). In all other cases an equilibrium arises between compost addition and breakdown processes. Furthermore, it is not as yet possible to make any reliable statement about the proportion of humus carbon that can be stored long-term, because none of the long-term studies that have been conducted involve a time frame of more than 20 years. There is considerable capacity for long-term storage in connection with landfilling. According to the calculation methodology used by the IPCC, only a certain proportion of the biogenic carbon in waste is broken down. This means that, conversely, the remainder is stored. However, the possibility cannot be excluded that further aerobic degradation takes place with formation of biogenic carbon dioxide. Above all, though, a one-sided crediting of the C sink from landfilling using default values would lead to significant asymmetries in the overall inventory and present the risk of resulting in wrong decisions. In this study, therefore, the carbon sink – where it is quantifiable – is stated only in sensitivity analyses.

For waste paper and wood, a **"credit for saved wood"** was introduced for the first time in the previous study (Öko-Institut/IFEU 2010). This involved estimating how the "forest carbon reservoir" changes (greater carbon accumulation in virgin forest). At the same time, in view of the significant pressure to use wood in Germany, the possibility was considered that primary timber "saved" through recycling does not remain in the forest but is used instead for energy. The resulting GHG abatement (substitution of fossil fuels) was credited to the material use that makes this possible. In this study this approach was not pursued further in the standard case. Carbon accumulation through wood saving is beset with considerable uncertainties and plays a minor role in the overall outcome (see Öko-Institut/IFEU 2010). Crediting of energy production from primary wood "saved" through recycling is only permissible if corresponding pressure of use has been shown to exist. However, in contrast to the situation for Germany, it is virtually impossible to arrive at a reliable estimate of the market situation and the demand for wood or the pressure on wood supplies for all the countries considered. It was therefore decided that a credit for saved wood would not be considered in the standard case. In the USA inventory a sensitivity analysis of "saved primary wood" was carried out by way of an example.

In the course of the project – prompted by the recommendations of the methodology workshop on 18 June 2012 (see footnote 5) – it was agreed that where possible **harmonised emission factors** would be used for substituted processes(major influence on results) in order to make the results more transparent and more comparable. Another advantage of such a procedure is that it permits to capture the contribution of national waste management to greenhouse gas mitigation. Using this approach, different estimates of substituted primary processes are standardised and the influence of the frequently large data uncertainties in connection with the description of the substituted primary processes is weakened. In the present study standard emission factors were in particular calculated and used for recycling wherever possible. This procedure is suitable for studies like the present one that aim to calculate the contribution of waste management to greenhouse gas mitigation for various countries. For other aims, by contrast, actual emission factors should be calculated and used. For example, this should be done if the effects of waste management on greenhouse gas mitigation in a particular region are to be identified and the results are to contribute to regional planning decisions. It would be inappropriate, for example, to use a global assessment e.g. for the co-incineration of refuse-derived fuels as a basis for providing these fuels if it were not then possible to use these fuels purposefully. Similarly, such global assessments should not be used as a reason to drive forward materials recovery without having suitable recycling structures in place. However, the aim of this study is not to provide a foundation for concrete planning decisions but to inform stakeholders about general opportunities for action which could be incorporated into political objectives. For the calculation of standard emission factors, available emission factors – especially those for dry recyclable materials – were first collated from the literature and analysed. In some cases it was possible to calculate a standard factor directly from this information; in others, further research was conducted and standard factors were recalculated using boundary conditions that were as broadly applicable as possible. The method of calculating and selecting harmonised emission factors for dry recyclables is described in the Annex (Section 11.1).

Finally, it was agreed that no imports or exports of waste would be considered, as information on the origin and destination of such waste is not available. However, the significance of imports and exports diminishes as harmonisation of the assessment increases. In principle, though, consideration could in future be given to assessing – as part of a sensitivity analysis – whether imported and exported waste for recycling is actually recycled. If it is not, disposal of this waste needs to be included in the inventory. If it is, it would be necessary to check whether the calculation based on standard emission factors is sufficient or whether country-by-country adjustment is needed.

4.1.3 Impact assessment of global warming potential

To assess the global warming potential, the individual greenhouse gases in the life cycle inventory are aggregated on the basis of their CO_2-equivalent values. The most important greenhouse gases and their CO_2-equivalent values for the 100-year time horizon used in this study are shown in Table 7. The CO_2-equivalent values are taken from IPCC (2007).

Table 7: Global warming potential of the most important greenhouse gases

Greenhouse gas	CO_2-equivalent value (GWP_i) [kg CO_2-eq/kg]	
Carbon dioxide (CO_2), fossil	1	1
Methane (CH_4), fossil*	27.75	21
Methane (CH_4), regenerative	25	18.25
Nitrous oxide (N_2O)	298	310
	(IPCC 2007, WG I, chapter 2, Table 2.14)	(IPCC 1995)

*Including the stoichiometrically calculated global warming effect of fossil CO_2 after conversion of the methane in the atmosphere (life of methane about 10 years)

Methane emissions are distinguished here according to their origin. Biogenic methane (from the conversion of organic substances) has a somewhat lower equivalence factor than fossil methane (from the conversion of fossil fuels), because the biogenic or regenerative carbon dioxide produced from the methane over time as a result of oxidation is treated as climate-neutral. For comparison purposes the table also shows the CO_2-equivalent values according to the IPCC (1995) that are used for national reporting under the Kyoto Protocol.

4.2 The emissions accounting procedure

The basic accounting procedure used for all country inventories is described below. Within the individual country inventories, those aspects that go beyond the basic procedure or differ from it in the case of a particular country are mentioned or described in detail. Deviations from the previous study are also described below.

4.2.1 Deviations from the previous study on account of the statistical reporting system

In contrast to the previous study (Öko-Institut/IFEU 2010), in this study it is not possible to produce separate inventories for the individual waste fractions. In addition, because of the lack of data, used wood is only taken into account in isolated cases. The main reason for the different accounting method is the classification system used in the statistical data on which the waste streams in the OECD inventory are largely based. To avoid double-counting, the statistics indicate only the final destination of waste that is disposed of. This means that the amounts reported as “recycled” are the amounts after pre-treatment in sorting plants or mechanical-biological treatment plants. In the previous study, the causal waste stream chain for each waste type in Germany was created by back-calculation from the statistical figures. Using known mass flow balances – in particular for water and degradation losses in mechanical-biological treatment (MBT) plants – it was possible to back-calculate the original input quantities. It was also possible to classify the destination of the output by calling on expert knowledge. This information cannot be gathered from the statistics themselves. In sources such as the national German waste statistics, the output from sorting plants or MBT plants is classed merely as “recovered”, “disposed of” and “other”. The type of disposal cannot be identified and the proportion used for energy recovery is only to a limited extent traceable.[7] For the OECD countries, it was not possible without excessive cost and effort to perform any reliable back-calculations in this study. This is also the reason why it was not possible to specify any treatment via MBT plants in the waste quantity details for the OECD countries. The quantity details were taken 1:1 from the statistical data; this means that the “incinerated” or “landfilled” quantities include not only directly delivered waste but also sorting residues from pre-treatment plants.

However, for treatment residues from further treatment or recovery plants, it is likely that this is only the case if these plants are licensed as waste treatment plants. Otherwise it is assumed that the treatment residues are not municipal waste but production waste and are therefore not included under “incinerated” or “landfilled” in the waste statistics. This is likely to be the case with glass, paper and metals used in glass and paper factories or steelworks. Plastics may be an exception here. In Germany sorted segregated plastics are also processed in plants classed as waste treatment plants. Accordingly the treatment residues arising from granulation that are recycled for energy should be included under “incinerated” in the municipal waste statistics. For the present study it is assumed that the quantity of recovered plastics identified from the statistics and also the quantities identified from the top-down information provided by USEPA correspond to the output of the treatment plants – in other words, that these recovered plastics comprise processed secondary granulate.

The above-mentioned considerations are important in connection with the emission factors for the material recovery of dry recyclables. Depending on the interface, the emission factors may need to be adjusted to the recycling quantities given in the statistics (see Section 4.2.4 and Annex, Section 11.1). In general, however, it should be noted that even with the most plausible adjustments, inaccuracies remain in the results on account of the uncertainties surrounding the actual mass flows. The statistical data are of only limited usefulness here and usable only for overarching purposes such as are addressed in this study. For more concrete issues or guidance

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[7] Waste code 191210 “combustible waste”, while e.g. 191212 “other wastes” is often also used for energy.

for planning decisions, more reliable information on waste streams is essential. This information should be available and documented following a material flow analysis approach.

4.2.2 Scope of the study and data basis

Like the previous studies for Germany and the EU27 (Öko-Institut/IFEU 2010 and 2005; IFEU 2006), the present study is confined to municipal solid waste (MSW). Where appropriate data are available, the waste is classified into types such as paper, glass, plastic and metal waste. The data for the OECD countries have been obtained mainly from the statistics of the OECD and Eurostat. Data on which the inventories of the individual countries, USA, India and Egypt are based are drawn mainly from the publications of public institutions such as USEPA and government ministries.

The statistics available from Eurostat and the OECD specify the amounts of waste collected via public waste collection systems, which largely correspond to the amounts of waste generated. In the OECD data the reported quantities of "generated" and treated ("total treatment") waste agree for two-thirds of the countries. For the remaining countries, the total quantity of treated waste is sometimes higher (Japan) than the quantity of generated waste and sometimes lower (Chile).[8] The inventory uses the treated waste quantities to which the details of waste management methods in the Eurostat figures refer.

The statistical data of the OECD countries vary in quality. For example, there are sometimes differing views on what should be classed as "municipal solid waste", and data collection methods may vary between countries.[9] It was not feasible to scrutinise or verify the statistical information and data for all of the 34 OECD and/or 28 EU countries for the purposes of the OECD inventory. Moreover, neither Eurostat nor the OECD reports the recycled quantities of individual waste fractions. Similarly, there are no details of the composition of residual waste. The attempt was made to obtain information on these issues from national publications. Often, though, the gaps had to be closed by plausible assumptions. In the light of these considerations, the OECD inventory (and the EU28 inventory) must be approached with some reservations with regard to data quality, and significant limitations also apply to the degree of accuracy possible in the individual country inventories. For the OECD and Eurostat data, more extensive harmonisation and an extension of reporting are desirable. Comprehensive information is available for the USA, and this was analysed in depth. In this case it was possible for the inventory to be considerably more detailed and amounts of waste that are not usually classed as MSW in Europe (used tyres, lead from starter batteries[10]) were excluded from the mass balance or at least from the assessment. Nevertheless, uncertainties also exist with regard to the data provided by USEPA. However, these were identified and examined in sensitivity analyses. Data from the USA inventory were used in the OECD inventory, but for reasons of symmetry not in the same degree of detail as in the USA inventory.

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[8] Explanations of the deviations are not available.

[9] E.g. in some countries rubble is included in MSW, and e.g. in Mexico overall waste generation is extrapolated from an estimate of per-capita generation.

[10] Under the German Waste Catalogue Ordinance (Abfallverzeichnis-Verordnung – AVV), end-of-life vehicles and their components come under AVV 16 01.

For India and Egypt the official figures on waste generation also specify the waste collected via the public waste collection system. In these countries, though, waste collection activities (e.g. doorstep collection) are also performed by the informal sector; a significant proportion of this waste is never captured but may be burned in the open, discarded wholesale or scattered. Because of the importance of the informal sector and the environmental impacts of the uncaptured amounts of waste, the attempt was made to quantify these amounts and include them in the inventories. This was done using estimated values from the literature. The results, while based on significant data uncertainties, provide some impression of these areas by comparison with formally collected MSW. For India and Egypt, information on the material recovery of individual waste types was deduced from information on the composition of waste in the literature and from details of the total amounts recycled.

In some cases the official data used for the country inventories differ significantly from information in other publications. For example, this is the case in India, where the most recent published figures relate to the year 2000 while other sources consider a more recent time horizon. In the USA the official data provided by USEPA contrast with other data that use a different method of recording national MSW generation. USEPA uses a top-down approach based on the products put into circulation, while in the contrasting bottom-up approach waste generation is calculated by questioning the waste authorities in the individual US states. The latter method produces significantly higher figures for waste generation in the USA. In these cases of deviating quantity data, sensitivity analysis was performed on the data collected by the two different methods (see the section on country inventories).

4.2.3 Collection

Waste collection covers both the actual collection method and transport to the waste treatment facility. The information on collection methods that is available for Germany was used for the OECD and USA inventories. This procedure can be justified on the grounds that waste collection accounts for only a very small proportion of the total emissions from waste disposal and the relevant data for the individual countries would be difficult and time-consuming to obtain.

In India and Egypt, some waste is not collected at all or is collected by the informal sector via hand-cart or doorstep collection. For the waste that is collected, no information on average transport distances is available. In principle it can be assumed here, too, that the transport emissions of waste collection have only a minor influence on the overall result. With the more comprehensive regular collection of waste that is desirable in future, the emissions would rise, but not to a significant extent. These emissions could/should be accepted, because they form the basis for optimised or regulated waste management.

To ensure that the influence of emissions from waste collection was not ignored, different methods were used in the inventories for India and Egypt: for India collection emissions were not included, while for Egypt a long transport distance of 100 km was assumed to apply generally.

4.2.4 Recovering dry recyclables

Harmonised emission factors

The derivation of the harmonised emission factors for dry recyclables that are used in this study is documented in the Annex (Section 11.1). The values thus obtained are shown in Table 8. The values relate to the quantities of recyclables sent to the actual recycling facility, either directly or after sorting; these quantities can be identified directly from the statistics (Eurostat, OECD) or the quantities of recyclables specified in the USEPA data can be used. Emissions from collection, transport and any sorting that takes place are not included in the emission factors but are calculated separately in the inventories. Debits and credits from disposal of the sorting residues are included in the landfilled or incinerated quantities (see above). For plastics this also applies to the processing residues arising from further treatment (granulation) (see Section 4.2.1).

Under this system the values for ferrous and non-ferrous metals relate to the quantities sent to the steelworks or aluminium smelter or for pyrolysis; they also include only the emissions and credits (substitution of primary steel/aluminium) that arise at this stage of the process. The values for paper and cardboard relate to the amounts sent (after sorting, if appropriate) to paper factories; similarly, the values for glass describe the amounts sent to glassworks. For textiles the values relate to segregated waste textiles suitable for re-use after manual sorting. This method is only considered if relevant information indicates that re-use occurs (e.g. doorstep collection in India). Otherwise this form of recovery is treated as a sensitivity in the USA balance.[11]

Table 8: Emission factors for dry recyclables used in this study

Waste or sorting fraction	Debit	Credit	Net
	in kg CO_2-eq/t waste by sorting facility or input to recycling plant		
Ferrous metals	338	-1,284	-945
Non-ferrous metals	406	-9,713	-9,307
Paper & cardboard	167	-960	-793
Glass	0	-514	-514
Textiles	32	-2,850	-2,818
Plastics	Special procedure – see text		

A special procedure was used for plastics. Plastic waste involves a variety of different types of plastic, usually collected as mixed plastic. In addition, some plastic products sent to waste contain several different types of plastic. The quality of recovery is therefore heavily dependent on how effectively the waste is sorted into segregated plastic types so that the mixed plastic fraction is as small as possible. The higher the degree of purity, the higher the quality of the use to which the secondary granulate that results from processing can be put in manufacturing. This quality of the secondary granulate is reflected in its substitution potential. In the LCA this is taken into account mathematically via the substitution factor (SF) or the amount that cannot be used as a substitute for primary plastic but only as a substitute for wood or concrete. In Germany this is relevant only to mixed plastic. To depict quality differences, three quality categories for plastic recycling were defined for this study. The quality distinctions

[11] A sensitivity analysis for the OECD balance showed hardly any difference in the results, since textile quantities are rarely reported separately for the OECD countries. A description of the sensitivity analysis has therefore been omitted.

are based on the substitution potential of different types of plastic and mixed plastic. The categories were defined as follows:

"high": SF = 1 for plastic types, SF = 0.9 for mixed plastics with 100% PE substitution

"medium": SF = 0.7 for plastic types, SF = 0.8 for mixed plastics with 32% PE substitution, otherwise wood and concrete substitution

"low": No plastic types, only mixed plastics as in "medium"

In Germany the SF of 1 for plastics in the "high" category is achieved in practice in areas such as PET recycling. For other plastic types the substitution potential is at least 0.8. The classification of mixed plastic as "medium" is in line with the current situation in Germany (Öko-Institut/HTP 2012). The "low" quality category is defined by the fact that the plastic waste is not sorted by plastic type and accordingly only a proportion of it is used to substitute primary plastics and other products such as those of wood and concrete. It was assumed that this is the case in India and Egypt. For the USA and OECD inventories the "medium" quality was assumed for the status quo because it was not possible to obtain more precise information on recycling. In the USA inventory the influence of high-quality recycling on the result was explored in a sensitivity analysis.

Computationally the emission factors for plastic recycling involve a debit for the electricity needed (country-specific emission factor) and a credit via the emission factor for primary production (Table 64). The three quality categories are obtained through calculation with the above substitution factors. The derivation of the emission factors is described in more detail in the Annex, Section 11.1.

Sorting and transport

As already mentioned, the sorting and transport of dry recyclables are not included in the above-mentioned emission factors. The sorting emissions are considered separately by back-calculating the original input quantity from the sorting efficiency and then calculating the electricity needed to sort this input quantity. This procedure is described for the USA in Section 5.9.2 (under the subheading "Recycling"). For India and Egypt this procedure does not apply, because in these countries dry recyclables are collected mainly by the informal sector, either via doorstep collection or by waste pickers who scavenge the waste collected at waste collection points or landfills for recyclables. In both cases sorting is performed mainly by hand.

The transport emissions incurred in transferring the output of sorting facilities either to recycling plants or to landfill or incineration sites for disposal of the sorting residues are also calculated separately. In connection with waste collection, it was not possible to obtain reliable information on the transport components in any of the countries considered. In developing countries and emerging economies the authorities usually have no details of actual quantities of recyclables and their fates. However, the same also applies to the USA, because recycling facilities are not required to report how much waste they have accepted and treated. For the OECD countries a decision was taken not to research transport because transport emissions are of minor importance in the overall result and obtaining the relevant data, if possible at all, would be difficult and time-consuming. In this study a standard transport distance of 200 km was assumed for the distance from sorting facility to recycling plant, landfill site or incinerator.

4.2.5 Landfill

The calculations relating to the depositing of waste are based in principle on (IPCC 2006). Key parameters for the calculation are:

- The degradable organic carbon (DOC) in the deposited waste
- The fraction of this that is dissimilated over time (DOCf)
- The methane content of the landfill gas (or the fraction of the degraded carbon that is converted to methane)
- The methane correction factor (MCF) to take account of the type of waste disposal site
- The oxidation factor (OX)
- The effective gas collection efficiency
- The type of gas use

It is a basic principle of life cycle assessment in waste management that all current and future emissions arising from the treatment of a particular quantity of waste are included in the inventory. This is particularly relevant in connection with landfilled waste because methane emissions are not simply produced immediately: they arise and are released over a period of several decades as biodegradation proceeds. All the environmental impacts incurred per tonne of waste must be included, since this is an essential basis for reliable decisions on whether it is better to landfill, incinerate or recycle waste. In the inventories, figures for the key parameters listed above are taken either from country-specific data (described in more detail in the sections on the individual countries) or from the IPCC's default values. In most cases the IPCC figures are used.

Normally no information is available on the **degradable organic carbon** (DOC) in deposited waste. The DOC is calculated from the country-specific waste composition and data on waste fractions (Table 13; the carbon levels quoted in (IPCC 2006) are included in this calculation).

Other default values according to (IPCC 2006) are:

DOCf = 50% (average value for all waste that may contain a proportion of lignin)

Methane content = 50 vol%

Methane correction factor (MCF):

managed landfill site – anaerobic	= 1
unmanaged open dump – deep (> 5 m waste) and/or high water level	= 0.8
unmanaged open dump – shallow (< 5 m waste)	= 0.4

Oxidation factor (OX):

default value	= 0%
covered (e.g. soil, compost), well-managed landfilled site[12]	= 10%

A country's **effective gas collection efficiency** is the product of the proportion of landfills that have gas collection systems installed and the average gas collection efficiency of these systems over the entire duration of the deposits. It is clear from (IPCC 2006) that gas collection should

[12] Default for OX according to IPCC is 0%; the value of 10% is justified for covered, well-managed landfills.

only be reported and taken into account if relevant evidence for it exists. For good practice the gas collection efficiency should be calculated on the basis of measurement of the total amount of gas collected or the total energy generated from the collected gas. Default values according to (IPCC 2006) are:

- The default gas collection value = 0% if no data are available
- The default gas collection value = 20% if gas collection is estimated on the basis of the installed gas collection systems

In the previous study (Öko-Institut/IFEU 2010) the 20% effective gas collection efficiency was used in the standard case for the EU27 and the three countries of Turkey, Tunisia and Mexico, and 40% was used in the sensitivity analysis. For Egypt and India this rate was retained for existing or possible future gas collection. For the OECD inventory the gas collection efficiencies in the EU National Inventory Report were re-analysed. The data used was that for 2010 in (EEA 2012). Following the method used in (EEA 2011) the reported gas collection efficiencies were not adopted 1:1; instead it was assumed that the maximum technically feasible effective gas collection efficiency was achieved. The maximum cap was set at 50% (in (EEA 2011) it was 45%). The resulting average gas collection for the EU-OECD countries is documented in Section 5.3.2. The 50% cap was also used for the standard case in the USA inventory, even though operators of landfill sites assume high gas collection efficiencies and higher figures are also to be found in the literature (Section 5.9.2). The influence of a high effective gas collection efficiency of 75% was considered in the sensitivity analysis.

The ways in which gas is used can range from venting and flaring to energy generation, e.g. in a combined heat and power (CHP) plant. For Egypt and India, in a procedure similar to that used in the previous study, it was assumed in the future scenarios that landfill gas that is collected in future will at least be flared off, resulting in oxidation of methane to CO_2 (climate-neutral, because biogenic). For the OECD inventory the process was simplified by using the landfill gas use rates calculated for the USA (50% CHP, 50% flaring, see Section 5.9.2). For inventory purposes, landfill gas use in the CHP plant is treated in the same way as biogas, using the method described in the next section, the only difference being the methane content, which is lower in landfill gas than it is in biogas.

Another aspect of landfilling is the permanent storage of carbon in the landfill body; this permanent storage can be assumed to occur for the non-degraded fraction (1 - DOCf). This C sink is treated as a sensitivity because its significance is disputed (Section 4.1.2) and according to (IPCC 1996, 2006) this amount is also not taken into account in the GHG inventory.

4.2.6 Composting, anaerobic digestion

Composting and anaerobic digestion were assessed using emission values according to (gewitra 2009). Table 9 shows the range of these values for various technologies for finished compost and composted digestate. The range of values arises mainly from different management methods. In simplified terms it can be stated that the minimum values represent best management practice, while the maximum values were recorded at sites characterised by poor professional practice. The emission values used in this study are based on this.

Table 9: Range of emission values for composting and anaerobic digestion – finished compost

Treatment	Min.	Mean	Max.
	in kg/t waste input		
Open composting			
Methane	0.47	1	2
Nitrous oxide (N_2O)	0.049	0.11	0.21
Closed composting (finished compost)			
Methane	0.3	0.71	1.5
Nitrous oxide (N_2O)	0.049	0.068	0.12
Anaerobic digestion with post-composting			
Methane	3.2	3.7	4.6
Nitrous oxide (N_2O)	0.038	0.12	0.19

Values according to (gewitra 2009)

For example, in India the residue of the collected mixed waste is composted after separation of dry recyclables. The composting usually involves piling the waste into high windrows that are neither ventilated nor deliberately irrigated. In most cases, too, the stacks are not under cover. These conditions increase the likelihood of anaerobic zones and correspondingly higher emissions of methane and nitrous oxide. For this mixed-waste composting in India the maximum emission values for open composting in Table 9 are therefore used. For the USA and the OECD countries, for which it was not possible to obtain any more specific details on composting conditions, the mean emission values were used.

The possible uses of the mixed-waste compost were assessed on a country-by-country basis. For example, mixed-waste composts in India are high in heavy metals and low in nutrients. Following the procedure of the previous study, no use was identified for this compost from mixed waste and so no credit was allowed. The C sink in the sensitivity analysis was also disregarded. The same applies to the OECD inventory.[13] For Egypt, on the other hand, it can be assumed that at least a few plants are averagely well managed and produce compost of acceptable quality. Here a proportional credit is allowed for compost produced from organic waste (after sorting from mixed waste) and the C sink is shown in the sensitivity analysis (see Section 7.2.4).

The debits and credits from the application of compost from source-segregated organic waste are measured in a standard manner. First, the simplifying assumption is made that all the compost produced is finished compost (because it is mainly green waste that is being composted). Simplified emission factors are then used for the direct emissions and the credit according to (IFEU 2012) (Table 10). The same procedure is used for the C sink in the sensitivity analysis.

[13] In the USA mixed-waste composting is practised on only a small scale and it has been ignored in the inventory.

Table 10: Average emission factors for compost application according to (IFEU 2012)

	Emission factors in kg CO_2-eq/t of compost
Green waste finished compost	
Debit	45
Credit	291
(C sink for sensitivity analysis)	19
Organic waste finished compost	
Debit	54
Credit	328
(C sink for sensitivity analysis)	16

Anaerobic digestion is not included in the status quo in any of the country inventories (in India and Egypt it does not occur; in the USA it applies to only very small fractions; for the OECD countries it is not reported in the statistical data). In the future scenarios anaerobic digestion was assumed in the inventories for the USA and the OECD countries for increased source-segregated organic waste. Ideally it was assumed that digestate is post-composted. For simplification, the emission factors for organic waste finished compost in Table 10 were used for the composted digestate. The emissions from anaerobic digestion with post-composting were calculated using the emission values in Table 9.

Otherwise, and as in the previous study, anaerobic digestion of biowaste is defined as follows:

- Average gas yield 100 m³/t biowaste, methane content 60 vol%
- Use in small-scale CHP with
  - 37.5% electrical efficiency (of which 20% for internal use)
  - 43% thermal efficiency (of which 25% for internal use), 20% of the surplus heat as usable heat
  - methane slip 1%

These parameters for biogas and biogas use were also used for biogas arising from anaerobic mechanical-biological treatment. The small-scale CHP data were also applied to the use of landfill gas.

4.2.7 Mechanical-biological treatment/stabilisation

Mechanical-biological treatment (MBT) or mechanical-biological stabilisation (MBS) in accordance with the technical concept that exists, for example, in Germany was included only in the future scenarios.

In the OECD countries, some treatment of this sort does occur in other countries apart from Germany, but the relevant quantities could not be reliably determined from the statistical data for the OECD countries as a whole (see Section 4.2.1). For simplification, therefore, emissions were only reported in the OECD inventory for the management method shown in the statistics. In the USA there is currently only composting of mixed waste from which recyclables have first been removed. This composting only takes place on a small scale and was ignored for the status quo. In India and Egypt, composting of mixed waste with prior sorting of recyclables (partly for energy recovery) takes place on a relatively large scale. However, the methods used are very simple, with manual pre-sorting and open composting of the residual fraction. The inventory process is described in more detail in the sections on the individual countries.

In the future scenarios, MBT and MBS were taken into account in the different countries for various reasons. In the **USA and the OECD countries** the status quo is characterised by significant waste fractions that are sent to landfill without treatment. In line with the German model it was assumed that a future reduction of landfilling in these countries will be easier to implement if at least a proportion of landfill capacities can continue to be used. For the medium scenario for the USA and the OECD countries an MBT strategy was therefore adopted under which an MBT residue for landfilling is produced. Because the relevant facilities would have to be built from scratch, it was assumed that all facilities would be anaerobic MBT plants. For OECD counties with small landfill fractions (e.g. Japan), no anaerobic MBT plants were included. The ideal scenario involves MBS instead of anaerobic MBT. The proportion of waste treated by MBT or MBS was set via the **ratio of waste incineration to MBT/MBS at 80:20**. This is based on the situation in Germany; treatment by MBT/MBS significantly in excess of the German ratio was not regarded as realistic. In principle, though, the potential for RDF production and co-incineration in power plants and cement works to reduce GHG emissions needs to be shown in the inventories.[14] For this reason a sensitivity analysis was carried out with a ratio of incineration to MBT/MBS of 50:50.

The mass balances for the anaerobic MBT and the MBS were obtained from information in (wasteconsult 2007) and (UBA 2011). Data on energy input and gas yield for the anaerobic MBT were taken from (Wallmann 2008). The use of the biogas from the anaerobic MBT was accounted for as described in the previous section. The mass balance data and the energy data for the two concepts are shown in Table 11. The anaerobic MBT produces surplus power and heat. For the surplus heat it is assumed as before that on a national average only 20% can actually be used. The natural gas requirement shown in the table is required for regenerative thermal oxidation (RTO) of the exhaust air. Deviating from the practice outlined above, the value for MBS is taken not from (Wallmann 2008) but from a presentation on mechanical-physical stabilisation (MPS) plants.[15] The residual gas potential for the MBT residue was accounted for in accordance with (IFEU 2012): the emissions amount to around 56 kg CO_2-eq/t MBT residue. The associated C sink included in the sensitivity analysis amounts to approx. 403 kg CO_2-eq/t MBT residue.

[14] In general, because coal of equivalent calofiric value is substituted, co-incineration of waste is more advantageous from a climate change mitigation perspective than MBT or use in an RDF power plant.

[15] ALBA AG: "Verwertung von Restabfällen nach dem MPS-Verfahren", Berlin 16 April 2009

Table 11: Mass balance and energy of anaerobic MBT and MBS for the USA and OECD inventories

	anaerobic MBT	MBS
Mass balance		
Metals	2.4%	3.4%
Refuse-derived fuels	35.2%	48.6%
Impurities to MSWI	7.6%	18.8%
MBT residue to landfill	31.3%	-
Inert fraction to landfill	-	7.8%
Losses (water, biological degradation)	23.5%	21.3%
Energy requirement / generation		
Electricity requirement kWh/t of input		81
Heat requirement		-
Electricity surplus in kWh/t	55	
Heat surplus in kWh/t	96	
Natural gas requirement (RTO) in kWh/t	52	27

For the RDF output it was assumed both for anaerobic MBT and for MBS that 50% of this RDF is used in RDF power plants and 50% is co-fired in power plants and cement works. In this connection it was also assumed that an RDF market exists or can be established. In the co-incineration, coal of equivalent calorific value is substituted (see Section 4.1.2). For the RDF power plant, efficiencies were estimated for the year 2030 (see Table 12). The characteristics of the RDF output were taken in simplified form from (UBA 2011):

RDF from MBT: calorific value = 13.2 MJ/kg; C fossil = 16.7%

RDF from MBS: calorific value = 13.4 MJ/kg; C fossil = 12.4%

In the medium scenario for **India and Egypt** an MBT concept was likewise assumed, although for these countries the concept is a significantly simpler one with aerobic treatment in an open setting without treatment of exhaust air by RTO. Improving on the status quo it was assumed that the composting is managed (smaller windrows, more frequent turning, avoidance of anaerobic zones) and that the separated RDF is actually used. Co-incineration in cement works was viewed as the most likely and appropriate option.[16] The mass balances were also adjusted to reflect the composition of waste in the two countries. Waste in India and Egypt has significantly higher inert and/or organic fractions than waste in the industrialised nations and contains very few recyclables (at least in the collected waste; generated waste quantity for Egypt considered in a sensitivity analysis). The assessment of MBT/MBS is described in more detail in the sections on the individual countries.

4.2.8 Waste incineration

For the GHG inventory the energy efficiency and the characteristics of the incinerated waste are relevant parameters of waste incineration in municipal solid waste incinerators (MSWIs) or RDF power plants. Table 12 summarises the net efficiencies used in this study. Egypt is not

[16] In India RDF is sometimes also co-fired in small tileworks that do not meet any emissions standards.

included in this table because waste is not incinerated there and waste incineration was not included in the future scenarios for that country.

Table 12: Net efficiency of thermal treatment plants in the status quo and future scenarios

<table>
<tr><th></th><th colspan="2">Status quo</th><th colspan="2">2030</th></tr>
<tr><th></th><th>Power</th><th>Heat</th><th>Power</th><th>Heat</th></tr>
<tr><td>MSWI USA power only</td><td>19%</td><td>0%</td><td rowspan="3">18%</td><td rowspan="3">42%</td></tr>
<tr><td>MSWI USA CHP</td><td>12%</td><td>30%</td></tr>
<tr><td>MSWI OECD</td><td>11.4%</td><td>31.6%</td></tr>
<tr><td>MSWI India</td><td>-</td><td>-</td><td>15%</td><td>0%</td></tr>
<tr><td>RDF power plant USA/OECD</td><td>-</td><td>-</td><td>25%</td><td>20%</td></tr>
</table>

The characteristics of the waste for incineration are country- and case-specific and are described in the individual sections. They change e.g. in the future scenarios since as a result of the assumption of increased source-segregated collection of recyclables the composition of residual waste changes; it is this composition that determines the fossil C content and the calorific value. The basic method of calculating the parameters is described below.

4.2.9 Characteristics of waste fractions

For the LCA, in particular in relation to landfilling and the incineration of waste, important characteristics are the calorific value and the biogenic and fossil carbon content. The literature usually contains at best occasional figures for calorific value; carbon contents are seldom quoted for the particular waste mix. Moreover, calorific values are frequently measured and published only for the source-segregated waste or the waste arriving at particular facilities. In order to nevertheless depict a consistent overall system on the basis of generated waste, the specified characteristics have been determined by drawing on corresponding characteristics by waste fraction. Mean values have already been calculated in the previous study (Öko-Institut/IFEU 2010); these are shown in Table 13.

Where the composition of the waste is known, these figures enable the characteristics of any mix of waste to be determined. In the LCA the waste composition is re-calculated at each stage of the material flow analysis. This means, for example, that starting from the composition of the generated waste, the composition of the residual waste is recalculated after recyclables have been removed and the corresponding characteristics are re-determined for this composition.

Table 13: Characteristics of waste fractions

Details in %	Total C kg/kg waste	Biogenic C % of total C	Calorific value kJ/kg waste
Organic and green waste	0.16	100	4,620
Paper and cardboard	0.37	100	13,020
Composites	0.43	49	18,017
Glass	0	0	0
Nappies	0.18	75	4,447
Plastics	0.68	0	30,481
Metals	0	0	0
Wood	0.38	100	13,250
Textiles, leather, rubber	0.39	56	15,020
Fine waste < 8 mm	0.13	65	5,133
Other wastes (incl. mineral waste)	0.21	53	7,800

Calculated values (Öko-Institut/IFEU 2010)

5 Waste management in OECD countries

5.1 Overview of member countries and regional classification

The 34 member countries of the OECD are divided into three regions. Turkey is assigned not to the "Asia" regional group but to "Europe, Turkey and Israel", because data for Turkey are often reported with the EU data. Israel has likewise been assigned to this group on account of its geographical location. Table 14 shows the regional distribution of the countries.

Table 14: Regional classification of the OECD member countries

Region	Regional group	Countries
America	North America South America	USA, Canada, Mexico Chile
Europe, Turkey and Israel	EU-OECD countries* OECD rest of Europe Israel and Turkey	EU-OECD countries* Norway, Iceland, Switzerland Israel, Turkey
Japan, South Korea and Pacific	Asia Pacific	Japan, South Korea Australia, New Zealand

*EU28 excluding Bulgaria, Croatia, Cyprus, Latvia, Lithuania, Malta and Romania, which are not members of the OECD

5.2 Waste generation and composition

For the countries of the "Japan, South Korea and Pacific" region and for Canada, Chile and Israel the data on waste generation were taken from the most up-to-date information of the OECD (OECD 2013) available at the time of the study. Data for the USA were taken from (USEPA 2013a) (see Section 5.9.1) and data for Mexico from (INECC 2012). For the EU-OECD countries and Norway, Iceland, Turkey and Switzerland the most recent data from Eurostat (Eurostat 2014a) were used. (OECD 2013) and (Eurostat 2014a) provide data both on the waste generated in the country as a whole and on the total waste treated. For two-thirds of the countries these waste quantities agree. The exceptions are Chile, where the waste treated was 5% less than that generated, and Japan, where 9% more waste was treated than generated.[17] The study uses the treated waste quantities to which the details of waste management method in the Eurostat figures refer. This yields the GHG debits and credits actually caused by waste management in the individual countries.

Table 15 summarises the waste generation of the individual countries and regions. Against each country name is shown the year to which the data refer. The "America" region has the largest waste generation at some 292 Mt (607 kg/(cap*a)). It is followed by "Europe, Turkey and Israel" with around 264 Mt. The "Japan, South Korea and Pacific" region has significantly lower generation at around 85 Mt. The difference is less marked if population size is taken into account. In this case the "Europe, Turkey and Israel" region treats only slightly more waste at 469 kg/(cap*a) than "Japan, South Korea and Pacific" at 421 kg/(cap*a). These quantities yield waste generation for the whole OECD of around 641 Mt (514 kg/(cap*a)).

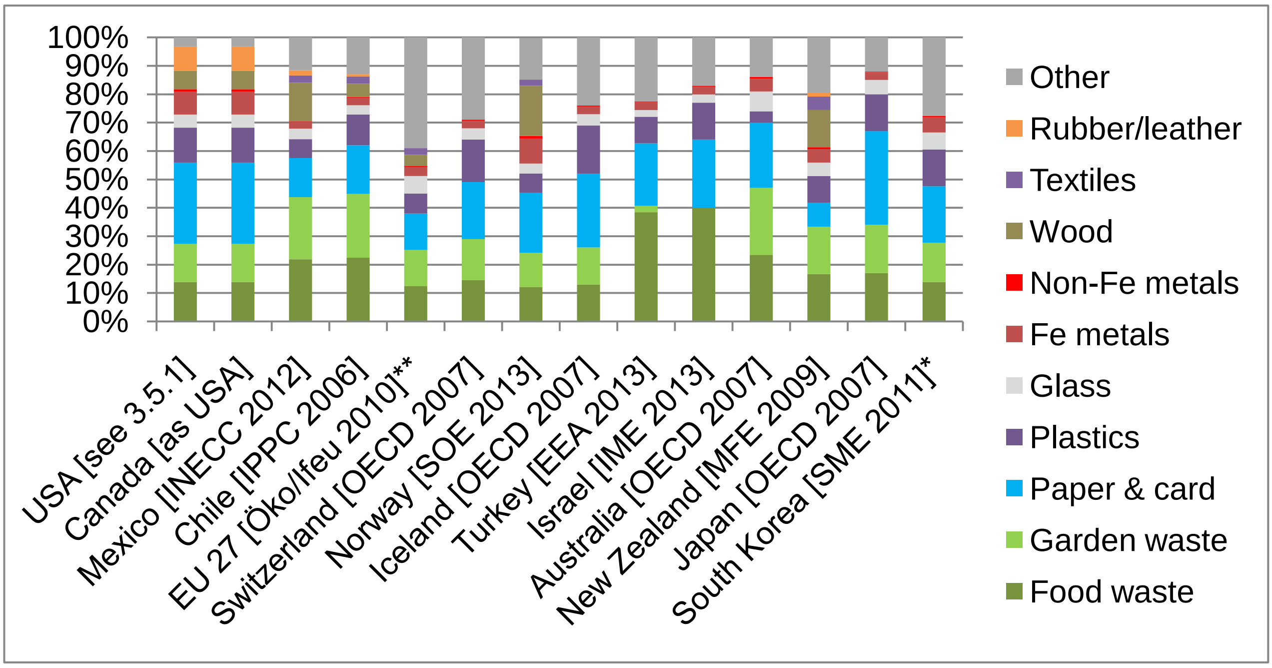
[17] No explanations of these deviations are available.

Table 15: Total treated waste by country and region, total waste and per capita

Region	Country	Total treated waste [1,000 t]	Total treated waste per capita [kg/cap*a]
America	USA	224,628	721
	Canada (2008)	34,345	984
	Mexico (2006)	26,350	227
	Chile (2009)	6,185	355
	Total	291,508	607
Europe, Turkey and Israel	EU (OECD) (2012)	227,200	487
	Switzerland (2012)	5,576	697
	Norway (2012)	2,343	469
	Iceland (2012)	108	350
	Turkey (2012)	24,730	330
	Israel (2009)	3,936	576
	Total	263,893	469
Japan, South Korea and Pacific	Australia (2009)	14,035	638
	New Zealand (2010)	2,126 (2,531)*	483
	Japan (2008)	50,597	397
	South Korea (2009)	18,581	380
	Total	85,339	421
OECD	**Total**	**640,740**	**514**

*Figure in brackets was reported incl. construction waste; here adjusted according to waste composition from (MFE 2009)

Figure 1: Percentage waste composition of the individual countries



*South Korea reports 0% plastic in waste. Since it can be assumed that this is a result of the methodology used in data collection, Japan's plastic percentage was applied to South Korea and the "other" category has been correspondingly reduced.
**Waste composition assumed for simplification to be similar to Germany (Öko-Institut/IFEU 2010)

Where possible the composition of the waste was calculated using national data for the individual countries. If the specific data were not available, data from other sources were used. In some cases the various sources did not distinguish between ferrous and non-ferrous metals (aluminium), or between food waste and garden waste. In these situations some assumptions were made: for metals a ratio of iron to aluminium of 9:1 was assumed, while the ratio of food waste to garden waste was taken to be 1:1. In the absence of data on the amount of wood, textiles, rubber and leather in the total waste, these fractions were included under "other wastes". Textiles are therefore not included in the GHG inventory. Wood, rubber and leather were in general not considered further.

5.3 Waste collection and management methods

Waste collection emissions were calculated in the same way for all OECD countries, because analysis of country-specific conditions was not possible without disproportionate time and effort (see Section 4.2.3). The direct emissions associated with collection were accounted for at a standard rate of 10 kg CO_2-eq/t. Transport direct emissions were treated in a similar way and were included in the inventory at a standard rate of 24 kg CO_2-eq/t waste. Where sorting of waste was taken into account, this too was included at a standard rate, with electricity usage of 40 kWh/t waste.

Information on the distribution of waste between the various disposal methods was taken in the first instance from the most up-to-date OECD or Eurostat data (OECD 2013, Eurostat 2014a). These data relate to the final destination of the waste. The reported methods are: incineration (with and without energy recovery), landfill, recycling and composting/anaerobic digestion. If the waste is pre-treated, for example by sorting or in an MBT facility, the output of this treatment is allocated to one or more disposal methods (Eurostat 2012). The quantities in the pre-treatment processes are not itemised.

Under the methodological guidelines for reporting statistical waste data to Eurostat, net recycling quantities must be reported (Eurostat 2012). In the present study, the waste quantities reported by Eurostat and the OECD are treated as net or output quantities, although it must be assumed that some member states do not report their data exactly in accordance with the methodological guidelines.

Incineration as a management method includes both thermal waste treatment in accordance with Article 3 (4) of Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the Incineration of Waste and co-incineration in accordance with Article 3 (5) of that Directive (Eurostat 2014b). In the present study the only form of incineration covered is thermal treatment in a MSWI. This applies both to the quantities reported by Eurostat and to those reported by the OECD, because no data are available on the quantities co-fired or on the specific conditions that apply to co-incineration.

For the USA, figures were taken from the USA inventory (see Section 5.9.2), for Mexico from (INECC 2012). Table 16 summarises the statistical data on waste management methods. For Mexico and the EU27 the sum of the totals differs from the total quantity in Table 15. For the EU27 this difference arises as an accounting gap in the statistics. In the case of Mexico the difference occurs because the waste streams of “disposal (scattered)” and “MSW n.b” are disregarded.

Table 16: Waste management methods in the countries and regions taken from statistical data (OECD 2013, Eurostat 2014a)

Region	Country	Recycling [1,000 t]	Composting incl. residual waste composting [1,000 t]	Landfill [1,000t]	Incineration (no energy) [1,000t]	Incineration (with energy) [1,000t]
America	USA[1)]	60,056	18,779	121,799		23,995
	Canada (2008)	6,034	2,439	24,578	1,294[2)]	
	Mexico (2006)	2,831	784	22,735		
	Chile (2009)	24	595[3)]	5,556	9	1
	Total	**68,945**	**22,597**	**174,668**	**216**	**25,083**
Europe, Turkey and Israel	EU (OECD) (2012)	64,332	35,492	70,521	9,022	47,833
	Switzerland (2012)	1,938	851			2,787
	Norway (2012)	620	333	44		1,346
	Iceland (2012)	39	6	54	2	7
	Turkey (2012)		190	24,540		
	Israel (2009)			3,936		
	Total	**66,929**	**36,872**	**99,095**	**9,024**	**51,973**
Japan, South Korea and Pacific	Australia (2009)	5,750		8,132		152
	New Zealand (2010)			2,126		
	Japan (2008)	9,776		821	2,878	37,122[4)]
	South Korea (2009)	11,112	249	3,457	323	3,440
	Total	**26,638**	**249**	**14,536**	**3,201**	**40,714**
OECD	**Total**	**162,512**	**59,718**	**288,299**	**12,441**	**117,770**

1) The data for USA deviate from those in Section 5.9, because the information there is quoted in short tons whereas here it is in metric tons.

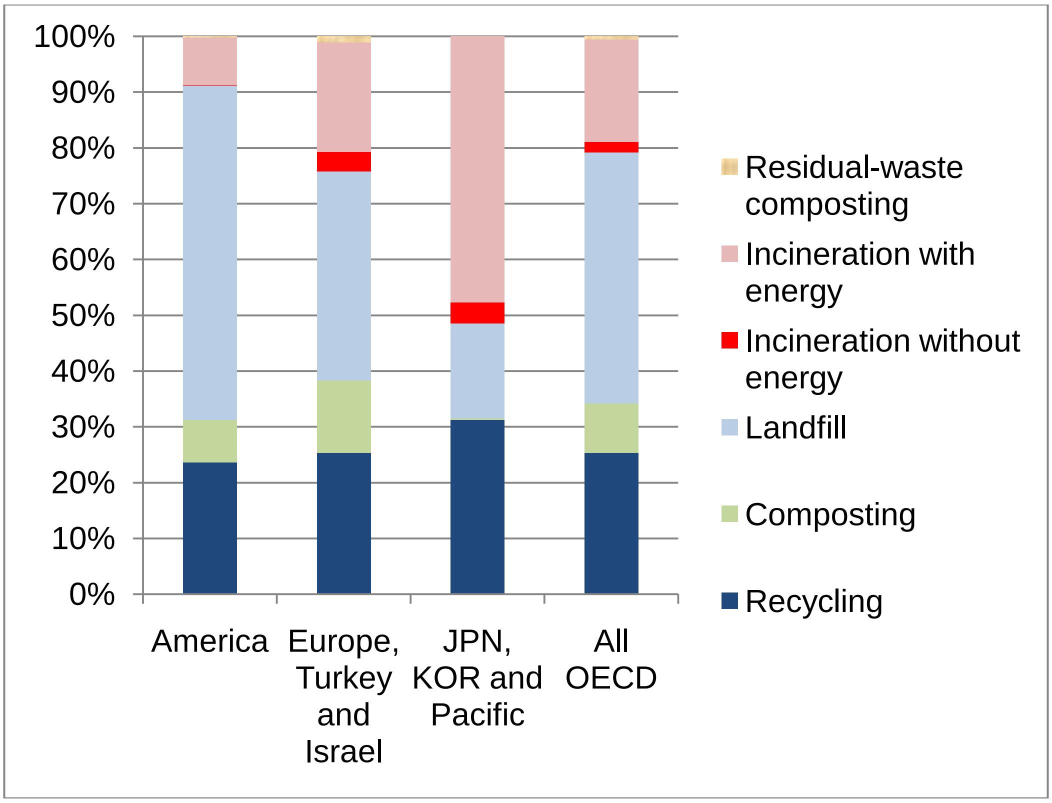
2) Incineration reported only as total incineration; for inventory purposes distribution as for EU-OECD countries: 16% no energy, 84% with energy.

3) For Chile in (OECD 2013) 564,000 t itemised as "other disposal". This quantity was allocated to residual waste composting ("composting" in this table). 2,000 t "other recovery (vermiculture)" was allocated to "composting".

4) For Japan, 1,723,000 t was itemised as "other recovery" in (OECD 2013). No concrete data on this are available. The amount was allocated to incineration with energy.

Figure 2 shows the percentage waste management methods in the three regions and the OECD as a whole. It is clear from this that in the "Japan, South Korea and Pacific" region incineration with energy recovery (48%) and recycling including composting (31%) are the most important methods. Recycling and composting are also an important method in "Europe, Turkey and Israel" (38%) and "America" (32%). In "America", though, landfill plays a more major role than in "Japan, South Korea and Pacific" and "Europe, Turkey and Israel". In "Europe, Turkey and Israel" 38% of waste is landfilled; in "America" the figure is 60%. Residual-waste composting is not of major importance in any region. The picture for the OECD as a whole is similar. The most important disposal method is landfill at 45%, followed by recycling/composting at 34%.

Figure 2: Percentage destination of waste according to management methods by region



5.3.1 Recycling and composting

In connection with recycling and composting, the OECD and Eurostat do not break waste down into individual fractions but simply quote the total amount recovered. For this reason fraction-specific recycling rates from national sources were used (Annex 11.2.1). In France, Spain and Portugal in particular, composting of unsorted, untreated waste is still widespread (Öko-Institut/IFEU 2010). For the present inventory the distribution of the composted amount of the EU-OECD countries and Turkey in (Öko-Institut/IFEU 2010) was used, according to which 1/13 comprises residual-waste composting and 12/13 is composting of source-segregated organic waste. For the EU-OECD countries and Turkey this yielded a residual-waste fraction in the composted amount of 8%. For Chile, (OECD 2013) designates part of the treated waste as "other disposal". It was assumed that this amount is attributable to residual-waste composting.

The emission factors for recycling are the agreed standard values described in Section 4.2.4. For the OECD countries a "medium" quality was assumed. The direct emissions of plastic recycling were calculated using the particular electricity mix of the countries concerned,[18] resulting in different specific net emission factors (Table 17). The lower the fossil share in the national electricity mix, the higher the net credits (see Table 19).

[18] Only for plastic recycling and incineration with energy are national electricity mixes taken into account when calculating the individual emission factors. For simplification, for all other treatment methods standard energy values are used.

Table 17: Specific net emission factors for plastic recycling in the OECD member states

America		Europe, Turkey and Israel		Japan, South Korea and Pacific	
Values in kg CO_2-eq/t plastic					
USA	-770	EU-OECD countries	-1,025	Australia	-500
Canada	-1,287	Switzerland	-1,497	New Zealand	-889
Mexico	-944	Norway	-1,494	Japan	-929
Chile	-977	Iceland	-1,494	South Korea	-929
		Turkey	-926		
		Israel	-820		

Composting was assumed to involve simple open composting (see Section 4.2.6). The direct emissions of residual-waste composting are the same as the direct emissions of segregated composting. Because of the high level of contaminants in the residual waste, no benefit is credited for the compost produced. The residual-waste composting thus produces a debit of 60 kg CO_2-eq/t_{input}.

5.3.2 Landfill

Both the OECD and Eurostat only provide data on total landfilling. For more precise differentiation between landfills with and without gas collection, data from National Inventory Reports (NIR) or other studies were used if available. If no details were available, the mean effective gas collection efficiency obtained for the EU-OECD countries was used.

The effective gas collection efficiency for the EU-OECD countries is calculated as the weighted mean of the individual countries' gas collection efficiencies reported in (NIR 2012f) with application of the 50% cap (see Section 4.2.5).The values reported in (NIR 2012f) and the figures used to calculate the weighted mean are documented in the Annex (Table 66). The relatively low gas collection efficiency of the EU-OECD countries of 34.6% is due not so much to the 50% cap as to the fact that the countries with high effective gas collection efficiencies now landfill only very small untreated quantities or none at all. For example, Germany[19] and Belgium with effective gas collection efficiencies of 45% and 50% respectively send virtually no untreated waste to landfill, while Spain and Poland with effective gas collection efficiencies of 20% and 17% respectively landfill larger amounts.

In Japan pre-treatment and landfilling methods have been developed to reduce methane emissions. Gas collection is not usual in Japan (NIR 2012d). Landfill methods include in particular semi-aerobic landfill, in which the formation of methane is reduced by ventilation. In 2010 63.5% of landfills for municipal waste operated on semi-aerobic principles (NIR 2012d). The ventilation of the landfill cuts the amount of methane produced by about half by comparison with anaerobic landfill. For this reason a weighted MCF of 0.68 was assumed for Japan – i.e. the remaining 36.5% of landfills continued to be rated as MCF=1. This method of calculation probably significantly overestimates the methane emissions of Japanese landfills.

[19] Ban on landfilling of untreated waste since 2005.

The effective gas collection efficiencies calculated or assumed for the OECD countries are shown in Table 18. For the three regions and the OECD as a whole this results in the following effective gas collection efficiencies:

"America"	43.7%
"Europe, Turkey and Israel"	31.0%
"Japan, South Korea and Pacific"	16.0%
All OECD	**37.9%**

Table 18: Gas collection efficiencies of the OECD countries

Country	Waste in landfill with gas collection*	Effective gas collection efficiency**	Source, comments
USA		50%	50% cap applied
Canada		40%	(EC 2014)
Mexico	40%	20%	(Öko-Institut/IFEU 2010)
Chile	40%	20%	assumption: as Mexico
EU-OECD countries		34.6%	calculated from (NIR 2012f)
Switzerland	-	-	no landfill
Norway		34.6%	assumption: as EU-OECD countries
Iceland		34.6%	assumption: as EU-OECD countries
Turkey	40%	20%	(Öko-Institut/IFEU 2010)
Israel		34.6%	assumption: as EU-OECD countries
Australia	11%	6%	(NIR 2012b)
New Zealand	63%	32%	(NIR 2012c)
Japan		0%	
South Korea		34.6%	assumption: as EU-OECD countries

**For countries for which no figure is shown, data were not provided or were not required because details of effective gas collection efficiency were available.

**For countries with no details of effective gas collection efficiency, assumptions were made; the maximum gas collection efficiency was set at 50%, even if countries have reported higher values.

For the majority of countries the composition of the landfilled waste is not known. For this reason the recycled or composted fraction was first subtracted from the original amount of the fractions. The remaining quantities were calculated in accordance with the ratio between landfill, incineration and residual-waste composting using the following formula:

$$m_{dep,i} = (m_{ges,i} - m_{rec,i}) * \frac{m_{dep,ges}}{(m_{dep,ges} + m_{ver,ges} + m_{rmk,ges})}$$

m_(dep,i)= landfilled amount of fraction i
m_(ges,i)= total amount of fraction i
m_(rec,i)= recycled/composted amount of fraction i
m_(dep,ges)= total amount to landfill
m_(ver,ges)= total amount incinerated
m_(rmk,ges)= total amount to residual-waste composting

The statistical data on the landfilled amount relate to landfill as the final treatment. This means that some of the quoted landfilled waste amounts are (biologically) pre-treated. For this reason the composition of the landfilled waste calculated in this inventory may not correspond to the actual composition. Since the size of the organic component is sometimes set too high, this leads to an overestimate of the methane emissions; because of the lack of information on the pre-treated amount, this overestimate is accepted as part of a conservative approach. The calculation for landfill uses the procedure for managed landfill described in Section 4.2.5. For the OECD member countries the default values in IPCC (2006) are used; country-specific figures were calculated only for the gas collection efficiency (see above). The calculation uses an MCF of 1 (exception: Japan, see above) and an oxidation factor of 0%.[20] Based on the results of the assessment of the USA (Section 5.9.2), it was assumed for the collected gas that 50% is used in small-scale CHP and 50% is flared off. The calculation of the use in small-scale CHP was performed as described in Section 4.2.5. In all countries, 50% of the heat obtained substitutes heating oil and 50% substitutes natural gas. The standard emission factor for heat was taken from (Öko-Institut/IFEU 2010) and amounts to 0.334 kg CO_2-eq/kWh.

Table 19: Emission factors (EF) for the electricity mix of the OECD countries

Country	Electricity mix EF [kg CO_2-eq/kWh]	Source
USA	0.775	(IFEU database)
Canada	0.230	(ecoinvent V3)
Mexico	0.592	(ecoinvent V3)
Chile	0.557	(ecoinvent V3)
EU (OECD)	0.507	(ecoinvent V3)
Switzerland	0.009	(ecoinvent V3)
Norway	0.011	(ecoinvent V3)
Iceland	0.011	assumption: as Norway[1)]
Turkey	0.611	(ecoinvent V3)
Israel	0.723	estimated[2)]
Australia	1.061	(ecoinvent V3)
New Zealand	0.650	(Philpott and Downward 2010)
Japan	0.608	(ecoinvent V3)
South Korea	0.608	assumption: as Japan[3)]

1) Similar electricity mix (ecoinvent by comparison with (NEA 2014)), both approx. 95% electricity from hydropower and geothermal.

2) Similar electricity mix to Australia (ecoinvent V3.1 2014): 75% coal, 15% natural gas, 10% other. Israel (Niv 2011): 65% coal, 33% natural gas, 2% other. Because of the smaller coal and larger natural gas fraction, the EF for Israel was reduced.

3) Similar electricity mix to Japan (ecoinvent V3.1 2014): 24% hard coal, 26% natural gas, 24% nuclear, 12% oil, 10% renewables, remainder other. South Korea (EIA 2014): 30% coal, 25% natural gas, 25% nuclear, 9% oil, 11% renewables.

[20] For some of the OECD countries it would be possible to apply an oxidation factor of 10% to well-managed landfills. However, this would require more extensive research. For simplification, the figure of 0% was applied as a conservative estimate when drawing up the inventory.

The emission factors for the electricity mix vary from country to country and are taken mainly from (ecoinvent V3.1 2014). For some countries, no values were available in ecoinvent. For these countries other sources had to be used or assumptions made. Table 19 shows the emission factors used for the electricity mix in the various countries. For the three regions these yield the following specific emission factors in the form of equal-weighted means. It is these that are used as standard values in the calculation of the direct emissions of sorting:

"America"	0.539 kg CO_2-eq/kWh
"Europe, Turkey and Israel"	0.312 kg CO2-eq/kWh
"Japan, South Korea and Pacific"	0.732 kg CO2-eq/kWh

Table 20 shows the results for landfill by country and region and for the OECD as a whole. The table gives the uncollected methane emitted into the atmosphere, the collected methane, the GHG credits for use of a proportion (50%) of the collected methane in small-scale CHP and the resulting net outcome in each case. The amount of emitted methane includes both the diffuse emissions from landfill and the 1% methane slip when the gas is used in small-scale CHP (see Section 4.2.6).

Table 20: Emitted and collected methane and GHG credits for small-scale CHP

Country/region	Methane emitted* [1,000 t CO_2-eq/a]	Methane collected [1,000 t CO_2-eq/a]	Credit small-scale CHP [1,000 t CO_2-eq/a]	Net [1,000 t CO_2-eq/a]
USA	80,075	79,677	-9,613	70,462
Canada	20,749	13,787	-879	19,870
Mexico	26,909	6,719	-682	26,227
Chile	5,921	1,478	-145	5,776
"America" overall	**133,654**	**101,661**	**-11,319**	**122,335**
EU (OECD)	59,712	31,535	-2,921	56,791
Switzerland	-	-	-	-
Norway	24	12	-0,51	23
Iceland	24	13	-0,52	23
Turkey	21,997	5,492	-569	21,428
Israel	3,878	2,083	-240	3,638
"Europe, Turkey and Israel" overall	**85,634**	**39,136**	**-3,731**	**81,903**
Australia	8,973	472	-71	8,902
New Zealand	1,411	937	-101	1,310
Japan	774	0	-	774
South Korea	2,261	1,214	-125	2,136
"Japan, South Korea and Pacific" overall	**13,419**	**2,624**	**-297**	**13,122**
OECD overall	**232,707**	**143,420**	**-15,347**	**217,360**

*diffuse emissions from landfill and methane slip from small-scale CHP (percentage < 0.5%)

5.3.3 Incineration

In OECD and Eurostat the waste streams for incineration are divided into incineration with and without energy recovery. An exception is Canada, for which no data on this distinction are available. For Canada the ratio of incineration "with energy" to that "without energy" in the EU-OECD countries was therefore used (84% with, 16% without).

For "Europe, Turkey and Israel" the net electrical efficiency of plants with energy recovery was taken from (CEWEP 2012). As the weighted mean of 314 plants studied, (CEWEP 2012) quotes gross production of 15% electrical energy and 37.1% thermal energy. The quoted values for internal use and auxiliary energy are deducted from this (3.6% for electrical energy and 5.5% for thermal energy). This results in net energy recovery rates of 11.4% for electricity and 31.6% for heat. For the USA the data in Section 5.9.2 were used. Because of a lack of specific data for Canada, Chile, Mexico, Japan, South Korea and Pacific, the efficiencies for Europe were used for these countries, as is done in some cases in (OECD 2012).

For the USA, the characteristics of residual waste used are those from Section 5.9.2. For "America" (excl. USA) and "Japan, South Korea and Pacific" the region-specific waste characteristics were calculated (based on the waste composition after removal of recycled and composted waste). For "Europe, Turkey and Israel" the waste characteristics for Turkey and Israel were likewise calculated on the basis of the waste composition and combined after weighting with the waste characteristics of the EU-OECD countries (see Table 37) from (Öko-Institut/IFEU 2010). The resulting values are shown in Table 21.

Table 21: Waste characteristics of residual waste in the three regions

	America (excl. USA)	Europe, Turkey and Israel	Japan, South Korea and Pacific
Calorific value [MJ/kg]	8.8	9.2	8.4
Total C [% SM]	23.3%	24.4%	22.3%
Fossil C [% SM]	8.9%	8.9%	9.3%
Biogenic C [% SM]	14.4%	15.5%	13.1%

These waste characteristics and efficiencies together with the heat/electricity emission factors yield the corresponding debits and credits for incineration. Table 22 shows the specific debits and credits and the resulting net outcome for incineration with and without energy recovery for the OECD countries. Negative net values indicate a net credit, positive ones a net debit. Where no figure is quoted, no waste is consigned to that incineration method in that country. The specific debits are identical for all the countries in a region, because identical boundary conditions were assumed (except USA). The specific credits vary depending on the particular proportion of incineration with energy recovery and the country-specific emission factor for the electricity mix (Table 19). For the OECD inventory, in contrast to the inventories for the individual countries of India, Egypt and the USA, the electricity mix and not the marginal electricity was used in offsetting the electricity generated, because the marginal electricity could not be reliably calculated for each country or for the region. The influence on the result is described in Section 5.9.3 using the USA as an example.

Table 22: Specific GHG debits and credits for incineration

	Incineration without energy			Incineration with energy		
	Debit	Credit	Net	Debit	Credit	Net
	[kg CO_2-eq/t waste]					
USA	-	-	-	393	-417	-25
Canada	356	0	356	356	-338	18
Mexico	-	-	-	-	-	-
Chile	356	0	356	356	-430	-74
EU-OECD countries	357	0	357	357	-431	-74
Switzerland	-	-	-	357	-287	71
Norway	-	-	-	357	-288	70
Iceland	357	0	357	357	-288	70
Turkey	-	-	-	-	-	-
Israel	-	-	-	-	-	-
Australia	-	-	-	370	-545	-176
New Zealand	-	-	-	-	-	-
Japan	370	0	370	370	-424	-55
South Korea	370	0	370	370	-424	-55

5.4 Results: waste management in the OECD

5.4.1 The standard case

Table 23 shows the results of the GHG inventory broken down by countries/regions and disposal methods. The specific net results for recycling are negative for all fractions in all countries and hence result in a net credit. In all countries, therefore, recycling contributes to GHG mitigation. By contrast, composting, landfill, incineration without energy recovery and residual-waste composting always result in net debits. The debits or credits as a result of incineration with energy recovery vary from country to country because each inventory was calculated using the appropriate country-specific emission factor for the electricity mix; for the USA country-specific efficiencies were also used. In Switzerland, Norway and Iceland the low electricity emission factors (and hence low electricity credits) result in a debit on account of the incineration with energy. However, in the three regions incineration with energy always results in a GHG credit overall.

Figure 3 shows the contribution of the individual disposal methods to the overall result (scaled to one t waste). From this it is particularly clear that landfill represents the largest GHG debit, while recycling produces the largest credit. Even in the event of high specific credits for electricity (as in "Japan, South Korea and Pacific"), incineration with energy recovery accounts for only about 9% of the total credit.

Table 23: Absolute results for global warming by disposal method and country

Region	Country	Recycling [1,000 t CO_2-eq]	Composting [1,000 t CO_2-eq]	Landfill [1,000 t CO_2-eq]	Incineration without energy [1,000 t CO_2-eq]	Incineration with energy [1,000 t CO_2-eq]	Residual-waste composting [1,000 t CO_2-eq]	Total [1,000 t CO_2-eq]
America	USA	-58,418	148	70,462		-593		11,599
	Canada	-5,994	19	19,870	74	19		13,988
	Mexico	-3,331	6	26,227				22,902
	Chile	-20		5,776	3	-0.07	34	5,793
Europe, Turkey and Israel	EU (OECD)	-71,321	280	56,791	3,225	-3,526	165	-14,386
	Switzerland	-2,137	7			197		-1,933
	Norway	-614	3	23		94		-494
	Iceland	-36	0.05	23	0.71	0.49		-12
	Turkey		1	21,428			0,88	21,431
	Israel			3,638				3,638
Japan, South Korea and Pacific	Australia	-4,474		8,902		-27		4,402
	New Zealand			1,310				1,310
	Japan	-8,698		774	1,064	-2,026		-8,887
	South Korea	-8,470	2	2,136	119	-188		-6,401
OECD	Total	-163,514	466	217,362	4,486	-6,049	200	52,951

The debits for collection, sorting and transport are not given here because they were calculated only for the regions as a whole and not for individual countries; the debits per region are shown in Table 28

Figure 3: Net contribution of the disposal methods to global warming (scaled to one t waste)

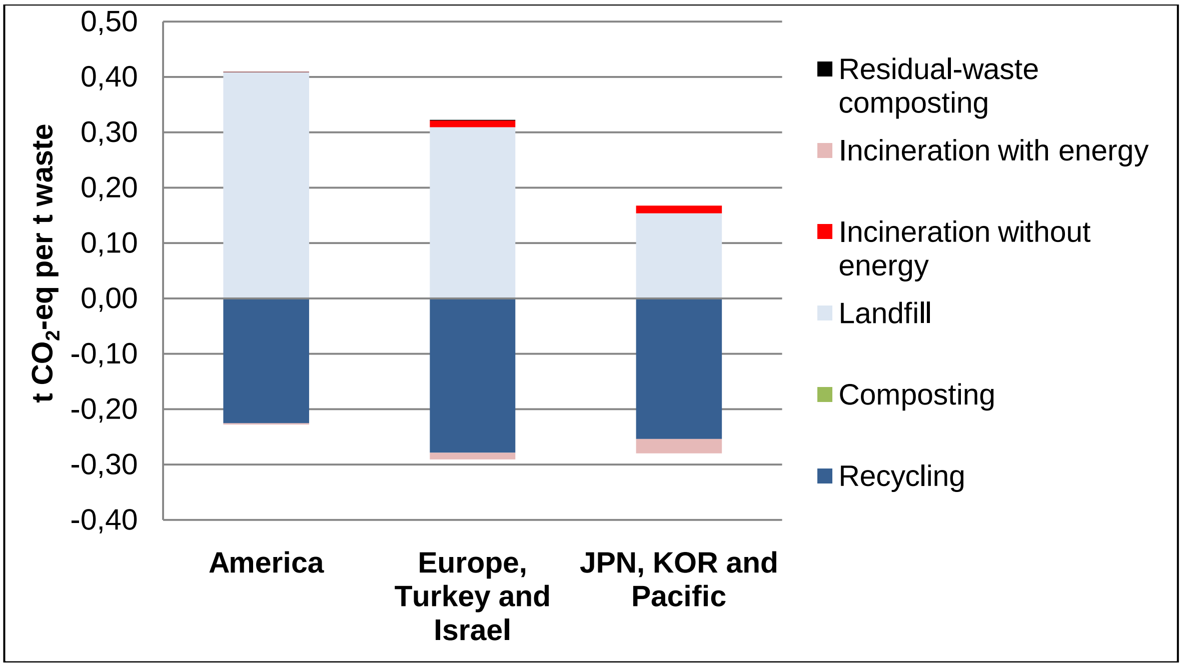


Table 24 and Figure 4 show the overall results for the countries/regions and for the OECD as a whole. They show the total emissions and the specific emissions per capita and per tonne of waste.

In all countries of the “America” region, the net result of waste management is a GHG debit. The total net debit (including collection, sorting and transport) in the “America” region is around 60 Mt CO_2-eq. In the “Europe, Turkey and Israel” region, too, the overall net result – despite the net credits in the EU-OECD countries, Switzerland, Norway and Iceland – is a debit of around 13 Mt CO_2-eq. In the “Japan, South Korea and Pacific” region the net result of the net credits in Japan and South Korea is an overall GHG credit of around -7 Mio. t CO_2-eq. In the OECD overall the net result of current waste management practices is a GHG debit of around 66 Mt CO_2-eq.

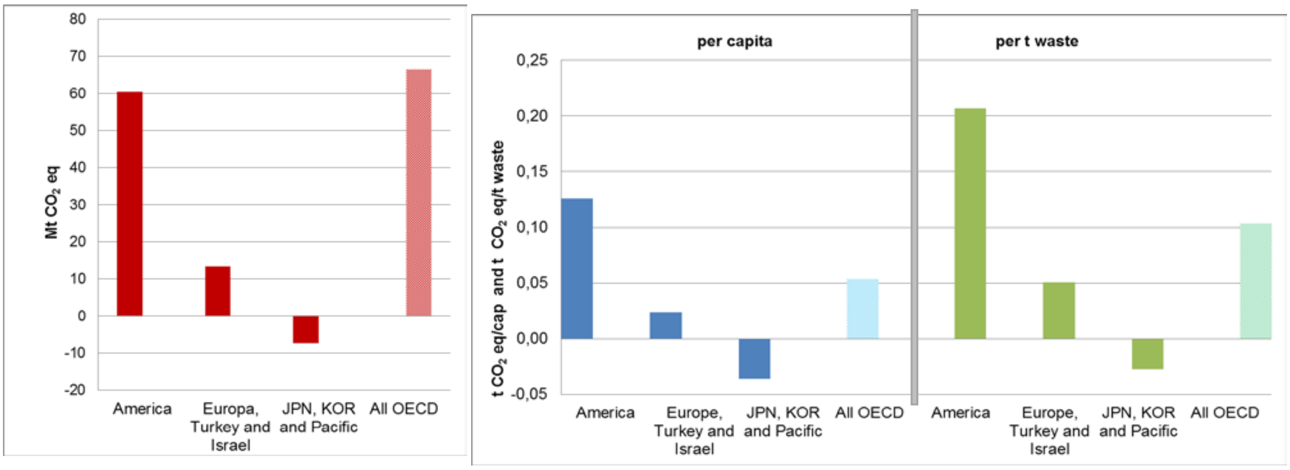
Per capita and per tonne of waste, Mexico and Chile have the highest specific net debits in the “America” region. In the “Europe, Turkey and Israel” region, only Turkey and Israel have specific net debits; all the other countries in this region have specific net credits. In “Japan, South Korea and Pacific”, waste management practices in Australia and New Zealand result in specific net debits, while South Korea and Japan yield specific net credits.

Table 24: GHG balance by country – net results in absolute terms, per capita and per tonne of waste

Region	Country	1,000 t CO_2-eq	t CO_2-eq/cap	t CO_2-eq/t waste
America	USA	11,599	0.037	0.052
	Canada	13,988	0.401	0.407
	Mexico	22,902	0.197	0.655
	Chile	5,793	0.333	0.937
Europe, Turkey and Israel	EU-OECD countries	-14,386	-0.031	-0.063
	Switzerland	-1,933	-0.242	-0.347
	Norway	-494	-0.099	-0.211
	Iceland	-12	-0.037	-0.107
	Turkey	21,431	0.286	0.968
	Israel	3,638	0.461	0.800
Japan, South Korea and Pacific	Australia	4,402	0.200	0.314
	New Zealand	1,310	0.298	0.616
	Japan	-8,887	-0.070	-0.176
	South Korea	-6,401	-0.050	-0.344
OECD	overall (without collection, sorting and transport)	**52,951**	**0.043**	**0.082**
OECD	overall (with collection, sorting and transport)*	**66,358**	**0.053**	**0.102**

* Direct emissions for collection, sorting and transport are included only in the OECD total; they were calculated for each region as a whole; the corresponding debits are shown in Table 28.

Figure 4: GHG balance by region – net results in absolute terms, per capita and per tonne of waste



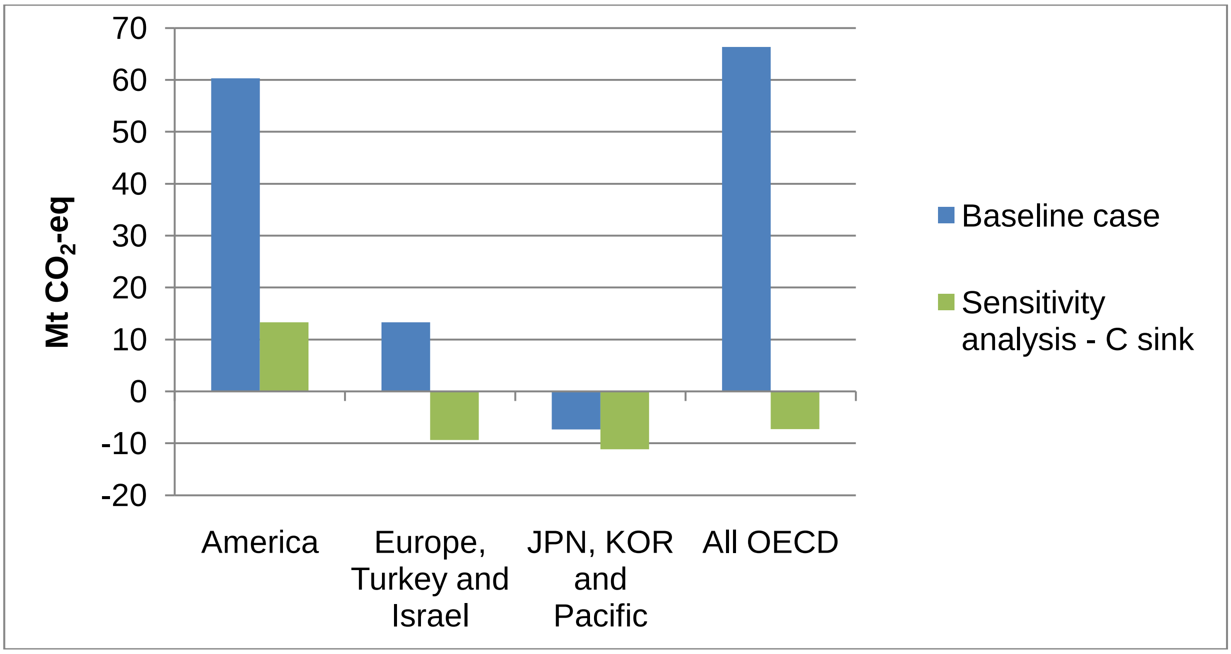
Direct emissions for collection, sorting and transport are included.

5.4.2 Sensitivity

As a sensitivity for the standard case, the results taking account of the C sink are shown. For the reasons mentioned in Section 4.1.2, the C sink is given for information only. The assessment covers the C sink in connection with landfill and with quality composts. The results are shown in Figure 5.

Consideration of the C sink results in a significant reduction in the net result for the "America" region and the "Europe, Turkey and Israel" region. In the "America" region there is a noticeably smaller net debit (around one-fifth of the value in the standard case), while for the "Europe, Turkey and Israel" region the result is reversed and becomes a net credit. The net credit in the standard case for "Japan, South Korea and Pacific" improves only slightly when the C sink is taken into account. The underlying factor is the amount of waste landfilled in the particular region. In "America" 60% of all waste is landfilled; in "Europe, Turkey and Israel" the figure is 38% and in "Japan, South Korea and Pacific" only 18%.

Figure 5: Net results taking account of the C sink by comparison with the baseline case



In the overall result for the OECD, consideration of the C sink also produces a reversal of the result, changing it from a significant net debit to a slight net credit. The debits as a result of landfill fall by about 33% as a result of the C sink.

5.5 Future scenarios to 2030

Two future scenarios for the year 2030 – a medium one and an ideal one – assess the climate change mitigation potential of changes in waste management in all OECD countries towards a closed-cycle management system. These scenarios are contrasted with the **business as usual (BAU)** scenario, which corresponds to an unchanged waste management system in 2030.

In the ideal scenario it is assumed that the landfilling of municipal waste ceases in all countries and that state-of-the-art technology for recycling waste is utilised. The medium scenario illustrates the climate change mitigation potential in the event that available recycling opportunities are not fully utilised. This scenario also represents the transition phase between BAU and the ideal scenario.

5.5.1 Description of the medium scenario to 2030

For the medium scenario the recycled and composted quantities in the individual fractions were increased. They are calculated roughly as the mean of the amounts in the BAU scenario and those in the ideal scenario (for the latter see Section 5.5.2). Conversely, it was assumed that the quantity of residual waste landfilled would be cut by 50%. At the same time, it was assumed that only landfills with gas collection will be used, with the result that the mean weighted effective gas collection efficiency in the OECD as a whole increases from 37.9% to 50% (50% cap). In addition, no more residual waste is composted. Of the remaining residual waste, 80% goes to incineration with energy recovery and 20% to anaerobic MBT with output of an RDF fraction.[21] The assumption that 20% of the waste remaining after recycling is treated in an MBT plant facilitates the progressive and controlled phase-out of the landfilling of MSW because around 30% of this waste quantity continues to be sent to landfill as MBT residue after biological treatment, but with significantly reduced residual methane emissions. In accordance with the system adopted in official statistics, the MBT is not designated as a final destination. This means that the output streams of the MBT plant are allocated to the relevant final disposal methods (recycling, landfill or incineration). Of the RDF fraction, 50% is used for energy in RDF-fired cogeneration plants and 50% as a substitute for coal in power plants and cement works. Both the emissions of the MBT facility and the credits through the use of the biogas and use of the RDF for energy are allocated in the results charts and tables to “incineration with energy”. The modelling of the MBT (mass flows, energy recovery, RDF characteristics) is described in Section 4.2.7.

The thermal efficiencies of incineration with energy recovery in municipal solid waste incinerators were increased (Table 25). Although modern MSWIs achieve net electrical efficiencies of up to 31% at maximum power generation and total thermal efficiencies of up to 87% at maximum heat generation, the realistic rate of increase in average efficiencies in plants of the future depends to a large extent on the conditions of heat output. In district heating grids with high feed-in temperature the electrical efficiencies fall noticeably. As realistic mean net efficiencies for MSWIs in the EU in the year 2030, Prof. Reimann, the author of (CEWEP 2012), gives figures of 18% for electricity and 42% for heat (Reimann 2014). These values can be applied across the entire OECD and are used for the future scenarios (Table 25).

Table 25: Net efficiencies of municipal solid waste incinerators in the status quo and future scenarios

	Electricity (BAU)	Heat (BAU)	Electricity (medium, ideal)	Heat (medium, ideal)
USA	19%	0%	18%	42%
all except USA	11.4%	31.6%	18%	42%

The increase in the recycled and composted amounts changes the waste composition and hence the characteristics of the waste after recycling and composting. These characteristics

[21] With the exception of Japan. In Japan incineration already accounts for 79% of the overall management methods in the standard case (with a recycling percentage of 19%). It cannot be expected that Japan will reduce its incineration capacities in favour of MBT plants. In the medium scenario for Japan all the residual waste therefore goes to incineration with energy.

were recalculated for the regions on the basis of the original waste composition less the waste recycled and composted in the medium scenario. An exception is the USA, for which the values from the USA inventory were used. The EU-OECD countries are another exception: for them the characteristics were retained in simplified form. Table 26 shows the calculated characteristics of the residual waste in the medium future scenario.

Table 26: Characteristics of residual waste in the medium future scenario (own calculations)

	America (excl. USA)	Europe, Turkey and Israel	Japan, South Korea and Pacific
Calorific value [MJ/kg]	9.0	9.4	8.6
Total C [% SM]	23.8%	24.5%	22.8%
Fossil C [% SM]	8.9%	9.0%	10.1%
Biogenic C [% SM]	14.9%	15.5%	12.7%

5.5.2 Description of the ideal scenario to 2030

In the ideal scenario, no waste is sent to landfill. As in the medium scenario, no composting of residual waste occurs. The recycling and composting rates[22] for the individual fractions are increased further. According to expert estimates, the following recycling and composting rates reflect the state of technology in some EU countries:

- food waste 70%
- garden waste: 80%
- plastics: 60%
- glass: 70%
- ferrous metal: 90%
- aluminium: 70%
- textiles: 50%

For food waste, source-segregated collection and 100% anaerobic digestion with post-composting are assumed. As in the medium scenario, of the waste remaining after removal of recyclables, 80% goes to incineration with energy recovery and 20% to mechanical-biological treatment. There is a change in this MBT in that instead of anaerobic treatment it involves mechanical-biological stabilisation (MBS), which is designed to produce a higher RDF output (see Section 4.2.7). As in the medium scenario, 50% of the RDF fraction is used for energy in RDF power plants and 50% as a substitute for coal in power plants and cement works. As in the medium scenario for the MBT plant, the operating emissions and credits associated with the MBS treatment are allocated to “incineration with energy” as a management method. In the ideal scenario the thermal efficiencies of incineration in the MSWI are increased as in Table 25.

[22] Biological treatment in the future scenarios is assessed partly as pure composting and partly as anaerobic digestion with post-composting of the digestate. All references to composting rates here cover both methods. All recycling rates relate to the output of the processing plants as a proportion of the total quantity of a waste fraction in the generated waste.

The increased recycling and increased composting and anaerobic digestion change the composition of the waste and hence its characteristics, which were recalculated for the ideal scenario as described above for the medium scenario (Table 27).

Table 27: Characteristics of "waste to incineration" in the ideal future scenario (own calculations)

	America (excl. USA)	Europe, Turkey and Israel	Japan, South Korea and Pacific
Calorific value [MJ/kg]	9.4	8.5	9.3
Total C [% SM]	25.1%	24.2%	24.2%
Fossil C [% SM]	9.2%	9.0%	11.4%
Biogenic C [% SM]	16.0%	15.2%	12.9%

The mass flows in the future scenarios by comparison with the BAU scenario in the three regions and for the OECD as a whole are shown in Table 67 in the Annex.

5.5.3 Results of the baseline comparison

Table 28 shows the results of the three future scenarios – namely "business as usual" (BAU), "medium scenario" (medium) and "ideal scenario" (ideal) – by disposal methods for the various regions. The BAU scenario corresponds to the standard case and depicts the outcome if nothing were to change. In the medium scenario the total debits of all OECD countries, amounting to around 66 Mt CO_2-eq per year, become a GHG credit of around -154 Mt CO_2-eq per year. The total GHG reduction is thus around 220 Mt CO_2-eq per year. In the medium scenario the 217 Mt CO_2-eq per year from methane emissions from landfill in the BAU scenario is already reduced by around 143 Mt CO_2-eq per year. The remaining and still significant GHG reduction in the ideal scenario by comparison with the medium scenario, at a good 132 Mt CO2-eq per year, is thus considerably smaller than the improvement in the medium scenario by comparison with BAU. Overall, through optimised waste management in all OECD countries a GHG credit of -287 Mt CO_2-eq per year can be achieved in the ideal scenario, representing an improvement of 353 Mt CO_2-eq per year by comparison with the BAU scenario.

In the medium scenario the reduction in GHG emissions from landfill is achieved partly through the fall in the amount of landfilled waste from approx. 288 to 155 Mt per year (see Table 67 in the Annex) and partly through the improvement in the mean effective gas collection efficiency from 37.9% to 50%. In addition, the pre-treatment of some of the waste to be landfilled leads to a reduction in landfill gas emissions. Alongside the prevention of methane emissions from landfill, an important role in the reversal from a significant GHG debit to a sizable GHG credit is played by the increase in recycled quantities from around 163 Mt per year in the BAU scenario to around 192 Mt per year in the medium scenario and 238 Mt per year in the ideal one. Even under BAU, recycling makes a contribution to GHG reduction of around -165 Mt CO_2-eq per year; in the medium scenario this increased by 31 Mt CO_2-eq per year and in the ideal scenario by a further 32 Mt CO_2-eq per year. It should be noted that incineration with energy not only generates credits for electricity and heat use or substitution of coal but also, through the shifting of quantities (incineration instead of landfill), contributes significantly to the prevention of methane emissions from landfill.

Table 28: Comparison of the results of the future scenarios to 2030 for the regions by disposal methods

Region	Future scenario	Recycling	Compost-ing	Anaerobic digestion	Landfill	Incineration without energy	Incineration with energy	Residual-waste composting	CLN, TSP, SOR	Total
		1,000 t CO_2-eq								
America	BAU	-67,764	174		122,336	77	-574	34	6,041	60,323
	medium	-88,950	357		47,541		-23,050		6,907	-57,195
	ideal	-110,523	279	-3,144			-38,758		8,002	-144,143
Europe, Turkey and Israel	BAU	-74,107	291		81,904	3,226	-3,234	166	5,094	13,339
	medium	-81,038	445		24,699		-19,115		5,313	-69,696
	ideal	-88,030	290	-2,734			-23,941		5,940	-108,475
Japan, South Korea and Pacific	BAU	-21,642	2		13,122	1,183	-2,241		2,271	-7,305
	medium	-24,629	89		2,441		-7,868		2,396	-27,572
	ideal	-27,917	93	-1,093			-7,896		2,525	-34,287
All OECD	BAU	-163,514	466	0	217,362	4,486	-6,049	200	13,407	66,358
	medium	-194,617	890	0	74,681	0	-50,034	0	14,616	-154,464
	ideal	-226,471	663	-6,970	0	0	-70,595	0	16,467	-286,906

Collection (CLN), transport (TSP) and sorting (SOR) were calculated collectively for the regions (see Section 5.3)

Figure 6 shows the specific net contributions of the disposal methods for the three future scenarios for the OECD as a whole. The fall in net debits from landfill and the increase in net credits from recycling are evident.

Figure 6: Net contributions to global warming of the disposal methods in the future scenarios for the OECD (per tonne of waste)

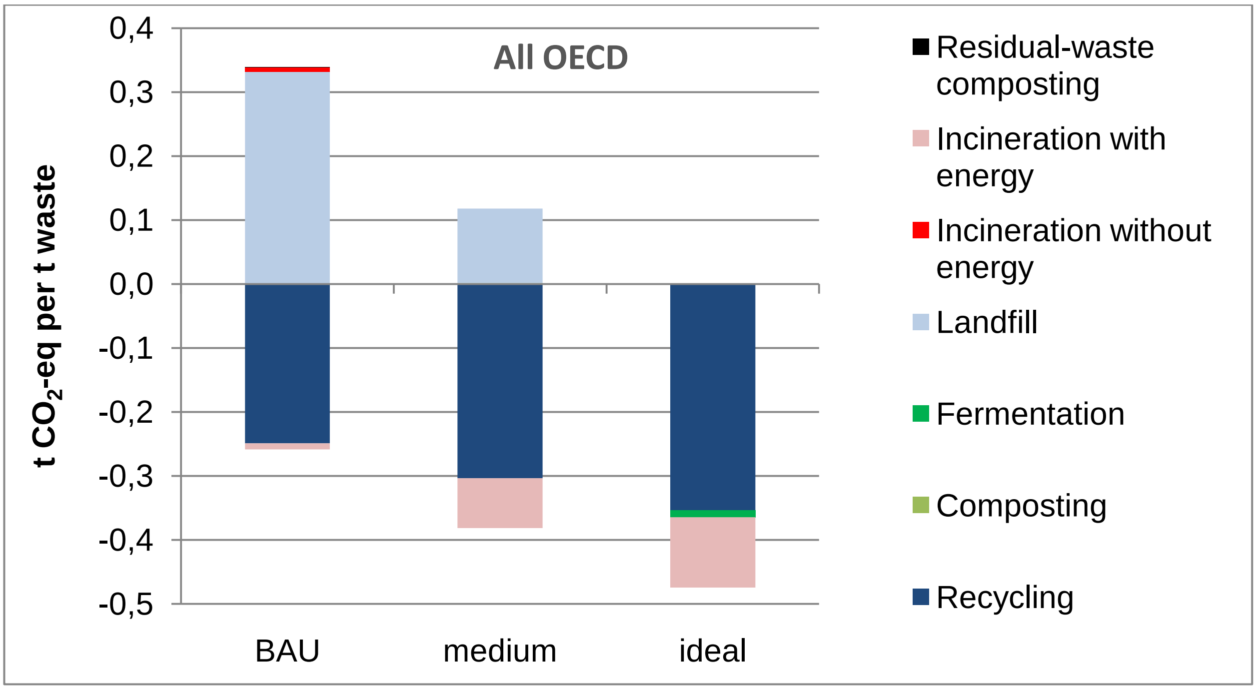
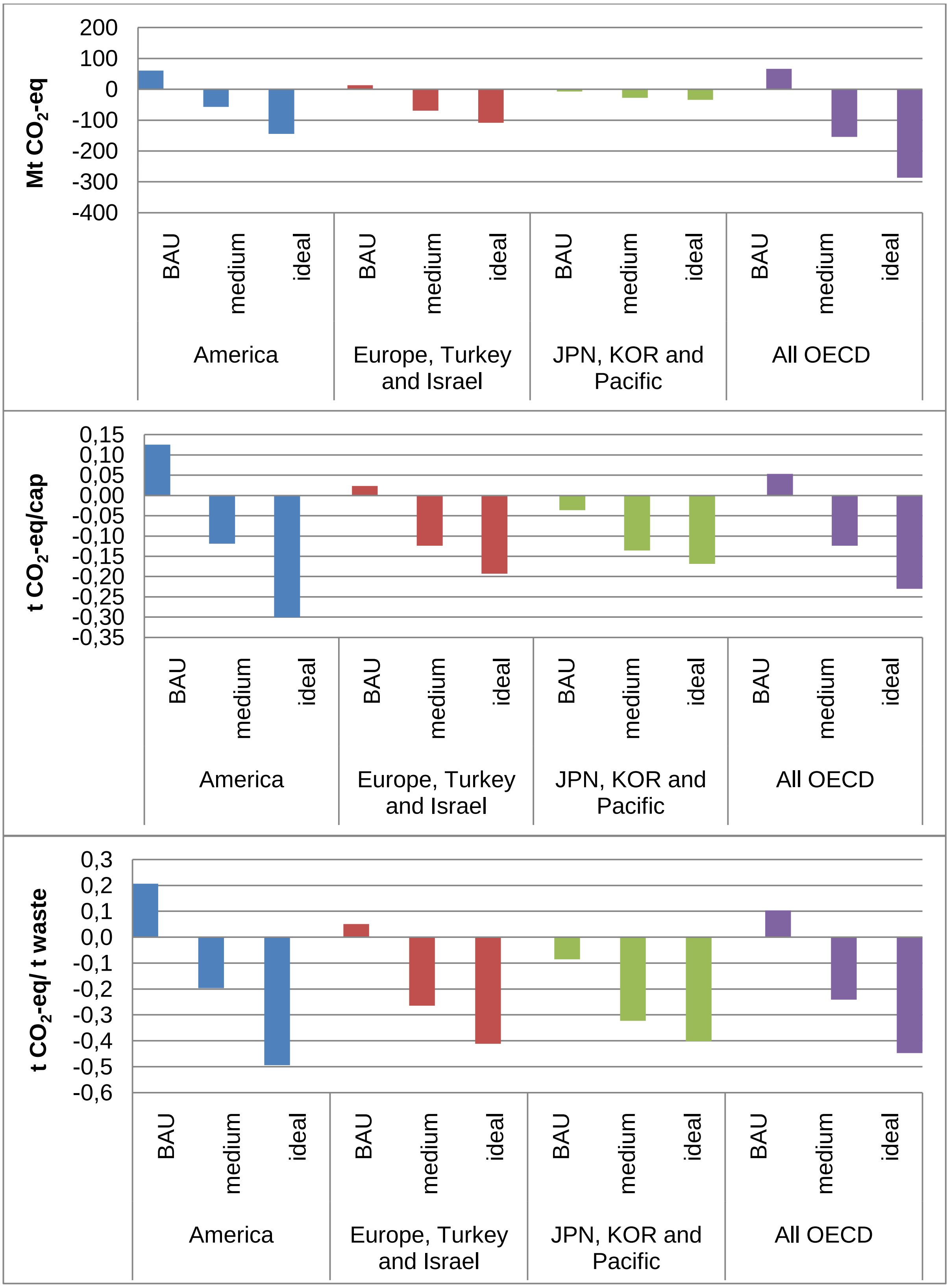


Figure 7 shows the net results of the three future scenarios by region. In addition to the absolute net results, the specific net results per capita and per tonne of waste are also shown. This method of depiction enables the results to be evaluated irrespective of the absolute volume of waste (results per tonne), and relating the results to population size (results per capita) enables the per-capita performance of waste management to be compared.

This shows that even in the ideal scenario, assuming that waste is handled in virtually the same way in all OECD countries, differences between the regions remain.

The specific results for incineration in the medium and ideal future scenarios and the specific results for anaerobic digestion in the ideal scenario, broken down by country, are given in the Annex (Tables 68 – 70).

Figure 7: Comparison of the results of the future scenarios – net results in absolute terms, per capita and per tonne of waste



Collection, sorting and transport are included

5.5.4 Results of the sensitivity analyses

For the future scenarios the following two aspects were considered as sensitivities and analysed as such:

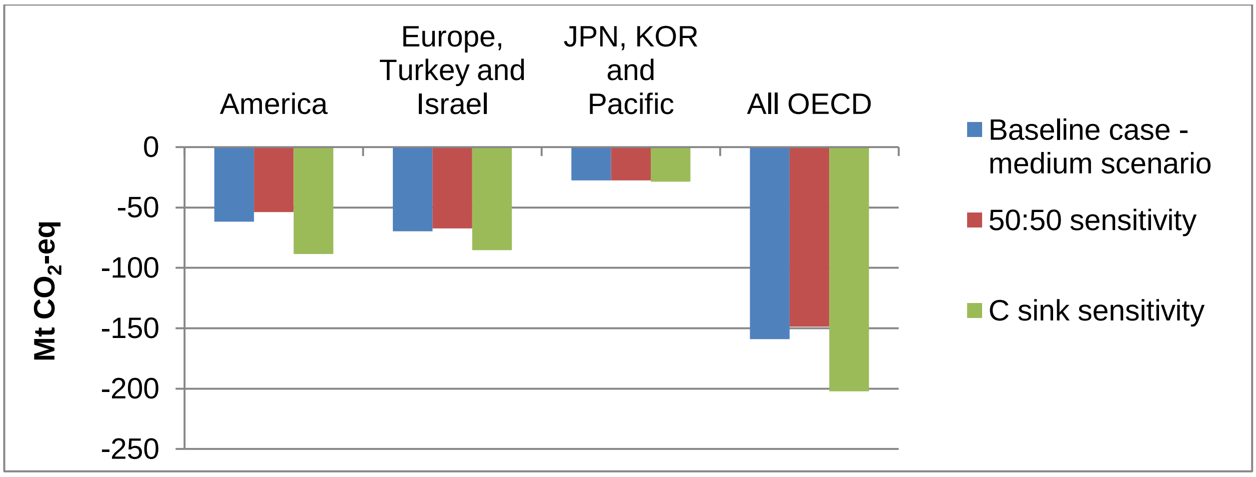
1. Result taking account of the C sink
2. Results with modified ratio between "incineration with energy" and treatment via MBT or MBS (50:50 instead of 80:20)

A decision was taken not to perform a sensitivity analysis for the OECD countries in which textile recycling was considered because very few member states report textiles separately and there would therefore be very little difference in the result.

Figure 8 shows the net result for the two sensitivities and the baseline case in the medium future scenario. Consideration of the C sink produces a slight improvement in the net result in the medium scenario in the "Europe, Turkey and Israel" region and the "America" region. For the "Japan, South Korea and Pacific" region it produces very little difference. There are two reasons for this: firstly, less waste is landfilled here than in other regions, and secondly no MBT was included in the assessment of Japan (footnote 23) so that there would be no MBT residue for which a C sink could be applied. Because of the reduction in landfill, the C sink has a significantly smaller impact on the result than in the baseline case (Section 5.4.2).

The shift in the MSWI/MBT ratio in the direction of more pre-treatment by MBT ("50:50 sensitivity") results in a slight worsening of the net result of the medium scenario for the "America" region and for "Europe, Turkey and Israel". In the "Japan, South Korea and Pacific" region the change is minimal. This is because no MBT was included in the analysis for Japan: instead, 100% incineration with energy is assumed (footnote 23).

Figure 8: Net results of the sensitivity analysis and baseline case in the medium scenario



The outcome of the 50:50 sensitivity analysis results from the overall quantities of waste sent to incineration and to landfill. The specific results for incineration and landfill improve in the sensitivity analysis. However, the quantity of waste sent to landfill increases and the quantity used for energy decreases. Table 29 shows this shift and its impact on the three regions. The absolute credits for incineration fall in the sensitivity analysis, while the absolute emissions of landfill increase. Overall the sensitivity analysis results in higher debits than the baseline case.

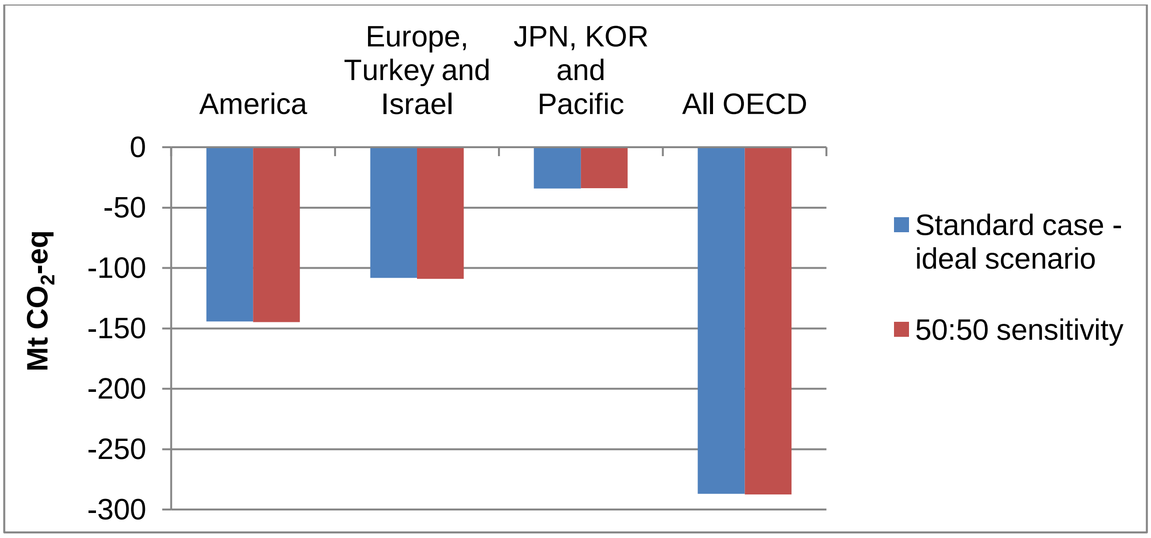
Table 29: Quantities, specific results and total emissions for incineration and MBT/landfill in the medium scenario (baseline 80:20) and the 50:50 sensitivity analysis for the three regions

	America		Europe, Turkey and Israel		Japan, South Korea and Pacific	
	Baseline 80:20	**Sensitivity 50:50**	**Baseline 80:20**	**Sensitivity 50:50**	**Baseline 80:20**	**Sensitivity 50:50**
Incineration - total [1,000 t]	62,410	50,317	76,020	61,290	36,438	34,828
Incineration - specific [kg CO_2-eq/t]	-369	-394	-251	-274	-216	-222
Incineration - absolute [t CO_2-eq]	**-23,050,314**	**-19,815,025**	**-19,115,008**	**-16,781,922**	**-7,868,491**	**-7,723,923**
Landfill - total [1,000 t]	91,745	98,363	54,921	62,982	8,058	8,939
Landfill - specific [kg CO2-eq/t]	518	487	450	399	303	279
Landfill - absolute [t CO2-eq]	**47,541,350**	**47,909,938**	**24,708,528**	**25,157,491**	**2,441,146**	**2,490,226**
Total (landfill plus incineration) [t CO2-eq]	**24,491,036**	**28,094,913**	**5,593,520**	**8,375,569**	**-5,427,345**	**-5,233,697**

Figure 9 shows the net result for the 50:50 sensitivity analysis and the baseline case in the ideal future scenario. The decision was taken not to show the results of the sensitivity analysis involving the C sink: this yields very little difference because no waste at all is sent to landfill and hence only quality composts contribute to the C sink (see Section 4.2.5).

The 50:50 sensitivity analysis produces a small improvement in the results of the ideal future scenario in the regions. Overall the change is not relevant.

Figure 9: Net results of the 50:50 sensitivity analysis and baseline case in the ideal scenario



5.6 Comparison with the OECD study of 2012

This section compares the assumptions and principal findings of this study with those of the study (OECD 2012). For the purposes of the present study the comparison is confined to the inventory of the OECD countries (Sections 5.1 – 5.7).

5.6.1 Regional classification of the member states

Both the regional classification and the countries considered differ between the two studies. The regional classification in (OECD 2012) is shown in Figure 67 in the Annex. In (OECD 2012) the “Europe region” was divided into “high recycling OECD Europe” and “low recycling OECD Europe”. The countries of Slovenia, Italy, Turkey and Israel – and also Chile – were not included in the inventory. In (OECD 2012) “OECD Asia” and “OECD Pacific” are sometimes treated as two regions. In this section the two are combined into one region for the purpose of comparing the two studies.

5.6.2 Emission factors

The emission factors for electricity used in the two studies differ in terms of methodology. In the present study accounting was performed largely by country (the exception being collection, transport and sorting); the calculations were made using the country-specific emission factors for electricity given in Table 19. The region-specific values for the present study given in Table 30 are equal-weighted means of the national values, which provide an approximate basis for comparison. By contrast, the region-specific emission factors in (OECD 2012) are the means of the countries assigned to a region, weighted by each country’s annual electricity generation; these were used as calculation factors in that study.

The main more relevant difference between the two studies lies in the way in which the emissions avoided as a result of the electricity generated are credited. In this study, the national specific emission factors are used (see Section 4.1.2); in (OECD 2012), by contrast, the marginal electricity defined for the region was credited. As a result of this difference alone, (OECD 2012) necessarily obtains higher credit effects as a result of incineration.

Table 30: Comparison of electricity emission factors in (OECD 2012) and the present study

	(OECD 2012) [kg CO_2-eq/kWh]			Present study [kg CO_2-eq/kWh]		
	America	Europe	Asia/Pacific	America	Europe	JPN, KOR and Pacific
Electricity mix	0.62	0.38	0.50	0.54	0.31	0.73
Marginal electricity	0.95	0.77	0.81	n.c.	n.c.	n.c.

*With regard to the electricity emission factor, (OECD 2012) does not distinguish between Asia and Pacific.
n.c. not considered

In both studies the emission factor for offsetting generated heat is taken from (Öko-Institut/IFEU 2010) and is 0.334 kg CO_2-eq/kWh.

Table 31 contrasts the emission factors for material recovery (recycling, composting or anaerobic digestion). In (OECD 2012) the different regions are sometimes treated differently, so that different values are used for “America” than for “Europe” and “Asia/Pacific”. The net emission factors for metals and plastics are higher in (OECD 2012); in the “America” region, in particular, significantly higher specific net credit values were used. For paper and cardboard, by contrast, the net emission factor for “America” is lower than in the two other regions in (OECD 2012) and also lower than in the present study. For food waste and garden waste the net debit values in (OECD 2012) are consistently somewhat higher and the net credit values for glass are consistently lower. Overall the net emission factors in (OECD 2012) are likely to result in somewhat higher credit effects than those in the present study.

Table 31: Comparison of net emission factors for material recovery in (OECD 2012) and the present study

	(OECD 2012) [kg CO_2-eq/t_{input}]			Present study [kg CO_2-eq/t_{output}]
Fraction	**America**	**Europe**	**Asia/Pacific**	**all regions**
Food waste	50	30	30	8 (-36)*
Garden waste	50	60	60	8
Paper/ cardboard	-550	-820	-820	-793
Plastic	-1,680	-1,060	-1,060	-937
Fe metals	-1,980	-1,000	-1,000	-945
Non-Fe metals	-15,020	-11,100	-11,100	-9,307
Glass	-310	-180	-180	-514

*Composting / value in brackets is the net emission factor for anaerobic digestion in the ideal scenario

In the assessment of landfill, (OECD 2012) includes not only methane emissions but also the emissions associated with operation of the site. However, the resulting difference is small. Considerably more significant is the fact that in (OECD 2012) an effective gas collection efficiency of 75% is applied generally to landfills with gas collection. In the present study, country-specific effective gas collection efficiencies were calculated, with an effective gas collection efficiency of 50% being applied as a maximum possible value (50% cap).

In relation to total landfilled amounts in the OECD, the effective gas collection efficiency is 42% (incl. landfills without gas collection) in (OECD 2012) and around 38% in the present study. In

(OECD 2012) 60% of the collected gas is flared off and 40% is used for electricity generation, with a net efficiency of 25%. In the present study 50% of the collected gas is flared off and 50% is used for energy generation (electricity and heat). In this case the net electrical efficiency is 37.5% and the net thermal efficiency 43%. In (OECD 2012) the oxidation factor is set at 10%; in this study it is 0%. As in (OECD 2012) no C sink was allowed for. From the stated boundary conditions of the two studies, roughly similar results for landfill are to be expected. Ultimately, however, the result also depends on the degradable organic carbon (DOC). (OECD 2012) uses the fraction-specific DOC values from (IPCC 2006); the resulting DOC of the landfilled waste could not be identified.

For incineration both studies considered thermal treatment in MSWIs as the standard scenario. The net efficiencies used in the two studies are shown in Table 32. Here the present study uses slightly higher efficiencies, because they relate to a more up-to-date time horizon. This results in slightly higher quantities of energy being generated. However, because of the offsetting of marginal electricity in (OECD 2012) it cannot be assumed that the net outcome of the present study is a more favourable result for incineration.

Table 32: Net efficiencies of MSWIs in (OECD 2012) and the present study

	(OECD 2012)	Present study
Net electrical efficiency	10%	11.4%
Net thermal efficiency	30%	31.6%

5.6.3 Waste quantities and disposal methods

Waste quantities and disposal methods in the two studies relate to different years and so necessarily differ. The differences are set out below so that the significance of differences in the results can be assessed.

Table 33 shows the quantities of the various fractions that are recovered or composted and the total quantities in both studies. For all fractions the quantities in the “America” region are comparable in the two studies; for all fractions other than food waste, garden waste and glass they are slightly higher in the present study. In total, an additional 39 Mt of waste is recovered or composted in “America" in the present study. The additional inclusion of Chile in the present study adds only 6.5 Mt waste to the total quantity. The total quantity in the “Europe” region is higher in (OECD 2012), although the present study additionally includes Turkey, Israel, Italy and Slovenia (see Section 5.6.1). However, in “Europe, Turkey and Israel” the amount recycled, composted or fermented is 24 Mt higher. In particular, significantly more garden waste is composted in the present study. This large difference is probably a result of the data basis. In (OECD 2012) data for Europe were taken from the OECD, while in the present study Eurostat data were used, supplemented in some cases by information from the individual countries. For Asia/Pacific, too, the total increases by 25 Mt. In the present study, significantly more paper and cardboard is recycled than in (OECD 2012).

Table 33: Comparison of the recycled quantities of the various fractions of domestic waste in (OECD 2012) and in the present study

	(OECD 2012)			Present study		
	North America [Mt]	Europe [Mt]	Asia/ Pacific [Mt]	America [Mt]	Europe, Turkey and Israel [Mt]	Japan, South Korea and Pacific [Mt]
Food waste	48	57	26	46	56	15
Garden waste	50	19	3	46	46	15
Paper/ cardboard	67	64	2	78	61	24
Plastic	25	25	10	36	28	10
Fe metals	11	12	3	19	12	3
Non-Fe metals	3	2	0	6	3	0,3
Glass	19	24	3	14	21	5
Total quantity	284	279	91	292	267	86

Table 34 contrasts the percentage waste management methods in the two studies. In the present study more goes on average to recycling and to incineration with energy and less to landfill.

Table 34: Waste management methods in (OECD 2012) and the present study

	(OECD 2012)*				Present study			
	America	**Europe**	**Asia/ Pacific**	**OECD**	**America**	**Europe, TUR, ISR**	**JPN, KOR, Pacific**	**OECD**
Recycling	20%	15%	24%	20%	24%	25%	31%	25%
Composting	10%	10%	6%	7%	8%	13%	0.3%	9%
Residual-waste composting	-	-	-	-	0.2%	1%	-	0.5%
Incineration without energy	0%	1%	0%	0.3%	0.1%	3%	4%	2%
Incineration with energy	11%	16%	28%	18%	9%	20%	48%	18%
Landfill	58%	51%	40%	50%	60%	38%	17%	45%

*The accounting gap represents other recycling methods such as pre-treatment

5.6.4 Future scenarios

To produce the future scenarios in (OECD 2012), the waste amounts were first extrapolated by applying a constant annual rate of increase. In the present study the original waste amounts were retained. In addition, in (OECD 2012) eight different future scenarios are analysed using the extrapolated quantities. Each scenario involves a particular technical improvement. The first scenario is based on increased recycling, the second on greater use of composting. Table 35 contrasts the rates used in the two studies.

Table 35: Recycling rates in the future scenarios in (OECD 2012) and the present study

Fraction	(OECD 2012)	Present study – ideal scenario OECD
Paper/ cardboard	85%	70%
Plastic	40%	60%
Fe metals	95%	90%
Non-Fe metals	87%	70%
Glass	85%	70%
Food waste	80%	70%
Garden waste	80%	80%

In a third scenario in (OECD 2012), 80% of food and garden waste is sent to anaerobic digestion. In the ideal scenario analysed in the present study, food waste is sent to anaerobic digestion. A further scenario in (OECD 2012) considers recycling rates from the first scenario in combination with MBT.[23] After removal of recyclables, 75% of the remaining waste is processed by MBT and 25% is sent to landfill and incineration in proportions equal to the baseline rates of landfill and incineration. The RDF output is used in cement works. In both future scenarios described in the present study 20% of the residual waste is processed by MBT – in the medium scenario by means of anaerobic MBT and in the ideal scenario by means of MBS. Of the RDF output, half is used in cement works and half in RDF-CHP. The remaining 80% is sent to incineration with energy recovery. The fifth scenario in (OECD 2012) assumes firstly that all landfills are equipped with gas collection systems. Secondly, a higher effective gas collection efficiency of 87% is assumed. The effective gas collection efficiency is thus increased to 87%, while in the present study it is capped at 50%. The sixth scenario in (OECD 2012) incorporates all the assumptions of the fifth one but additionally assumes that 100% of recovered landfill gas is utilised for electricity generation. In the sixth scenario the waste remaining after recycling and composting is incinerated with energy recovery. In addition the net efficiency is increased to 16% for electricity and 50% for heat. In the present study the optimised net efficiencies are 18% for electricity and 42% for heat. The final scenario in (OECD 2012) involves waste prevention, referred to as “source reduction”. For each fraction the quantity is reduced by 30%. For the reasons specified in Section 4.1.1, source reduction is not considered in the present study.

5.6.5 Results

In (OECD 2012) the results are presented as the difference between the baseline scenario and the particular future scenario. This makes it possible to identify the most effective optimisation. Both the specific results per tonne of waste and the overall reduction are described. The most effective measures per tonne of waste are source reduction and recycling (recycling at 1.3 – 2.7 t CO_2-eq per t recycled waste). The largest reductions within the regions are achieved in “North America and “OECD-Pacific”, because in the baseline scenario in these regions very much more

[23] (OECD 2012) makes no mention of biogas in connection with this scenario; it must therefore be assumed that the scenario involves aerobic MBT.

waste is sent to landfill without gas collection than in the other regions. In the present study the greatest savings are also achieved through reduction of the landfilled quantity and improved recycling. The smallest specific reduction is achieved through composting and anaerobic digestion.

According to (OECD 2012), the largest overall reductions are obtained in the scenarios involving incineration with optimised efficiencies, improved energy recovery through use of the collected landfill gas, and recycling in combination with MBT. Although the specific reduction in these scenarios is lower, the large quantity reductions in these situations result in significant GHG mitigation. By far the largest reductions arise in “America”. The same is true of the present study.

To take account of both effectiveness and quantity changes, an integrated scenario was analysed in (OECD 2012). In the integrated scenario the most effective scenarios (optimised recycling and composting and source reduction) are combined with optimised incineration. Because of the source reduction involved, the results are not comparable with the values obtained in this study.

In summary, the differences in the results are attributable to the following factors: Firstly, in the baseline scenario in (OECD 2012) more waste is landfilled than in the present study. For (OECD 2012) a shift in waste quantities away from landfilling to more recycling and incineration thus results in a larger reduction. In addition, for metals and plastic the recycling rates and net emission factors for recycling in (OECD 2012) are significantly higher than in the present study, so that here again there is a larger reduction in the future scenarios (see Table 31). By comparison with the present study, the net thermal efficiency of incineration is somewhat higher in the future scenarios in (OECD 2012), but electrical efficiency is lower. A significant factor is that in (OECD 2012) marginal electricity is used as the substitution process for energy recovery, but in the present study the country-specific electricity mix is used.

5.7 Conclusions for the OECD countries

The results of the GHG inventory for the OECD countries show that landfilling (despite a medium effective gas collection efficiency of 38%) causes the most GHG emissions. Recycling results in the largest GHG credits but cannot adequately compensate for the GHG debits as a result of landfilling. Other disposal and treatment methods have little effect on the result. The bottom line, therefore, is that current waste management practices in the OECD countries result in a significant GHG debit. A sensitivity analysis that takes account of a C sink turns the result into a slight net credit. However, the C sink cannot be reliably proved and is not included in greenhouse gas inventories produced by the IPCC method.

In the two future scenarios that were explored, waste management under the modified conditions results in noticeable GHG credits. In particular, the reduction or prevention of landfilling of untreated waste makes a significant contribution to reduction of the GHG debit. In addition, improved recycling rates and use of RDF from MBT or MBS in RDF-CHP or as a substitute for coal in power plants or cement works result overall in higher GHG credits. This is especially true in the case of the ideal scenario.

The following recommendations are made with regard to the development of waste management in the OECD countries (insofar as they have not already been implemented in individual countries):

- Introduction or expansion of segregated collection of recyclables, especially plastics and organic matter. This enables higher-grade plastic recycling to be achieved. Segregated collection of organic matter enables anaerobic digestion capacities for biogas production to be increased and at the same time permits production of quality compost that can replace the use of peat.
- In countries in which large amounts of waste are currently landfilled, plans for the gradual reduction of landfilling should be drawn up and systematically implemented. Anaerobic MBT or MBS can be used as bridging technologies if required. For efficient use of MBT or MBS, a market for RDF must exist or be created.
- Recycling rates should be increased. It is not only the quantity but also the quality of recycling that is important. This applies in particular to plastic, the recycling of which is often low-grade. Unavoidable sorting and processing residues should be co-fired for energy generation or treated thermally in MSWIs with efficient energy recovery.

5.8 EU28

In the analysis of waste management in the OECD member states, the EU was represented by the 21 EU-OECD countries. The entire EU28[24] is considered separately in this section.

5.8.1 Waste generation, composition and management methods

Waste generation in the EU and the distribution among disposal methods are shown in Table 36. The recycling and composting rates are the same as for the EU-OECD countries (see Section 5.2 and Annex 11.2.1). As in the OECD inventory, the waste characteristics of the EU28 have been taken from (Öko-Institut/IFEU 2010) (Table 37).

Table 36: Waste generation of the EU28 in 2012 and distribution among disposal methods (Eurostat 2014a)

	Recycling	Composting	Landfill	Incineration without energy	Incineration with energy	Total	Total per capita
	1,000 t						kg/cap
EU28	65,596	35,724	80,733	9,209	48,818	240,181	476

Table 37: Waste characteristics of the EU28 (own calculation)

Calorific value [MJ/kg]	Total C	Fossil C	Regenerative C
9.2	24.5% SM	9.0% SM	15.5% SM

These characteristics can be used to calculate the specific emissions for waste incineration in a MSWI (Table 38). The net efficiencies of the MSWI are those used in the OECD analysis (11.4% for electricity, 31.6% for heat, see Section 4.2.8). The specific emissions for recycling and

[24] EU-OECD countries plus Bulgaria, Croatia, Cyprus, Latvia, Lithuania, Malta and Romania.

composting and the emission factors for electricity and heat are those of the EU-OECD countries or the EU27.

Table 38: Specific debits and credits of incineration in MSWIs in the EU28

Incineration without energy			Incineration with energy		
kg CO_2-eq/t			kg CO2-eq/t		
Debit	Credit	Net	Debit	Credit	Net
360	-	360	360	433	-73

The average effective gas collection efficiency for landfilling in the EU28 is 33% (taking account of the 50% cap). The debits and credits from landfilling are shown in Table 39.

Table 39: Emitted and collected methane and GHG credits for small-scale CHP plants in connection with landfilling in the EU28

Methane emitted* [1,000 t CO_2-eq]	Methane collected [1,000 t CO_2-eq]	Credit small-scale CHP [1,000 t CO_2-eq/a]	Net [1,000 t CO_2-eq/a]
61,572	29,666	-2,748	58,824

*diffuse emissions from landfill and methane slip from small-scale CHP (percentage 0.2%)

5.8.2 Results for the EU28

The baseline case

Tables 40 and 41 show the absolute results in terms of the global warming effect and the results per tonne of waste and per capita.

Table 40: Absolute results – global warming effect by disposal methods

	Recycling	Composting	Landfill	Incineratio n without energy	Incinera-tion with energy	Residual-waste composting	CLN, TSP, SOR
[1,000 t CO_2-eq]	-72,722	260	58,824	3,317	-3,557	166	5,291

Table 41: Overall net result – absolute, per tonne of waste and per capita

Net absolute [1,000 t CO_2-eq]	Per tonne of waste [t CO_2-eq/t waste]	Per capita [t CO_2-eq/cap]
-8,421	-0.017	-0.035

Figure 10: Net contribution of the disposal methods to global warming (scaled to one t waste)

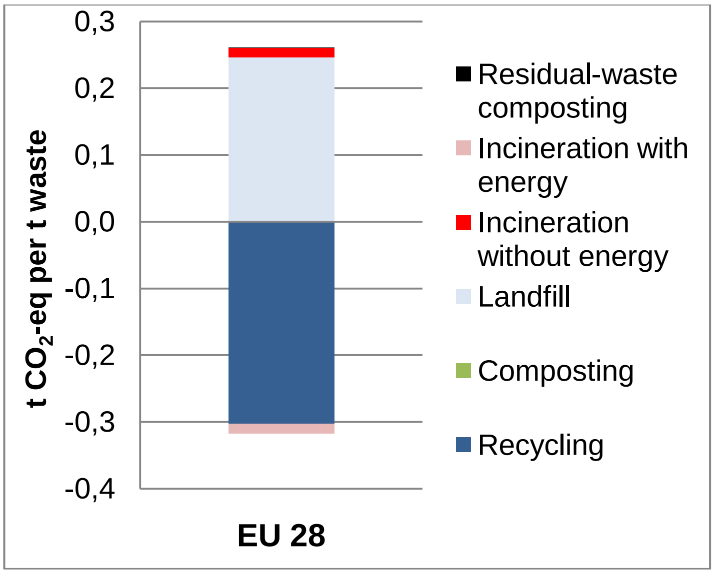
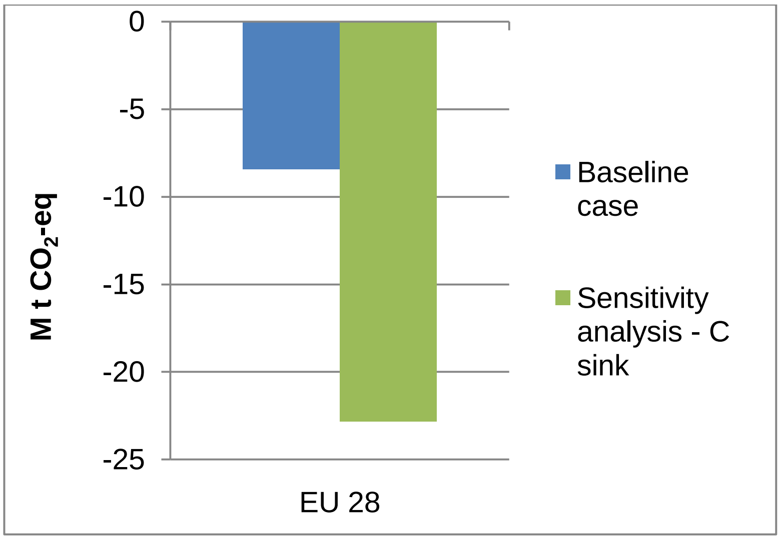


Figure 10 shows the net contribution of the different disposal methods to global warming, scaled to one t waste. From this it is particularly clear that landfill represents the largest GHG debit, while recycling produces the largest credit. Incineration with and without energy recovery contributes only a very small amount to the credits/debits. The direct contributions of residual-waste composting and composting of source-segregated organic matter are virtually insignificant. However, segregated collection of organic waste plays a significant part in reducing methane emissions from landfilling.

Sensitivity

As a sensitivity for the baseline case, the result taking account of the C sink is reported. For the reasons mentioned in Section 4.1.2, the C sink is given for information only. The calculation covers the C sink in connection with landfill and with quality composts. The results are shown in Figure 11. Taking the C sink into account improves the overall result by around 14 Mt CO_2-eq.

Figure 11: Net results taking account of the C sink by comparison with the baseline case



5.8.3 Future scenarios to 2030

The future scenarios were calculated in accordance with the definitions in Section 5.5. For simplification, the characteristics of the waste consigned to MSWI or to landfill were kept constant. The characteristics of the RDF produced are as given in Section 4.2.7.

Baseline comparison

Table 42 shows the net contributions of the three future scenarios to 2030 – "business as usual" (BAU), "medium scenario" (medium) and "ideal scenario" (ideal) – by disposal methods. In the medium scenario the total net credit of around -8 Mt CO_2-eq per year increases to around -65 Mt CO_2-eq per year. The total GHG reduction is around 57 Mt CO_2-eq per year. In terms of the overall result the ideal scenario improves on the medium one by a further 34 Mt CO_2-eq. By comparison with BAU the ideal scenario achieves a GHG reduction of around 91 Mt CO_2-eq.

Table 42: Net contributions of the future scenarios to 2030 by disposal methods for the EU28

	1,000 t CO_2-eq								
	Recycling	Composting	Anaerobic digestion	Landfill	Incineration without energy	Incineration with energy	Residual-waste composting	CLN, TSP, SOR	Total
BAU	-72,722	260		58,824	3,317	-3,357	166	5,291	-8,421
medium	-78,002	406		24,262		-16,874		5,497	-64,711
ideal	-83,274	295	-2,385			-20,353		6,183	-99,533

Figure 12: Net contributions to global warming of the disposal methods in the future scenarios for the EU28 (per tonne of waste)

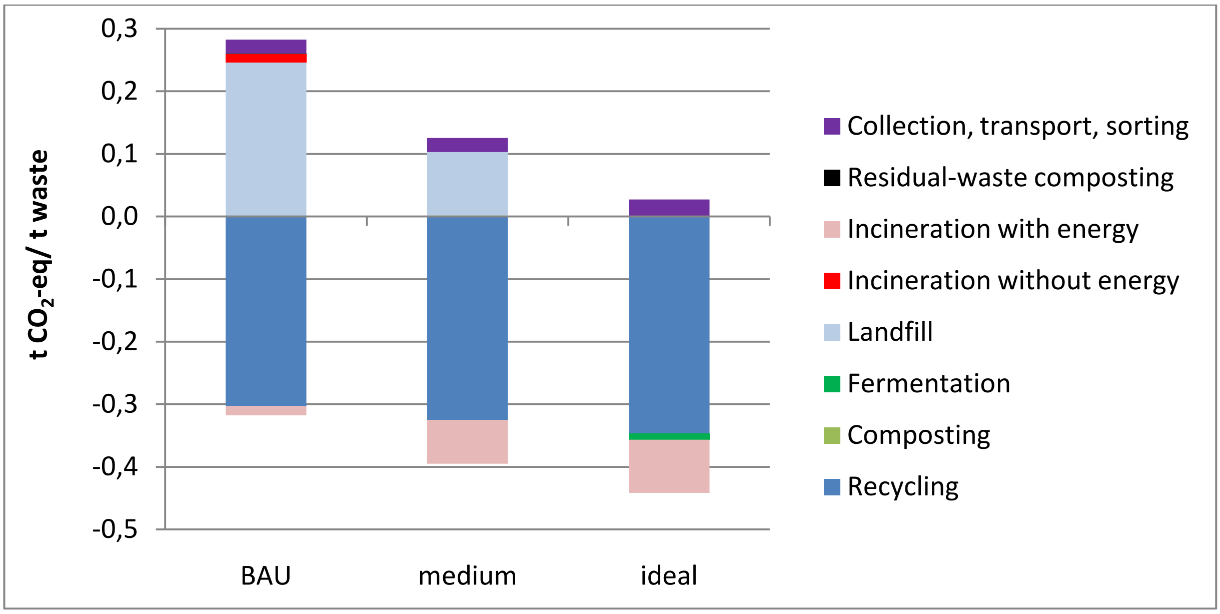


Figure 12 shows the net contributions of the various disposal methods for the three future scenarios per tonne of waste. Whereas in the BAU scenario landfill still represents a considerable GHG debit, in the medium scenario this is significantly reduced on account of the smaller proportion of waste that is landfilled and the higher effective gas collection efficiency. The GHG credits from incineration with energy recovery and from recycling improve in the medium scenario and again in the ideal scenario. In the ideal scenario anaerobic digestion also makes a small contribution to the GHG credit. Composting, on the other hand, plays very little part in any of the scenarios.

Sensitivities

For the future scenarios the following two aspects were considered as sensitivities:

1. Result taking account of the C sink
2. Results with modified ratio between “incineration with energy” and treatment via MBT or MBS (50:50 instead of 80:20)

Figure 13 shows the net results of the sensitivity analysis by comparison with the baseline case for the medium scenario. Including the C sink produces a slight improvement in the result of around 10 Mt CO_2-eq, while the shift in the MSWI:MBT ratio produces very little improvement.

Figure 13: Net results of the sensitivity analysis and baseline case in the medium scenario, EU28

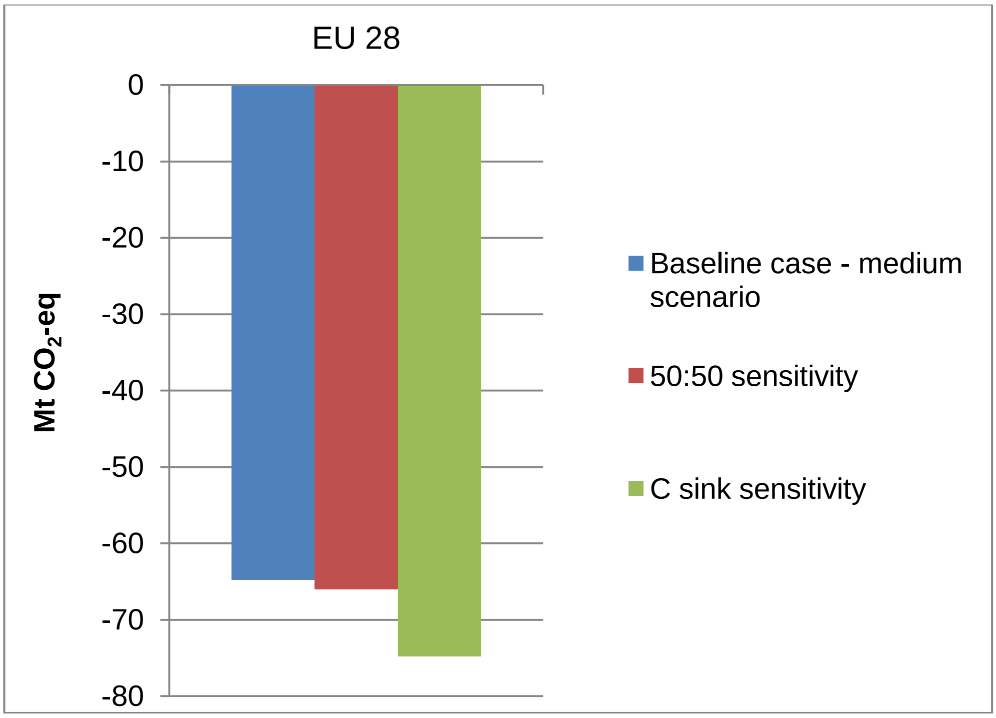


Figure 14: Net results of the 50:50 sensitivity analysis and baseline case in the ideal scenario, EU28

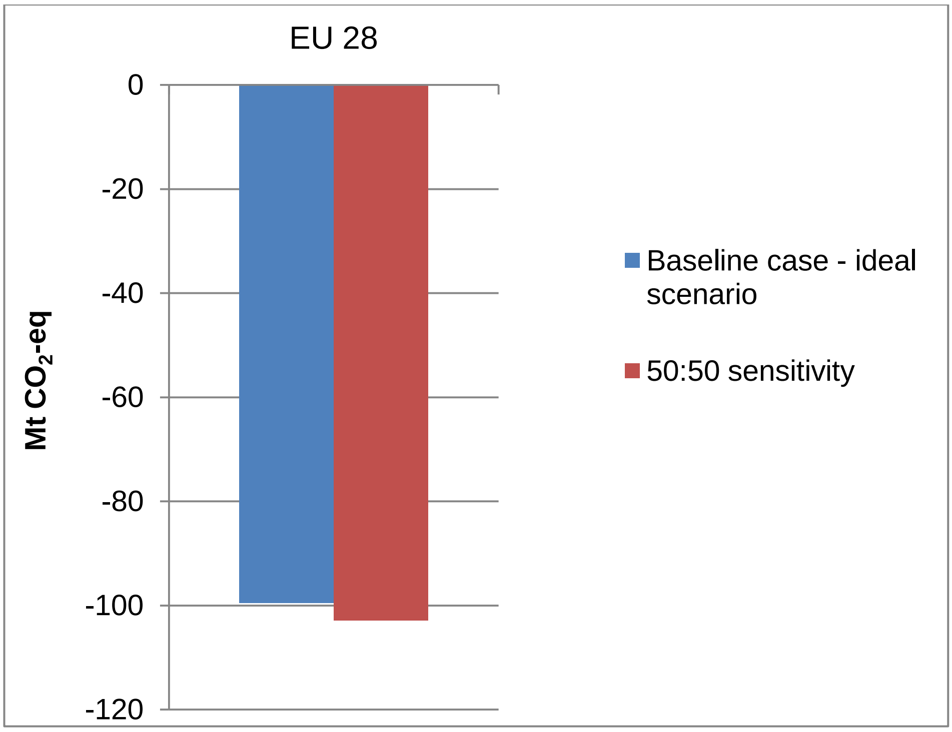


Figure 14 shows the net result for the 50:50 sensitivity analysis and the baseline case in the ideal future scenario. The decision was taken not to quote the results of the sensitivity analysis involving the C sink, because this yields very little difference. The 50:50 sensitivity analysis produces a small improvement in the results of the ideal future scenario. Overall the change is not relevant.

5.8.4 Conclusions – EU28

The picture with regard to waste management in the EU28 is similar to that for the EU-OECD countries and the OECD as a whole. The landfilling of waste causes the most GHG debits, while recycling results in the largest credits. Other disposal methods have very little effect on the result. Overall, current waste management practices in the EU28 result in a slight net credit in the GHG inventory.

For the EU28, waste management under the modified conditions again results in noticeable GHG reductions in the two future scenarios that were explored. In particular, the reduction or prevention of landfilling of untreated waste makes a significant contribution to reduction of the GHG debit. In addition, improved recycling rates and use of RDF from MBT or MBS in RDF-CHP or as a substitute for coal in power plants or cement works result overall in higher GHG credits.

For the development of waste management in the EU28, the recommendations are similar to those for the OECD as a whole, namely introduction or expansion of segregated collection of recyclables, especially plastics and organic matter, adoption and systematic implementation of plans for the gradual reduction of landfilling, and an increase in recycling rates.

5.9 Waste management in the USA

Data on waste volumes and disposal methods are in principle taken from the information published by the responsible public institutions (see Section 4.2.2). In the case of the USA the basis is the statistics on MSW volumes, composition and management methods published by the United States Environmental Protection Agency (USEPA). For the inventory the figures for 2011 published in (USEPA 2013a and 2013b) were analysed. (USEPA 2013b) is a fact sheet that is published annually. A full report is published by USEPA every two years, the most recent one being (USEPA 2013a). Supplementary information on the situation in the USA is drawn from an expert discussion (Thorneloe 2012) and other literature sources. Where possible, data from the calculation tools developed by various departments of USEPA were also analysed. One of these tools is the Waste Reduction Model (WARM 2013), a simple tool with default emission factors and data that are described in various documents. The other is the Municipal Solid Waste – Decision Support Tool (MSW-DST 2013), a very complex tool for which comprehensive further information is published in various documents.[25]

5.9.1 Waste volume and composition

According to (USEPA 2013a) the volume of municipal solid waste generated in the USA in 2011 was around 250 million US short tons, which was similar to previous years (Figure 15).

The **quantities quoted by USEPA are given in US short tons.**
1 short ton = 0.907185 metric tons

To ensure that the values can be tracked, quantities have not been converted into metric tons for the USA balance.

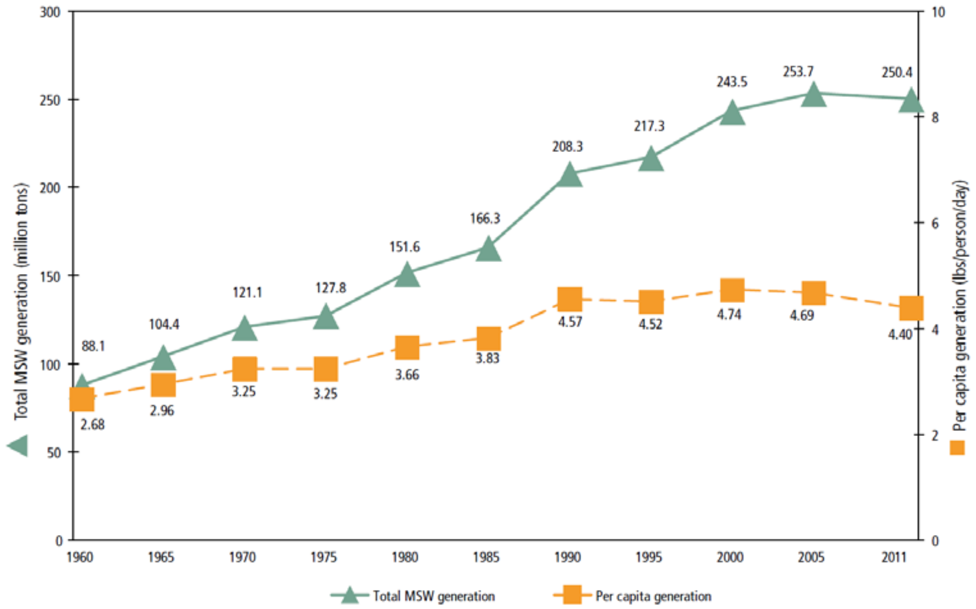
Unless otherwise stated, all quantities in this section are given in US short tons; metric tons
are reported in megagrams (Mg).

By contrast, the figures for the USA in the OECD balance in Sections 5.1 – 5.7 are given in metric tons: this ensures comparability with the other OECD states.

USEPA calculates waste amounts and management methods by means of a top-down approach (see also Section 4.2.2). The method used is that of material flow analysis, which is based on production data (by mass) for materials and products. To identify waste amounts, the values are corrected for imports and exports and for materials transferred to the construction sector. Other adjustments involve product lifetimes. The quantities of food waste and green waste are taken from studies of waste collection.

[25] https://mswdst.rti.org/resources.htm

Figure 15: Waste generation over time in the USA (USEPA 2013b)



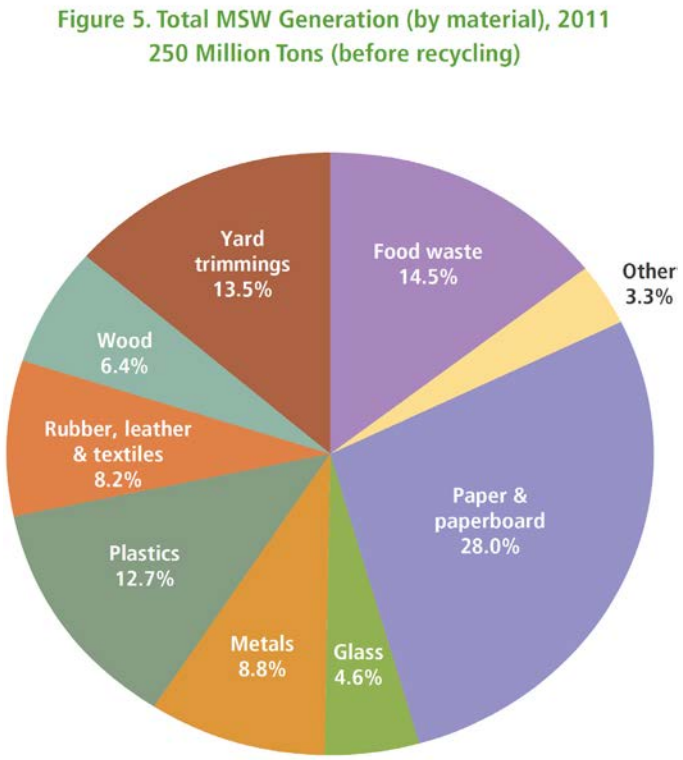
A contrasting bottom-up approach is also used in the USA to measure waste generation and disposal; the bottom-up approach is based on actual waste amounts, which are identified using information supplied by the waste authorities in the individual US states. This survey, entitled "The State of Garbage in America" (SOG survey), is conducted regularly by the Earth Engineering Center (EEC) at Columbia University and the journal *BioCycle*.[26] However, the relevant data is not available for all US states – weighing of waste is compulsory only at landfills and incinerators, and the proportional mass of the MSW is often estimated rather than weighed separately. For these reasons, therefore, there are uncertainties about actual waste volumes and management methods.

The analysis was based on the official USEPA figures using the material flow approach. The figures from the alternative SOG survey were compared in a sensitivity analysis.

According to (USEPA 2013b), the composition of the waste generated in 2011 is as shown in Figure 16. Paper and paperboard comprise the largest percentage at 28%. This is followed by food waste at 14.5% and garden waste at 13.5%. Plastics at 12.7% are one of the larger waste fractions. The other fractions of textiles, wood, metals and glass each account for less than 10% of the total.

[26] http://www.biocycle.net/2010/10/26/the-state-of-garbage-in-america-4/

Figure 16: Composition of municipal solid waste in the USA in 2011 (USEPA 2013b)



5.9.2 Waste collection and management methods

In the USA, green waste (yard trimmings) and dry recyclables are collected separately. The majority of waste, though, is collected as mixed waste. By far the largest proportion of mixed waste is sent directly to landfill; some is incinerated and smaller proportions go to sorting facilities or mixed-waste composting facilities. According to (USEPA 2013a) the number of waste facilities in the USA in 2011 was as follows (quantities in short tons):

- 1,908 landfills
- 86 municipal waste-to-energy projects with a total capacity of 96,164 t/d (around 35 million t/a)
- 633 materials recovery facilities (MRF) with an estimated total throughput of 98,449 t/d (around 36 million t/a)
- 43 mixed-waste processing facilities with a total throughput of approx. 46,700 t/d (around 17 million t/a)
- 12 mixed-waste composting facilities with a total throughput of approx. 1,400 t/d (511,000 t/a).
- 3,090 yard waste composting facilities (YWCF) with an estimated total throughput of approx. 52,900 t/d (approx. 19 million t/a).

The destinations of municipal solid waste are documented in detail in (USEPA 2013a). The table in Figure 17 details the total waste generated in various years and the amounts landfilled, incinerated, recovered and composted. This shows that materials recovery increased significantly in the 1980s and '90s. The current overall recovery rate (recycling plus composting) is 34.7%.

Figure 17: Waste generation and management methods in the USA (USEPA 2013b)

Table ES-1. Generation, Materials Recovery, Composting, Combustion with Energy Recovery, and Discards of Municipal Solid Waste, 1960 – 2011

(In thousands of tons and percent of total generation)

	Thousands of Tons									
	1960	**1970**	**1980**	**1990**	**2000**	**2005**	**2007**	**2009**	**2010**	**2011**
Generation	88,120	121,060	151,640	208,270	243,450	253,730	256,500	244,270	250,500	250,420
Recovery for recycling	5,610	8,020	14,520	29,040	53,010	59,240	63,100	61,640	64,960	66,200
Recovery for composting*	Neg.	Neg.	Neg.	4,200	16,450	20,550	21,710	20,750	20,170	20,700
Total Materials Recovery	5,610	8,020	14,520	33,240	69,460	79,790	84,810	82,390	85,130	86,900
Discards after recovery	82,510	113,040	137,120	175,030	173,990	173,940	171,690	161,880	165,370	163,520
Combustion with energy recovery**	0	400	2,700	29,700	33,730	31,620	31,970	29,010	29,260	29,260
Discards to landfill, other disposal†	82,510	112,640	134,420	145,330	140,260	142,320	139,720	132,870	136,110	134,260
	Pounds per Person per Day									
	1960	**1970**	**1980**	**1990**	**2000**	**2005**	**2007**	**2009**	**2010**	**2011**
Generation	2.68	3.25	3.66	4.57	4.74	4.69	4.66	4.36	4.44	4.40
Recovery for recycling	0.17	0.22	0.35	0.64	1.03	1.10	1.15	1.10	1.15	1.16
Recovery for composting*	Neg.	Neg.	Neg.	0.09	0.32	0.38	0.39	0.37	0.36	0.37
Total Materials Recovery	0.17	0.22	0.35	0.73	1.35	1.48	1.54	1.47	1.51	1.53
Discards after recovery	2.51	3.03	3.31	3.84	3.39	3.21	3.12	2.89	2.93	2.87
Combustion with energy recovery**	0.00	0.01	0.07	0.65	0.66	0.58	0.58	0.52	0.52	0.51
Discards to landfill, other disposal†	2.51	3.02	3.24	3.19	2.73	2.63	2.54	2.37	2.41	2.36
Population (thousands)	179,979	203,984	227,255	249,907	281,422	296,410	301,621	307,007	309,051	311,592
	Percent of Total Generation									
	1960	**1970**	**1980**	**1990**	**2000**	**2005**	**2007**	**2009**	**2010**	**2011**
Generation	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%
Recovery for recycling	6.4%	6.6%	9.6%	14.0%	21.8%	23.3%	24.6%	25.2%	25.9%	26.4%
Recovery for composting*	Neg.	Neg.	Neg.	2.0%	6.7%	8.1%	8.5%	8.5%	8.1%	8.3%
Total Materials Recovery	6.4%	6.6%	9.6%	16.0%	28.5%	31.4%	33.1%	33.7%	34.0%	34.7%
Discards after recovery	93.6%	93.4%	90.4%	84.0%	71.5%	68.6%	66.9%	66.3%	66.0%	65.3%
Combustion with energy recovery**	0.0%	0.3%	1.8%	14.2%	13.9%	12.5%	12.5%	11.9%	11.7%	11.7%
Discards to landfill, other disposal†	93.6%	93.1%	88.6%	69.8%	57.6%	56.1%	54.4%	54.4%	54.3%	53.6%

* Composting of yard trimmings, food waste and other MSW organic material. Does not include backyard composting.
** Includes combustion of MSW in mass burn or refuse-derived fuel form, and combustion with energy recovery of source separated materials in MSW (e.g., wood pallets and tire-derived fuel). 2011 includes 25,930 MSW, 520 wood, and 2,810 tires (1,000 tons)
† Discards after recovery minus combustion with energy recovery. Discards include combustion without energy recovery.
Details may not add to totals due to rounding.

Of the 250 million tons of waste generated in 2011, 53.6% was landfilled, 26.4% recycled, 8.3% composted and 11.7% incinerated. The quantity incinerated includes 2.81 million tons of used tyres (9.6% of the volume of waste incinerated). Used tyres were not included in the previous studies of the climate change mitigation potential of waste management and they were excluded from the analysis of the USA.

The remaining volume of waste considered amounts to 247,610,000 t. Of this adjusted volume of waste, around 35% was recycled and 65% disposed of as residual waste. Of this residual waste, around 84% was landfilled. These waste streams formed the basis of the GHG inventory for the USA (Figure 18).

Figure 18: Material flow diagram for the waste streams on which the USA analysis is based

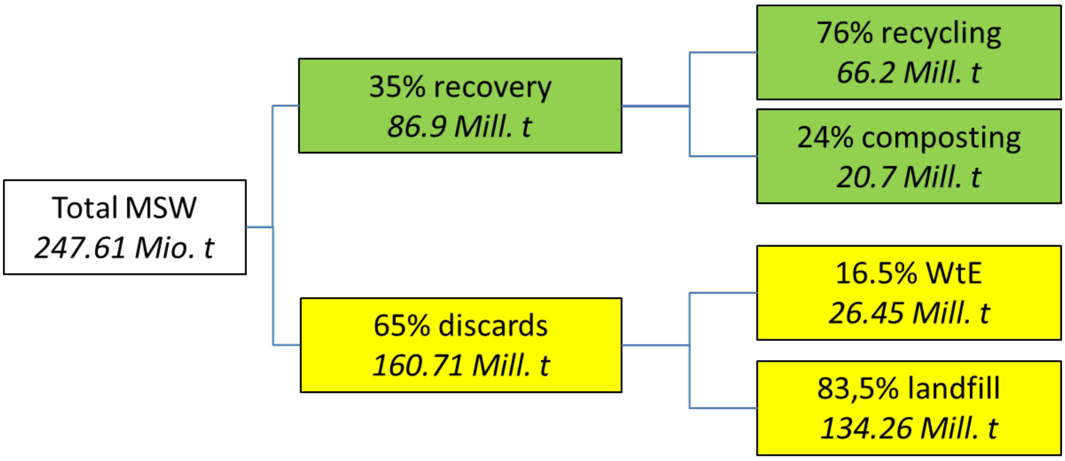


Figure 19: Generation and recovery by waste types in the USA (USEPA 2013b)

Table 1. Generation, Recovery, and Discards of Materials in MSW, 2011*
(in millions of tons and percent of generation of each material)

Material	Weight Generated	Weight Recovered	Recovery as Percent of Generation	Weight Discarded
Paper and paperboard	70.02	45.90	65.6%	24.12
Glass	11.47	3.17	27.6%	8.30
Metals				
Steel	16.52	5.45	33.0%	11.07
Aluminum	3.47	0.72	20.7%	2.75
Other nonferrous metals†	1.96	1.34	68.4%	0.62
Total metals	**21.95**	**7.51**	**34.2%**	**14.44**
Plastics	31.84	2.65	8.3%	29.19
Rubber and leather	7.49	1.31	17.5%	6.18
Textiles	13.09	2.00	15.3%	11.09
Wood	16.08	2.38	14.8%	13.70
Other materials	4.59	1.28	27.9%	3.31
Total materials in products	**176.53**	**66.20**	**37.5%**	**110.33**
Other wastes				
Food, other‡	36.31	1.40	3.9%	34.91
Yard trimmings	33.71	19.30	57.3%	14.41
Miscellaneous inorganic wastes	3.87	Negligible	Negligible	3.87
Total other wastes	**73.89**	**20.70**	**28.0%**	**53.19**
Total municipal solid waste	**250.42**	**86.90**	**34.7%**	**163.52**

* Includes waste from residential, commercial, and institutional sources.
† Includes lead from lead-acid batteries.
‡ Includes recovery of other MSW organics for composting.
Details might not add to totals due to rounding.
Negligible = Less than 5,000 tons or 0.05 percent.

A further breakdown of recovery volumes is contained in a table from (USEPA 2013b) (Figure 19). In this table the waste generated is subdivided into fractions and for each fraction the amount recovered and the amount of residual MSW ("waste discarded") is shown. According to these figures, particularly high recovery rates are achieved for other nonferrous metals (68.4%),[27] paper (65.6%) and garden waste (57.3%).

Table 43 shows the waste composition used for this study (for simplification, the volume of incinerated tyres was deducted from the "other" waste) and the composition of the residual waste after removal of recyclables. Particularly striking is the significant reduction in the amount of paper in the discarded waste and conversely the increase in plastic waste as a result of the corresponding recycling rates (Figure 19). The percentage of paper waste falls from around 28% to 15%, while the percentage of plastic waste rises from around 13% to around 18%, which is significantly higher than in other country studies. For example, the percentage of plastic in German residual waste in (Öko-Institut/IFEU 2010) was just under 6% and for India this study calculated the percentage of plastic in residual waste to be around 5-6% (Table 53). The high percentage of plastic has a significant influence on the fossil carbon content of the residual waste.

Table 43: Waste composition before and after removal of recyclables

	Waste generated		Waste discarded	
	in million tons	in %	in million tons	in %
Organic waste	70.02	28.3	49.32	30.7
Wood	16.08	6.5	13.70	8.5
Paper	70.02	28.3	24.12	15.0
Plastic	31.84	12.9	29.19	18.2
Rubber and leather	7.49	3.0	6.18	3.8
Ferrous metals	16.52	6.7	11.07	6.9
Nonferrous metals	5.43	2.2	3.37	2.1
Glass	11.47	4.6	8.30	5.2
Textiles	13.09	5.3	11.09	6.9
Other	5.65	2.3	4.37	2.7
Total	**247.61**	**100.0**	**160.71**	**100.0**

Characteristics of the waste streams

The composition of the total waste generated and of the residual waste after recovery of recyclables provides the basis for calculation of the key characteristics needed for the GHG inventory. The calculation uses the characteristics of waste fractions give in Table 13. The only exception is the plastic fraction. Because there is a high percentage of plastic in the residual

[27] According to further information in (USEPA 2013a), the quantities given under "other nonferrous metals" consist entirely of lead from lead batteries. In the GHG inventory a special procedure was adopted for these amounts (see the "Recycling" section).

waste and this has a strong influence on the result, the calorific value and fossil carbon content (C content) of plastic for the USA was calculated from country-specific information.

As a first step, a detailed analysis of the composition of this fraction by plastic type was carried out. Figure 20 shows a summary from (USEPA 2013a) of generation and management methods by plastic type. According to this, the plastic fraction in the residual waste ("discards") is 42% PE, 24% PP, 12% PET, 7% PS, 3% PVC and 11% other. The average calorific value was calculated from this and from details of the calorific value of plastic types in (Ecoinvent 2007). The average fossil C content was determined in the same way. Deviating from the values in Table 13, this amounts to around 73% (instead of 68%) and the calorific value amounts to 35.5 MJ/kg (instead of 30.5).

Figure 20: Generation and recycling of plastic waste by plastic type (USEPA 2013a)

Table 7 (continued)

PLASTICS IN PRODUCTS IN MSW, 2011

(In thousands of tons, and percent of generation by resin)

Product Category	Generation (Thousand tons)	Recovery (Thousand tons)	Recovery (Percent of Gen.)	Discards (Thousand tons)
Total Plastics in MSW, by resin				
PET	4,280	830	19.4%	3,450
HDPE	5,590	550	9.8%	5,040
PVC	900			900
LDPE/LLDPE	7,520	370	4.9%	7,150
PLA	50			50
PP	7,180	30	0.4%	7,150
PS	2,170	20	0.9%	2,150
Other resins	4,150	850	20.5%	3,300
Total Plastics in MSW	**31,840**	**2,650**	**8.3%**	**29,190**

HDPE = High density polyethylene; PET = Polyethylene terephthalate; PS = Polystyrene
LDPE = Low density polyethylene; PLA = Polylactide; PVC = Polyvinyl chloride
LLDPE = Linear low density polyethylene; PP = Polypropylene

‡ Other plastic packaging includes coatings, closures, lids, PET cups, caps, clamshells, egg cartons, produce baskets, trays, shapes, loose fill, etc.
PP caps and lids recovered with PET bottles and jars are included in the recovery estimate for PET bottles and jars.
Other resins include commingled/undefined plastic packaging recovery.
Some detail of recovery by resin omitted due to lack of data.

Finally, the biogenic and fossil carbon content and the calorific value of the total volume of waste and the residual waste ("discards") were calculated from the composition of the waste and the characteristics of the individual waste fractions. The results are shown in Table 44.

Table 44: Calculated characteristics of waste streams

	Total MSW generated, calculated	Total residual waste, calculated	Input to incineration according to Covanta	Input to landfill, calculated
Biogenic carbon in %	19.3	16.2	17.6	15.9
Fossil carbon in %	11.0	15.3	9.9	16.4
Calorific value in kJ/kg waste	11,836	12,770	10,200	13,276

Details of the calorific value of the different waste fractions are available for the USA (Kaplan et al. 2009b), but without corresponding data on the carbon content. These calorific values for the USA are consistently higher than the values shown in Table 13. If the calorific value of the residual waste were to be calculated using these figures, it would amount to around 13.3 MJ/kg (instead of 11.8 MJ/kg, see Table 44). For the plastic waste fraction this calculation used the value previously obtained and described for the USA, because (Kaplan et al. 2009b) only reports calorific values for PE, PET and mixed plastic.[28]

The fossil carbon content calculated for the residual waste is comparatively high. As already mentioned, the main reason for this is the large proportion of plastic in the residual waste. The fossil C contents obtained in the above-mentioned country studies were significantly lower – 9% in (Öko-Institut/IFEU 2010) for Germany and 5% for residual waste in India (Table 53). Further information on the characteristics of the residual waste were also provided by the Covanta Energy Cooperation,[29] one of the main providers and operators of waste-to-energy (WtE) facilities in the USA. Covanta has records going back many years of the average calorific values of waste input and data on fossil carbon content going back to 2007. The fossil carbon data are obtained by measuring the CO_2 in the exhaust gas and determining the fossil and biogenic components by means of the radio carbon method in accordance with the guidelines of the US EPA Greenhouse Gas Reporting Program (GHGRP).[30] The values do not fluctuate greatly over time. The characteristics according to Covanta shown in Table 44 were obtained from the figures for 2011.[31]

On the basis of the composition reported by USEPA for the generated waste and for the residual waste remaining after removal of recyclables, the above figures from Covanta mean that the proportion of residual waste sent to landfill must contain larger percentages of plastic than the residual waste sent to incineration. This again would mean that the residual waste must be deliberately collected and sent to landfill or incineration, which seems fairly unlikely. A more plausible explanation is that there are weaknesses, at least in respect of the plastic fraction, in the USEPA data obtained by the top-down method.

Nevertheless, the figures from the USEPA survey have been retained for the standard case analysis. However, because the Covanta data are based on measurements, they are regarded as reliable and they are also used for the GHG inventory. In consequence it is assumed – although it is unlikely – that the large volume of plastic waste that remains is mainly sent to landfill. This results in different figures for residual waste to incineration and residual waste to landfill (Table 44). The impact on incineration of the high fossil C content calculated for the residual waste is investigated in a sensitivity analysis (sensitivity – incineration).

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[28] In (Kaplan et al. 2009b) PE and PET are described as having the same calorific value, namely 19,000 BTU/lb (approx. 44 MJ/kg). By contrast, according to (Ecoinvent 2007) the calorific value of PET is around 23 MJ/kg and that of PE is around 42 MJ/kg. This corresponds with the fossil C content of the two types of plastic (PET approx. 55%, PE approx. 82%).

[29] Email of 23 May 2014

[30] http://www.epa.gov/ghgreporting/

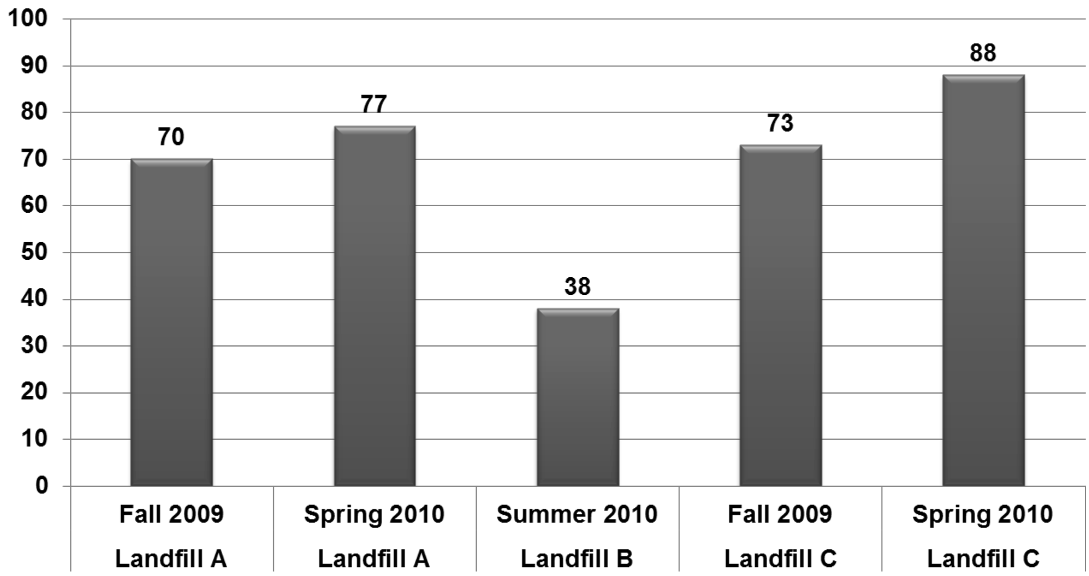
[31] In 2011 the total C content of the waste input amounted to 27.5% SM; the biogenic proportion of this was 64%.

Landfill

For various reasons, landfilling in the USA occupies a special position. Firstly, in recent years the USA has witnessed a trend towards the dumping of waste in "wet landfills" (Thorneloe 2012). These wet landfills, which need no leachate treatment, aim to accelerate biological decomposition in the landfill by adding liquid and recirculating the leachate. Measurements show that methane emissions also rise sharply. However, no reliable data are available. According to the recommendations of an expert (Thorneloe 2012), these landfills should not be considered separately. This recommendation has been followed and all landfilling is balanced as described in Section 4.2.5.

Secondly, another special feature of landfilling in the USA is the gas collection efficiencies quoted, which are relatively high. The majority of landfills in the USA are operated by two large companies, one of which is Waste Management Inc. An expert (Thorneloe 2012) states that the efficiency of the gas collection systems used in landfills varies. Operators postulate the "CO_2-neutral landfill" with 95% gas collection efficiency. According to measurements performed by USEPA (ORD), these gas collection efficiencies are unrealistic. Measurement programmes at three landfills yielded the gas collection efficiencies shown in Figure 21.[32] However, these apply only to the landfilling period that was considered or investigated. There are no data on effective gas collection efficiencies over the entire storage period, which should be considered to last 100 years.

Figure 21: Measurement of gas collection efficiencies at three US landfills



Source: Presentation by Susan Thorneloe at a methodology workshop on 18 June 2012 in Berlin
*calculated as CH_4 collected / (CH_4 collected + CH_4 emissions); conventional collection efficiencies can include soil oxidation in the denominator which would lower the efficiencies.

[32] At the landfill with 38% gas collection efficiency a well has now been installed and the gas collection system has been enlarged (Thorneloe 2012).

WARM (2013) likewise assumes relatively high gas collection efficiencies for the entire storage period. The tool gives the user the option to distinguish between three cases: the typical case, the worst case and an aggressive case.

- typical = Years 0-2 0% Year 3 50% Years 4-7 75% Years 8-100 95%
- worst case = Years 0-5 0% Years 6-7 75% Years 8-100 95%
- aggressive = Year 1 25% Years 2-3 50% Years 4-7 75% Years 8-100 95%

The effective gas collection efficiency over the 100-year time span is not given; it depends on the annual methane formation rate that is applied. However, it can be assumed that the effective gas collection efficiency, if calculated, would be over 80% and hence significantly higher than is generally postulated under the current state of scientific knowledge. It is for this reason that (EEA 2011), for example, does not adopt the high gas collection efficiencies reported by some EU countries; instead, a maximum technically feasible effective national gas collection efficiency of 45% is assumed, even if all landfills have gas collection systems.

In the USA the Clean Air Act (CAA)[33] in principle requires all large landfills to install gas collection systems within five years of the depositing of waste. These systems are extended successively to new areas of the landfill as soon as more waste is dumped. The landfill gas that is collected is then either flared off or used for energy recovery. In 2005, according to information in (Kaplan et al. 2009a), 427 out of 1,654 municipal landfills in the USA practised landfill gas collection with subsequent energy recovery with a total capacity of 1,260 MW.

Detailed up-to-date statistics on landfills in the USA were provided to UBA by USEPA in the form an Excel file.[34] Analysis of this data indicated 1,504 landfills classed as “open”. 1,034 of these (69%) have a gas collection system. According to annual waste acceptance rates, 93% of landfill capacity is equipped with a gas collection system and 97% of this also has gas flaring. In the light of this it was assumed for this study that all landfills have gas collection systems installed.

Figure 22: Gas collection and oxidation rates in the MSW-DST (Kaplan et al. 2009a)

TABLE 1. Inputs to the Landfill Process Model

	LFG collection system efficiency [a] (%)	oxidation rate (%)
during venting	0	15
during first year of gas collection	50	15
during second year of gas collection	70	15
during third year and on of gas collection	80	15

[a] We assumed efficiency of the collection system based on the year of the operation and the ranges stated in U.S. EPA's AP-42 (*8*).

[33] http://www2.epa.gov/laws-regulations/summary-clean-air-act

[34] Landfill Methane Outreach Program/Global Methane Initiative, Climate Change Division, US Environmental Protection Agency (date saved 27 January 2014); email of 12 February 2014

The gas collection efficiencies specified by landfill operators and in the WARM tool are also viewed critically within USEPA (Thorneloe 2012). While the MSW-DST likewise tends to assume high gas collection efficiencies (see Figure 22, 80% from the 3rd year of gas collection), (Kaplan et al. 2009a) points out conversely that landfilling also results in significant methane leaks on the scale of 60-85%.

According to the inventory report of USEPA (2012), the amount of methane captured in 2010 was only 57%. However, this figure cannot be used directly for comparison of the status quo, because the methane emissions in national reports include all emissions from old deposits (usually since 1950); these emissions may arise from old landfill cells without gas collection systems. In general, though, the scientific evidence suggests that a national average effective gas collection efficiency of up to 80% over the lifetime of the deposits is not realistic. In this study the maximum effective gas collection efficiency considered to be technically possible was set at 50% (Section 4.2.5); this is the figure used here for the standard case in the USA balance. To demonstrate the influence of a high gas collection efficiency, an effective gas collection efficiency of 75% was analysed as a sensitivity. This figure roughly corresponds to the measurements in Figure 21.

The calculation for landfill generally uses the procedure for managed landfill described in Section 4.2.5. For the USA balance the default values in (IPCC 2006) are again used. The gas collection efficiency used, though, differs from this (see above) and in a further deviation the DOC is calculated on the basis of the country-specific waste composition. The following figures were used in the calculations:

- DOC = 15.9% (see Table 44)
- DOCf = 50% (IPCC default value)
- Methane content = 50 Vol% (IPCC default value)
- MCF = 1 (IPCC default value for managed landfills)
- Effective gas collection rate = set at 50% (sensitivity 75%)
- OX = 10% (IPCC default value for well-managed landfills)

The oxidation factor (OX) of 10% according to the IPCC is justified for well-managed landfills (see Section 4.2.5). According to Figure 22, the MSW-DST tool assumes an oxidation rate of 15% for the USA. According to (IPCC 2006), OX values higher than the 10% figure should be very well documented and referenced and supported by national statistics. The available publications do not provide any such evidence for the higher oxidation rate. The analysis therefore uses the figure of 10% recommended for well-managed landfills in (IPCC 2006). In a departure from this, the OECD balance uses a symmetrical and conservative oxidation factor of 0% for all countries, because the situation in the 34 individual OECD countries could not be identified.

Information on the use of landfill gas is contained in the Excel file supplied by USEPA. According to this information, 44% of open landfills with gas collection use the landfill gas in a small-scale CHP unit, 23% provide no information, 13% use the landfill gas directly or for heat generation, 6% use it in gas turbines, 5% practise co-incineration, 4% produce biomethane, 3% use the landfill gas to evaporate the leachate, and the remaining 3% report “other” use. This information is not linked to landfill gas quantities and so cannot be analysed in terms of the quantity of landfill gas collected. According to the inventory report of USEPA (2012), in 2010 around half of the collected landfill gas was flared off and half was used for gas-to-energy. It is assumed that gas-to-energy involves use in small-scale CHP units. As with biogas use, a methane slip of 1% of the methane input is assumed and the efficiencies are set at 37.5% for electricity and 43% for heat (see Section 4.2.6). For heat it is generally assumed that on a national average 20% of the surplus heat can be used externally. If surplus electricity is produced, it is treated for substitution purposes as marginal electricity (see Section 4.1.2).

The same assumptions on landfill gas use were made as in the OECD balance. However, in that balance surplus electricity was credited via substitution of the average electricity mix, not substitution of marginal electricity. In the OECD balance this applies to all electricity generated. For the USA balance, the influence of marginal electricity versus average electricity is considered in a sensitivity analysis of waste incineration.

Composting (incl. residual-waste composting)

Information on composting in the USA was taken from (USEPA 2013a) and documentation relating to the MSW-DST (USEPA 2000a). Most composting in the USA involves source-segregated green waste (predominantly from kerbside collection), which is composted in yard waste composting facilities (YWCF). Anaerobic digestion of organic waste is rare and is ignored in the status quo. For simplification, the small quantity of recovered food waste ("Food, other") shown in Figure 19 is included in green waste composting.

More detailed information on the technical systems used in composting facilities is not available. The MSW-DST (USEPA 2000a) models a simple, open windrow composting facility, because it is assumed that these predominate in the USA. In line with this, this study also assumes a simple, open composting system. The electricity requirement is assumed to be 10 kWh/Mg input and the diesel requirement 3 l/Mg. Assessment of the emissions from composting is based on the mean values for open composting according to (gewitra 2009) (Table 9). No information is available on the compost produced and the uses to which it is put. The analysis assumes production of finished compost, half of which is used in agriculture and half in horticulture. The emission factors for the credits for compost use are given in Section 4.2.6.

In addition to green waste composting, mixed-waste composting is also practised in the USA. (USEPA 2000a) distinguishes between two types of facility: the low-quality compost facility (LQCF) and the high-quality compost facility (HQCF). In the case of the LQCF the aim of biological treatment is to reduce the volume of waste. Coarse removal of impurities takes place before composting; these materials are later transferred to landfill with the mixed-waste compost that has been produced. In the high-quality compost facility both impurities and recyclables are removed; impurities are sent to landfill, while recyclables go to recycling facilities. The material that passes through the screen is transferred to mixed-waste composting. After composting it is again passed through a screen and then used as compost for soil improvement and landscaping purposes and in farms, nurseries and mines. Overall, though, mixed-waste composting is of minor importance in the USA. According to (USEPA 2013a), in 2011 around 511,000 tons of mixed waste was treated in 12 mixed-waste composting facilities. This represents 0.2% of the total volume of waste generated. Because of this very low percentage, mixed-waste composting is ignored in the analysis.

Recycling

In the USA dry recyclables arise in three main ways: from segregated collection, from sorting of mixed recyclables and from sorting of mixed waste. Sorting facilities differ in the equipment that they use. According to (USEPA 2013b), many materials recovery facilities (MRFs) are considered low technology, meaning that the materials are on the whole sorted manually. MRFs also include recycling yards at which recyclables received from households and businesses are sorted manually (in some cases by the owner of the waste upon delivery). Modern automated MRFs use eddy currents, magnetic pulleys, optical sensors and air classifiers. Precise details of the number of automated or simple sorting facilities are not available. It was also not possible to obtain any information on the mass flows in the sorting facilities. However, (USEPA 2000c) contains data on sorting efficiency. Figures are given for different types of waste entering the facility, such as mixed waste, pre-sorted recyclables, mixed recyclables, etc. The efficiency of the sorting process varies between 70% and 100%, depending on the waste type and fraction; the lower rate of 70% applies to mixed-waste sorting facilities. The sorting efficiency of materials recovery facilities ranges from 90% to 100%, depending on the method of collection and the technology used. For the analysis it is assumed that 100% sorting efficiency is achieved only for recyclables collected on a segregated basis (direct recycling). For materials recovery facilities s a sorting efficiency of 90% is used as a standard conservative estimate.

The percentages collected on a segregated basis or treated at materials recovery facilities or mixed waste sorting facilities are calculated from information on the capacities of the sorting facilities. For simplification it is assumed that the facilities operate at full capacity. According to (USEPA 2013a, p.137) the total capacity of the 633 MRFs was 98,449 t/d. The capacity of the 43 mixed-waste sorting facilities is stated to be around 46,700 t/d (USEPA 2013a, p.139). Using these sorting efficiencies the relevant output quantities can be calculated and expressed as a percentage of the total recycled quantity of 66,200,000 t (Figure 17).[35] Table 45 shows the figures used in this calculation and the results. The calculated percentages are used for the analysis.

Table 45: Characteristics of materials recovery and mixed-waste sorting facilities

	Materials recovery facilities	Mixed-waste sorting facilities	Segregated collection
Number	633	43	
Daily throughput in t/d	98,449	46,700	
Sorting efficiency in %	90%	70%	100%
Output of recyclables in t/a	32,340,497	11,931,850	21,927,654*
Percentage of recycled volume (2011: 66.2 Mt)	49%	18%	33%

*Calculated as the difference between the total recycled quantity and the outputs of the sorting facilities

No information is available on the electricity requirements of the facilities. For simplicity and because the technology used varies widely, from simple to fully automated, an average

[35] Under the data collection system used, the recycled volume that is shown is the recycled volume after sorting (secondary products sold plus net export); e.g. the volume of paper is the amount purchased by paper mills (plus net export).

electricity requirement of 40 kWh/Mg waste input is assumed.[36] This is roughly equivalent to average values for mechanical-biological treatment plants in Germany and is applied also to the mixed-waste sorting facilities. The electricity requirement relates to the input quantity, which is back-calculated from the sorting efficiency. The sorting residues that result from the sorting process are included in the amounts shown as being sent to landfill and incineration (reporting system based on final destination).

The dry recyclables – either source-segregated or sorted – correspond to the quantities shown as "weight recovered" in Figure 19. The fractions recorded in that table that are not included in the analysis (assigned a zero rating in the GHG balance) are "rubber and leather" and "other materials", and in the standard case also "textiles". Recycling of textiles is included in this study only if textiles are collected at the doorstep or if other information is available that enables the quality of the textiles and their suitability for recycling to be assessed. That is not the case here. However, the influence of recycling is considered in a sensitivity analysis for the future scenarios to 2030. "Rubber and leather" and "other materials" were also assigned a zero rating in the previous study. An additional factor in relation to the USA is the fact that the quantities of recycled rubber include rubber from used tyres. It was not possible to remove the total quantity of recycled used tyres from the calculation. The same applies to the quoted recycled quantity of other nonferrous metals (1.34 Mt). According to further information in (USEPA 2013a), all of this is lead from lead batteries. Using the available data it was not possible to remove this lead from the calculation, because no information was available on the percentages of plastics and other materials (e.g. electrolytes). The quoted quantity of recycled plastic was retained unchanged in the analysis. However, in a departure from the procedure otherwise used for nonferrous metals in the analysis, the quantity of lead was not treated as equivalent to aluminium recycling but was rated as zero in the GHG balance.

General information on management methods for recyclables can be found in (USEPA 2013a, p. 39ff). According to this source, paper waste is used primarily in the paper industry, with small amounts being used for insulation or as pet litter. Glass waste goes to glass factories, with some also being used for insulation and in road building. Ferrous metals go to the steel industry, while aluminium and nonferrous metals go to the metal industry. Regardless of this, the further processing and re-use of the dry recyclables is inventorised using the standard emission factors given in Section 4.2.4 and explained in more detail in the Annex (Section 11.1).

For plastics a medium quality of recycling is assumed. Although information on the composition of plastic waste broken down by plastic types is available (Figure 20) and was used for the analysis,[37] this information results from the top-down survey approach of USEPA. Concrete details of the ways in which plastics are actually recycled are not available. Because of the lack of data, the conservative assumption is made that recycling is of medium quality. The effects of high-quality plastic recycling are considered in a sensitivity analysis for the future scenarios to 2030. The system of assigning plastic recycling to three quality categories is described in Section 4.2.4. The following emission factors are obtained for the USA:

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[36] State-of-the-art sorting facilities can be assumed to have an electricity requirement of around 80 kWh/Mg.

[37] The composition involved ("recovery") is 35% PE, 31% PET, 1% PP, 1% PS and 32% other.

(Details in kg CO_2-eq/Mg plastic, here in accordance with the system for reporting granulate)

"Medium plastic recycling"	Debit 538	Credit -1513	Net -974
"High-quality plastic recycling":	Debit 538	Credit -2392	Net -1854

Waste incineration

Up-to-date information on waste incineration and waste-to-energy (WtE) plants in the USA was taken from the report of the Energy Recovery Council[38] (ERC 2010). The report states that in 2010 there were 86 WtE plants in operation in 24 states, with a total throughput of more than 97,000 tons of MSW per day. This is more or less equivalent to the figures in (USEPA 2013a) for 2011.

The USA has no municipal solid waste incinerators without energy generation. The term "waste-to-energy plants" applies to both municipal solid waste incinerators and RDF power plants. Municipal solid waste incinerators in the USA are also termed municipal waste combustion (MWC) units. Irrespective of the input material, and as in Germany, all plants must meet the same emission standards. According to (ERC 2010) all WtE facilities meet the maximum achievable control technology (MACT) standards of USEPA and therefore place a smaller burden on the environment per unit of electricity produced than other electricity generating plants.

The majority of WtE plants produce only electricity. The article "Burn or bury" (Kaplan et al. 2009a) assumes that only electricity is generated and that the efficiency of the plant is 19%. In a sensitivity analysis the article considers a range of efficiencies between 15% and 30%, it being assumed that WtE facilities could achieve efficiencies that are closer to those of conventional power plants. In this study options for boosting efficiency are explored in the future scenarios. However, these scenarios assume combined heat and power generation, because from the point of view of climate change mitigation this is preferable to pure conversion into electricity.

The above-mentioned electrical efficiency of pure conversion into electricity was verified by analysing an Excel file of the ERC[39] provided to the UBA by Covanta. The gross capacity-weighted mean efficiency of the plants producing only electricity is 21.4%. The difference of 2.4% between this and the 19% according to (Kaplan et al. 2009a) is of a plausible size for internal electricity consumption. In addition, it is clear from the ERC Excel file that in addition to plants that produce only electricity there are also combined heat and power (CHP) plants and plants that produce process steam. In terms of capacity share the pure electricity generation plants predominate at 84%. CHP use accounts for 12% and process steam generation for 4%. However, no information on temperature levels is available; the quantity of steam produced is simply shown in lb/h.

In view of the data uncertainties, the USA balance in the standard case assumes pure conversion to electricity with a net efficiency of 19%. The influence of a proportion of CHP use

[38] The ERC is a national trade group or association, presumably comparable to the interest group for thermal waste treatment plants in Germany, the Interessensgemeinschaft der Thermischen Abfallbehandlungsanlagen (ITAD).

[39] Email of 14 February 2014

is considered in a sensitivity analysis. For simplicity the small proportion of process steam generation is included in the inventory as CHP (total 16%). For the proportion of CHP use the available data yields a capacity-weighted gross electrical efficiency of 15.4%. This is similar to the average value for Germany.[40] On this basis the net electrical efficiency was set at 12% in the sensitivity analysis and the thermal efficiency at 30%.

In accordance with the agreed methodology (Section 4.1.2), electricity generation was credited using the marginal approach. In the USA around 50% of electricity is generated from coal; this suggests that the substituted marginal electricity should be electricity from coal. This is also in line with the recommendation of the methodology workshop on 18 June 2012 and the method used by USEPA in its own inventories (Thorneloe 2012).

It was not possible to use the marginal approach for the OECD balance; for these countries electricity generation was credited on a standardised basis using the average national electricity mix of the 34 OECD countries. This is a further reason why the result of the USA balance cannot be directly compared with that of the OECD balance. The influence of the electricity credit under the marginal versus the average approach is compared in a sensitivity analysis for incineration.

The total incinerated amount of 26.45 Mt in 2011 includes 0.51 Mt of wood waste.[41] Because this is a small proportion of the total (1.9%), it is not considered separately. The characteristics used for incineration are the values for incineration shown in Table 44 ("Input to incineration according to Covanta"). The case of equal distribution of the residual waste to incineration and landfill and hence equal characteristics is considered in a sensitivity analysis.

5.9.3 Results: waste management – USA

The standard case

The standard case was based on the waste volume and management methods detailed in USEPA (2013a, 2013b). The bar chart in Figure 23 shows waste management methods in the USA.

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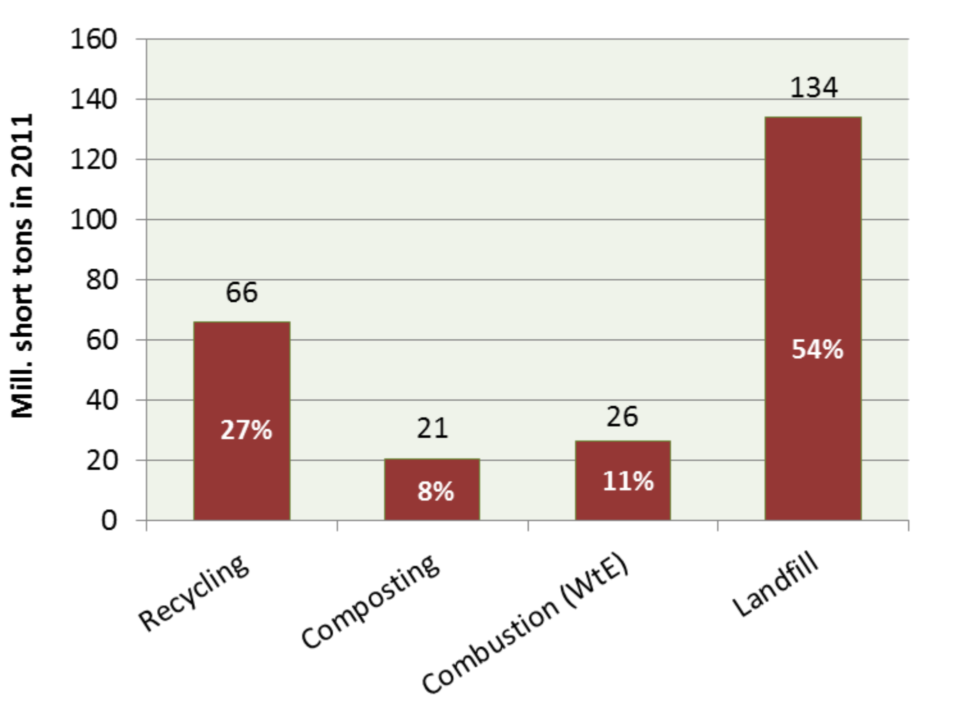
[40] Gross electrical efficiency 15%, heat use 30% according to ITAD data for (IFEU 2012); 3% of electricity assumed to be self-used

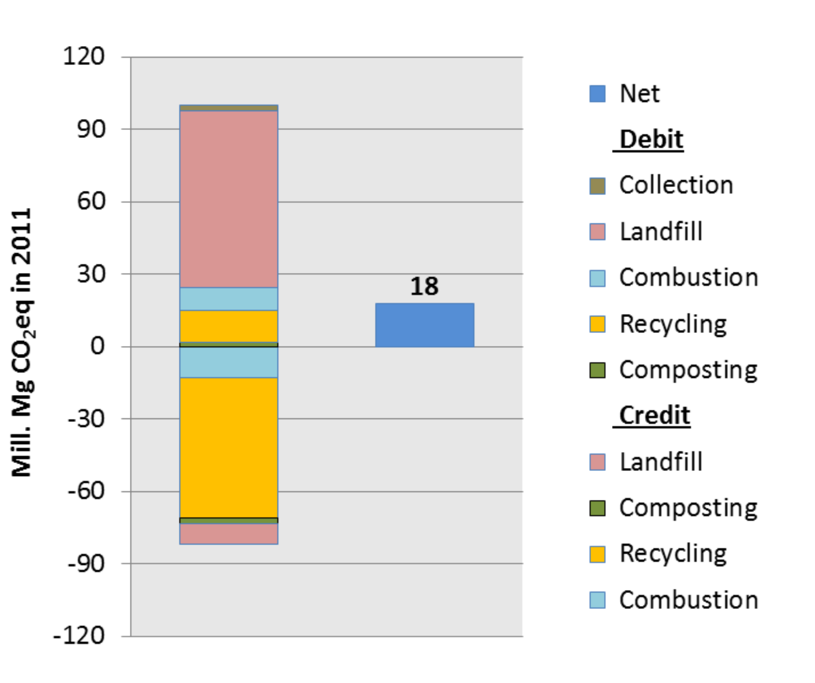
[41] The quantity of used tyres likewise included in the original volume was excluded from the analysis (see the start of the section).

Figure 23: Waste management methods in the USA according to USEPA data



The result of the GHG balance for the current waste management situation in the USA is shown in Figure 24. This indicates that waste management in the USA causes **GHG debits totalling around 18 million Mg CO_2-eq annually**. The net debit is mainly the result of methane emissions from the landfilling of waste. The GHG emissions thus caused amount to around 74 million Mg CO_2-eq. Some of this debit is offset by the use of landfill gas in CHP plants (see Section 5.9.2), but this amounts to only about one-eighth of the debit caused.

Figure 24: Results of the GHG balance for the status quo in the USA



The relatively good overall outcome is due to the significant credit from recycling. Incineration of waste results in a net credit, but because of its limited size this is of minor importance. Composting also results in a net credit but is barely visible in the overall result.

Sensitivities

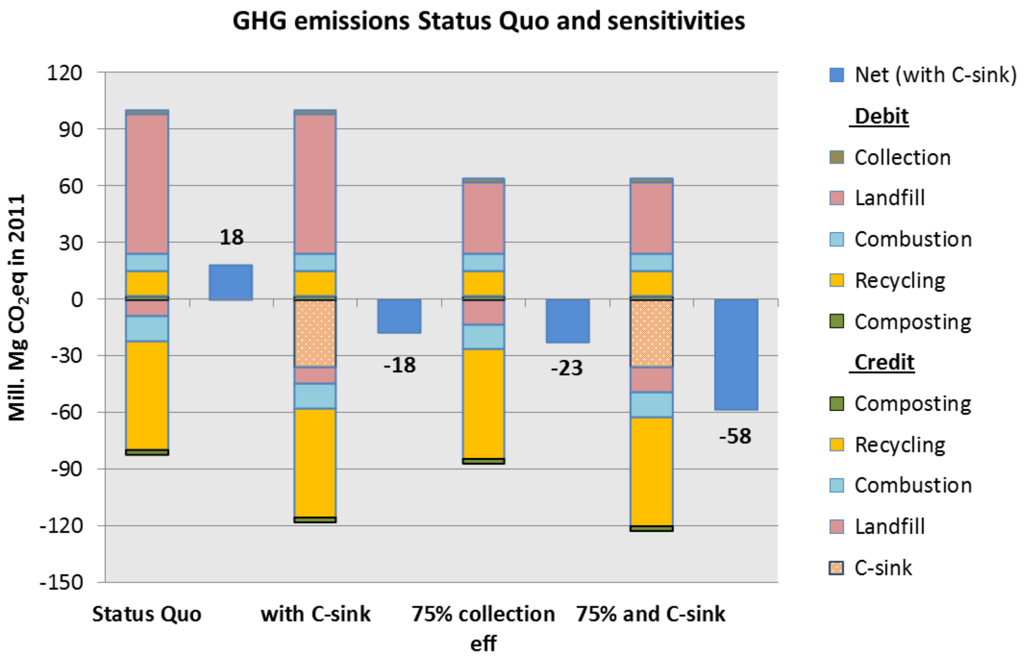
The following aspects were explored as sensitivities:

3. Result taking account of the C sink and a high effective gas collection efficiency of 75%.
3. Result of equally distributed characteristics of residual waste (“Cfoss high”) and for the electricity mix instead of marginal electricity as the equivalence process (“electricity mix”) using incineration as an example
4. Result based on waste generation in accordance with the SOG survey (EEC 2014)
5. “Wood credits”: wood saving and assumption that saved wood is used for energy

1. C sink and high effective gas collection efficiency of 75%

For the reasons mentioned in Section 4.1.2, the C sink is shown in a sensitivity analysis for information only. Figure 25 shows the result “with C sink” and also the result for a high effective gas collection efficiency of 75% (instead of 50% in the standard case) and for the combination of C sink and high gas collection efficiency (“75% with C sink”). The C sink includes the storage of humus C in compost, although the amount involved is low. The C sink is dominated by the landfilled organic component of the residual waste that in computational terms has not yet degraded.

Figure 25: Comparison of GHG balances for the status quo and sensitivities



The influence of the C sink is somewhat smaller than that of the high effective gas collection efficiency. By comparison with the standard case, both reverse the result, producing a net credit of −18 and −23 million Mg CO_2-eq respectively. In combination the two would result in a net credit of around –58 million Mg CO_2-eq. However, it should again be pointed out that the C sink is associated with considerable uncertainties and a national effective gas collection efficiency of 75% over the entire duration of the deposit is not considered realistic. A more reliable way of preventing methane emissions is to reduce the dumping of organic waste.

2. Sensitivity analysis – incineration

Using incineration as an example, the influence of the following aspects was explored:

- Result taking account of a proportion of CHP use ("with 16% CHP")
- Result with equally distributed characteristics of residual waste ("Cfoss high")
- Result with electricity mix instead of marginal electricity as the substitution process ("electricity mix")
- Combined result of equally distributed characteristics of residual waste and electricity mix instead of marginal electricity ("electricity + Cfoss")

The sensitivity "with 16% CHP" relates to the proportion of CHP use (12%) and process steam generation (4%) identified from the ERC Excel file. For the sensitivity analysis these two figures were combined, because no information on the thermal efficiency or temperature level of the process steam is available. The net electrical efficiency is set at 12% and the thermal efficiency at 30% (see Section 5.9.2, "Waste incineration").

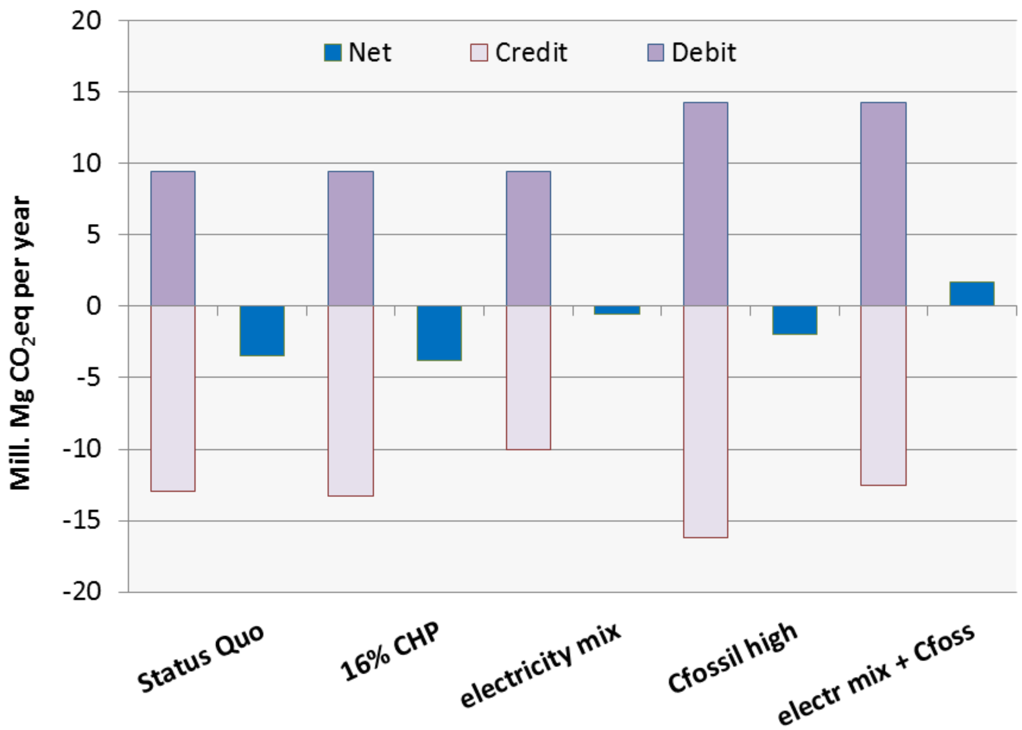
The sensitivity analysis for equally distributed characteristics of residual waste uses the characteristics obtained mathematically from the composition of the residual waste (Table 44) instead of the figures from Covanta. The higher fossil C content (15.3% instead of 9.9%) and higher calorific value (12.77 MJ/kg instead of 10.2) only affect waste incineration and are therefore only explored for incineration.

The assumption in relation to "average electricity versus marginal electricity" for generated electricity is likewise relevant mainly to waste incineration. Otherwise in the status quo electricity is generated only from landfill gas use, although in significantly smaller quantities.

Figure 26 shows the result of the sensitivity analysis. In the standard case – as already described – this results in a net credit on account of the waste incineration ("status quo"). This is increased by the proportional CHP use that was considered ("with 16% CHP"). However, because the CHP share at 16% is low, the difference is relatively small. The "electricity mix" variant is similar to the "status quo" but uses the emission factor for the USA electricity mix (775 g CO_2-eq/kWh) instead of that for marginal electricity with 100% coal as the substituted fuel (1002 g CO_2-eq/kWh). The credit for the electricity generated from waste incineration is correspondingly lower. The result is a small net credit. A high fossil C content also reduces the net credit by comparison with the status quo, despite higher calorific value ("Cfoss high"). If both were to be combined – the high fossil C content and the calculation with the emission factor for the electricity mix instead of marginal electricity – waste incineration in the USA would result in a net debit ("electricity + Cfoss").

The sensitivity analyses show the significant influence both of the characteristics and of the selected equivalence process. The marginal approach is justified for life cycle assessments in waste management provided that it is agreed that the electricity generated from the incineration of waste is "additional" electricity, which is usually the case. However, the fossil C content and the calorific value of the waste are crucial for waste incineration. If the characteristics calculated from the USEPA data apply, it is definitely recommended for the USA that the currently low share of plastic in recycling is increased. This is essential if incineration of waste is not to result in a GHG debit. However, the requirements of the US EPA Greenhouse Gas Reporting Program (GHGRP) (see footnote 32) provide an excellent instrument for monitoring the fossil CO_2 emissions of waste incineration and this should be used to introduce more extensive waste separation measures promptly.

Figure 26: Results of the sensitivity analysis - combustion



3. Waste volume according to (EEC 2014)

In Section 5.9.1 it was explained that in addition to the top-down survey approach of USEPA there is also a bottom-up survey. This survey has been conducted by the Earth Engineering Center (EEC) of Columbia University and the journal *BioCycle.* Every two years the waste authorities of the US states are asked to provide information about waste volumes and management methods in the USA; these data are compared with the top-down figures of USEPA in a BioCycle report entitled “The State of Garbage in America” (SOG). The sensitivity analysis uses the figures from the report recently published by the EEC (EEC 2014) which relate to the year 2011. (EEC 2014) is a continuation of the SOG survey produced by agreement with BioCycle. For 2011 no response was received from nine of the 50 US states surveyed; three of these stated that they did not have the relevant data. Ten of the 41 participating states were unable to provide any statistics on recycling or composting. In general little information on these areas is available, since neither composting facilities nor sorting and recycling plants are obligated to provide details of the quantities handled. The situation with regard to incinerators and landfills is different, since they must report the tonnage received. The data gaps were closed using values from the 2008 SOG survey, adjusted for an annual increase in line with population growth. For 2011 the SOG survey calculated that the total volume of waste arising in the USA was around 389 million short tons. Table 46 compares the USEPA and EEC figures.

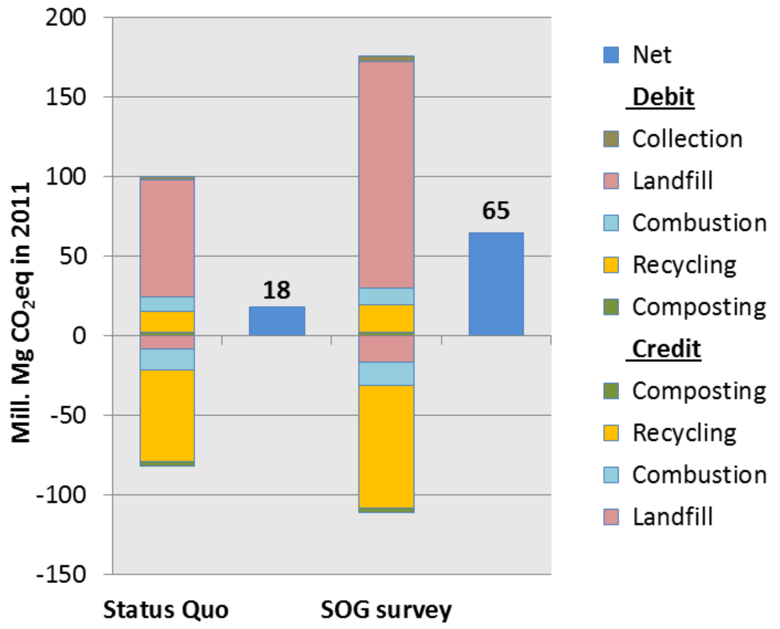
Table 46: Comparison of waste volume and management methods according to USEPA (2013) and EEC (2014)

	USEPA (2013a,b)		EEC (2014)	
Total waste arising	247,610,000 t*		388,959,390 t	
Recycling	66,200,000 t	27%	87,808,128 t	23%
Composting	20,700,000 t	8%	24,646,893 t	6%
Incineration	26,450,000 t	11%	29,507,191 t	8%
Landfill	134,260,000 t	54%	246,997,177 t	64%

*without incinerated tyres

This shows very clearly that most of the additional volume identified in the SOG survey was landfilled (64%). In absolute terms the figures in the SOG survey are higher than the USEPA ones in all areas – recycling, composting, incineration and landfill. In percentage terms, though, they are lower in all areas except landfill.

Figure 27: Comparison of the GHG balance for the status quo and the SOG survey



The data from the SOG survey yields a significantly less favourable result for the USA's waste management. According to Figure 27 the effects of composting and incineration are virtually unchanged. Recycling results in a larger credit on account of the larger quantities. However, this is more than offset by the 1.8 times larger quantity of landfilled municipal waste. The emissions from landfilling are roughly twice as high and the net result is that the GHG debit is about 3.6 times higher than with the USEPA figures.

4. "Wood credits" for wood saving and assumption that saved wood is used for energy

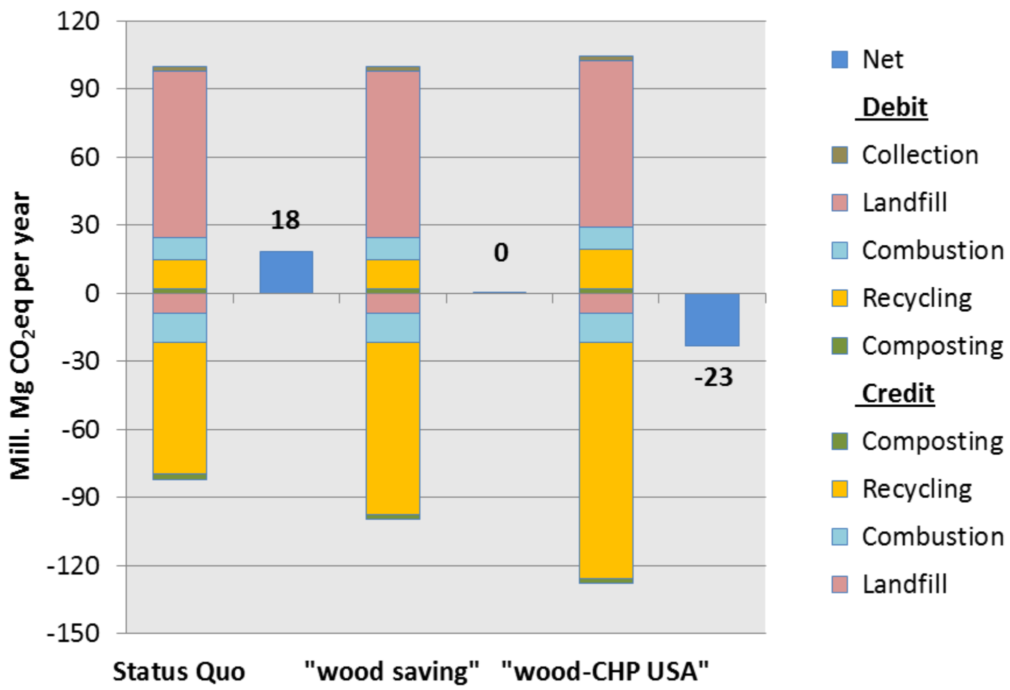
A **"credit for saved wood"**, reflecting the amount of wood saved through paper recycling, was introduced for the first time in the previous study (Öko-Institut/IFEU 2010). Because this is based on uncertain data (quantification of C accumulation in virgin forest) and depends to a large extent on the national situation (pressure of use for saved wood), it was agreed that for this study the "wood credit" would be considered only in a sensitivity analysis (see Section 4.1.2). The following three cases are considered:

- "Wood saving": Increased C accumulation in virgin forest
- "Wood CHP USA": Assumption that saved wood is converted to energy in a wood-fired CHP plant.

For wood saving, the C accumulation calculated in the previous study was used. In line with IFC Consulting (2006), a conservative estimate of 0.8 Mg C/Mg industrial wood was applied to wood left in the forest and written down over the usual period of 20 years. Converted into CO_2 this is equivalent to a credit of −147 kg CO_2/Mg fresh wood. Sorted waste paper corresponds to an input quantity of fresh wood about 2.8 times greater.

For the scenario involving conversion of the saved primary wood to energy, no information on existing wood-fired CHP plants in the USA was available. It seems that wood is very rarely used for CHP in the USA. In view of the high percentage of pure conversion to electricity at municipal solid waste incinerators, it is assumed that where wood is used for energy it is also only electricity that is generated. In a departure from the previous study, the net electrical efficiency is assumed to be 15% and it is assumed that there is no thermal use. As Figure 28 shows, the effect is nevertheless high, because electricity from coal is substituted (marginal approach).

Figure 28: Sensitivity of "Credit for saved wood" for recycling of paper and wood



Taking "wood saving" into account reduces the net debit to nearly zero. The cause is the very large quantity of recycled paper, which constitutes almost 70% of the tonnage recycled. The "wood saving" calculated here is in principle equivalent to the offset for C sequestration in (USEPA 2006). However, the emission factor calculated here for the credit is significantly lower than the credits in USEPA (2006) (see Table 61). Because of the major uncertainties with regard to the actual C storage, these credits should always be specified separately and not as part of the standard balance.

If saved wood were used for energy in the USA (“wood CHP USA"), this would result in a noticeable net credit. At this point it should be pointed out that the USA has the second-highest paper consumption in the world (after China) and that its per-capita consumption of around 230 kg is also relatively high (as at 2011; China 72 kg, Germany 247 kg).[42] This highlights the fact that while the credit for wood saving rewards recycling, it also rewards this careless consumption behaviour. Given that the world’s forests are dwindling, this sensitivity for the USA sends the wrong signal. It view of these findings it is recommended that the credit for the conversion to energy of wood saved through material recycling is not used in future inventories.

5.9.4 Future scenarios to 2030

Baseline comparison

A medium and an ideal scenario were drawn up for each of the countries considered. It was decided that a comparison on the basis of a higher total tonnage of waste would not be performed, because experts (Thorneloe 2012) do not consider that any significant change in waste management in the USA is likely in future. A moderate increase in waste generated is probable, with the additional quantities that arise being landfilled. The effects of such a trend can be seen in the sensitivity analysis based on the figures from the SOG survey (Figure 27).

It was not possible to draw on national plans or programmes in compiling the future scenarios, because no corresponding recommendations for future waste management exist in the USA. The country merely has resource conservation targets set by USEPA, implementation of which is the responsibility of the individual states. The calculations performed in relation to possible developments in the USA are “what ifs”, with no programmatic background. Similarly, the Climate Action Plan of the US government published in March 2014 (The White House 2014) states that methane emissions make up 9% of total GHG emissions in the USA and that landfills are the world’s third-largest source of methane emissions, but it relies on voluntary measures and stakeholder processes (page 5):

> “EPA will release a proposed update to its current standards for new municipal solid waste landfills in the summer of 2014, including assessing opportunities for further minimizing emissions when landfills are built or modified. Since there may be an even bigger opportunity for reducing methane emissions at existing landfills, EPA will also issue an Advanced Notice of Proposed Rulemaking (ANPRM) by June 2014 to engage industry and stakeholders on a range of approaches for cutting methane-rich landfill gases currently being emitted by existing facilities.”

Other measures mentioned are the further involvement of USEPA in voluntary LFG projects through the Landfill Methane Outreach Program, and the call from USEPA and the US

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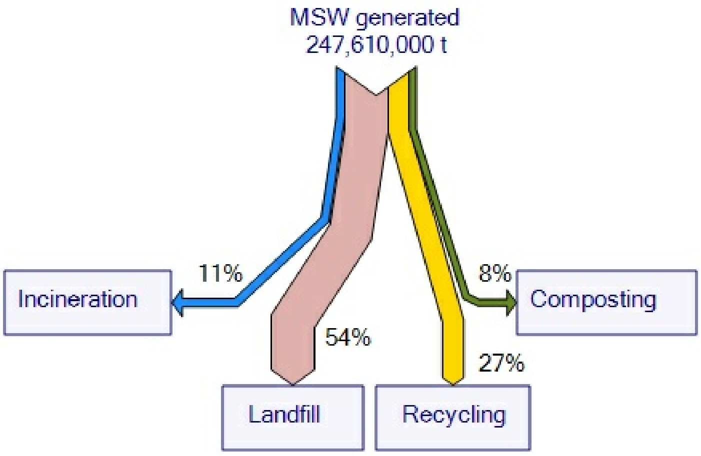
[42] http://de.statista.com/statistik/daten/studie/5959/umfrage/verbrauchsmenge-von-papier-in-ausgewaehlten-laendern/

Department of Agriculture (USDA) to reduce or reuse food waste through the US Food Waste Challenge.

In the light of this it was agreed that the future scenarios for the USA should be based on rising recycling rates. The results are compared with the status quo (Figure 29), which can be interpreted as a "business as usual" scenario in accordance with the prediction of unchanged waste management in 2030.

Figure 29: The status quo for waste management methods in the USA as a starting point for the future scenarios to 2030



Taking as a starting point the overall recovery rate (sum of recycling and composting) in the status quo of 35%, the medium future scenario assumes a moderate increase in this rate to 45%, while the ideal scenario assumes that it rises to 60% in line with the requirements of the EU Waste Directive 2008.[43] In addition, the ideal scenario assumes – for all countries equally – that landfilling is abandoned. Overall the following waste management methods are assumed for the future scenarios to 2030:

2030 medium:	45% recovery, 25% incineration, 30% landfill
2030 ideal:	60% recovery, 40% incineration, 0% landfill

The increased recovery rates were obtained using plausibly increased fraction-specific recovery rates (Table 47). In this process the waste fractions of "rubber, leather", "other nonferrous metals" and "other", which are not considered further, were assumed to remain constant. The fraction-specific recovery rates in the ideal scenario largely correspond to those assumed for the OECD inventory. The exceptions are food waste, wood and paper. Because kitchen waste is rarely segregated at source in the USA, the rate was set to a mere 30% (instead of 70%). Wood is not specified for the OECD countries; for the USA the rate was increased from 15% to 30% in the ideal scenario. In the medium scenario the rates for both wood and food waste were kept constant. The recovery rate for paper in the USA is already high in the status quo at 66%. For

[43] Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives (Official Journal of the European Union of 22 November 2008 No. L 312 p. 3); last amended on 26 May 2009 (OJ EU of 26 May 2009 No. L 127 p. 24).

the medium scenario a slight increase to 70% was assumed; in the ideal scenario a further increase to 80% was applied, which is higher than for the OECD balance (where it is 70%).

Table 47: Recovery rates of the waste fractions and overall for the status quo and future scenarios to 2030

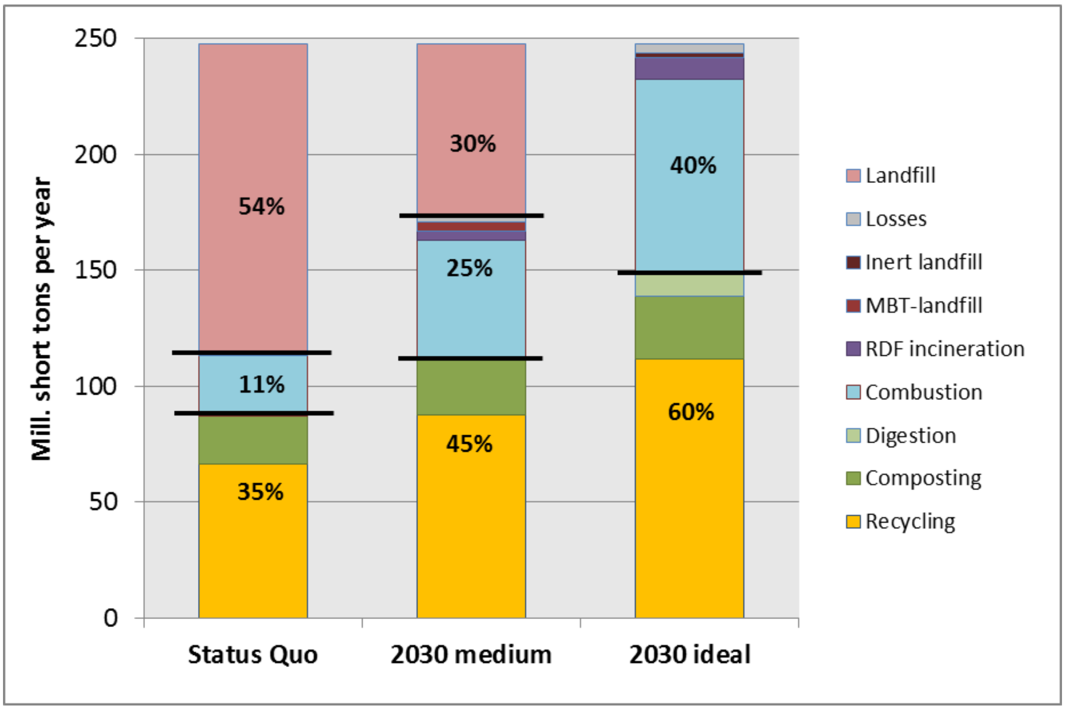
	Status quo		2030 medium		2030 ideal	
	Recovered (t)	Rate	Recovered (t)	Rate	Recovered (t)	Rate
Food waste	1,400,000	4%	as status quo		10,771,035	30%
Green waste	19,300,000	58%	23,399,145	70%	26,741,880	80%
Wood	2,380,000	15%	as status quo		4,824,000	30%
Paper	45,900,000	66%	49,014,000	70%	55,016,000	80%
Plastic	2,650,000	8%	12,736,000	40%	19,104,000	60%
Ferrous metals	5,450,000	33%	9,912,000	60%	14,868,000	90%
Aluminium	720,000	21%	1,388,000	40%	2,429,000	70%
Glass	3,170,000	28%	5,735,000	50%	8,029,000	70%
Textiles	2,000,000	15%	constant, considered as sensitivity			
Nonferrous metals, other*	1,340,000	68%	constant, not considered further			
Rubber, leather	1,310,000	18%	constant, not considered further			
Other	1,280,000	16%	constant, not considered further			
Total recovered and overall recovery rate in relation to total waste generated						
	86,900,000	**35%**	**111,894,145**	**45%**	**148,712,915**	**60%**

*Lead from lead batteries, not included in GHG balance

As another aspect of the future scenarios it was also assumed for the USA balance that a proportion of the residual waste is treated by means of anaerobic MBT (2030 medium) or in MBS facilities (2030 ideal) (see Section 4.2.7). Based on the relevant percentage of “incineration” (25% in the medium scenario, 40% in the ideal scenario), a split of 80% direct incineration and 20% treatment via MBT/MBS was applied in the baseline case. A split of 50:50 was explored in a sensitivity analysis.

The resulting waste management methods in the future scenarios by comparison with the status quo are shown in Figure 30. In this diagram final destinations are shown based on the output streams of the MBT/MBS. *“RDF incineration”* represents the RDF fraction produced: 50% of this is used in RDF power plants and 50% is co-fired (in power plants/cement works). *“MBT landfill”* represents the anaerobically treated MBT residue from the MBT plant, which is landfilled with the release of only small residual methane emissions. *“Inert landfill”* is the inert fraction that is separated out and landfilled after biological stabilisation in the MBS. This produces no further GHG emissions. *“Losses”* are water losses and losses from biological degradation in the MBT or MBS. *“Waste incineration”* includes both the “usual” waste incineration and the fraction “Impurities from MBT to MSWI” (see Table 11).

Figure 30: Waste treatment – status quo and scenarios to 2030



As a result of the increased recovery rates in the future scenarios, the composition of the residual waste and hence also the characteristics of that waste also change. Deviating from the status quo, the composition of the residual waste was assumed to be identical for the incinerated and landfilled waste volumes in the future scenarios. With increasing percentages going to incineration in the future scenarios, the assumption that plastics are predominantly landfilled cannot be plausibly retained. The composition of the residual waste in each of the future scenarios is shown in Table 48; the resulting characteristics calculated for the GHG balance are shown in Table 49.

Table 48: Composition of residual waste after recovery in the future scenarios

	Medium scenario 2030		Ideal scenario 2030	
	Mt	%	Mt	%
Organic waste	45.22	33.3	32.51	32.9
Wood	13.70	10.1	11.25	11.4
Paper	21.01	15.5	14.00	14.2
Plastic	19.10	14.1	12.74	12.9
Rubber and leather	6.18	4.6	6.18	6.2
Ferrous metals	6.61	4.9	1.65	1.7
Nonferrous metals	2.08	2.0	1.66	1.7
Glass	5.73	4.2	3.44	3.5
Textiles	11.09	8.2	11.09	11.2
Other	4.37	3.2	4.37	4.4
Total	**135.71**	**100.0**	**98.9**	**100.0**

Table 49: Calculated characteristics of residual waste in the future scenarios

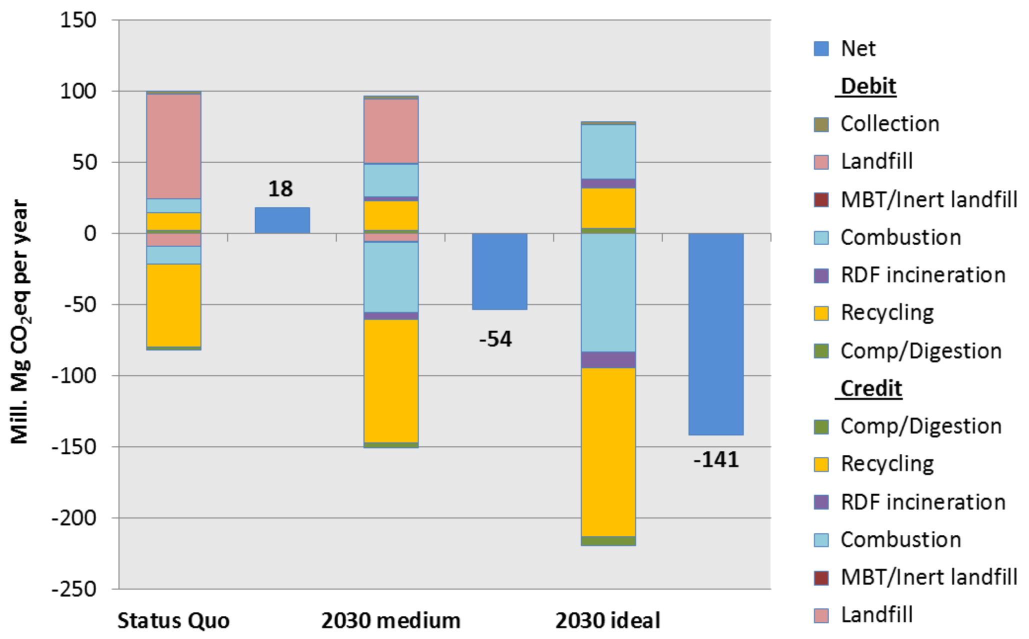
	Medium scenario 2030	Ideal scenario 2030
Biogenic carbon (%)	17.9	19.0
Fossil carbon in (%)	12.7	12.8
Calorific value (kJ/kg waste)	12,048	12,406

The other boundary conditions for the future scenarios to 2030 are described in more detail in Section 4.2.; the most important are listed again below:

- CHP operation for waste incineration, net efficiencies: 18% electrical, 42% thermal
- Net efficiencies for RDF power plants: 25% electrical, 20% thermal
- Substitution of coal for RDF co-incineration on the basis of equivalent calorific value
- Anaerobic digestion as described in Section 4.2.6, composting unchanged
- Mass flows for anaerobic MBT and MBS and RDF characteristics see Section 4.2.7
- Emission factors for dry recyclables recovery constant; high-quality plastic recycling in sensitivity only (see Section 4.2.4)
- Landfilling unchanged (OX = 10%, effective gas collection efficiency 50%)

Overall this therefore involves only a few technical improvements, which in the main address incineration. In consequence the differences by comparison with the status quo shown in Figure 31 are attributable mainly to the redirection of the waste stream (reduction in landfilling and an increase in recovery and incineration) and to CHP use where waste is incinerated.

Figure 31: Results of the GHG balance for the status quo by comparison with the future scenarios to 2030



The results show that the reduction in the landfilled proportion from 54% to 30% in the medium scenario is accompanied by a corresponding reduction in methane emissions. There is at the same time an increase in the emissions from increased recovery and incineration, but these are more than offset by the associated credits. Although different characteristics were used for incineration in the future scenarios, they involve a similar ratio of calorific value to fossil C content, so that the changes in the scenarios to 2030 (significantly higher credits than debits) are attributable mainly to the CHP use. The specific net result for waste incineration increases in the medium scenario from −131 kg CO_2-eq/t[44] waste to −520 kg CO_2-eq/t waste. In the ideal scenario the specific result is even higher at −547 kg CO_2-eq/t waste; this is because of the altered characteristics of the residual waste (see Table 49).

As a result of the cessation of dumping and in particular the corresponding increase in recovery, the ideal scenario again produces a net credit 2.6 times larger than the net credit in the medium scenario. In addition, the waste incineration is also beneficial here on account of the CHP use instead of pure electricity generation. The amounts treated by anaerobic MBT in the medium scenario or by MBS in the ideal scenario are comparatively small and thus make a less noticeable contribution to the net credit.

Sensitivities

As sensitivities for the scenario comparison the following aspects were considered:

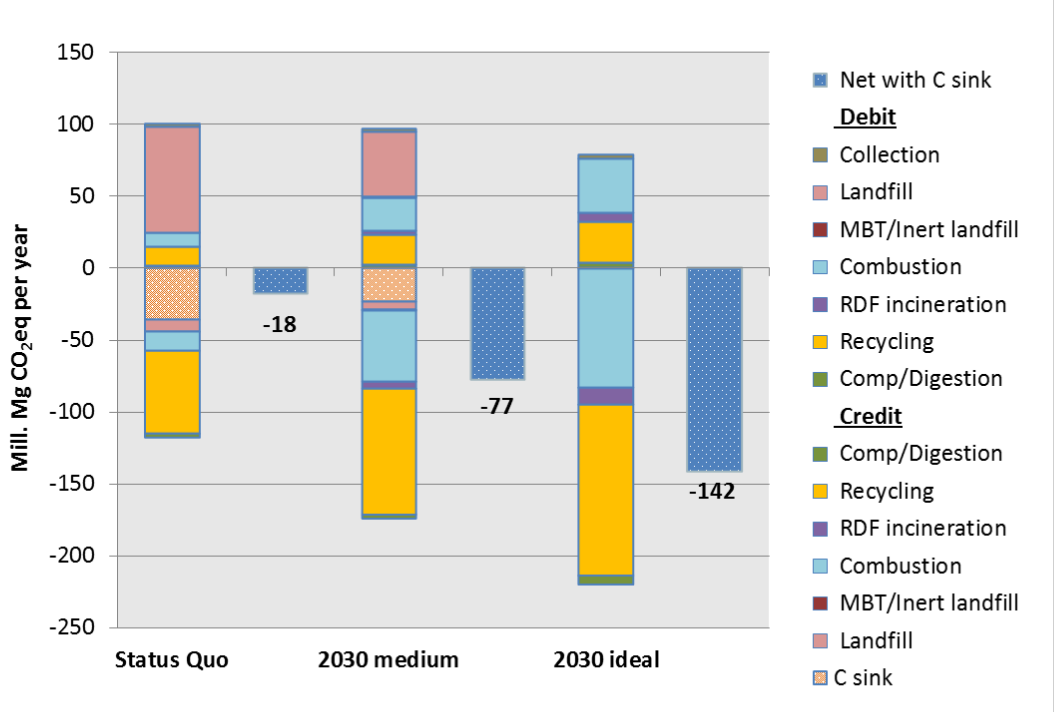
- Result taking account of the C sink
- Result assuming high-quality rather than medium plastic recycling and taking account of textile recycling
- Result with a 50:50 (instead of 80:20) split between waste incineration and MBT/MBS

Accounting for the C sink

For the reasons mentioned in Section 4.1.2, the C sink is considered in a sensitivity analysis for information purposes only. Figure 32 shows the result of the scenario comparison “with C sink”. For the status quo the result is as shown in Figure 25. In each of the future scenarios the influence of the C sink falls as a result of the reduction or cessation of landfilling. In the ideal scenario the C sink arises only from the use of compost and composted digestate; it produces only a slight increase in the net credit.

[44] Net value with marginal electricity as substitution process; for electricity mix the value would be around −20 kg CO_2-eq/t waste

Figure 32: Results of the sensitivity analysis – GHG balances in the scenario comparison with C sink

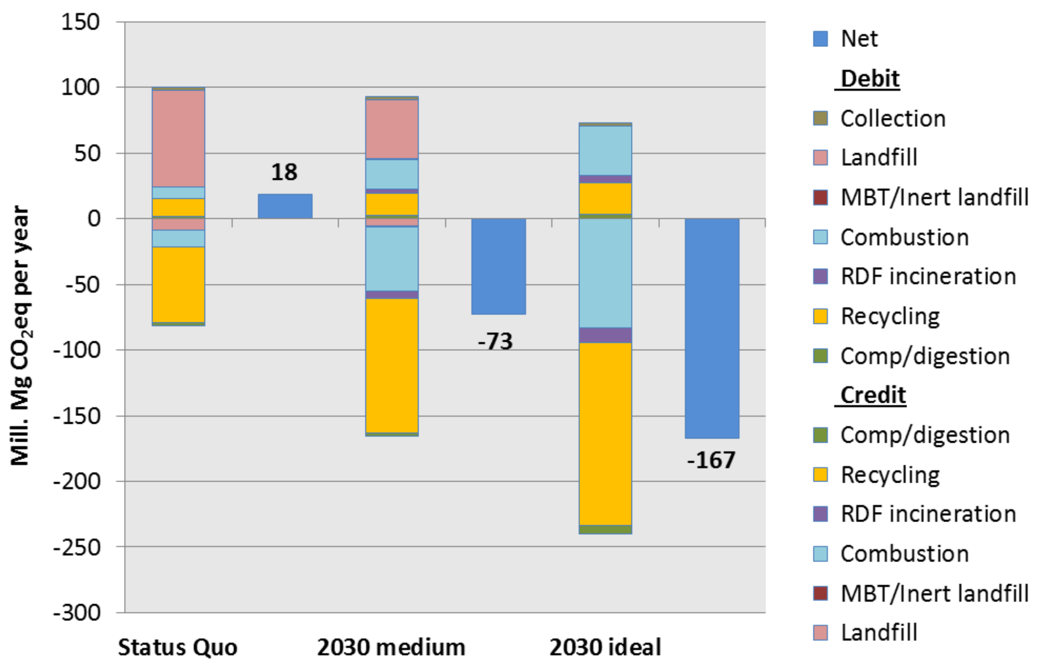


Including textile recycling and high-quality plastic recycling

Textile recycling features in the USEPA figures but without further details of the methods of collection and reuse. Because the USEPA figures are derived from the production statistics, it is not safe to assume that all the textiles are actually reused. For this reason the impact of reuse is considered in this sensitivity analysis. For plastics recycling there are likewise no further details of type and quality. The plastic types detailed in the USEPA figures (Figure 20) are also obtained using USEPA's top-down survey method. For the standard case the conservative assumption was made that recycling is of medium quality. The impact of high-quality plastic recycling is considered in this sensitivity analysis. The emission factors for the medium and high-quality recycling of plastic waste in the USA are given in Section 5.9.2.

The results in Figure 33 show that the sensitivity analysis yields a significantly higher net credit in the future scenarios. In the medium scenario the net credit is around 19 million Mg CO2-eq higher; in the ideal scenario, where recycling rates are increased further, the net credit is as much as some 26 million Mg CO2-eq higher. Eighty percent of this increase is attributable to the high-quality plastic recycling that is assumed. The textile recycling that was also considered results in a higher specific net credit (–2810 $_2$-eq/Mg recycled compared with –1854 C high-quality plastic recycling), but the volume of recycled plastic waste in the analysis is significantly higher (around 12 Mt as against 2 Mt textiles) and makes a correspondingly higher contribution to the absolute result.

Figure 33: Results of the sensitivity analysis – GHG balances in the scenario comparison with recycling as the sensitivity



50:50 split between waste incineration and MBT/MBS

For the future scenarios the baseline case for the USA and the OECD countries assumes a moderate percentage of residual waste that is treated by means of anaerobic MBT or MBS. The ratio of incineration to treatment via MBT/MBS was set at 80:20. The background is that at present there are only a few countries in which MBT/MBS technology is widely used and there is no expectation that a significantly higher percentage than in Germany would be achieved. A ratio of 50:50 was considered in the sensitivity analysis.

Figure 34: Mass flows with an 80:20 and 50:50 split – medium scenario to 2030

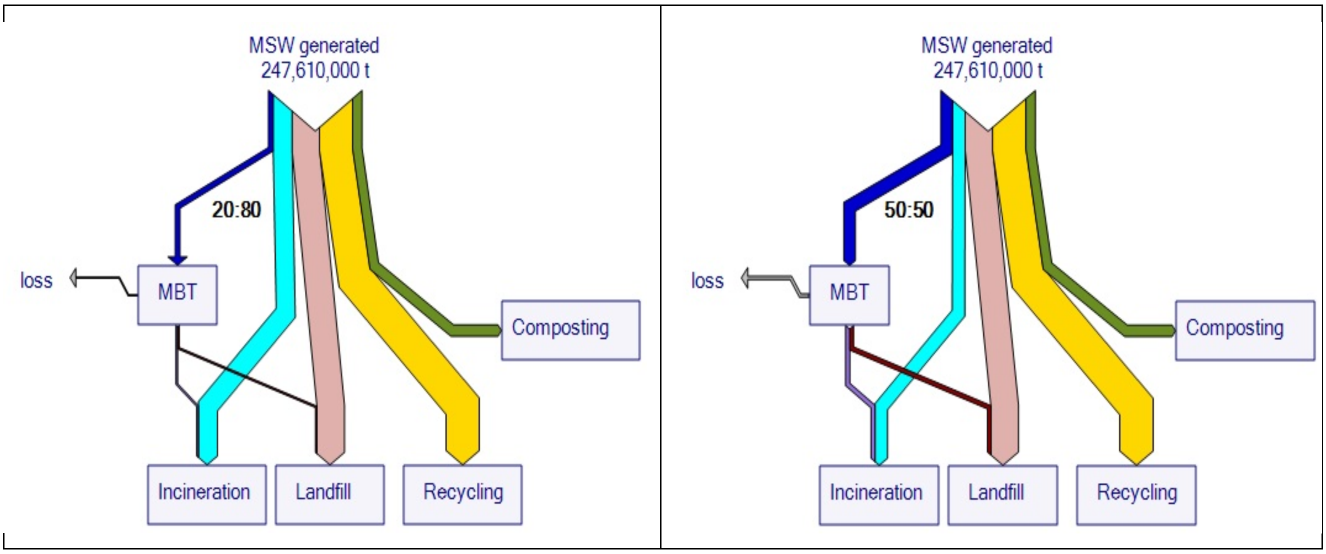
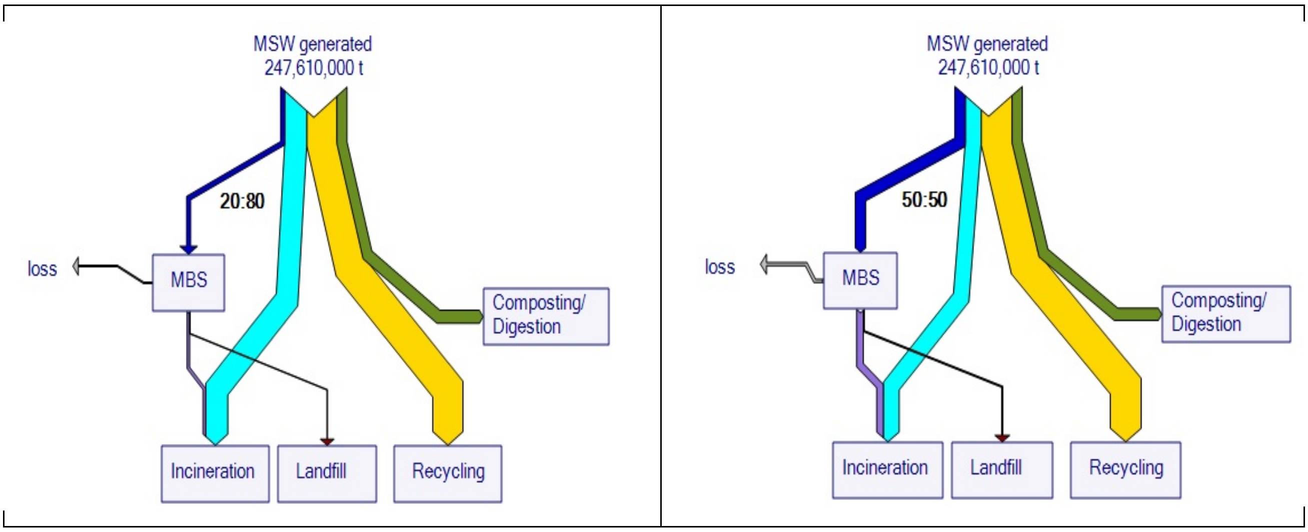


Figure 35: Mass flows with an 80:20 and 50:50 split – ideal scenario to 2030



Figures 34 and 35 show the mass flows for the ratios in the baseline case (80:20) and the sensitivity analysis (50:50) for the medium and ideal scenarios. In both scenarios it is clear that the amount treated by means of MBT/MBS is relatively small and the amount of RDF produced for energy generation is even smaller. In the case of MBS it is only this amount that produces the achievable credit effect; in the case of anaerobic MBT biogas use also plays a part. For MBS the RDF fraction amounts to around 49% of the input; for anaerobic MBT it is around 35% (see Table 11). The remainder consists of impurities that are sent to incineration (included there in the result), metals (small proportion, assigned to “recycling” in the classification system based on final destination), losses, inert materials with no GHG impact and in the case of MBT also MBT residues with low residual methane emissions from landfilling of these residues.

In the 50:50 sensitivity analysis the proportions treated by MBT/MBS increase at the expense of the proportion sent to incineration. The quantities for recycling and composting (anaerobic digestion) and the amounts of residual waste sent to landfill (only relevant to the medium scenario) are unaffected by the sensitivity analysis. For the RDF fraction produced it was generally assumed that 50% is used in an RDF power plant and that 50% is co-fired in power plants or cement works. The characteristics of the RDF from MBT and MBS are given in Section 4.2.7. These are average values for Germany taken from the literature.

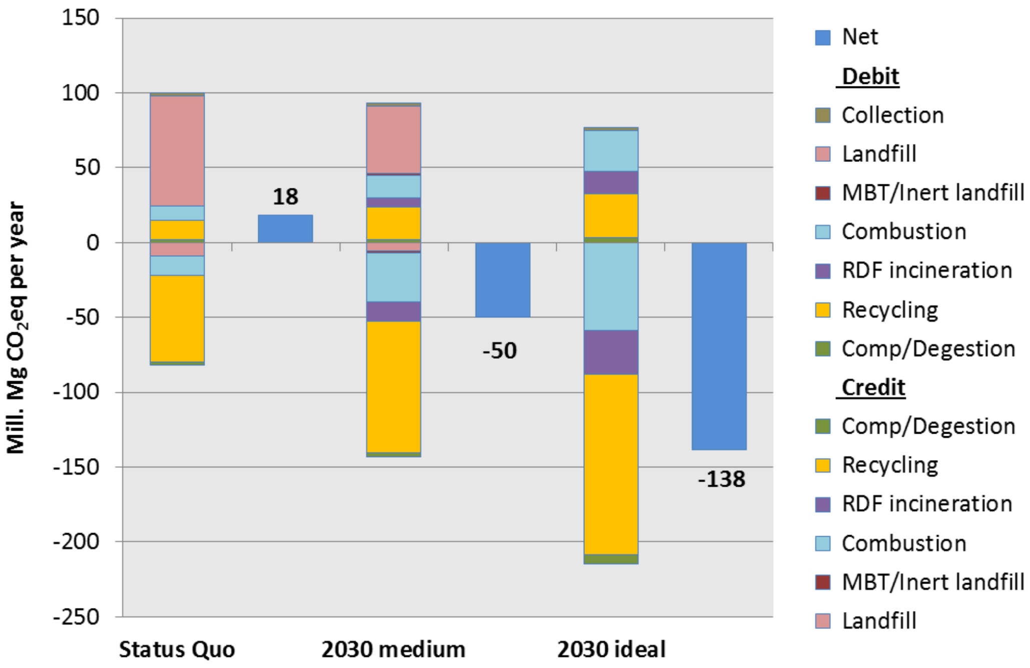
The result in Figure 36 shows that as a result of the increased treatment by MBT or MBS instead of incineration the net credit by comparison with the baseline case (Figure 30) is reduced – in the medium scenario by around 4.1 million Mg CO_2eq and in the ideal scenario by around 3.3 million Mg CO_2eq. The background is that incineration leads to a better specific net result than the treatment by MBT or MBS, as the values in Table 50 show.

Table 50: Specific results for incineration and treatment by MBT or MBS in the future scenarios

Values in kg CO_2eq/t input	Debit	Credit	Net
Incineration 2030 medium	453	−973	−520
Anaerobic MBT 2030 medium	241	−474	−233
Incineration 2030 ideal	455	−1,002	−547
MBS 2030 ideal	308	−587	−279

The different values for incineration in the medium and ideal scenarios are caused solely by the somewhat different characteristics (especially calorific value, see Table 49). The treatment via MBT/MBS involves lower debits but also results in lower credits by comparison with incineration.

Figure 36: Results of the sensitivity analysis – GHG balances in the scenario comparison for the split between incineration and MBT/MBS



It should be noted, however, that the reliability of these results is limited, since the characteristics of the RDF had to be taken from the literature. It is entirely possible that other, more favourable figures could be obtained for RDF from MBT or MBS in the USA's waste composition. The key factor here is the relationship between calorific value and fossil C content. This is more favourable for RDF from MBS than for RDF from MBT, because RDF from MBS contains organic waste components. On the other hand, anaerobic MBT has the advantage of biogas production and use. In the specific net result the two concepts are relatively close to each other.

To achieve a specific net result equal to incineration, the calorific value of the RDF from MBT, for the same fossil C content, would have to be at least 22 MJ/kg (instead of 13.2 MJ/kg), while for the RDF from MBS it would have to be 19 MJ/kg (instead of 13.4 MJ/kg). It is not possible to judge whether higher calorific values in the RDF for the same fossil C content are plausible for the USA. For the given characteristics the treatment by means of MBT or MBS would yield a somewhat better result if 100% of the RDF were used for co-incineration in power plants or cement works instead of 50% of it being used in RDF power plants.

5.9.5 Conclusions – USA

The results of the GHG analysis for the USA show that the landfilling that is currently still the predominant practice results overall in a GHG debit. A sensitivity analysis with a higher effective gas collection efficiency and/or inclusion of a C sink reverses the net result, but in neither case can the effect be reliably proven. Methane emissions are reliably avoided by reducing or avoiding the dumping of organic waste. Both the future scenarios that were considered result in a very clear improvement by comparison with the status quo.

While the above findings are unambiguous, uncertainties remain with regard to the actual volume and composition of the waste and the resulting characteristics of the incinerated and landfilled waste. This is a result of USEPA's data collection methods, which in all probability do not produce an accurate picture of the actual waste streams. Uncertainties with regard to the quantities of municipal waste that are landfilled and incinerated also arise in connection with the alternative method of collecting primary data from the waste authorities of the US states (the SOG survey), since the fractions are not necessarily weighed separately upon delivery. In addition, there are uncertainties with regard to the recycled and composted amounts, because neither recycling facilities nor biological treatment facilities are required to document and report their activities.

The uncertainties are particularly relevant to the volume and management methods of plastic waste: according to the USEPA survey, this constitutes a relatively high percentage (13%) of MSW and only 8% of it is recycled. From the climate change mitigation perspective, the emphasis should wherever possible be on high-quality recycling of plastic waste. Thermal treatment does not result in a GHG credit unless high efficiencies are achieved and it is mainly electricity from coal that is substituted. By contrast, high-quality material recycling of plastic has high credit potential.

To improve the data situation and facilitate a more precise assessment of the GHG reduction potential for the USA, it is desirable to:

- introduce a general requirement to document the treatment of municipal solid waste; quantities should be recorded by weighing, including at recycling and biological treatment facilities
- conduct an analysis of the residual waste sent to landfill, to include determination of the calorific value and the fossil and biogenic C content.

To promote further development of closed cycle management, leading ultimately to full utilisation of the GHG reduction potential, the following general recommendations can be made:

- Strengthen recovery structures, in particular by promoting an infrastructure for material recycling of plastic waste involving sorting techniques that sort plastics by type and subsequent high-quality processing into secondary granulate.
- Draw up a strategy for the progressive reduction of landfilling, with anaerobic MBT or MBS as a bridging technology wherever appropriate.
- Long-term goal: high recycling rates with high-quality recycling and high-quality conversion to energy of the unavoidable sorting and processing residues in WtE plants or if possible through co-incineration.

In this context "high quality" means "high potential for substituting primary resources" – and "while ensuring good hygienic and sanitary conditions" for workers (including those in the informal sector in developing countries and emerging economies).

The above recommendations represent the requirements for achieving the goal. Realistic knowledge of waste streams and waste characteristics is essential for planning and appropriate management of material flows. It is also crucial for development of the infrastructure required for high-quality processing and recycling. Without details of the actual volume of waste and its characteristics, it is not possible to make meaningful plans for the number or type of treatment facilities to be used as alternatives to landfilling. Efforts to establish a market for the secondary products that arise (in particular compost, secondary granulate, RDF where appropriate) are also important. The high quality of the input must be emphasised in order to create the corresponding demand. If this succeeds, closed cycle management will become economically viable.

6 Waste management – India

Data for the analysis for India are drawn from official information and other relevant publications. However, as is the case with many developing countries and emerging economies, only limited reliable information on waste volumes and management methods is available for India. The data obtained for India and the assumptions made, including those relating to the future scenarios, were discussed with representatives of government agencies and other experts at a workshop in New Delhi on 7 November 2012 organised by GIZ. The workshop not only yielded important information for the assessment of emissions but also highlighted some important concerns of the stakeholders and some suggestions for improvements. For a summary and background information on the workshop see the Annex (Section 11.3).

6.1 Waste generation and composition

With 1.2 billion inhabitants, India is the most populous country in the world after China. According to the Indian census of 2011, around one-third of the population or approximately 377 million people live in urban areas. The urban population is larger than the total population of the USA (308.7 million).

The actual volume of waste generation in India is not known. In (WBI 2008, p.17) the volume of waste generation in urban areas is put at 42 Mt. According to (Annepu 2012, p.36) it is generally assumed that around 50 Mt of waste is generated annually in urban India. On the basis of his own analysis of 366 cities (representing 70% of the urban population) the author arrives at an annual volume of waste in 2011 of almost 70 Mt for urban areas. For rural areas only isolated details are available. According to (MoEF 2010, p.4), urban areas generate significantly larger quantities of waste than rural ones: this is on account of economic development and rising consumption. Some figures for waste generation in rural areas are cited in (Annepu 2012, p.31), but because they are not representative of rural areas as a whole use of these figures is not recommended. Estimates of waste volumes for India overall can be found in (MoEF 2010). On the basis of a study conducted by the Central Pollution Control Board (CPCB) and census data for 2008, waste generation for India as a whole is estimated here to be 0.573 Mt/d, which is equivalent to around 209 Mt/a.

The total volume of waste generated in India can also be extrapolated from data on average per-capita generation. On this point, too, differing information is available. According to the Central Public Health and Environmental Engineering Organization (CPHEEO), average per-capita generation in cities varies between 0.2 and 0.6 kg/(cap*day) (WBI 2008, Kumar et al. 2009). In (CPCB 2000) the average per-capita generation in cities is put at 0.376 kg/(cap*day). In India's Second National Communication to the UNFCCC (MoEF 2012, p.76), calculations of greenhouse gas emissions are based on assumed average per-capita generation for India as a whole of 0.55 kg/(cap*day). On the basis of the population figures in the Indian census for 2011, these two average values would correspond to annual waste volumes for the whole of India of around 166 Mt and 243 Mt respectively.

The problem of quantifying waste generation was one of the subjects raised at the workshop in Delhi. Participants confirmed the difficulty of obtaining reliable figures. For information on waste volumes, a new study recently published by the World Bank was recommended (WBI 2012). This is a global review of solid waste management containing data on all the countries covered. The waste generation figures quoted for India are taken from a World Bank study of 2006 and result in urban waste generation of 40 Mt/a. The waste composition is based on data for 2004. For the calculations in this study the data in (WBI 2008) was used in both cases, since it is more up to date and more comprehensive.

The absolute volume of waste generation is ultimately relevant only for comparative purposes such as categorisation of the results by comparison with the GHG calculations for the UNFCCC or categorisation in the global context. For the latter purpose in particular use of a range was agreed with the 42 Mt reported in (WBI 2008) as the lower value and the 243 Mt calculated from the communication to the UNFCCC (MoEF 2012) as the upper value. The lower figure is used for the baseline case; the upper one is used in a sensitivity analysis.

Differing details of waste composition are also found, as Table 51 shows. In principle the official figures were used where possible. However, the most recent figures are contained in (CPCB 2000). Comparison of the older and more recent data in Table 51 reveals that a significant difference lies in the reduction of the inert waste fraction. This trend is also confirmed in (MoEF 2010), according to which Indian waste typically consists of 51% organic waste, 17% recyclables, 11% hazardous waste and 21% inert material. The inventory uses the data for 2005 in (WBI 2008). The advantages over (Annepu 2012) are twofold: firstly, the World Bank study is to a large extent officially accepted and, secondly, it differentiates between various dry recyclables. In addition, the data in (Annepu 2012) apply only to collected waste (delivery to landfill) and not to the total waste generated.

Table 51: Waste composition in India according to various sources

Details in %	(WBI 2008) for 2005 *Values used in this study*	(WBI 2008) for 1995	(Annepu 2012) for 2011 (collected waste)	(Sharholy et al. 2008) CPCB 2000 for Metrocities
Compostable waste	47.43	42.21	51.3	41.8
Paper	8.13	3.63	17.48	3.5
Plastic, rubber	9.22	0.6		3.9+0.8
Metals	0.5	0.49		1.9
Glass	1.01	0.6		2.1
Textiles	4.49	-		5.7
Inert material	25.16	45.13	31.21	40.3
Other	4.016	-		
Water content			47	30
Calorific value (MJ/kg)			7.3	7.433

6.2 Waste collection and management methods

Waste collection in India is not universal. Separate public collection of recyclables and organic waste does not occur or is limited to a few pilot schemes. On average it is assumed that the collection rate is 50-90%; in some Indian cities, though, the collection rate is only 25% (MoUD/CPHEEO 2005). In (MoEF 2010, p.3) the average collection rate for India as a whole is cited as 60%. The remaining 40% is not collected but is discarded wholesale or scattered. The activities of the informal sector are not included in the data.

However, a system of informal collection of recyclables has become established in India. Newspapers, glass, cans, plastic bags and old clothes are collected separately in homes and businesses and sold to waste purchasers who go from door to door (WBI 2008, p.133). The quantity of waste collected in this way is not recorded. (Annepu 2012, p.70) estimates that this quantity may be up to four times larger than the quantity sorted and recycled from formally collected waste. (Annepu 2012, p.70) assumes that the informal sector sorts and recycles 20.7% of recyclables. According to (Annepu 2012), recyclables comprise 17.48% of waste; doorstep collection is thus responsible for recycling around 15% of the generated waste (20.7%*17.48%*4 = 14.5%).

According to (WBI 2008), around 4 Mt (9.5%) of the total generated waste in urban areas of 42 Mt is taken for recycling and a further 4 Mt is dumped in unmanaged landfills. This yields an average collection rate for cities of 81%. Although the collection rate is likely to be lower nationally, as stated in (MoEF 2010), the data in (WBI 2008) are nevertheless used for the inventory, because they are the only source of a complete picture of the situation. The effects of a higher proportion of scattered waste of 40% are considered in a sensitivity analysis.

In (WBI 2008, p.11) it is furthermore stated that 94% of collected waste is disposed of in landfills that do not use state-of-the-art technology. The percentages are taken from the official data on the status of waste disposal for 1997 in (MoUD/CPHEEO 2005, p.7) for cities (Table 52). According to this, the remaining quantities are disposed of via composting or "other".

Table 52: Status of waste management in 1997 (MoUD/CPHEEO 2005, p.7)

Cities	Class I	Class II
Cities	300	345
Mode of collection		
- Manual	50%	78%
- Trucks	49%	21%
- Other	1%	1%
Disposal		
- Dumping	94%	93%
- Composting	5%	6%
- Other	1%	1%

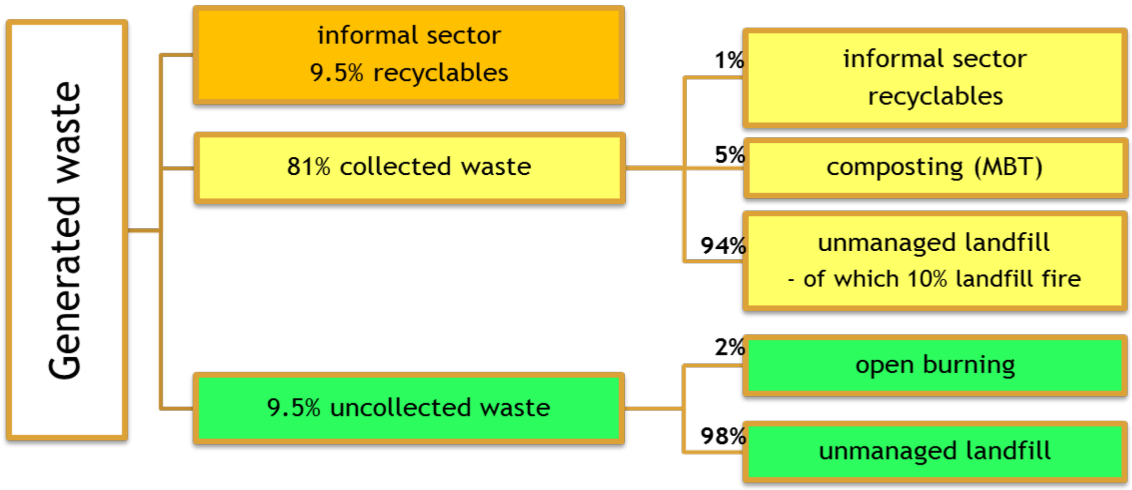
In (Annepu 2012, p.24) it is stated in addition that 91% of formally waste collected is deposited in unmanaged dumps and up to 6% is sent to a "composting" facility. The remaining amount – around 3% - is the proportion of collected recyclables (see above). On the basis of the information in the literature, it was assumed for the inventory that the collected waste is disposed of by the following methods:

- 94% dumped in unmanaged landfills
- 5% composted
- 1% other

It is assumed that the 1% "other" consists of recyclables sorted by waste pickers at waste collection and transfer sites and landfills.

The uncollected waste is mainly scattered or dumped, while a small proportion is burned in the open. In (Annepu 2012, p.24) the proportion of waste burned in the open is estimated to be 2%. No other data on this issue are available in the literature, so that the figure of 2% was used in the calculations. The values used to depict waste streams in India in the inventory are shown in Figure 37.

Figure 37: Material flow diagram for the waste streams on which the India inventory is based



As a result of waste collection conditions in India – roadside collection, overflowing containers where containers are used and the fact that waste is sometimes simply discarded in the street – animals are commonly found in places where waste accumulates, and they eat the organic waste. In addition to dogs and cats, cows are often involved. In GIZ India it is assumed that cows and other animals eat up to 60% of the organic waste. However, this figure cannot be discounted. It is also difficult to know how this aspect should be assessed. In principle it raises serious hygienic concerns and officially (MSW Rules 2000) stray animals should therefore be kept away from waste. It is not clear what would happen to the animals if they were unable to feed on this organic waste. Cows, which are sacred in India, are usually privately owned: alternative means of feeding them would have to be found, so that substitution potential cannot be entirely rejected. However, because this is a hygienically undesirable and inadmissible practice and no reliable estimate of its extent is possible, this aspect is not included in the inventory.

6.2.1 Calculating the characteristics of the waste streams

The removal of recyclables by the informal sector changes the composition of the waste and hence also its calorific value and carbon content, which are important parameters in calculation of the inventory. This is taken into account in the calculations.

For India the volume of waste generated is described via the composition of waste according to (WBI 2008) in Table 51. The volume of waste collected by the informal sector at the doorstep was then deducted from this. In the absence of detailed information it was assumed that the dry recyclable fractions of paper, plastic, metals, glass and textiles were collected in equal quantities. For each of these fractions this results in a collection rate of 41%. Textiles are included in doorstep collection here because they are cited in (WBI 2008, p.16 and p.133) as a fraction of doorstep collection. According to (WBI2008), the reselling of newspaper, bottles, plastic bags, old clothes and glass is a well-established tradition in India. Old clothes are usually sold on directly.

The waste stream remaining after doorstep collection contains correspondingly fewer recyclables (Table 53). The calorific value and the biogenic and fossil carbon content of this resulting waste were calculated using the characteristics of the individual waste fractions shown in Table 13.

Table 53: Composition of various waste streams

Details in %	Generated waste	Waste after doorstep collection	Waste after waste picking
Compostable waste	47.43	52.4	53.0
Paper	8.13	5.3	4.9
Plastic, rubber	9.22	6.0	5.6
Metals	0.5	0.3	0.3
Glass	1.01	0.7	0.6
Textiles	4.49	2.9	3.0
Inert material	25.16	27.8	28.1
Other	4.016	4.4	4.5
Carbon, biogenic		11.3	11.2
Carbon, fossil		5.0	4.8
Calorific value (kJ/kg waste)		5,744	5,595

The same procedure was used for the waste remaining after waste picking. Here, too, the volume of additional recyclables removed was deducted. The only difference is that it cannot be assumed that this waste contains usable textiles. The 1% waste picking rate for India was therefore applied only to the fractions of paper, plastic, metals and glass. Here again it was assumed in the absence of detailed information that these waste fractions were collected in equal quantities. In relation to the volume of recyclables remaining after doorstep collection this yields a collection rate for each of these fractions of 8%.

Taking doorstep collection and waste picking together, this means that for the generated waste 45% of each of the recyclable fractions paper, plastic, metals and glass and 41% of the textile fraction is collected by the informal sector and recycled. 10.3% of the generated waste is thus removed in the form of dry recyclables (9.5% doorstep collection, rest waste picking). The figure is relatively low on account of the high proportion of organic matter and inert waste in Indian municipal waste, which together make up 75% of the quantity of waste.

In the analysis the waste composition resulting after waste picking and the associated characteristics (Table 53) were applied equally to the 94% of waste that is landfilled and the 5% that is composted. This is unlikely to be a precise picture of the reality, but in view of the general uncertainties it is an appropriate simplification. Conversely, waste picking already takes place at collection and transfer sites and not only at dumps. This means that it is entirely possible for the composition of the waste sent for composting (after transfer) to be roughly similar to that of the waste that is dumped.

6.2.2 Landfill

Most waste disposal sites in India arose in an uncontrolled manner, with no attention paid to basal liners or maintaining an adequate distance from the water table. There is no leachate or gas collection, there are no monitoring systems, and the waste that is emplaced is not compacted but at best levelled with bulldozers. According to official data (CPCB 2010, cited in Annepu 2012, p.24), 10% of the deposited waste is burned in the open or in landfill fires.

The legal framework

The methods of dealing with waste that have been described – scattering, open burning and uncontrolled dumping of collected waste – have been illegal in India since 2000. The parlous state of the waste management system, and in particular the severe and growing lack of space as a result of unauthorised and unmanaged dumping sites, led to the introduction of stringent legal regulations with a highly ambitious timetable for implementation of the provisions. The Municipal Solid Waste (Management and Handling) Rules entered into force in September 2000 (MoEF 2000).

They cover in particular the following aspects:

- Waste collection:
  must take place, ban on burning, requirement to keep stray animals away from waste, etc.
- Dumping:
  should – mainly for reasons of space – usually be restricted to non-biogenic waste, rules on siting, emplacement, monitoring, etc.
- Provisions on leachate, compost and burning of waste:
  Rules on treatment facilities (sealed surface, leachate capture, covering of waste, minimising and monitoring noise emissions), guidelines on concentrations of heavy metals in compost, limit values for leachate, rules on incineration of waste (complete incineration, chimney height, emission limits, ban on incinerating PVC).

Mainly because of a lack of funding, but also because of poor coordination at ministerial level and inadequate information on how the rules are to be implemented, progress towards reforming waste management in India in line with these rules has so far been rudimentary. The MSW Rules were revised in 2011.

Calculating GFG emissions

The GHG emissions from uncontrolled dumping were calculated using the procedure described in Section 4.2.5. Irrespective of whether waste is formally collected or not, it is disposed of in uncontrolled dumpsites without any form of protection or monitoring, so that for the purposes of the inventory there is no difference between the scattering of uncollected waste and the dumping of collected waste.

The following values were used to calculate the GHG debits for India:

- DOC = approx. 11% (see Table 53)
- DOCf = 50% (IPCC default value)
- Methane content = 50 vol% (IPCC default value)
- MCF = 0.4 (MoEF 2012, see below)
- no gas collection
- OX = 0% (IPCC default value)

This procedure conforms well with that described in India's Second National Communication to the UNFCCC (MoEF 2012, p.76). There, too, the default values are used for the DOCf and methane content, and it is assumed that there is no gas collection and no methane oxidation. Even the DOC, which in (MoEF 2012) is estimated at 11% on the basis of the waste composition, corresponds to the values calculated in this study (Table 53). The only relatively minor deviation lies in the fact that in (MoEF 2012) it was assumed that 70% of the volume of waste generated is ultimately landfilled, while according to the waste streams on which this study is based the landfilled proportion of the generated volume of waste amounts to 86%.

The procedure in (MoEF 2012) was used to assess the type of disposal. (MoEF 2012) assumes unmanaged, shallow disposal sites and the methane correction factor (MCF) is therefore set at 0.4. This assumption has a major influence on the result, because the methane formation potential of the dumped waste falls to 40% as a result of the increased aerobic activity that is assumed to take place.

When comparing the status quo with the future scenarios a sensitivity analysis is therefore performed to explore the emissions situation if it is assumed that only 50% of the waste is dumped in shallow sites and that the remainder is piled higher, with correspondingly increased anaerobic conditions. In this case the IPCC default value for unmanaged deposits > 5 m is used (MCF = 0.8).

The future scenarios assume managed landfilling. This means that the methane emissions inevitably increase, because the IPCC default value of MCF = 1 is used for managed landfill sites. A further element of the future scenarios is that gas collection is assumed to take place: in the baseline case the IPCC default value of a 20% effective gas collection efficiency is applied. In addition, the result with a higher effective gas collection efficiency of 40% is described in a sensitivity analysis.

6.2.3 Composting (mechanical-biological treatment)

Almost all the "composting" that takes place in India involves mixed MSW. Of the more than 80 treatment facilities in India, only two treat source-segregated organic waste (Annepu 2012). There are also a few vermicomposting facilities for separately collected organic waste – mainly market waste and food waste from restaurants. Biomethanation takes place only in small-scale units, sometimes at household level, that use green waste and market waste. The biogas produced is used entirely for heat generation. The only attempt to establish a central biomethanation plant was made in 2003; according to (Annepu 2012) it failed because organic waste is not separated at source in India, and similar attempts in future would fail for the same reason. Because of their minor contribution, biomethanation and vermicomposting are not considered in this study.

The waste is composted in plants that can be described as simple mechanical-biological treatment (MBT) systems. Usually the waste delivered to the plant is first separated mechanically and the sieved fraction, which should be < 100 mm, is composted. Before composting an inert fraction is also removed. According to information provided for (IFEU 2011), this can theoretically be used for brick-making, but the extent to which this actually occurs cannot be assessed. As a conservative estimate, it was assumed for the purpose of the inventory that this inert waste is landfilled. The volume subsequently sent to composting amounts to around 50-60% of the input (IFEU 2011). The sieve overflow – consisting mainly of plastic, textiles, scraps of paper, adherent organic matter and inert material – can either be segregated again (plastic and paper) and sold to recyclers or it may be processed into RDF; alternatively, all the sorting residue may be landfilled. More precise information on this is not available. For the inventory it was assumed that an RDF fraction is produced.

According to (Annepu 2012), India has five or six plants that produce refuse-derived fuel (RDF). GIZ estimates that the number is in the region of 20, but this has not been confirmed. The capacity of the plants ranges from 100 to 800 tonnes of waste per day, while the amount of RDF produced is between 30 and 250 tonnes per day. The RDF produced is sold either to RDF power plants or to cement works. In (Annepu 2012) RDF is regarded as a good opportunity for converting waste to energy in India. Because of the substantial inert and organic fractions, direct incineration is not usually appropriate. However, there are virtually no RDF power plants in India[45] and cement works usually have little interest in RDF because the works have historically been built close to coal-mining areas and coal is cheap because it is subsidised. The use of RDF in smaller plants such as brickworks is undesirable, because these plants usually fail to meet the minimum standards for emission limits.

For the separated RDF fraction it was assumed that it cannot all be converted to energy. In view of the fact that there are probably only two RDF power plants in India and that cement works are reluctant to use RDF, there are likely to be considerable difficulties in finding a sales market. It was reported by the GIZ project (IFEU 2011) that RDF cannot be sold and is simply stored on-site, which over time is equivalent to unmanaged landfilling. For the inventory it was assumed that 30% of RDF is co-fired in cement works and 70% is dumped in unmanaged sites.

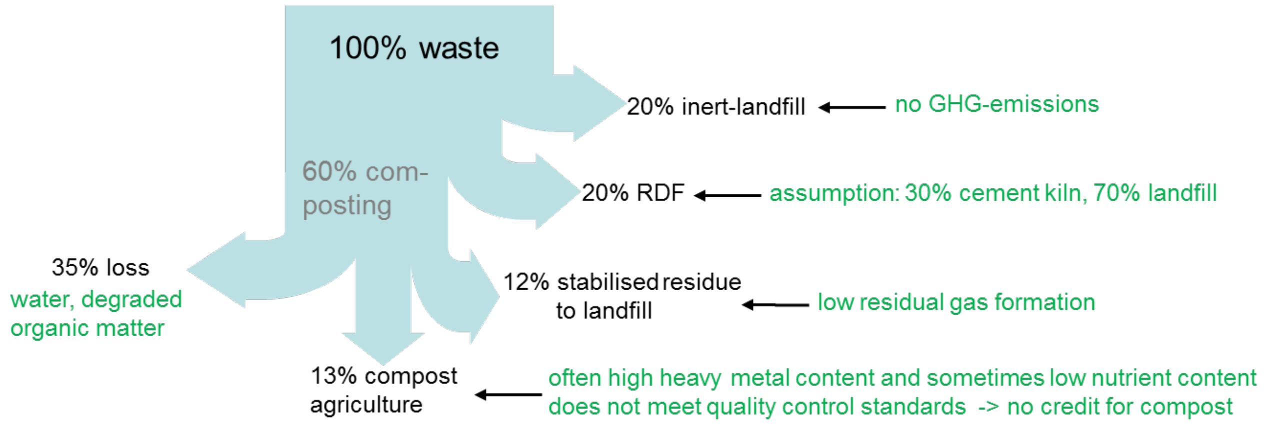
Composting of the sieved fraction extracted from the mixed MSW usually takes place in windrows – in the best case in triangular windrows (approx. 1.5 x 3 m), which provide an optimum ratio of surface area to volume, as described in (Annepu 2012). When combined with the addition of bulking material and regular aeration of the windrows by turning, this is a good management method and one that minimises emissions of the greenhouse gases methane and nitrous oxide. Often, though, large table windrows are created on unsealed surfaces; they are rarely or never turned and sometimes stand in water when it has rained. In estimating the greenhouse gas emissions it is assumed that for India as a whole these unfavourable conditions are more likely to be the norm. The upper value of the range in (gewitra 2009) for open composting is used as an approximation to the status quo (Table 9).

After composting the waste is sieved again, because compost sold to farmers must have a grain size of less than 4 mm. Composting of waste therefore results in only a small mass stream,

[45] According to (Annepu 2012, p.51) there are two such plants in Hyderabad and Vijayawada. Agricultural residues such as rice husks are co-fired at these plants.

amounting to 10-15% of the total input (IFEU 2011), that can be sold as compost. In (Annepu 2012, p.50) the percentage that is recovered as compost is put even lower, at 6-7%. The author also assumes that all the rest of the waste is landfilled. Allowing for water loss and loss through biodegradation, the total amount landfilled amounts to 60% of the total input.

Figure 38: Material flow diagram for composting of mixed waste ("simple MBT")



On the basis of the given information, an average material flow model of mechanical-biological treatment in India was produced (Figure 38) and presented at the workshop in Delhi. The above-mentioned details from (Annepu 2012) were considered too extreme for the baseline case and were not included in this form. In accordance with the general information received it is assumed that after mechanical separation of inert waste and recyclables, 60% of the volume of waste is composted. For the 40% of separated waste it was assumed for simplification that this is divided equally, consisting of 20% inert waste that is landfilled and 20% plastic etc. which is assumed to be converted into RDF. Segregation of individual dry recyclable fractions is considered unlikely in the baseline case.

For the composted fraction, which consists mainly of organic matter, a degradation rate of just under 60% is assumed, since this is typical for composting. For the composted residue it is assumed that further sieving results in roughly equal amounts of compost and stabilised coarse-grained residue. These assumptions yield the mass flows shown in Figure 38. According to this, 13% of the input is used as compost in agriculture, 6% (20% x 30%) is co-fired as RDF in cement works and 46% is landfilled (20% inert waste, 14% RDF, 12% stabilised residue).

The assessment of the landfilled fractions corresponds in principle to that described under "Landfill". However, inert waste does not undergo biological degradation and it can be assumed that biodegradation of the stabilised compost residue is significantly reduced. Like MBT residue, this residue was therefore assessed as having a DOCf of 10% and a methane content of 40 vol% (IFEU 2012). It was again assumed that no methane oxidation takes place for the material dumped in unmanaged sites.

Identifying the characteristics of the RDF fraction involved making assumptions. The possible composition of the RDF was calculated. It was assumed for this purpose that all the recyclables remaining in the mixed waste are transferred to the RDF fraction together with the other wastes and components of the inert fraction. On the basis of this model composition, the fossil carbon content was set at 20% and the biogenic carbon content at 15%. For the calorific value the recommended minimum calorific value for RDF in India was used (3200 kcal/kg RDF, equivalent to around 13.4 MJ/kg). The co-incineration of the RDF in cement works substitutes coal, for which in India an average calorific value of 17 MJ/kg is assumed (Annepu 2012, p.93). Standard emission factors were used to calculate the GHG emissions of avoided extraction and combustion of coal.

It is assumed that the compost produced from composted waste is always contaminated with pollutants. It has been confirmed that this is the case in India. (Annepu 2012) refers to a study conducted by the Indian Institute of Soil Science (IISS) in Bhopal, which found that compost produced from MSW in India contains high concentrations of heavy metals and has low nutrient value. Composts from most of the 29 cities in the study exceeded the limits on heavy metals laid down in the MSW Rules 2000 and failed to meet Indian quality standards for potassium and phosphorus content, TOC and moisture content. As in the previous project, it is assumed for the purpose of the inventory that there is no use for compost from MSW.

6.2.4 Recycling

Recycling of the dry recyclables removed by the informal sector is assessed using the standard emission factors derived in Section 4.2.4. It has already been explained in Section 4.2.4 that while India has an integrated paper production system, recycled paper fibre is used there too. It is also not possible to say to what extent coal is used for energy generation. There are currently around 500 paper mills in India – most of them SMEs – with an installed capacity of 12.75 Mt. There are fewer than 25 mills with a capacity of 50,000 t/a, and some six paper mills produce almost 90% of the country's newsprint. Current paper and cardboard production is estimated at 10.11 Mt per annum, while domestic consumption is put at 11.15 Mt (9.3 kg/cap) annually. After two years of stagnating demand the Indian Paper Manufacturers Association IPMA[46] expects demand for paper to rise to 14 Mt by 2015 and 20 Mt by 2020.

In general there is an acute shortage of wood in India. The law therefore prohibits Indian paper manufacturers from maintaining large plantations for primary paper fibre. Because of the shortage of pulpwood, the mills are making increased use of agricultural wastes such as bagasse and waste paper. According to the IPMA, the input to Indian paper production is made up of 44% waste paper fibre, 35% chemical pulp and 21% agro-residues. Some of the waste paper that is used is imported; only 47% has been recovered in India. Between 850,000 and 1,000,000 tonnes of paper are recycled annually in India, corresponding to a recycling rate of 20%. One of the reasons why this recycling rate is so low is that paper is often re-used for purposes such as packaging. Because of these shortages, the Indian paper industry is calling for the lifting of restrictions on plantations and establishment of a well-organised pro-active system for collecting, sorting, classifying and using recyclable paper in order to limit imports.

After organic and inert waste, plastic waste is the third-largest waste fraction in Indian MSW, although considerably smaller in terms of volume (Table 53). In the present study it was assumed that 41% of plastic waste – the same percentage as for other recyclables – is collected at the doorstep, so that relatively small amounts remain in MSW. Thin plastic bags have been posing a problem at rubbish tips and landfills. The Recycled Plastics Manufacture and Usage Rules 1999 attempted to address this by introducing a minimum thickness of 20 μm for plastic bags. The idea is that the bags are then more likely to be picked out by waste pickers, because thin plastic bags are of lower value and segregating them is difficult. However, it is difficult for

[46] Indian Paper Manufacturers Association; http://www.ipma.co.in/

manufacturers to measure the thickness of their bags, so the standard is not usually implemented. Indian states have responded to this by specifying a greater thickness for plastic bags. This varies between 30 and 70 μm in different states.

In general it can be assumed that the plastic waste purchased through doorstep collection is suitable for higher quality recycling. According to (MoEF 2010), 80% of plastic waste in India consists of recyclable thermoplastics such as PET, PE, PVC, PP and PS and 20% consists of duroplastics such as PUR, alkyl, epoxy, ester, formaldehyde and silicone, which are not easily recyclable. However no breakdown of the data according to recyclable plastic is available, and there is likewise no information on recycling methods after doorstep collection. Experts assume (IFEU 2011) that plastic waste is usually processed into secondary granulate through “plastics to pelletisation”. This granulate is mostly used for relatively thick-walled plastic products such as planters. According to European experience, such uses tend to replace wood and concrete: only some of it replaces primary plastic. In terms of the three-tier system of plastic recycling described in this study (see Section 4.2.4), this corresponds to low-value plastic. The derivation of the three quality categories for plastic recycling is documented in the Annex (Section 11.1).

For India, in contrast to the method used for the OECD and the USA, the inventory relates to the collected volume estimated on the basis of the available data. It is assumed that processing residues that arise in India are landfilled. In the inventory this does not give rise to any additional GHG emissions. Transport emissions are included on a standard basis (see Section 4.2.4).

The emission factors calculated for India – in CO_2-eq per tonne of collected plastic waste – are:

“low”	Debit 418	Credit -511	Net -94

6.2.5 Other technologies

Technologies other than composting of MSW / mechanical-biological treatment (MBT) are not currently used in India or play only a minor role.

Municipal solid waste incinerators (MSWIs)

Only a few MSWIs or WtE plants have been built in India, and the country’s experience of them has not been good. One of the first plants was built in Timarpur, New Delhi, in the 1980s, but it failed. At the workshop in India a representative of the Ministry of New and Renewable Energies (MNRE) stated that the reasons it was closed was that the energy content of the waste was too low. The next project was not attempted until many years later, but this too failed, this time because the percentage of organic matter in the waste was much lower than expected (11-19%). As at 2012, 4-5 projects were under way. One of these projects involves a plant in Bangalore: GIZ has stated that because of miscalculation this plant will probably not be completed. Another plant, built in the Okhla district of New Delhi, commenced operation in 2012. It is ultimately designed to have a capacity of around 2,000 t/d. The plant apparently came on stream according to plan.[47] According to news reports,[48] the plant is in stable

[47] Inspection by Jürgen Giegrich im December 2012.

operation but is emitting levels of dioxins and furans more than 30 times above the permitted limit, despite the legal requirement for rubber and PVC waste to be removed before incineration to prevent this pollution. The permitted limit is 0.1 mg/m³ (for comparison, in Germany under the 17th Federal Immission Control Ordinance [BImSchV] it is 0.1 ng/m³). Because of previous bad experience and concerns about health risks, incineration of waste is highly controversial among the Indian public. For example, construction of the plant in Okhla met with considerable public opposition as a result of concerns about risks to health from the plant's emissions. However, because of the high organic content (up to 70% including paper) of mixed MSW, the Indian government classes WtE as a "renewable energy technology" (Annepu 2012, p.45).

In this study, waste incineration is included in the future scenarios. Aspects of this relevant to the GHG inventory are the fossil carbon content of the waste, the calorific value and the amounts of energy produced on combustion. A special feature of India is that incineration of PVC waste is prohibited. For the future scenarios it was assumed for simplification that all plastic waste is segregated mechanically from the mixed waste and recycled. The recycling was assessed as described above. For the remaining waste it was assumed for simplification that removal of the plastic halves the fossil carbon content and reduces the calorific value to 80% of the initial value. Because the main purpose of the future scenarios is to highlight potential, these simplifications are considered appropriate. In a similar simplification, a net electrical efficiency of 15% was assumed for the energy generation. It is assumed that no heat extraction takes places, because there is no certainty that the heat can be used externally.

Past problems with "new" technology

In addition to the problems with MSWIs described above and the opposition from citizens' initiatives, India has experienced many problems of a similar sort with other plants. The first MSWIs in India were built in 1975-76 (10 plants). These and two RDF plants built in 2003 in Hyderabad and Vijayawada are no longer operating. The main reasons for the failures were:

- The plants were designed for larger volumes of waste than it was actually possible to obtain.
- Operating costs were not taken into account in the calculations.
- The imported technology did not take account of local conditions.

In connection with the WtE plants, in particular, the following problems arose:

- No spare parts were available for the imported semi-automatic machinery.
- The shredders were not adapted to the Indian mixed waste and were frequently blocked by lumps and fragments of plastic; blades broke on fragments of glass and metal contained in the waste.
- The continuous electricity supply that was required could not be guaranteed.
- The plant was unable to operate in the rainy season.
- There was no market for the products that resulted.

[48] http://e360.yale.edu/feature/out_of_indias_trash_heaps_a_controversy_on_incineration/2716/ und http://timesofindia.indiatimes.com/city/delhi/Waste-to-energy-plant-poisoning-air-Study/articleshow/20358451.cms?referral=PM

Lessons were learned from these problems, but there are still plants that face these and other difficulties. For example, RDF is still being produced for which there is no market and which is therefore ultimately landfilled. Another example is the new MSWI in Okhla, which is thought to be over-sized. In the opinion of GIZ, it was also not clear that the waste supplied was suitable for incineration. High levels of organic and inert material and high water content could mean that incineration requires large additional quantities of heating oil. It was not possible to either confirm or exclude these issues on a tour of the plant.

6.3 Results: waste management – India

6.3.1 The baseline case

In the baseline case the lower range value of 42 Mt according to (WBI 2008) was used for the waste volume. The waste management methods shown in the flow diagram in Figure 37 are also depicted in Figure 39 in the form of a bar chart. According to this, most MSW in India is collected formally and landfilled on an unmanaged basis.

Figure 39: Destination of waste generated in India

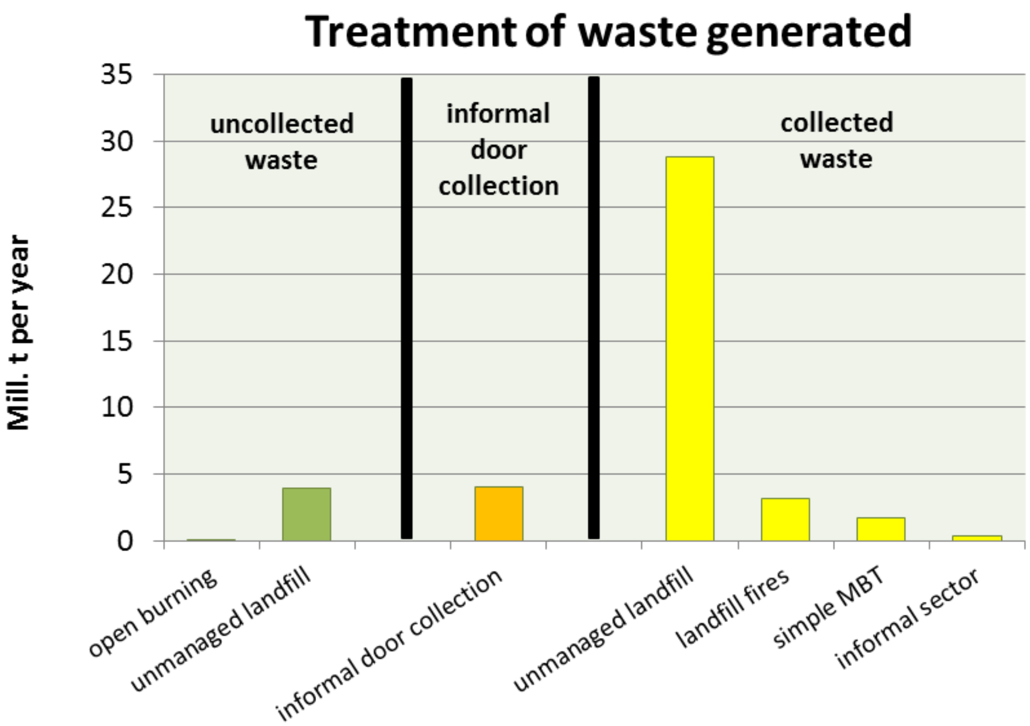
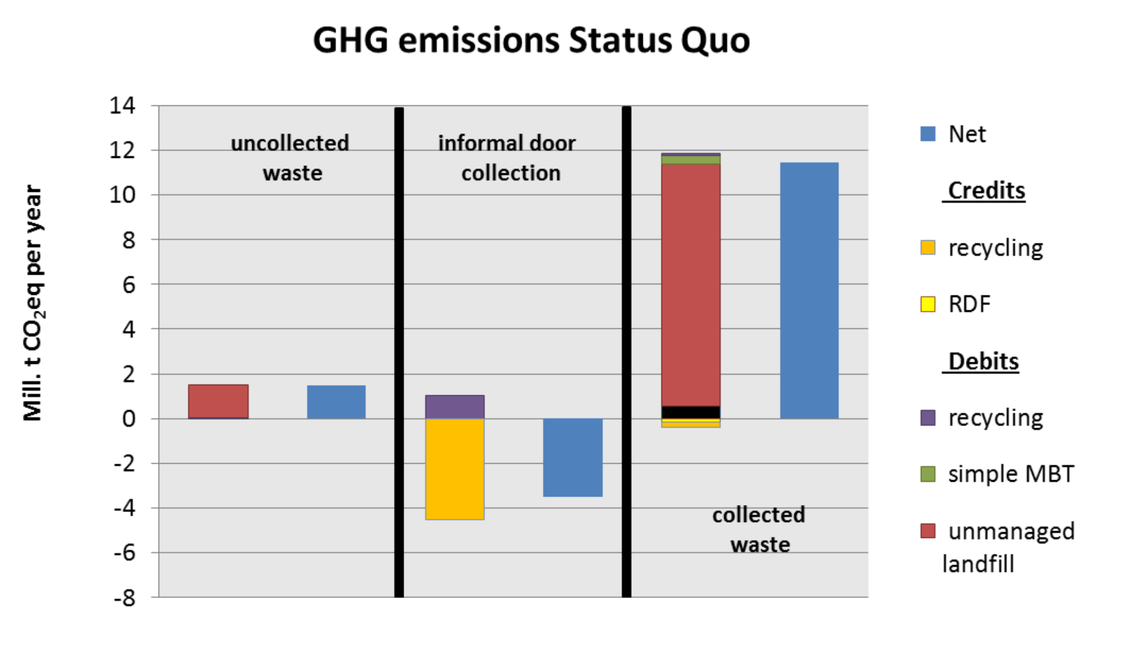


Figure 40: Results – GWP of the status quo



The result of the assessment of waste management in India for the status quo is shown in Figure 40. In total, waste management in India causes **annual GHG emissions amounting to around 9.4 Mt CO_2-eq.** In the diagram this result is divided into the GHG emissions that arise from the uncollected, scattered and openly burned waste, those from doorstep collection with recycling and those from formally collected waste and its treatment.

The calculation with regard to unmanaged dumping does not distinguish between collected and uncollected waste (Section 6.2.2). The dumped waste results in total in emissions of around 12.3 Mt CO_2-eq. Because of the small amount of waste involved, the emissions from open burning are by comparison of minor importance. Only the activities of the informal sector (doorstep collection, waste picking and recycling of dry recyclables) result in a net GHG credit; this amounts to 3.6 Mt CO_2-eq. For doorstep collection this is clearly visible in the results chart; for waste picking the additional volume collected by this means is relatively small, so that the credit is less distinct.

6.3.2 Sensitivities

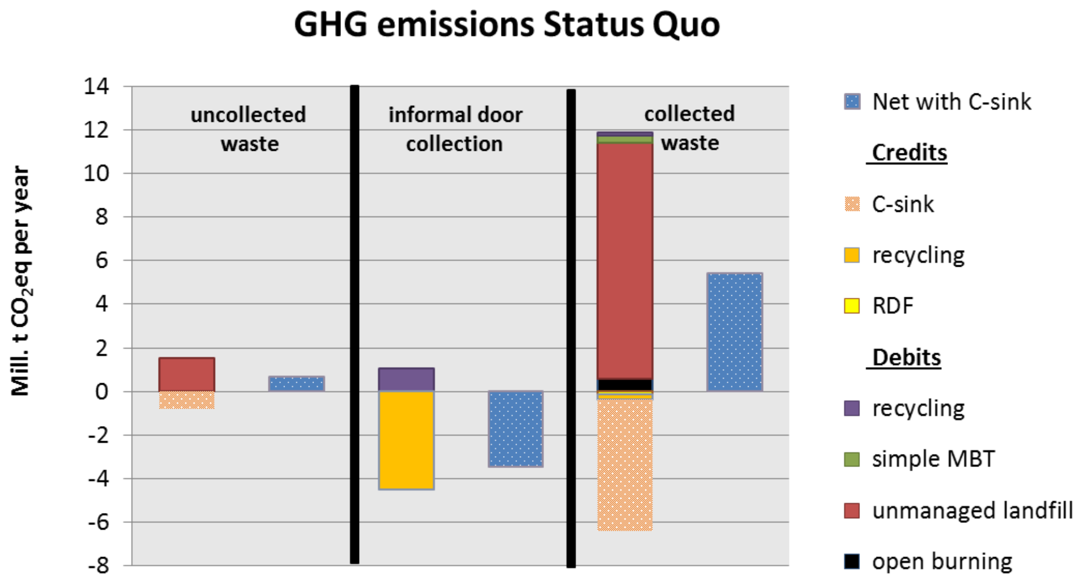
The following aspects were explored as sensitivities:

- Result taking account of the C sink
- Result for a collection rate of 60% instead of 81%
- Result for upper range waste volume of 243 Mt

C sink

In Figure 41 the result for the baseline case is extended to include the sensitivity analysis of the C sink. When this is taken into account, the net emissions are reduced by more than half. The overall outcome still involves net debits, but these are now only around 2.6 Mt CO_2-eq as against around 9.4 Mt in the baseline case. However, the C sink is affected by uncertainties (Section 4.2.5) and is therefore presented only as a sensitivity analysis.

Figure 41: Sensitivity analysis: GHG balance for the status quo with C sink

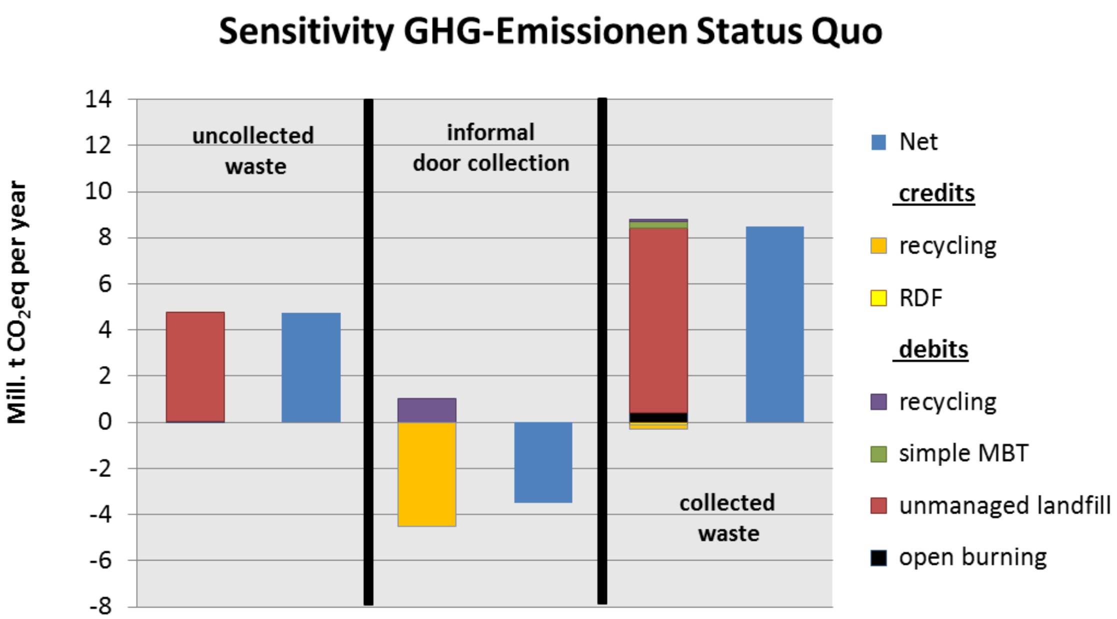


Collection rate of 60% instead of 81%

It is clear from the literature that one of the larger data uncertainties is the collection rate. The rather higher values quoted in some sources relate to urban areas. Only in one source is a countrywide collection rate of 60% cited, and this source provides no further information on waste management methods and type of treatment (Section 6.1). The deviating countrywide collection rate is therefore considered here as a sensitivity. In other respects the distribution of the waste streams is the same as in the baseline case.

Figure 42 shows the results of the sensitivity analysis, subdivided into the three categories of uncollected, doorstep collection and collected. With the lower collection rate there is a noticeable shift of the GHG emissions towards the uncollected waste. In the overall result, though, the difference is small, because the unmanaged dumping of uncollected and of collected waste results in the same GHG emissions. A small difference arises from the fact that no waste picking is assumed to take place for the uncollected waste and in line with this fewer recyclables can be removed by the informal sector from the collected waste. In consequence the overall net debit for the 60% collection rate is around 9.7 Mt CO_2-eq (without C sink), as against the 9.4 Mt CO_2-eq in the baseline case.

Figure 42: Sensitivity analysis: GHG balance for lower collection rate



Upper range waste volume

The upper range for the waste volume was calculated from data in (MoEF 2012) and the Indian census. On this basis the volume of waste generation in India in 2011 would amount to 243 Mt (Section 6.1). For this volume of waste the overall net debit amounts to around 54.5 Mt CO_2-eq, as against the roughly 9.4 Mt CO_2-eq in the baseline case with 42 Mt waste. In line with the increase in the volume of waste, it is thus almost six times higher.

In calculating the implications of the upper range value of the volume of waste, it was assumed that the waste composition and management methods are the same as in the baseline case with the lower range value of 42 Mt waste. However, the proportions of recyclables could be over-estimated. For example, it is reported in (MoEF 2010) that India's consumption of plastics in 2008 was put at 8 Mt; the resulting annual volume of plastic waste was thought to be 5.7 Mt. However, calculations based on the information in (WBI 2008) and the upper range value of the waste volume would result in an annual volume of plastic waste of 22.4 Mt. In the baseline case and hence also in the scenario involving a high volume of waste, 45% of this is collected and recycled by the informal sector. According to (MoEF 2010), this percentage would in turn be an under-estimate. It is reported in (MoEF 2010) that 60% of the generated plastic waste is recycled; the remaining 40% is not collected but is simply thrown away.

A similar situation exists with regard to waste paper. According to the Indian Paper Manufacturers Association, up to 1 Mt of waste paper is recovered annually, which is equivalent to a recycling rate of 20% (Section 6.2.4). This would correspond to a generated annual quantity of waste paper of 5 Mt. However, the upper range value of the volume of waste and the given waste paper percentage according to (WBI 2008) would yield an annual quantity of waste paper of 19.7 Mt.

These two examples illustrate the problems of data uncertainty and the corresponding urgency – not only in India but generally in countries in similar situations – of establishing not only an organised waste management system but also a monitoring system to track waste streams. If a ban on dumping is sought in India, management measures cannot be effective unless the actual potentials of the recyclable fractions and the overall volumes of waste are known.

6.3.3 Contrast with the figures in the communication to the UNFCCC

In India's Second National Communication to the UNFCCC (MoEF 2012, p.43 and 78), the country's calculated GHG emissions for the year 2000 are specified as follows:

Total:	1,523,777,440 t CO_2-eq
Waste sector:	52,552,290 t CO_2-eq
Solid waste disposal:	10,251,990 t CO_2-eq

Solid waste disposal thus accounts for only 0.7% of India's total GHG emissions. According to information in the communication, the calculation was made for the total volume of waste in India (basis: 0.55/kg/(cap*d)), for which it was assumed that 70% is ultimately landfilled.

The communication to the UNFCC includes information on annual emissions. The calculation (first order decay method, FOD) involves an analysis over time of the methane emissions released over the decades from the landfilling of waste. The reported emissions for a particular year reflect the real emissions of all waste deposited previously (according to IPCC recommendations since 1950) and in the reporting year. This method of calculation is used to monitor annual national GHG emissions. Comparison with the total emissions – direct and future – arising from the dumping of a certain volume of waste, as in the life cycle inventory approach, is not appropriate.

On the basis of the assumption stated in the communication (70% of 243 Mt waste landfilled), absolute GHG emissions amount to around 52 Mt CO_2-eq (characteristics as described in Section 6.2.2).

6.4 Future scenarios to 2030

6.4.1 Baseline comparison

As agreed, the future scenarios involved drawing up a medium and an ideal scenario. These take account of the special situation in India. The MSW Rules (2000) define the country's objectives as:

- an organised waste management system with managed waste collection and treatment
- as far as possible landfilling only of non-biogenic, non-recyclable waste.

In line with this, 100% formal collection of MSW was assumed for both future scenarios for the year 2030.

However, this does not affect doorstep collection, which already makes an important contribution to waste management in India. The removal of a proportion of dry recyclables from the formally collected waste by waste pickers also remains unchanged in the future scenarios. This informal collection of recyclables also contributes to waste management, although it usually takes place under conditions that give rise to serious concerns about health and hygiene. If these activities are to be expanded, it is essential that the general conditions are improved. There are signs that this is occurring, for example in Mumbai. There are three organisations for waste pickers there that issue members with ID cards (in exchange for an annual fee) and negotiate with official agencies in an attempt to improve working conditions and ensure integration of the informal sector (GTZ 2008). In the future scenarios it was decided not to reflect possible future changes in the informal sector.

For India the various bad experiences with new technology are also relevant (Section 6.2.5). The official agencies and other stakeholders are currently adopting a reticent position on this issue. The official agencies stipulate that where new technology is involved, waste treatment methods must be assessed and approved by the Central Pollution Control Board before they can be authorised in cities. In practice this is not always the case. For example at the workshop in New Delhi it was reported from one city that mobile waste incineration units had recently been installed there that the agency representatives knew nothing about. In addition, by comparison with new and not yet established technologies, GIZ India regards it as more expedient at this stage to expand the "composting" (simple MBT) that is already practised instead of landfilling waste.

In consequence it is assumed as a medium scenario that of the collected waste, less the recyclables collected by the informal sector, 50% is sent to managed landfill and 50% to mechanical-biological treatment. However, in contrast to the MBT material flow shown in Figure 38, it is assumed for the future scenario that is actually possible to sell the RDF that is produced and that all the RDF is co-fired in cement works. However, with regard to the quality of the compost that is produced and the way it is assessed, no change by comparison with the status quo is applied, because the process still involves composting of mixed waste.

In the ideal scenario, unlike in the medium scenario, new technology was taken into account. An element of the ideal scenario – as in all the countries considered – is the assumption that the practice of landfilling waste is phased out completely. The main alternative options for treating mixed waste are incineration and technically more demanding mechanical-biological treatment. Gasification and pyrolysis of waste are not considered, despite the fact that these processes are currently in vogue in India – that is, private investors are offering these processes to municipal agencies. However, they are processes that have not been tried and tested in practice and that have failed in countries such as Germany.[49]

Figure 43 summarises the selected future scenarios. For the ideal scenario to 2030 it was again assumed that the mixed waste was assigned in equal amounts to two different treatment methods. In is assumed that 50% is incinerated in a MSWI and 50% is treated by mechanical-biological stabilisation (MBS). Waste incineration and the way in which the emissions from it are calculated has already been described in Section 6.2.5.

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[49] Examples are the two gasification facilities using the Thermoselect process in Karlsruhe and in Ansbach, which operated for a few years but then had to be closed on account of continuing problems with the inhomogeneous mixed waste. Similarly, the Siemens facility (pyrolysis with afterburning) in Fürth never went into normal operation following sealing problems (gas leakage) during testing.

Figure 43: Overview of future scenarios to 2030

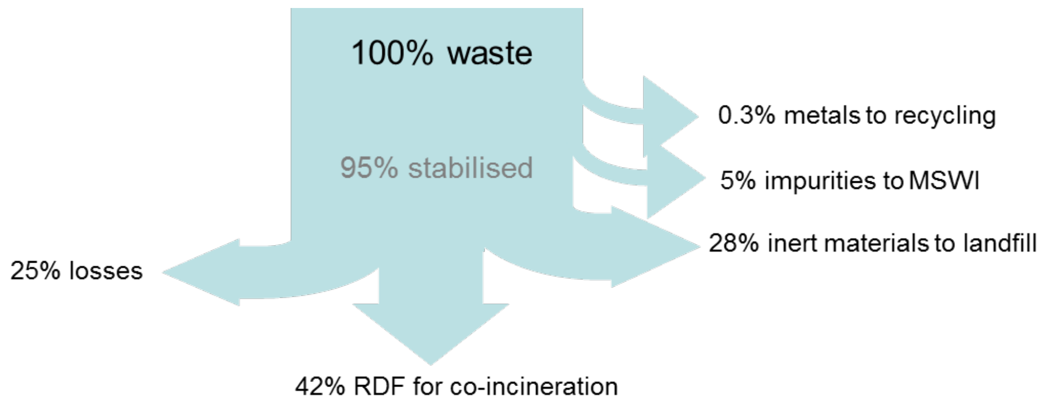
2030 medium	**2030 ideal**

- no change to the informal sector
- otherwise 100% formal collection

50% managed landfill, 20% gas collection efficiency; flaring 50% simple MBT, 20% RDF, co-incineration cement kiln, still no credit for compost	50% MSWI, plastics pre-sorted, recycled 50% MBS, 42% RDF fraction, co-incineration cement kiln

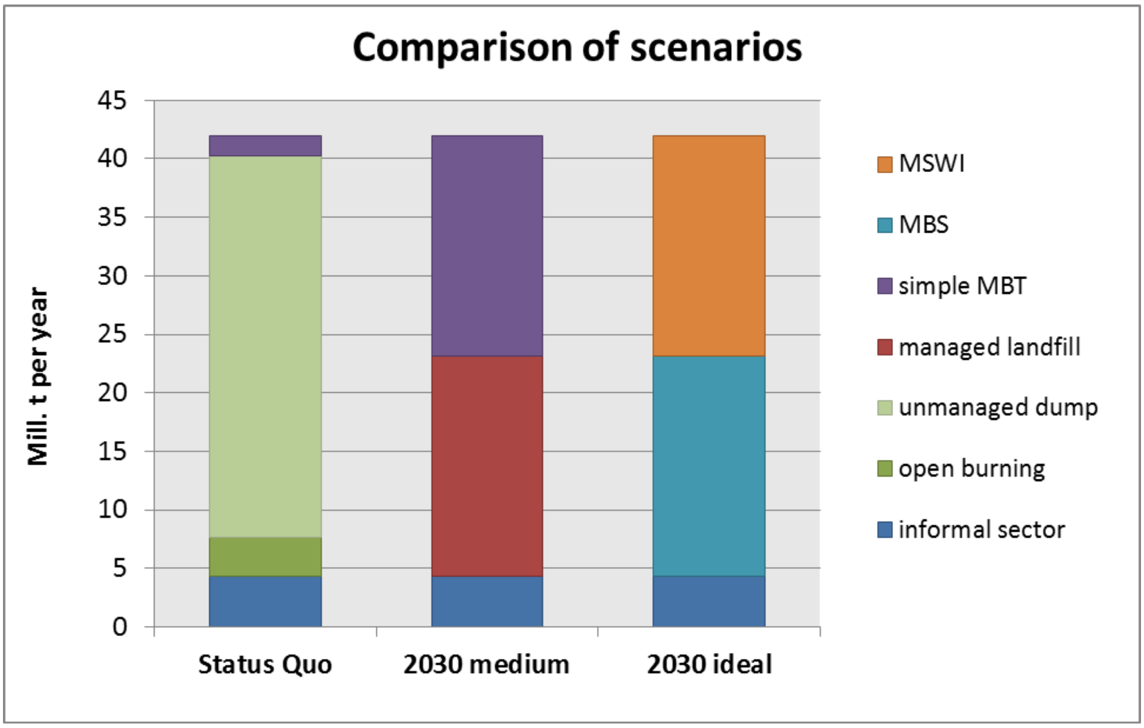
The mechanical-biological stabilisation (MBS) is modelled on the pattern of German MBS facilities. However, adjustments were made to the description in Section 4.2.7 to allow for the larger organic and inert fractions in Indian waste. It is assumed that mechanical separation of impurities, metals and inert components takes place first. By comparison with the details in Table 11 a smaller percentage separation of metals and impurities is applied, but the inert fraction that is landfilled is assumed to be larger (inert and metal percentages set on a model basis at 100% of the content in the waste). Losses are put at 25%. On this basis the RDF fraction is calculated to be 42% of the input. The mass flows of MBS as adjusted for India are shown in Figure 44.

Figure 44: Material flow diagram - MBS



In Figure 45 the final destination of the waste in the defined scenario to 2030 is contrasted with the destination in the status quo. This shows very clearly that the unmanaged landfilling in the status quo no longer exists in the future scenarios.

Figure 45: Waste treatment: status quo and future scenarios to 2030

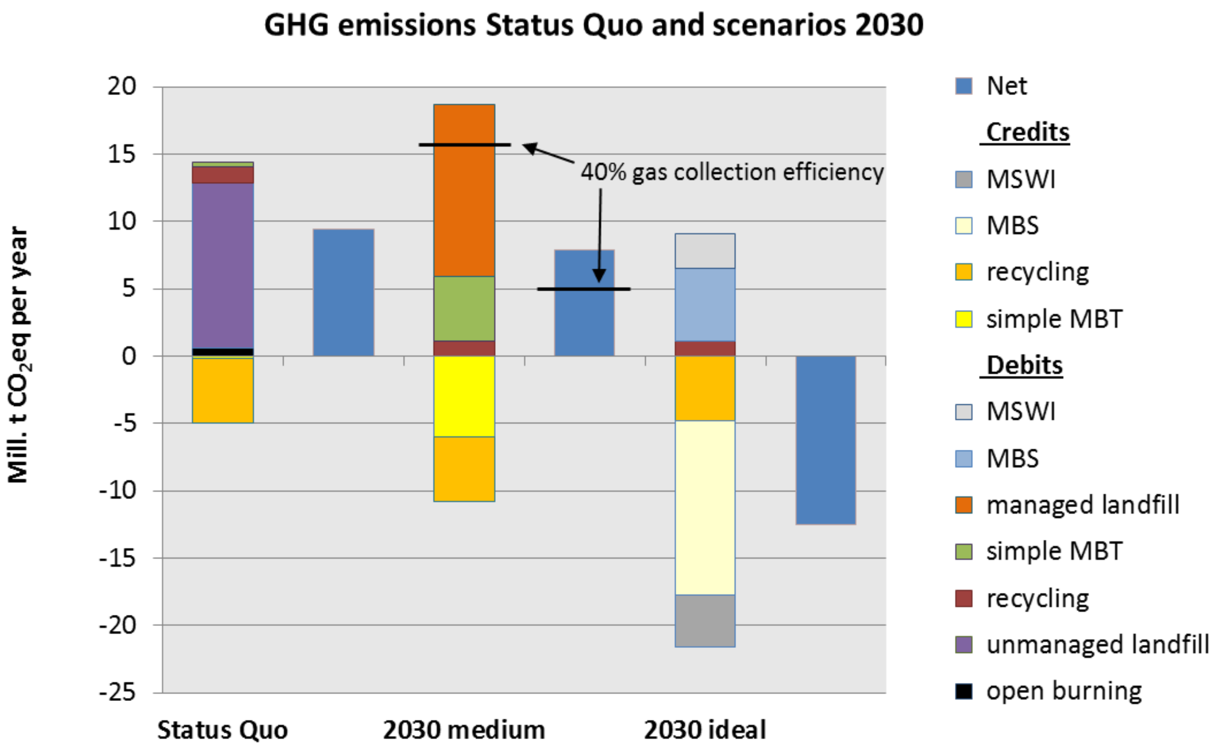


The results of the inventory of the future scenarios are shown in Figure 46. This makes it clear that by comparison with the status quo, the net result in both future scenarios is a reduction in GHG emissions as a result of organised waste management. In the medium scenario to 2030 this is due mainly to the fact that half of the collected waste is treated using simple MBT. For landfilling the special feature of assessment using the IPCC method for the conditions under which waste is landfilled is apparent. According to this method, the managed landfilling that is assumed to occur in future leads initially to greater formation of methane: the managed landfilling under anaerobic conditions is assessed using a methane correction factor of 1. For shallow unmanaged landfilling (status quo), by contrast, it is assumed that on account of more aerobic conditions only 40% of the methane formation potential is actually effective (Section 4.2.5).

This 100% methane formation potential actually results in a worsening of the GHG balance – and this despite the fact that the medium scenario assumes that gas collection systems are installed with an effective gas collection efficiency of 20% and that the methane collected in this way is subsequently flared off. However, it is also assumed that only 50% of the collected waste is landfilled, as opposed to 78% in the status quo. This is the only reason why, for example, there are equally high GHG debits from landfilling in this scenario and in the status quo. If the same quantity were sent to managed landfill in the medium scenario to 2030 as in the status quo (with correspondingly less waste handled via MBT), the net debit would be around 16 Mt CO_2-eq instead of the 9.4 Mt CO_2-eq in the status quo; even with an assumed higher effective gas collection efficiency of 40% it would still be around 11 Mt CO_2-eq.

It is very clear from this that, while a transition to managed landfilling represents major progress in terms of health and hygiene, it does little to contribute to climate change mitigation. Indeed, it can result in higher emissions, so that from a climate change mitigation perspective alternatives to landfilling need to be sought. As already mentioned, the medium scenario that is considered results in a net GHG reduction of 16%, because 50% of the waste is treated via MBT. If an effective gas collection efficiency of 40% is assumed, a GHG reduction of 50% can be achieved.

Figure 46: Result for GWP: status quo versus future scenarios



The situation in the ideal scenario is quite different. Here the net result is a credit. The negative value shows that waste management helps to prevent GHG emissions elsewhere in the economic system. In the ideal scenario to 2030 this is mainly in the energy sector. Because the incineration of waste results in production of electricity and because the co-incineration of RDF in cement works substitutes coal, the corresponding primary production of electricity and the combustion of coal of an equivalent calorific value can be avoided, thus avoiding the GHG emissions that would otherwise result from this. Because these avoided emissions are higher than the GHG emissions arising from the incineration of waste and the production of RDF, the net result is a negative value.

In a departure from the method used for the OECD and USA inventories, the calculation here is performed – as in the previous study – by waste fractions, because the figures on volume and management method relate to the generated waste and not, as in the official statistics, to the final destination. This is only relevant in the ideal future scenario, in which recyclables are separated in both treatment by MSWI and by MBS. In this inventory the benefit from the recycling of these materials (plastics in MSWI and metals in MBS) is assigned to the treatment method ("MSWI" or "MBS"). In both cases, however, the percentage arising from recycling is of minor importance (for segregation of plastic prior to MSWI 7%, for metals from MBS 1%).

The ideal scenario produces a noticeable improvement in the GHG balance, but on account of the problems associated with acceptance of these technologies it must be regarded as difficult to implement. A general reservation attaching to waste incineration is that this technology has not so far proved its worth in India. In this connection operation of the facility in Okhla, New

Delhi, needs to be monitored further in order to explore future opportunities for waste incineration. However, there are reservations generally – and not only in the population – about thermal treatment on account of the emissions of pollutants with which it is associated. For example, the final point in the recommendations for dealing with packaging waste listed in (MoEF 2010, p.30) is "Strategies for recovering energy by incineration of packaging waste should be discouraged and banned". India's experience of automated mechanical-biological treatment facilities has also been unfavourable. Apart from this, there are also reservations about producing fuel from organic waste. This was made clear at the workshop in New Delhi, where it was stated that organic waste should always be used first to produce compost and that only if this is not possible should processing into RDF be considered.

6.4.2 Sensitivities

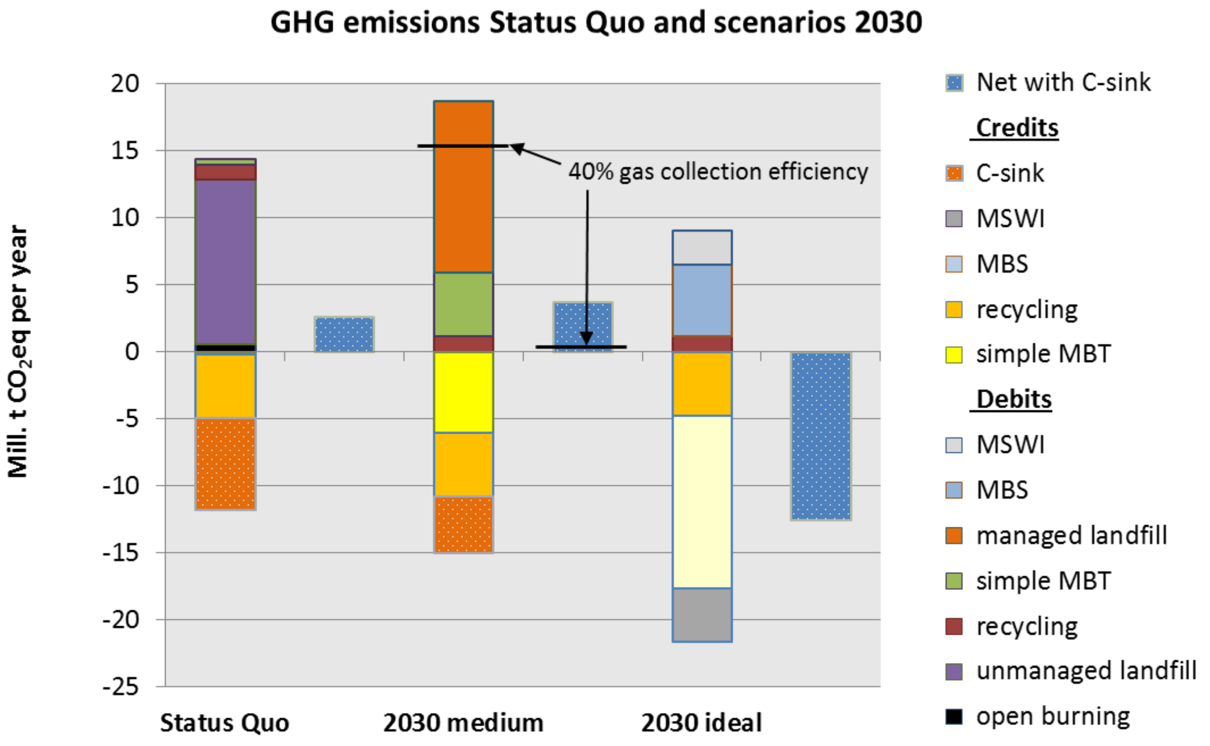
As sensitivities for the scenario comparison the following aspects were considered:

- Result taking account of the C sink
- Result assuming a higher methane correction factor in the status quo

C sink

In Figure 47 the result for the baseline case is extended to include the sensitivity analysis of the C sink. If this is considered as a credit, it produces a deterioration in the net result for the medium scenario to 2030 by comparison with the status quo. The reason for this is the higher percentage of landfilled waste in the status quo (78%), which results in a larger credit for the C sink. An improvement in the medium scenario to 2030 by comparison with the status quo only arises if an effective gas collection efficiency of 40% is achieved.

Figure 47: Result for GWP: status quo versus future scenarios with C sink



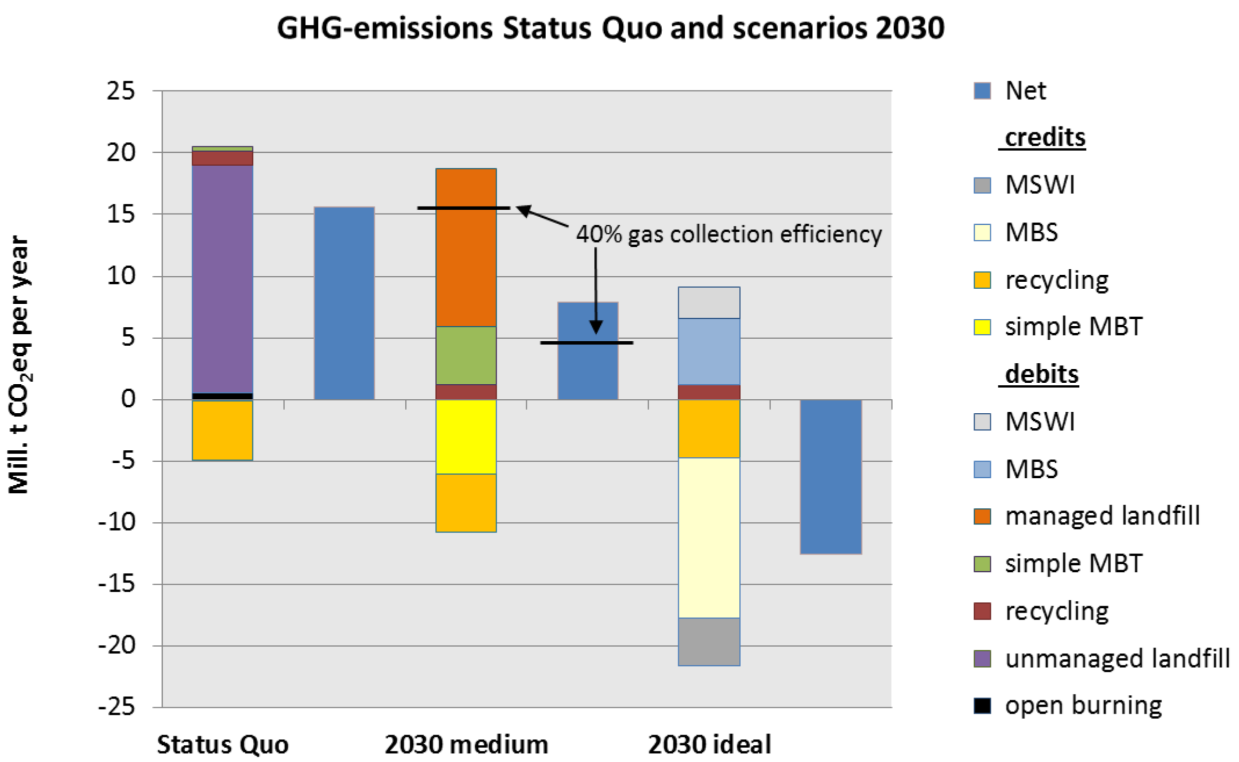
Higher methane correction factor

In the discussion of results for the baseline comparison it was explained that future managed landfilling actually causes the GHG balance to worsen by comparison with shallow, unmanaged landfilling, because in shallow unmanaged landfilling as described by the IPCC the aerobic conditions mean that only 40% of the methane formation potential is actually effective. Mathematically an effective gas collection efficiency of 60% would be required to balance out the increased emissions of managed landfilling under anaerobic conditions.

The classification of landfill deposits in India as not higher than five metres was taken from the statement in India's Second National Communication to the UNFCCC (MoEF 2012). However, this is likely to be a qualitative assessment. It is conceivable that there are also deposits > 5 m, or that there are deposits that are standing in water. For the purpose of this sensitivity analysis it was assumed that 50% of waste deposited in unmanaged landfills falls into one or both of these categories. According to the IPCC a methane correction factor of 0.8 needs to be applied to deposits of this type (Section 4.2.5).

The result of these assumptions is shown in Figure 48. Under these conditions the GHG emissions from unmanaged landfilling in the status quo increase from around 12.3 Mt CO_2-eq to around 18.4 Mt CO_2-eq. In these circumstances managed landfilling with a 20% effective gas collection efficiency would be a small improvement, even if the same amount of waste is landfilled in the future scenario as in the status quo (Section 6.4.1). A 40% effective gas collection efficiency would produce a larger improvement.

Figure 48: Sensitivity analysis: GHG balance with higher MCF in the status quo



6.5 Conclusions – India

Estimation of the GHG balance of India's current waste management system results overall in a GHG debit. Because of the uncertainties attending the data on waste volumes and management methods, this debit cannot be reliably quantified in absolute terms. While the sensitivity analysis that takes account of a C sink reduces the net emissions, it does not change the debit into a credit.

The GHG balance is improved through the collection and recycling of recyclables, which in India is performed by the informal sector. Although these activities are beneficial in terms of climate change mitigation, they are carried out by waste pickers under conditions that pose major risks to health.

The future scenarios that were explored show that the main ways in which the situation can be improved are through comprehensive organised collection of municipal solid waste (while retaining the informal collection of recyclables) and a reduction in the landfilling of organic waste. The introduction of managed landfilling is not in itself sufficient to cut GHG emissions; indeed, it can lead to increased emissions because managed landfilling has greater methane formation potential than unmanaged dumping. Treating mixed waste by means of technologies such as MSWI and MBS, which was considered as an alternative, results in a net credit, but in India this can only be considered with reservations on account of the country's experience with failed projects involving these technologies. The population, and in some cases also government agencies, are sceptical about waste incineration.

To improve the data situation, which ultimately provides the basis for a reliable estimate of GHG reduction potentials and better decision-making in connection with waste management planning and governance, it would be desirable to:

- Identify actual waste streams
- Identify the composition of waste and important characteristics
- Identify the actual potential of the recyclable fractions

To improve and expand an integrated waste management system that enables India's GHG mitigation potential to be fully utilised, it would be relevant to:

- Develop the capacities of official bodies at national and regional level in relation to implementation and monitoring of legal requirements and sound assessment of planning proposals with due regard to regional circumstances.
- Expand treatment capacities as an alternative to landfilling – for reasons of acceptance initially for low-cost and simple treatment methods such as simple mechanical-biological treatment of MSW with the involvement of the informal sector.
- Establish joint agreements on objectives with a progressive implementation plan with the Ministry of Environment & Forests (MoEF), the Ministry of New & Renewable Energy (MNRE) and the Ministry of Urban Development (MoUD).

The promote the implementation of measures, the following aspects should be noted and considered:

Countries such as Germany can support this process partly through the transfer of knowhow at technology level but in particular by providing support at the level of the official agencies, which urgently need to strengthen their capacity.

Funding is currently available primarily for the private sector, usually for individual facilities or single projects. By contrast, NAMAs (Nationally Appropriate Mitigation Action) provide an opportunity to finance the development of an integrated waste management system. So far, however, there are only rudimentary specific suggestions on how NAMAs for the waste management sector could or should be planned, monitored and verified.

Measures are urgently required, not least because the annually increasing volumes of waste mean that "no measures" results in a steady increase in GHG emissions.

Measures must be planned and implemented to harmonise with the activities of the informal sector. Important considerations are: conservation of the vital source of income and improvement of the health and hygiene aspects of working conditions.

Measures would ultimately not only reduce GHG emissions: introduction of an integrated waste management system with as much recycling of dry materials as possible would also make a significant contribution to resource conservation and the reduction of further environmental impacts.

7 Waste management - Egypt

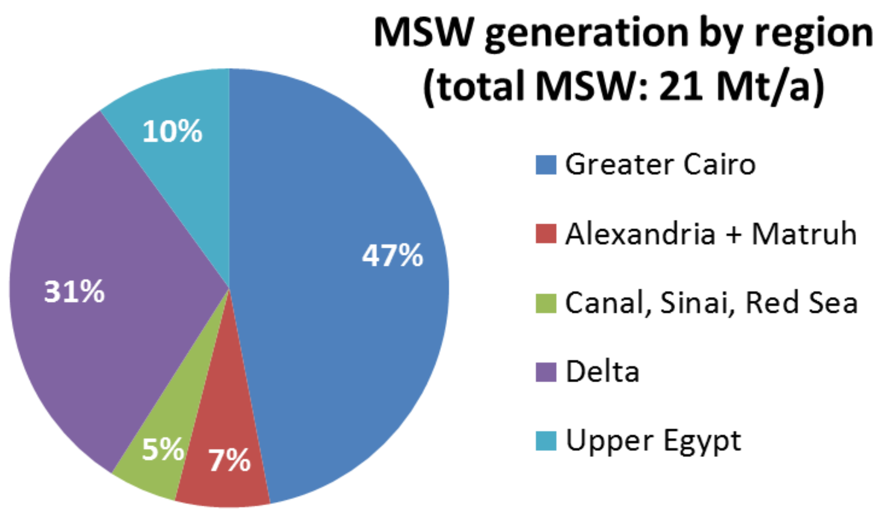
7.1 The current situation

7.1.1 Waste generation

According to the Egyptian office of statistics, Egypt has a population of approximately 84 million (CAPMAS 2013). 43% of the population is classed as urban.

Figures for waste volumes are published by Sweep-Net, a regional network for the exchange of information on integrated waste management in the MENA region[50] (Sweep-Net 2012). For 2010 the total volume of MSW is put at 21 Mt. The per-capita volume of waste is given as 0.7-1.0 kg/(cap*day) in urban areas and 0.4-0.5 kg/(cap*day) in rural areas. Applying the upper figures in these ranges to a population of 80 million in 2010 yields a total annual volume of MSW of 21 Mt. Figure 49 shows the volume of MSW by region.

Figure 49: Volume of waste in Egypt by region (Sweep-Net 2012)



It is clear from this that approximately half of Egyptian MSW arises in the Greater Cairo area (governorates of Cairo, Giza and Qalyubia). In addition, almost one-third arises in the Nile Delta area. The amount of waste generated in the arid regions (Upper Egypt, South Sinai, Red Sea) is relatively small. Figure 50 shows the location of the Egyptian governorates and the corresponding precipitation profiles.

Other figures broken down by governorate are published in (EEAA/METAP 2005, Ministry of Trade & Industry 2008) using information obtained from the Egyptian Environmental Affairs Agency (EEAA). The present assessment is based on data from the (Ministry of Trade & Industry 2008) for the year 2006 (Table 54, left). However, the figure used for the total volume of waste

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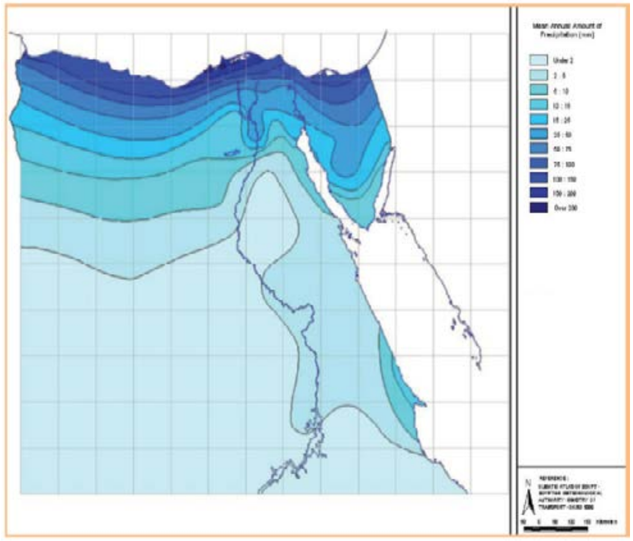
[50] Sweep-Net is a project supported by the German Federal Ministry for Economic Cooperation and Development (BMZ); the published information relates to waste disposal and waste management in the Middle East and North Africa (MENA) region.

is that of 21 Mt for the year 2010 given in (Sweep-Net 2012). To take account of the larger volume of waste, the figures have been extrapolated from 16 Mt/a (2006) to 21 Mt/a; the percentage distribution between governorates is assumed to be constant (Table 54, right).

Figure 50: Location of the 27 Egyptian governorates (left) and the corresponding precipitation profiles (right, in mm per year)

[Source: http://english.ahram.org.eg/NewsContent/3/12/56122/Business/Economy/Egypt-to-create–new-governorates-to-push-decentra.aspx]



[Source: 2. UNFCCC Communication 2010]

From the summary in Table 54 (left) it is clear that around three-quarters of MSW in Egypt arises in urban areas and one quarter in rural areas.

Table 54: Municipal solid waste generation in Egypt by governorate

Governorate	Waste generated (min. I+T 2008)			Collection rate			Collection figures for (ref. year 2010)	
	Urban (Mt/a)	Rural (Mt/a)	Total (Mt/a)	Urban (Sweep-Net 2010/12)	Rural (Messery et al. 2009)	Total (Min. I+T 2008)	Collected (Mt/a)	Uncollected (Mt/a)
Cairo	3.94	0.00	3.9			62%	3.22	1.97
Giza	1.33	0.35	1.7			64%	1.42	0.80
Qalyubia	0.96	0.30	1.3			50%	0.83	0.83
Alexandria	0.95	0.00	1.0			77%	0.97	0.29
Beheira	0.37	0.42	0.8	60%	27%		0.44	0.60
El Wadi El Jadid	0.02	0.01	0.0	60%	27%		0.02	0.01
Qena	0.17	0.20	0.4	60%	27%		0.21	0.28
Red Sea	0.08	0.00	0.1			52%	0.05	0.05
Marsa								
Monufia	0.18	0.28	0.5	60%	27%		0.24	0.37
Al Gharbia	0.37	0.30	0.7			50%	0.44	0.44
Kafr El	0.40	0.23	0.6	60%	27%		0.40	0.43
Damietta	0.27	0.11	0.4	60%	27%		0.26	0.26
Sohag	0.10	0.22	0.3	60%	27%		0.15	0.26
Aswan	0.16	0.05	0.2			41%	0.12	0.17
Asyut	0.14	0.20	0.3	60%	27%		0.18	0.26
Dakahlia	0.95	0.45	1.4	60%	27%		0.91	0.93
North Sinai	0.08	0.01	0.1			33%	0.04	0.07
South Sinai	0.10	0.02	0.1	60%	27%		0.08	0.07
Port Said	0.25	0.00	0.3	60%	27%		0.20	0.13
Ismailia	0.17	0.08	0.3	60%	27%		0.16	0.17
Luxor	0.06	0.00	0.1			45%	0.03	0.04
Suez	0.15	0.00	0.2	60%	27%		0.12	0.08
Al Sharqia	0.20	0.43	0.6	60%	27%		0.31	0.52
Beni Suef	0.20	0.09	0.3	60%	27%		0.19	0.19
El Minya	0.19	0.26	0.4	60%	27%		0.24	0.35
El Faiyum	0.11	0.11	0.2	60%	27%		0.12	0.16
Total	**12**	**4**	**16**				**11.4**	**9.7**

7.1.2 Waste collection

Egypt's waste collection system is not comprehensive. Collection rates are generally higher in urban areas than in rural ones and higher in affluent districts than in poor ones. (Sweep-Net 2012) cites collection ranges of 40-85% for urban areas and 0-35% for rural districts. Specific figures for some governorates are given in (Ministry of Trade & Industry 2008). (El-Messery et al. 2009) cite a collection ra*te of 27% as the average for rural areas.

The present study is based on the specific figures published by (Ministry of Trade & Industry 2008); for the remaining governorates global figures of 60% (urban) and 27% (rural) are used in the calculations (Table 54, centre).

The waste generation figures calculated for Cairo and Alexandria of 14,000 t/d and 3,500 t/d respectively in 2010 (Table 54 right) accord well with the data of (Zaki 2013), who quotes waste generation figures for 2012 of 15,000 t/d and 4,000 t/d respectively. The total uncollected fraction is calculated to be 46%. This figure accords well with the data of the National Solid Waste Management Programme (NSWMP 2013), which assume that 40% of waste is uncollected.

Various stakeholders are involved in the collection of MSW in Egypt. An overview of the structures in both the formal and informal sectors can be found in publications such as (CID/GTZ 2008).

Stakeholders in the informal sector (according to (CID/GTZ 2008))

- Mixed waste collection

In Greater Cairo in particular the traditional informal waste collection system plays a significant part in the collection of MSW; this informal system was established long before attempts were made to set up a formal waste collection system. These garbage collectors, the Zabbaleen, collect mixed municipal waste door-to-door. In return for payment of a licence fee, the responsible local authority – the Cairo Cleansing and Beautification Authority (CCBA) or the Giza Cleansing and Beautification Authority (GCBA) – grants them a licence to collect in a particular area. This permits them to collect waste in these areas and to charge householders a fee. The waste is transported to the "garbage cities" where the Zabbaleen live, where it is sorted manually and some of it is processed into secondary raw material. The processing activities range from simple dismantling, shredding and washing to granulation and extrusion. According to the data available for 2008, the organic fraction was used as animal feed and approximately 20% was considered no longer recyclable and taken to dumps.

- Collection of recyclables for exchange/payment

Throughout Egypt there exists a recycling system whereby street dealers known as Sarriha collect recyclable waste items (mainly scrap metal and plastic) from the inhabitants in exchange for money or household items. In some cases they also purchase source-segregated waste from commercial waste generators. Other stakeholders, the Robabekia and the Saxonia peddlers, operate in Greater Cairo and most other governorates, dealing in old and secondhand household goods. The Robabekia trade in old appliances, housewares, clothing, paper, books, glass bottles and scrap metal. The Saxonia peddlers, whose name refers to the hard porcelain from Saxony that they traditionally sold, specialise in old clothing and also trade in dishes, plates, bowls and tubs. According to (Zaki 2013), as a percentage of the total volume of waste the volume of recyclables collected by the roaming traders is virtually negligible.

- Selective waste picking from mixed waste

Recyclables are collected selectively from mixed waste in waste containers placed in the street by formal disposal companies, at waste collection points and at landfill sites. This waste picking takes place in most urban areas of Egypt (Zaki 2013). Recycling activities in rural areas are minimal because of the lower recyclable content there (El Messery et al. 2009). With regard to calculation of the GHG emissions of the waste sector, it must be assumed that the waste materials collected by waste pickers enter the recycling system, as there would otherwise be no incentive to collect them. In addition, relevant quantities must be collected by these methods, since the waste collected via the formal waste disposal system contains only small amounts of recyclables (Sherif 2012, I+U/GTZ 2006).

Stakeholders in the formal sector

There is at present no specific waste management legislation in Egypt (Sweep-Net 2012). More general legislation relevant to waste management includes the Law on General Public Cleanliness of 1967 and the Law on Protection of the Environment of 1994, with their various amendments. The 1994 law regulates the Ministry of the Environment with the Egyptian Environmental Affairs Agency (EEAA) as the executive arm (CID/GTZ 2008).

The task of negotiating contracts with private-sector waste management companies is the responsibility of the governorates. Municipal authorities are responsible for the cleanliness of the towns and cities and for licensing small local businesses (CID/GTZ 2008).

In the cities, private national and international companies have been commissioned to undertake waste disposal (Zaki 2013). The city of Cairo employs not only Egyptian companies but also multinational Italian and Spanish companies whose contracts run until 2017 (Viney 2013). However, implementation of the contracts that have been entered into is posing significant problems, especially with regard to the multinational companies. These problems are described for example in (Iskander 2009), who outlines the development of the formal waste disposal system in Cairo since the 1980s and the associated problems.

In the remaining governorates formal waste collection in urban areas is undertaken by the local authorities. In some rural areas, especially in villages, non-governmental organisations support the scanty collection system. There are some villages in Egypt in which no waste collection system has been established at all (Zaki 2013). (El-Messery et al 2009) report that private contractors are employed in some cases, but they are very few in number because the communities are poor and the volume of recyclables in their waste is very small. These authors also describe the problems of waste collection and disposal in rural areas. They state that while theoretically 75% of districts are covered by a disposal service (71% via local authorities, 24% via the private sector and 5% via civil society), this is only functional for 27% (with 25% of local authority services, 71% of private sector services and 100% of civil society services being functional).

Separate collection of recyclables or organic waste does not take place on any significant scale. However, the goal of separating waste into “wet” organic waste and “dry” waste (the remainder) was laid down in the National Strategy for Integrated Municipal Solid Waste Management of 2000 (EEAA 2005). According to (CID/GTZ 2008) it specifies a target of 40% segregated collection by 2005, although this has not yet been achieved. By means such as education campaigns in households and schools, non-governmental organisations in particular are attempting to persuade the population to separate “wet” and “dry” waste, largely with a view to improving the working conditions of manual sorters (Zabbaleen) in the waste sector (CID/GTZ 2008).

7.1.3 Waste composition

The information on waste composition diverges, as Table 55 shows. For the composition of the arising municipal waste, (Sweep-Net 2012) cites an organic fraction of around 50%. The percentage of recyclables is relatively high. This composition is based on data from the Waste Management Central Department of the EEAA and is a pure estimate (Zaki 2013).

By contrast, the Infrastruktur und Umwelt consultancy – commissioned by GTZ – has conducted two studies of the composition of Egyptian MSW (I+U/GTZ 2006). These studies relate to the formally collected quantities in the governorate of Kafr El-Sheikh, which lies in the relatively water-rich Nile Delta, and in the governorate of Qena in the dry region of Upper Egypt (see also Figure 50). A striking feature of the waste composition in both cases is the very low percentage of recyclables and the very large organic fraction. With regard to the organic fraction it should be borne in mind that this may include materials such as dust and sand included in street sweepings, since it was not possible to segregate these before measurement.

Table 55: Waste composition in Egypt according to various sources

Details in %	Sweep-Net 2012	I+U/GTZ 2006 (Kafr El-Sheikh)	I+U/GTZ 2006 (Qena)
Compostable waste	56	88	70
Paper	10	2	10
Plastic	13	4	9
Metals	2	0.5	1
Glass	4	0.5	2
Textiles		0.5	2
Other	15	4.5	6
Water content	30-40	50 (in organic material)	30 (in organic material)

The difference between Kafr El-Sheikh and Qena is partly explained by the lower water content of the organic waste in Qena. On the other hand, (informal) recycling activities are likely to be less extensive in more rural Upper Egypt, so that the recyclable fraction there is somewhat larger. There is a general trend for the waste arising in rural areas to contain fewer recyclables, but this results in turn in fewer recycling activities (El-Messery et al. 2009). In addition, the more advanced recycling infrastructure is better established in the Delta (EEAA/METAP 2005). The study of (I+U/GTZ 2006) also concludes that recycling activities in cities in the Delta are very efficient.

A study for the Bill & Melinda Gates Foundation undertook a detailed analysis of the composition of waste in Cairo (BMGF 2012). However, because the categorisation used in that study differs from the “classic” division into organic waste, recyclables and other, these results were not taken into account in the present study.

New studies of the volume and composition of waste in Egypt, commissioned by GIZ, are currently in progress: publication of the final results is still awaited (Stretz 2013).

7.2 Modelling for greenhouse gas emissions inventory

7.2.1 Disposal methods & waste composition

On the basis of the description of the structure of Egyptian waste management, the following conditions have been defined for the present inventory of the associated GHG emissions.

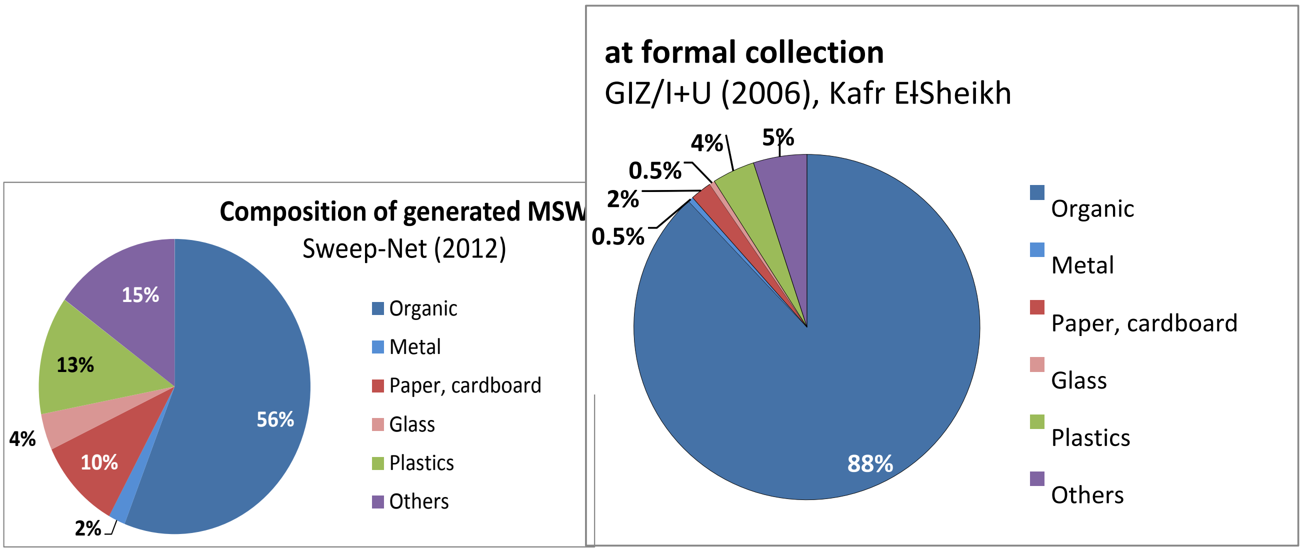
Consideration of the informal sector

With regard to the informal sector, the present study considers only the collection of mixed municipal waste by the Zabbaleen in the Greater Cairo area (GCA). Because this collection takes place directly at the doorstep, it is assumed that the collected mixed waste conforms to the composition according to (Sweep-Net 2012). Within the boundaries of the general uncertainties, the informally collected quantities for the GCA can be quantified: according to (Zaki 2013) the amount collected in Cairo is approximately 4,500 t/d, which is equivalent to 30% of the municipal waste arising there. For the two other governorates in Greater Cairo (Giza and Qalyubia), collection rates of 20% have been applied. In the rest of Egypt no informal waste collection is taken into account. For the activities of the Zabbaleen, quantitative information on disposal methods is available. (CID/GTZ 2008) states that after manual sorting around 80%-85% of the collected waste is recycled. For the present study it is accordingly specified that 80% of waste is either fed to animals (organic waste) or recycled (dry recyclables). A mass-weighted equal distribution is assumed, which leads to a recycling rate of 92.5% for each fraction (organic matter, metals, paper and cardboard, glass, plastic, textiles) (Figure 51). The remaining 20%, consisting of the “other” category and the sorting losses from the above categories, is landfilled. It is assumed that landfilling involves unmanaged dumps, since at present by far the largest part of Egyptian municipal waste goes to unmanaged dumps (Sweep-Net 2012).

Figure 52 summarises the quantities collected informally by the Zabbaleen and the breakdown between the corresponding disposal methods. According to (Zaki 2013) the volume of recyclables collected by purchase or exchange at the doorstep is negligible. The volumes of reusable materials collected and recycled by waste pickers are also ignored in the assessment of the status quo, because quantifying these volumes is attended by major uncertainties.

A general estimate of the potential was obtained by comparing the generated waste composition (Sweep-Net 2012) with the composition identified by (I+U/GTZ 2006), for example for the governorate of Kafr El-Sheikh (Figure 51, see also Section 7.1.3).

Figure 51: Comparison of the generated waste composition according to (Sweep-Net 2012) and the waste composition for the governorate of Kafr El-Sheikh according to (I+U/GTZ 2006)



On the basis of these data, the attempt was made to estimate the recyclable streams presumably collected by waste pickers from the difference between the composition of the generated waste and the formally collected waste. In this connection it should be borne in mind that the data on the composition of the generated waste are pure estimates that could be inaccurate (Zaki 2013). Similarly, the means for the formally collected waste composition from the governorate of Kafr El-Sheikh are not representative of all parts of the country. For example, the second study by (I+U/GTZ 2006) for the governorate of Qena arrived at different figures (see Section 7.1.3).

The estimate yielded a total recycling volume (metal, paper and cardboard, glass, plastic) of approximately 5 Mt/a, which is equivalent to about 25% of the total waste generated in Egypt. This figure is ten times higher than the recycling rate of 2.5% quoted by (Sweep-Net 2012). For plastics this estimate resulted in a potential plastic recycling volume of a good 2 Mt/a. This contrasts with a recycling volume of approximately 0.3 Mt/a identified in the national study of the plastic recycling sector commissioned by the Ministry of Trade & Industry (Ministry of Trade & Industry 2008).

Because no plausible link can be created between the waste compositions in the two data sources and the quantification described above is too uncertain, it was decided that the materials collected and recycled through waste picking would not be included in this study. It should be noted, though, that these activities make a significant contribution to the recycling output of the Egyptian waste management sector (see also EEAA/METAP 2005, CID/GTZ 2008).

Waste picking is indirectly taken into account in the calculation of the emissions of the formal waste management system since the formally collected quantities of waste are based on the waste composition according to (I+U/GTZ 2006), which contains only very small percentages of recyclables. However, this means that the extent of actual (informal) recycling is underestimated. To illustrate the effect on the climate change mitigation potential of waste management of including all recycling activities, the ideal future scenario (Scenario 2), as well as considering the inventory of the overall waste management system on the basis of the waste composition in accordance with (I+U/GTZ 2006), considers in a sensitivity analysis the balance based on the composition of the generated waste in accordance with (Sweep-Net 2012).[51]

Consideration of the formal sector

For the rest of Egypt outside the Greater Cairo area it is assumed that all collected waste is collected by the formal sector. After deduction of informal collection, the formal collection rates for the Greater Cairo Area are 32% (Cairo), 44% (Giza) and 30% (Qalyubia). It follows that in total 43% of the municipal waste generated in Egypt is treated by the formal waste management sector (Figure 52).

The breakdown between disposal methods is based on the data in (Sweep-Net 2012):

- 9% composting
- 2.5% recycling
- 5% sanitary landfill
- 83.5% open dump

It is assumed that the composition of the formally collected waste stream is the average composition determined by (I+U/GTZ 2006) for the governorate of Kafr El-Sheikh, as recyclables are removed before formal collection. For simplification this assumption is applied to all governorates, since by far the largest proportion of the waste arises in regions whose climate is fairly similar to that of the Delta (GCA, Alexandria, Delta; see Figure 50). Around 15% of the waste arises in very arid, rural areas (Upper Egypt, South Sinai, Red Sea) where the composition of the waste is likely to be more similar to that of Qena. However, the waste arising on the Red Sea is presumably strongly influenced by tourism and this will mean that the waste is of different composition again. Because of the relatively small volume involved and the general data uncertainties associated with the inventory, the issue has not been explored in such detail.

Consideration of the uncollected waste

Around 46% of the municipal waste generated in Egypt is not collected. For the proportion arising in urban areas (approx. 28%), it is assumed that all of this is scattered. According to (Zaki 2013), other practices such as open burning are very rarely used. (El Messery et al. 2009) interviewed key individuals and representatives of local councils on the subject of uncollected waste in rural areas. They found that

- approx. 50% of the population use the combustible components as fuel for cooking
- approx. 80% of farmers feed the organic components to animals or compost it
- approx. 50% of the population scatter waste (on open land or canal banks)
- approx. 30% burn MSW openly.

[51] An initial sensitivity analysis in which all waste is considered on the basis of the composition according to (Sweep-Net 2012) has already been carried out in connection with the assessment of the status quo. However, this does not reflect the inclusion of recycling activities, because the distribution between different forms of collection and the recycling performance of the formal sector are initially kept constant by comparison with the baseline case. Irrespective of the composition, this corresponds to a very small recycling volume.

Direct use of the combustible and organic components is ignored in the assessment of the status quo, because waste that is reused directly is not generally considered to form part of waste management and does not feature in the waste statistics. For the basic scenario it is therefore assumed that 70% of the uncollected waste is scattered and the remaining 30% is burned in the open.

In addition, a sensitivity analysis is carried out to illustrate the effect of direct use as a contribution to climate change mitigation. It is assumed for this purpose that organic waste is not simply dumped or burned but that 20% of it is fed to animals and 20% is composted. A further 25% is used as fuel for cooking. Restricting this type of use to the organic fraction is a simplification, as according to (El Messery et al. 2009) other combustible components (including plastic, because of ignorance of their harmful effects on health) are also used as cooking fuel. However, because organic matter constitutes by far the largest proportion of combustible waste, this simplification seems justified.

Figure 52: Material flow diagram of the waste streams assumed for Egypt

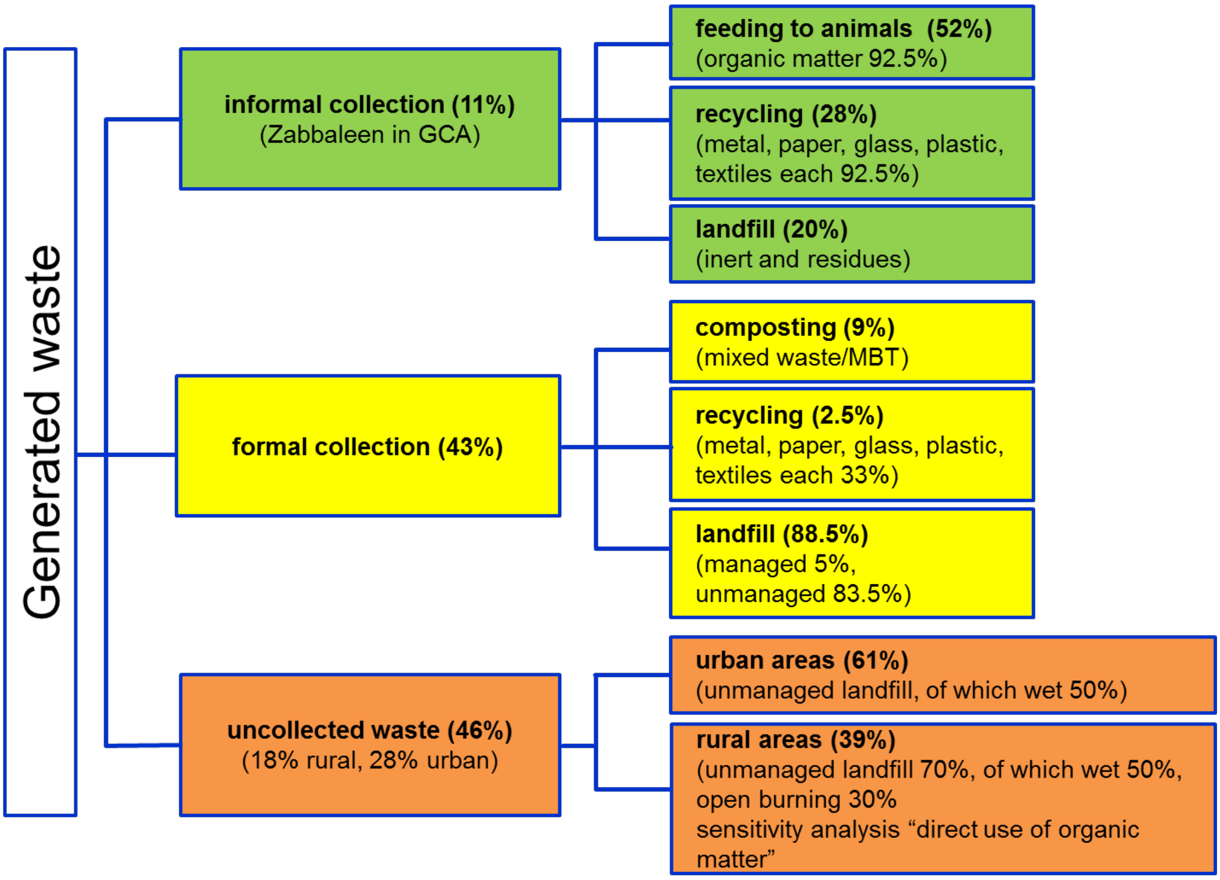
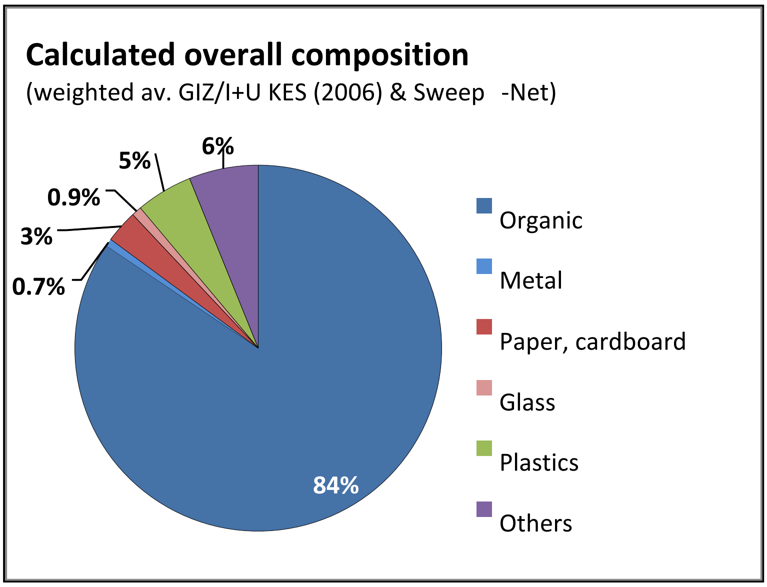


Figure 52 summarises the conditions that have been specified. For the purpose of the inventory the uncollected waste is regarded as having the composition specified in (I+U/GTZ 2006), because it is assumed that recyclables in open dumps are waste picked and/or that the waste disposed of in rural areas has a low recyclable content. In a sensitivity analysis, the composition according to (Sweep-Net 2012) is applied for all forms of collection (including formal and uncollected).

7.2.2 Calculating the characteristics of the waste streams

The inventory of the Egyptian waste management sector is based for informally collected municipal waste on the composition according to (Sweep-Net 2012) and for formally collected and uncollected waste on the composition according to (I+U/GTZ 2006). For the basic scenario this yields the overall composition shown in Figure 53.

Figure 53: Calculated overall composition for the assessment of the Egyptian waste management system



To quantify the textiles, the proportion of 10% of the “other” category in (I+U/GTZ 2006) was also applied to the composition according to (Sweep-Net 2012). The mean degradable organic carbon (DOC) content is thus 16% (Sweep-Net 2012) or 20% (I+U/GTZ 2006). If changes occur as a result of upstream treatment processes, the compositions are adjusted accordingly.

For the informal sector the composition of the segregated waste that is ultimately dumped is calculated by deducting the organic matter fed to animals and the reusable materials removed for recycling. Because a large proportion of the organic matter is removed, the material that is sent to landfill has a relatively low degradable organic carbon content (DOC 6%).

Composting and recycling activities also change the composition of the formally collected waste that is sent to landfill. However, because of the small amounts that are composted and recycled, this effect is so small that the DOC content of the dumped waste remains at around 20%.

For the uncollected fraction the composition in the basic scenario remains unchanged: the DOC in the waste sent to landfill is thus 20%, while the fossil C content (from plastics and textiles) of the openly burned waste is 3%. In the sensitivity analysis, consideration of the direct use of some of the organic material at the place of origin (fuel, animal feed, composting in rural areas, see Section 7.2.1) results in a reduced organic content and hence in slightly modified characteristics (DOC_{Sens1} 19%, $C_{fossil,Sens1}$ 4%).

7.2.3 Landfill

A distinction is made between managed and unmanaged landfill. The scattering of uncollected waste corresponds in the model to unmanaged landfill.

Modelling of the emissions associated with landfilling is based on the default values in (IPCC 2006) (see Section 4.2.5). For the scattering of uncollected waste, 50% of unmanaged landfilling is modelled as shallow (MCF=0.4). The other 50% is assumed to be unmanaged landfilling with a high water level (MCF=0.8), because dumping on canal banks is common practice in Egypt. In this case the methane emissions are correspondingly increased. For the unmanaged landfilling of collected MSW, 100% shallow landfilling is assumed. In view of the very dry climate it can be assumed that the methane emissions from landfilling are below the values calculated using the IPCC default values. This opinion was shared by Egyptian stakeholders, but because it could not be reliably verified the IPCC default values were retained as conservative estimates.

The effect of landfill fires is not considered, because no quantitative details are available.

7.2.4 Composting

Since the end of the 1990s the construction of composting facilities has been driven forward in Egypt, because such facilities were regarded as a means of recycling municipal waste, which consists mainly of organic mattter (Sherif no year, Iskandar 2009). According to (EEAA/METAP 2005) there are 56 composting facilities to which mixed waste is delivered. However, according to (Sherif no year, Iskandar 2009) major problems have been encountered in operating these facilities, so that many of them are standing idle or being used only occasionally.

The composting takes place after segregation of the organic matter. Despite using mixed waste, a few facilities nevertheless produce compost of acceptable quality (Zaki 2013). However, all the compost is sold as being of satisfactory quality – sometimes at correspondingly low prices – because awareness of the negative impacts of poor quality is low (Zaki 2013).

The emissions from these facilities are calculated from the average methane and nitrous oxide emissions for open composting according to (gewitra 2009) (Table 9) and the resource consumption for transporting the compost (for 20 km) and applying it (as in (Öko-Institut/IFEU 2010)). Because (Zaki 2013) states that a few companies produce compost of acceptable quality, a credit (as in (Öko-Institut/IFEU 2010)) is assigned for 10% of the compost produced. The results are presented without a C sink in the baseline case and with a C sink as a sensitivity. Where the C sink is included, it is applied to the entire composted quantity.

Because at least some of the composting facilities involve mechanical-biological treatment (MBT), the emissions arising from the MBT are also estimated for guidance purposes using the average value for Germany (27.7 kg CO_2-eq/t_{input}) calculated in (Öko-Institut/IFEU 2010). Because of the way the data is presented in the statistics (see Section 7.2.1, Sweep-Net 2012), a more nuanced assessment of the emissions of composting facilities as MBT is not possible.

In addition to the composting of some of the formally collected MSW described above, a percentage of home composting is also considered in the sensitivity analysis of the direct use of organic matter in rural areas. In this case the emissions are also calculated from the mean methane and nitrous oxide emissions for open composting in (gewitra 2009). It is assumed that because of the dry climate home composting tends not to result in very high methane emissions. No emissions are assessed for transport and application. 100% of the compost produced is credited.

7.2.5 Recycling

The Egyptian recycling sector is described in (EEAA/METAP 2005). According to this report, emissions for recycled materials arise in Egypt in connection with paper and cardboard, plastic, metals, glass, textiles and bones. In general the waste undergoes a two-stage sorting process and is then traded countrywide via an (informal) network of dealers. While some of the materials are reused, the majority are processed in workshops and factories to produce marketable products for local needs. There are specialised recycling centres for the various materials where the recyclables are brought together and processed. Most of these centres are in Cairo and the Nile Delta. The standard emission factors described in Section 4.2.4 are used to assess the contribution of recycling to climate change mitigation. In some cases special features of the Egyptian situation are taken into account.

- Plastic

PE, PP and PS are recycled in many cities. PVC is also recycled but is not relevant to the present study since it is not a significant component of municipal waste. The most highly specialised centres are located in Cairo, Dakahlia and El Minya. Moqattam, one of the “garbage cities”, is the site of Egypt’s largest plastic trading and recycling centre. According to (Ministry of Trade & Industry 2008), the difficulties associated with applying for a licence mean that the vast majority of plastic recyclers operate without a licence; as a result, only 2% of the existing facilities are registered with the Industrial Development Authority (Ministry of Trade & Industry 2008). The equipment used is mainly crushers, granulating machines and agglomerators. The plastic that is to be recycled is first sorted by type and colour and then washed and dried before granulating. Injection moulding, blow moulding and extrusion are used to process the granulate into products. Because of uncertainties with regard to origin and possible contaminants, the recycled material should not find its way into medical applications, toys or products that come into contact with food. PET is another material that is much in demand and one that according to (I+U/GTZ 2006) is not collected formally. (Zaki 2013) states that PET, which is mainly imported in the form of bottles (Ministry of Trade & Industry 2008), is usually exported for recycling. As for India, plastic recycling is assessed using the emission factor for “low” substitution effect. It should be noted, however, that because of the upstream sorting that takes place in Egypt it is likely that secondary granulate can be used in higher quality applications.

- Paper and cardboard

In Egypt waste paper is classified as newspaper, magazines, white paper, cardboard or mixed. Important products are grey cardboard and kraft paper, which are used to produce kitchen roll, toilet paper, paper towels, etc. Major trade centres are located in the Greater Cairo area (Moqattam, Qalyubia). Most of the recycling works are located in Cairo (6th October Industrial Area) and the governorate of Sharquia (10^{th} Ramadan Industrial Area) that adjoins the GCA to the north. Egypt’s largest paper factory (Rakta), which processes both virgin fibre (from rice straw) and waste paper, is in Alexandria. The assessment of the emissions of paper recycling is based on the standard emission factors at fibre level (see Section 4.2.4).

- Glass

Glass is sorted by colour: white glass is the most valuable and brown the least. Some recycled glass is made into jewellery and craft products sold to tourists; the rest is used as an input material by various glass factories. There are recycling plants in Alexandria and Qalyubia. The emission factors used are the standard emission factors described in Section 4.2.4.

- Metals

Tinplate is recycled in steel mills and aluminium in aluminium smelting plants, with wire or strips produced as the end product. The factories are located mainly in the Delta (Mit Ghamr, governorate of Dakahlia) and in the Greater Cairo area. Mit Ghamr processes 70% of Egypt's aluminium scrap (both old and new scrap). Exports go to Libya and the Sudan. In Mit Ghamr there are also copper and steel recycling works. Lead is also recycled in Egypt, but this study of the recycling of municipal solid waste (excluding electronic waste) does not consider either copper or lead recycling. In the assessment of the contribution of metal recycling to climate change mitigation it is assumed that that the metal mix in MSW comprises 87% ferrous metals and 13% aluminium; the standard emission factors set out in Section 4.2.4 are used.

- Textiles

Textiles are separated into cotton and synthetic fibres and/or by colour. The end products are mainly rag rugs and stuffing materials for mattresses and cushions. These recycling activities take place predominantly in the poorer areas of Cairo and Alexandria and in Upper Egypt. For example, there are textile processing works in the Greater Cairo area (Moqattam and Qalyubia) and in two governorates south of Giza (Fayoum, Bany Sweif). The textile recycling that has been described is assigned a value of zero in the GHG inventory (see Section 4.2.4).

- Bones (not included)

Bones are used for glue for the wood industry, for active carbon for water filters, oil factories and sugar refineries, in the production of calcium powder as a feed additive, and as a substitute for ivory in craft work. Fats are used in the production of cosmetics. There are processing works in Qalyubia and other districts. Animal waste is not a component of MSW and is not considered in this study.

Both for formal composting and for the amount that according to (Sweep-Net 2012) is formally sent to recycling, the emissions for the sorting of recyclables – some of which takes place at composting facilities – are estimated for guidance purposes using the average value for MBT in Germany (27.7 kg CO_2-eq/t_{input}) identified in (Öko-Institut/IFEU 2010). No emissions are assigned to the sorting of informally collected waste, because all such waste is sorted manually.

7.2.6 Other technologies

In addition to the disposal and recycling methods described above, the present study also considers the use of organic waste as animal feed. The organic material separated out from informally collected municipal waste is fed to animals (predominantly pigs). However, because this practice raises hygiene issues, it is assigned a value of zero in the GHG balance. Animal feeding is, though, included in the sensitivity analysis of the direct use of organic waste in rural areas because it is assumed that source-segregated organic waste is used here. For the emission factor the assessment uses for indicative purposes the substitution of fodder beet and/or soya, as originally developed for the feeding of organic matter to pigs. The substitution takes place 1:1 on the basis of calorific value. The nutritional value is set at 0.02 kg CO_2-eq/MJ: this value is derived from Ecoinvent data (v2.2) on fodder beet in Switzerland and soya growing in the USA and Brazil (including land clearance). In this sensitivity analysis organic matter (and other combustible waste components) is also used as a cooking fuel. Because some plastics are burned in this process, which thus raises serious health concerns, this practice is assigned a value of zero (neither credit nor debit) in the GHG balance.

In the countryside some of the waste is burned in the open. In this case the fossil CO_2 emissions from the burning of plastics and textiles are included as greenhouse gas emissions.

The ideal future scenario (Scenario 2) includes some production of biogas from an (in future) source-segregated organic fraction. This option is also considered in the scenarios for reducing GHG emissions from the waste sector in Egypt's Second National Communication to the UNFCCC (2nd UNFCCC Communication 2010), and in (TOC 2010) it is regarded as a very promising technology. In the present study the biogas that is produced substitutes natural gas. Because no data on natural gas production were found for Egypt, the study uses data on Algerian production for the year 2030 (GEMIS v4.8). The emission factor including combustion is 2.6 kg CO_2-eq/kg natural gas. The losses in upgrading biogas to natural gas quality are set at 10%. According to (Holmgren 2012), this is roughly the loss incurred by Swedish plants in 2008. In addition, a loss of 5% is included for supply of the energy needed for upgrading. This results in a substitution factor of 0.85 on the basis of calorific value. For simplification the emissions associated with the production of biogas are calculated in accordance with (Öko-Institut/IFEU 2010). The debit thus includes a 1% methane slip and a debit of 53 kg CO_2-eq/t input for the digestate. In accordance with (Öko-Institut/IFEU 2010) the credit for the use of the digestate is 40.6 kg CO_2-eq/t input and the C sink is 20.1 kg CO_2-eq/t input.

In addition, in ideal future Scenario 2 all the residual waste (after segregated collection of a large part of the organic waste) is regarded as being disposed of via MBT. The mass balance of the MBT is adjusted for conditions in Egypt. It was assumed that 60% of the dry recyclables (constituting 12% of the input) can be separated and recycled. For the remaining recyclable quantities it was assumed that 90% is converted into an RDF fraction, which accounts for 11% of the input (dry recyclables plus a 44% organic fraction). 13% is separated out in an inert fraction. The remaining quantity is biologically stabilised (MBT residue 26% of total input, losses 38% of total input). In calculating the calorific value of the RDF fraction it was assumed that all the components are present in a dry state: given the climate conditions in Egypt, this can easily be achieved through solar energy.

RDF is already being produced in Egypt, although in connection with MSW it seems that this is still at the pilot project stage. Larger-scale projects tend to involve the use of agricultural residues (ENTAG/ECARU; Sweep-Net RDF). To assess the climate change mitigation contribution of RDF use, the fossil CO_2 emissions arising from the incineration of plastics and textiles and the emissions associated with transporting the RDF a distance of 200 km to the cement works are first calculated on the debit side. These are offset by a credit on the basis of calorific value for the substitution of natural gas, which is currently the main source of energy for cement works in Egypt. To take account of upgrading losses, a substitution factor of 0.9 is applied.

The MBT emissions are calculated on the basis of electricity and diesel use (10 kWh/t_{input} and 9.6 kg CO_2-eq/t_{input} respectively). The GHG emissions from stabilisation are calculated using the mean methane and nitrous oxide emissions for open composting according to (gewitra 2009) (see Table 9). For the stabilate, including the inert fraction, emissions for transport to the landfill (30 km) and dumping (electricity requirement 2 kWh/t, mechanical energy 9 kg CO_2-eq/t) are also added. To calculate the landfill gas emissions from the stabilate, it is assumed in accordance with (IFEU 2012) that 40% of the original DOC remains in the stabilate after stabilisation, with a DOCf of 10% and a reduced methane content of the landfill gas of 40%. The values for the assessment of the MBT including disposal of the stabilate and the inert fraction are harmonised with the assessment for India.

Municipal solid waste incinerators are not considered for Egypt, since at present no such plants exist (except for incineration of clinical waste) and as far as is known none are planned for the future. For example, (CID/GTZ 2008, p. 12) cites the objectives of the National Strategy for Integrated Municipal Solid Waste Management, which make no mention of MSWIs. Similarly, MSWIs are not considered as an option in the scenarios for reducing GHG emissions from the waste sector described in Egypt's Second National Communication to the UNFCCC (2nd UNFCCC Communication 2010).

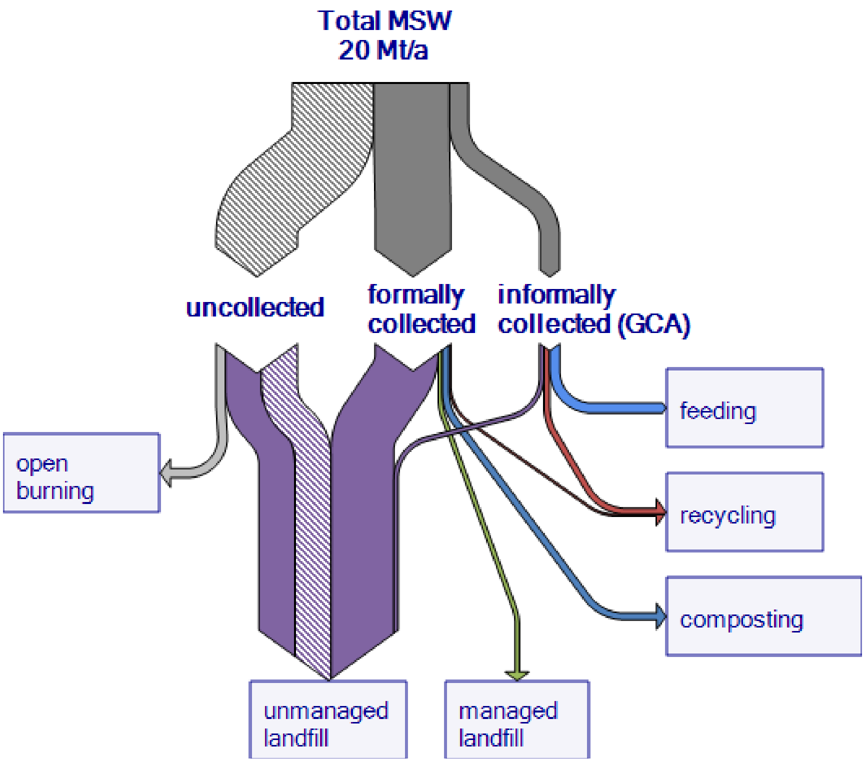
The GHG emissions associated with transport of the collected MSW are estimated at a standard rate. The transport distance is set at 100 km. Transport emissions for Egypt (as for India) are calculated at a standard rate of 230 g CO_2-eq/tkm, which corresponds to the emissions of commercial vehicles in Germany in the 1980s.

For average electricity generation (electricity mix) in Egypt, an emission factor of 500 g CO_2-eq/kWh is applied. The literature yields a range of values for this: 501 g CO_2-eq/kWh (Brander 2011), 450 g CO_2-eq/kWh (Climate Registry 2013) and 490-510 g CO_2-eq/kWh (Blodgett, no year).

7.3 Results: waste management – Egypt

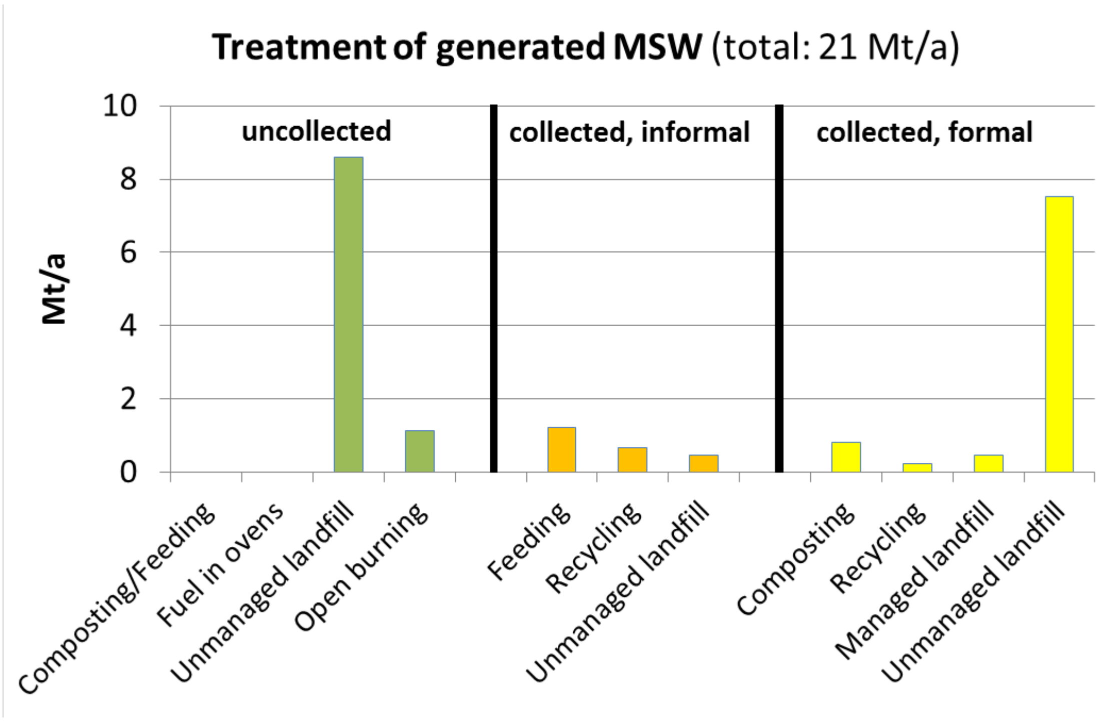
The conditions and assumptions described in Section 7.2.1 result in the division of waste streams between the various collection and disposal methods shown in Figure 54. It is clear from this that unmanaged landfill accounts for by far the largest proportion: approximately 25% goes to wet unmanaged landfill, which includes the disposal of uncollected MSW on canal banks.

Figure 54: Waste streams in the various management methods



Although informal collection is taken into account only for the Greater Cairo area, it nevertheless makes a relevant contribution at approx. 10%. Waste from this sector goes mainly to feeding and recycling. Only a small residual stream is landfilled. By contrast, composting and in particular recycling play only a minor part in the formal sector. Figure 55 shows the generated waste quantities in million tonnes per year.

Figure 55: Destinations of waste generated in Egypt (total 21 Mt/a)

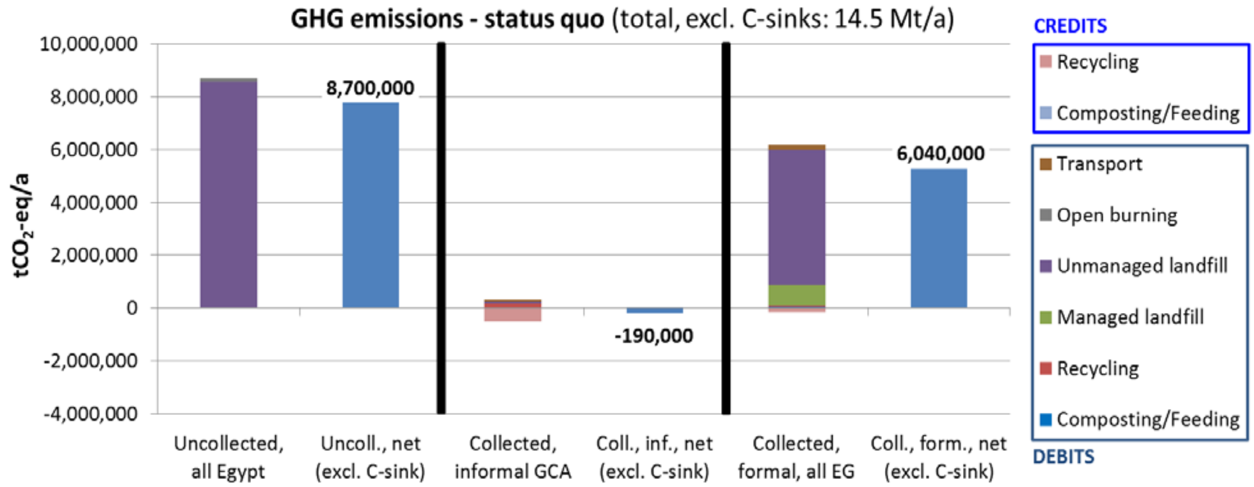


The direct recycling (composting/feeding/fuel) of the organic matter in uncollected MSW is considered only in the sensitivity analysis described below.

Figure 56 shows the GWP for the status quo of the Egyptian waste management system, broken down according to collection categories. The main contribution is made by the emissions from unmanaged landfill, although as a result of the methane correction factor these are significantly lower in specific terms than the emissions from managed landfill. For formally collected and uncollected MSW, other practices do not play a significant part in generating emissions.

As a result of the small mass flows involved, the indicative estimate for the emissions from mechanical-biological treatment for composting and recycling in the formal sector yields a negligibly small contribution of approx. 0.03 Mt CO_2-eq/a.

Figure 56: Results – GWP of the status quo without C sink



In the informal sector the credit for recycling outweighs the debits associated with disposal. If feeding were also credited here using the above-mentioned emission factor for the substitution of beet/soya, the credit would be about 50% higher again.

The total GHG emissions for the status quo amount to approx. 14.5 Mt CO_2-eq/a. They are thus on the same scale as the values quoted in Egypt's Second National Communication to the UNFCCC (2nd UNFCCC Communication 2010), which gives a figure of 12 Mt CO_2-eq/a for dumping on land in the year 2000. However, the figures are only roughly comparable, because annual emissions from the depositing of waste were calculated and reported for the National Communication (2nd UNFCCC Communication 2010), while the life-cycle inventory allocates all emissions over the entire life of the landfill to the year in which the waste was deposited. The two calculation methods also differ in the underlying composition of the waste; (2nd UNFCCC Communication 2010) uses the composition of generated waste based on data from EEAA (approx. 50-60% organic, 10-25% paper), while this study uses the composition according to (I+U/GTZ 2006), which involves 88% organic matter and 2% paper.

As a sensitivity analysis, Figure 57 shows the GWP taking account of the C sink in landfills and through compost use. This reduces net total emissions by around 40% to somewhat more than 8 Mt CO_2-eq/a.

Figure 57: Results of sensitivity analysis – GWP of the status quo with C sink

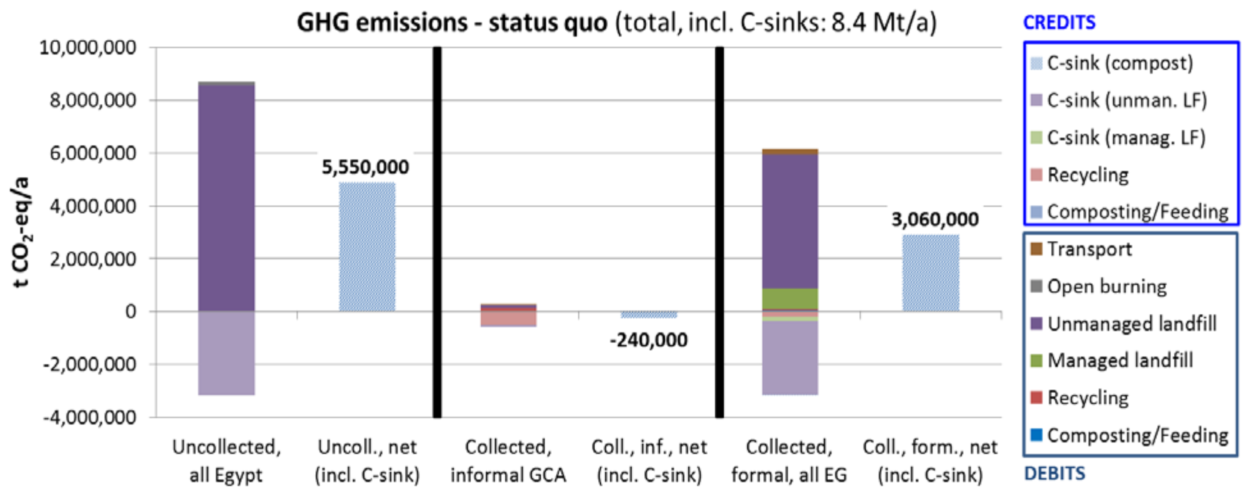
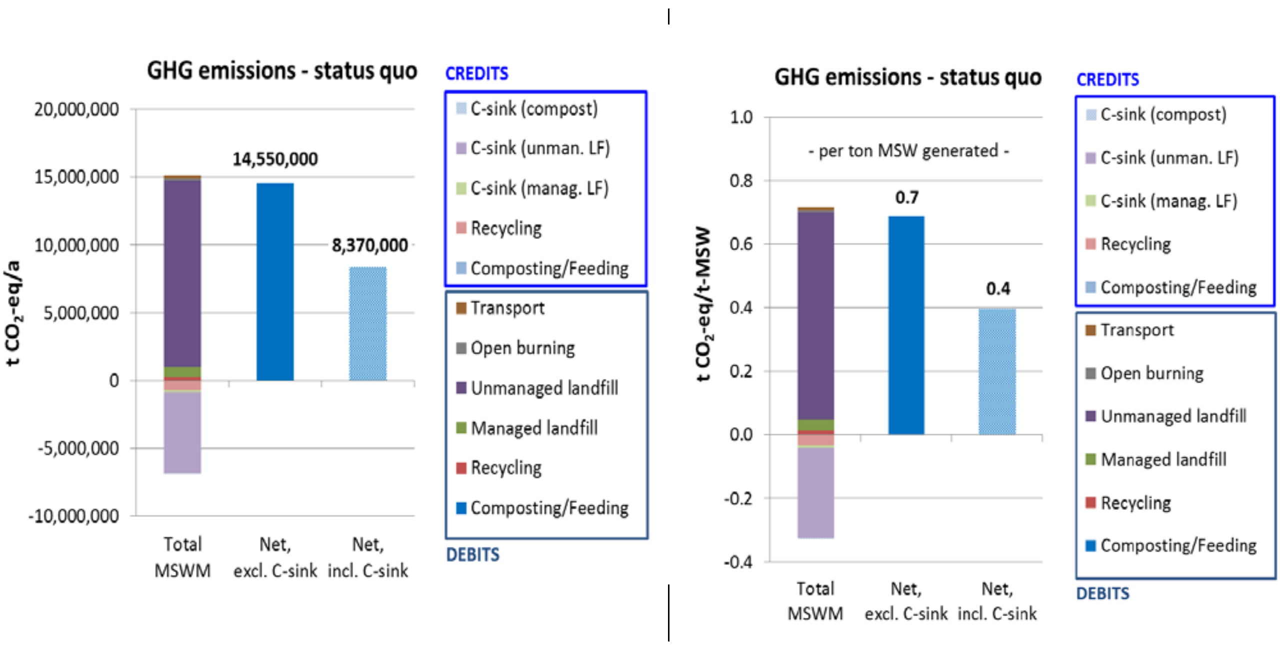


Figure 58 provides an overview of the entire waste management sector. This shows that each tonne of MSW generated in Egypt produces 0.7 t CO_2-eq (without C sink) or 0.4 t CO_2-eq (with C sink)

Figure 58: GHG emissions with and without C sink: absolute annual emissions and specific values.



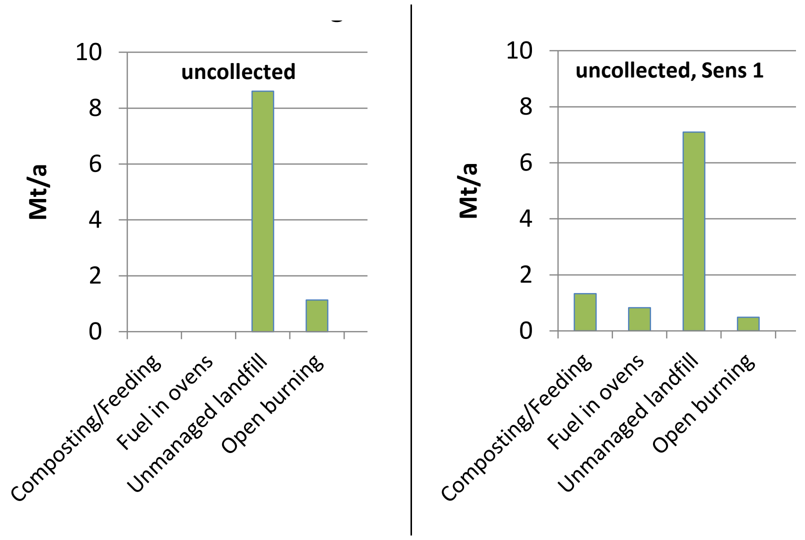
Sensitivity analyses

Two further sensitivity analyses were performed in relation to the status quo:

- "Sens 1": Direct use of organic matter in uncollected MSW

For rural areas the baseline scenario considers only unofficial dumping (unmanaged landfill) and burning of uncollected waste. According to (El-Messery et al. 2009), other practices are home composting, use as animal feed and use as fuel in stoves. The modelling of these practices is described in Sections 7.2.1 and 7.2.6. Figure 59 shows the resulting distribution of the waste streams between disposal methods according to sensitivity analysis 1 (Sens 1, right) by comparison with the baseline scenario (left).

Figure 59: Destination of uncollected waste – status quo and Sens 1

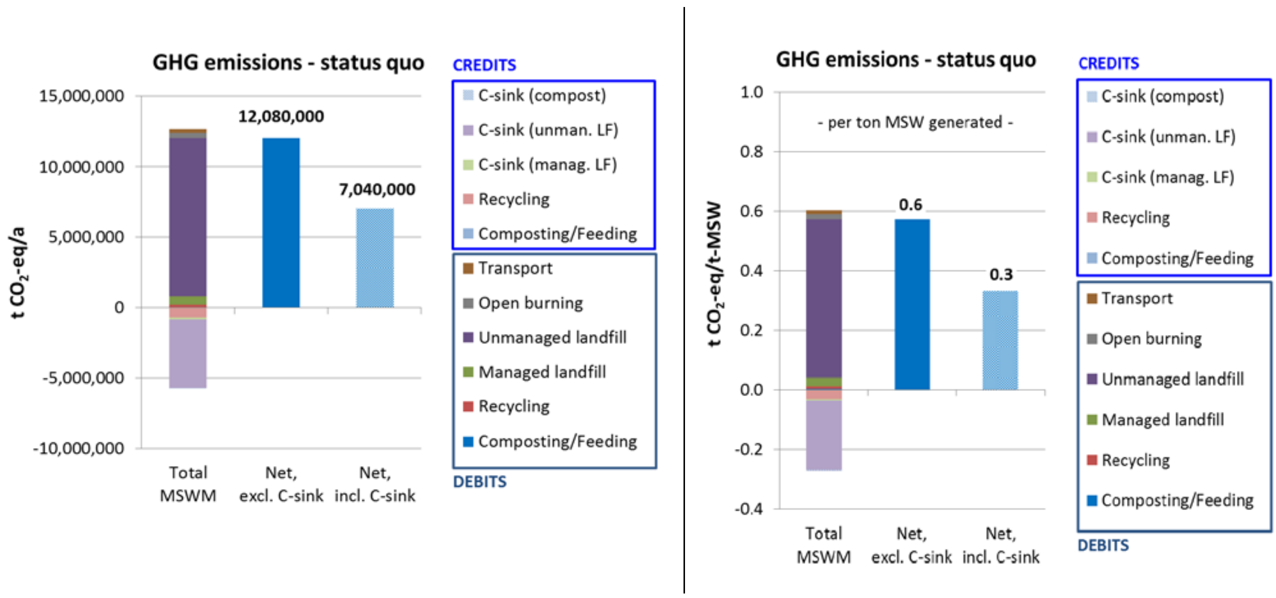


This reduces the landfilled quantity by almost 20%. In addition, credits are allocated for the composting and feeding. This reduces the GHG emissions of the entire waste management sector by around 12% and 14% respectively to 12.7 Mt CO_2-eq/a (without C sink) and 7.1 Mt CO_2-eq/a (with C sink).

- “Sens 2”: Variation in composition

This sensitivity analysis uses the waste composition according to (Sweep-Net 2012) for the formally collected and uncollected waste (see Sections 7.1.3, 7.2.1, 7.2.2) in order to illustrate the uncertainty with regard to the assumption of higher organic content in (I+U/GTZ 2006). The altered composition (in particular 56% organic instead of 88%, 10% paper instead of 2%) results in reduced GHG emissions (Figure 60, reduction of approx. 17%).

Figure 60: Results of sensitivity analysis – absolute annual emissions and specific values, Sens 2



The credits for recycling remain unchanged in this sensitivity analysis, because the percentage of formally collected waste that is recycled (2.5% according to (Sweep-Net 2012)) is not changed. The credit for formal composting falls, because although 9% mixed waste still goes to composting, the organic fraction of this is smaller. However, the contribution is negligibly small. The effect of the altered composition, and especially of the increased recyclable content, only becomes relevant if recycling rates are increased – i.e. if the increased recyclable content is actually recycled. This is illustrated in the sensitivity analysis of ideal future Scenario 2 (see Section 7.4.2).

7.4 Future scenarios to 2030

7.4.1 Description of the scenarios

The volume of MSW in Egypt is cited as growing at 2% p.a. in (Sweep-Net 2012) and at 3.4% p.a. in (Sweep-Net 2010) and other older publications. This means that the waste generated in 2030 will be at least 30 Mt/a. Another future trend that is described is the rising percentage of recyclables as a result of the increased consumption of packaging materials. However, in the future scenarios considered here both the total quantity and the composition of the waste are kept constant in order to facilitate a comparison of different disposal options (see Section 4.1.1).[52]

As with the other countries, the future scenarios considered for Egypt are a "medium" Scenario 1 (SC 1) and an "ideal" Scenario 2 (SC 2). The scenarios are based on the disposal options discussed in Egyptian sources. For example, according to (EEAA/METAP 2005), the National Strategy for Integrated Municipal Solid Waste Management takes account not only of collection, transfer and temporary storage but also of recycling, composting and sanitary landfilling technologies. (CID/GTZ 2008) cites the objectives of this strategy as increased collection rates, dumping in managed rather than unmanaged landfills (80%), composting of organic material (50%), recycling (20%), segregated collection (40%) of "wet" (organic) and "dry" (other) materials, and prevention of waste. Prevention is not considered in this study (see Section 4.1.1). The percentages in brackets relate to the targets set for 2005: according to (CID/GTZ 2008) these were not actually achieved.

In relation to the segregated collection of organic matter and other wastes, (CID/GTZ 2008) mentions pilot campaigns run by the NGO A.P.E., which resulted in 65% of the population of two residential districts of Cairo continuing to sort their waste over a period of two years. Other studies in the Delta and Upper Egypt have shown that in Egypt waste is "traditionally" sorted into organic and non-organic fractions; however, among the population that has lived for some time in urban areas and is relatively affluent, this trend is disappearing. The (2nd UNFCCC Communication 2010) considers not only the technologies described above but also the production of RDF and biogas as options for preventing GHG emissions from the waste sector.

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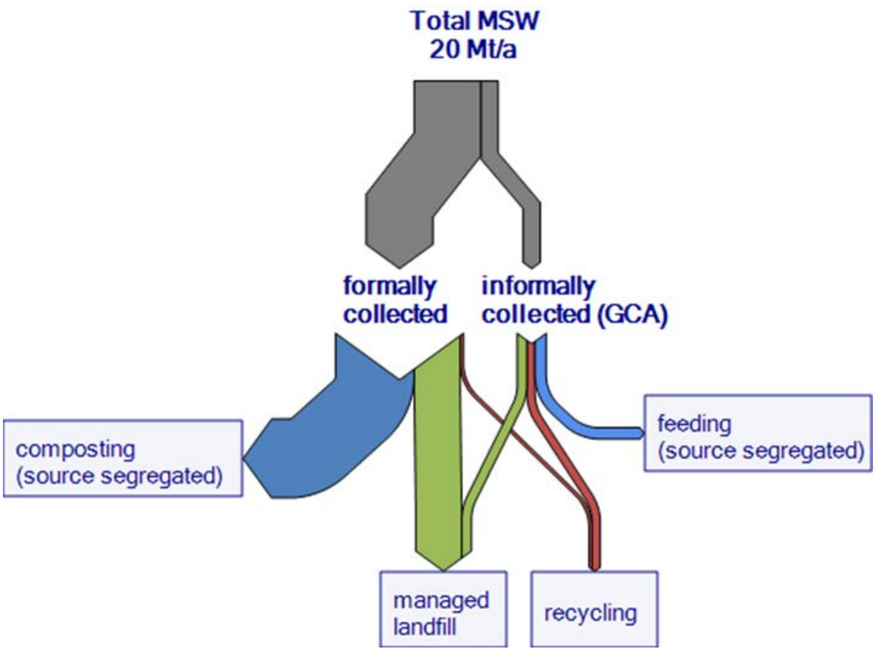
[52] The emission factor for electricity is also kept constant (see Section 4.1.2). However, this does not play a major part in the Egyptian inventory, because standard emission values are used e.g. for the recycling and composting emissions. The marginal electricity is relevant only in future Scenario 2 in the conversion of biogas to electricity.

Scenario 1: Segregated collection

Based on the objectives described above, Scenario 1 envisages a collection rate of 100%[53] and implementation of the objective of segregated collection. A target of 70% is set for 2030. It is also assumed that managed landfilling is fully implemented. For this landfilling the effective gas collection efficiency is assumed in the basic scenario to be 20%, while one of 40% is considered in a sensitivity analysis. The source-segregated organic matter is either fully composted or used by the informal sector as animal feed. On account of the segregated collection, the credits for animal feed described in Section 7.2.6. are awarded. However, steps should be taken to explore whether sanitisation of the organic waste should be required. Compost from the composting of source-segregated organic waste is credited in full. In this scenario, recycling rates in the formal sector are still low (2.5% of formally collected waste). In the informal sector they remain unchanged at over 90% for each recyclable. The additional volume of waste collected (46%) is divided between the formal and informal sectors in equal proportions. The resulting waste streams and the corresponding disposal and recycling methods are shown in Figure 61.

Figure 61: Waste streams in the various disposal methods in Scenario 1



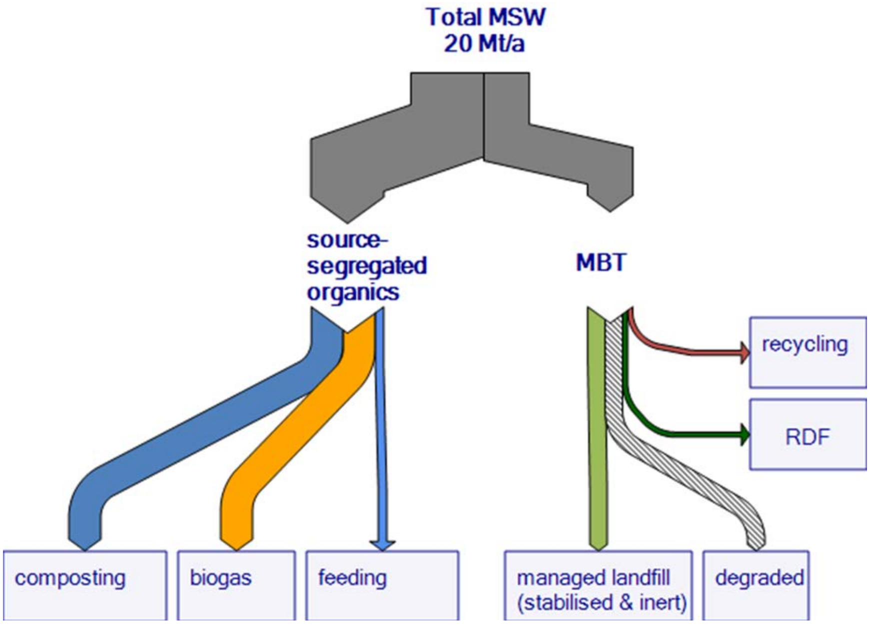
Scenario 2: Segregated collection & technology

Scenario 2 no longer distinguishes between informal and formal collection. The baseline case assumes that the composition of the overall volume of waste is as in (I+U/GTZ 2006). As in Scenario 1, 70% of the organic waste is collected separately. The amount fed to animals is the same as in Scenario 1. Of the remaining source-segregated organic matter, 50% is composted

[53] The direct use of organic matter on the land remains of course a viable alternative. The quantities used in this way are not then classed as part of the waste management system: this has the end result of reducing the volume of waste.

and 50% is used in biogas plants. All the residual waste is treated by MBT. It is assumed that in this MBT 60% of each recyclable fraction is separated out and sent to recycling. 90% of the combustible recyclables (paper & cardboard, plastics, textiles) remaining in the sorting residue are channelled into an RDF fraction. This RDF fraction is enriched with 44% organic matter. The remaining waste is biologically stabilised and, like the inert fraction, is sent to landfill ("managed landfill" in the description of the results). The corresponding emission factors are described in Section 7.2.6. The resulting distribution between the disposal and recycling methods is shown in Figure 62.

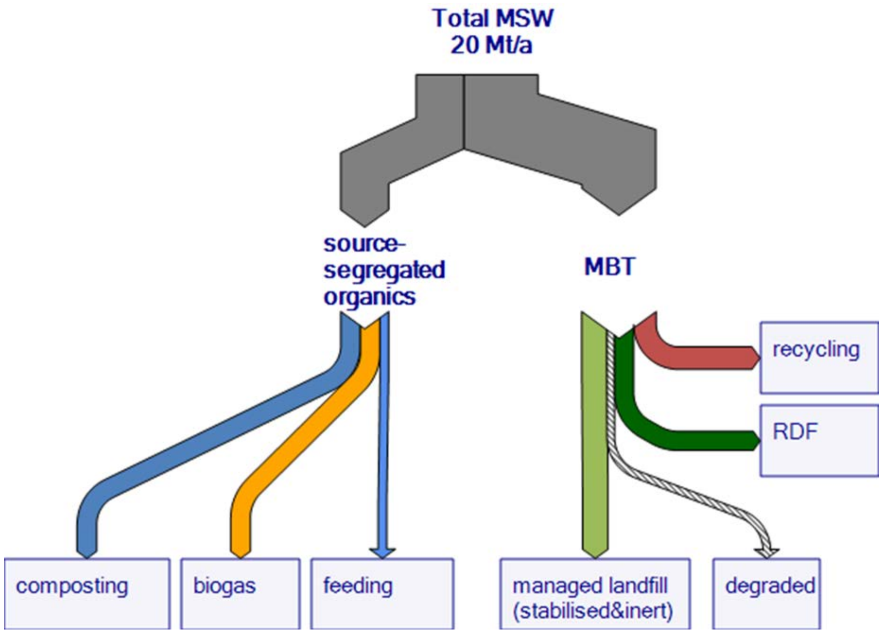
Figure 62: Waste streams in the different disposal methods in Scenario 2 – composition according to GIZ/I+U



Sensitivity analysis – Scenario 2

The sensitivity analysis in connection with Scenario 2 uses the waste composition according to (Sweep-Net 2012) rather than that according to (I+U/GTZ 2006). This means that the recyclables currently collected by waste pickers are included in the inventory (for further explanation see Sections 7.1.2 and 7.2.1). Figure 63 shows that as a result of the smaller organic fraction (70% of which continues to be segregated at source), the emphasis shifts from the recycling of organic matter to MBT. Because of the higher percentages of recyclables in the waste, a significantly larger stream (of recyclables) is now recycled. In addition, the amount of RDF that can be produced increases, because the increased recyclable content (with constant recycling rates) means that more organic matter can be added to the RDF fraction (up to 44%). More is landfilled; as a result of the pre-treatment of the organic fraction (stabilisation) the degradation losses are smaller.

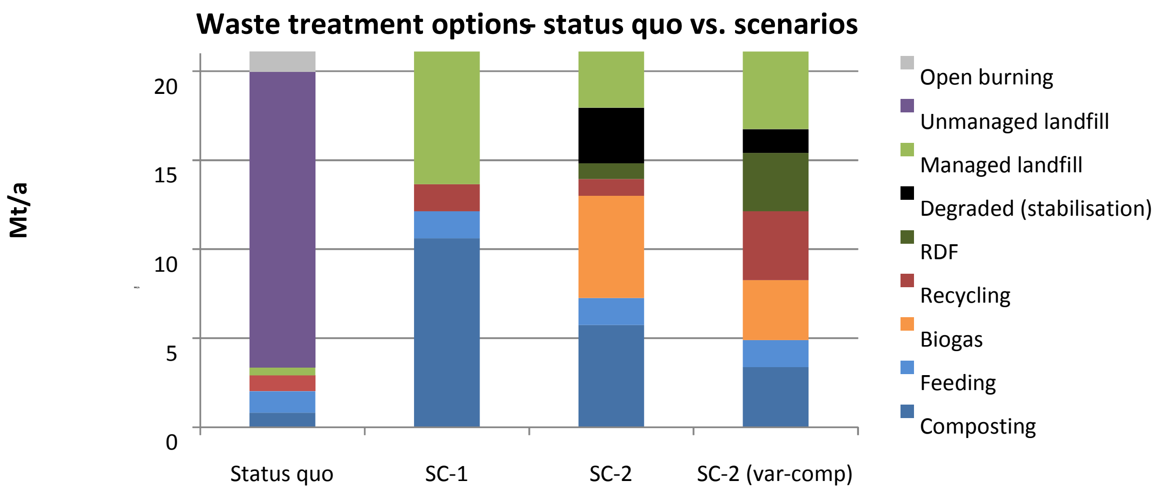
Figure 63: Waste streams in the different disposal methods in Scenario 2 – composition according to Sweep-Net 2012



7.4.2 Results

Figure 64 shows the disposal methods in the status quo and in the future scenarios. Most of the fraction sent to unmanaged landfill in the status quo disappears in the future scenarios, being replaced by recycling of source-segregated organic matter and managed landfilling, and in Scenario 2 by landfilling of the MBT residue ("degraded (stabilisation)") and the inert fraction ("managed landfill"). Recycling also plays a part, as does the production of RDF in Scenario 2.

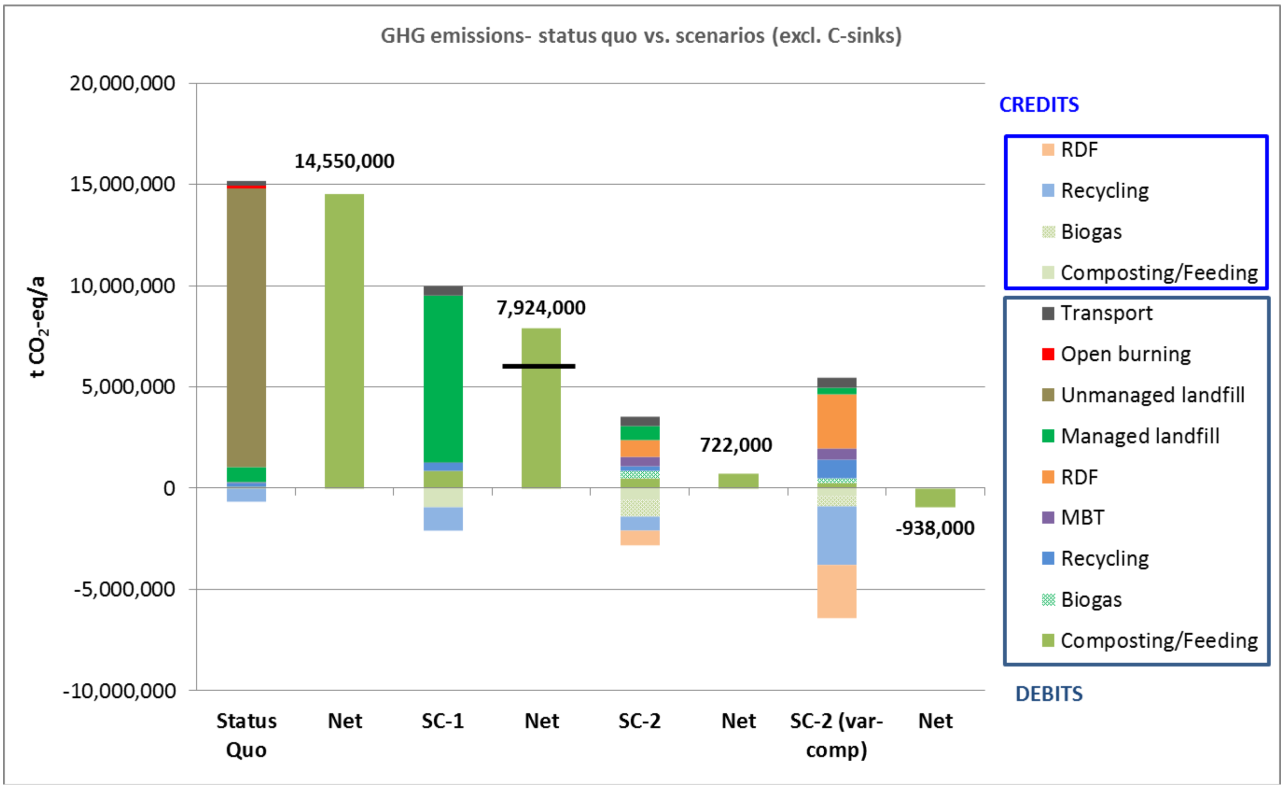
Figure 64: Waste treatment: status quo and future scenarios to 2030



In the comparison of the baseline case and the sensitivity analysis ("var-comp") in Scenario 2, the effect of the significantly smaller organic fraction described above is clear. This effect is also evident in the comparison of the source-segregated organic volumes in Scenarios 1 and 2: because the composition of the informally collected amount in Scenario 1 is as described in ((Sweep-Net 2012), organic fraction 56%) but the composition of all the collected waste in Scenario 2 is as in ((I+U/GTZ 2006), organic fraction 88%), Scenario 1 yields a slightly lower source-segregated organic quantity (for a segregated collection rate of 70% in both cases). For the same reason, recycling is higher in Scenario 1 (larger recyclable fraction (and greater recycling efficiency) in the informal sector).

The results in terms of the GHG inventory are compared in Figure 65. GHG emissions are already significantly reduced in Scenario 1 as a result of the segregated collection and recycling of 70% of the organic matter. In addition there are credits for composting/feeding and slightly increased recycling as a result of complete collection. However, despite the gas collection efficiency of 20%, the specific emissions from managed landfilling are higher than those from unmanaged landfilling. This is a result of the full methane formation potential (MCF = 1). The effect of a gas collection efficiency of 40% is shown in Figure 65 by the black line across the net bar of Scenario 1: this reduces total net emissions by around 25%.

Figure 65: Result for GWP: status quo versus future scenarios to 2030 including sensitivity SC 2

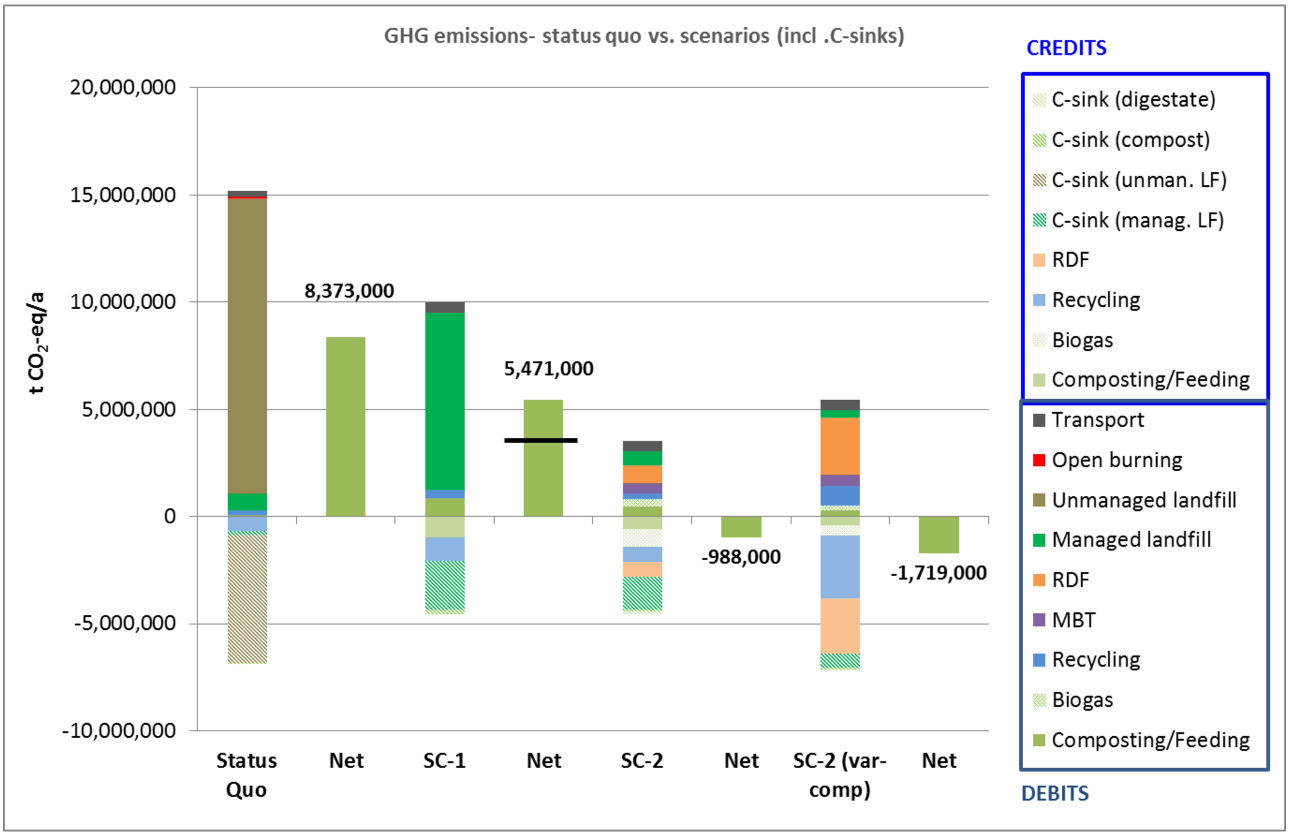


As described in Section 7.2.5 for the status quo inventory, in Scenario 1 the emissions for formal recycling of materials from residual waste are estimated indicatively using a standard emission factor for MBT according to (Öko-Institut/IFEU 2010). At approx. 0.01 Mt CO_2-eq/a the contribution is negligible.

In Scenario 2 the net result is under 1 Mt CO_2-eq/a. In the associated sensitivity analysis based on a different composition of the waste (lower organic but higher recyclable content) a net credit is achieved. This is mainly attributable to the credits for biogas use and recycling. The production of RDF does not result in any net credit: this is because of the direct fossil CO_2 emissions from the incineration of plastics and textiles. The emissions from managed landfilling (MBT residue and inert waste) are significantly reduced by the biological stabilisation of the organic fraction.

Figure 66 contrasts the GHG emissions in the various scenarios taking account of the C sink.

Figure 66: Sensitivity analysis – GHG emissions in the status quo versus future scenarios to 2030 with C sink



The net results are significantly reduced when the C sink is included, so that a net credit is achieved even in the baseline case of Scenario 2 (higher contribution of the C sink from landfilling by comparison with the sensitivity analysis on account of the larger organic fraction).

7.5 Conclusions – Egypt

The GHG inventory of Egypt's current waste management system shows a net debit. The analysis shows clearly that the unmanaged dumping of uncollected waste and the landfilling of formally collected waste account for the largest proportion of GHG debits. Even the inclusion of a C sink does not turn the net result into a credit in the status quo. The recycling of informally collected waste is the only item that contributes to a GHG credit. It should be borne in mind at this point that GHG reduction effected by the informal sector usually involves work carried out in conditions that are harmful to health.

The future scenarios show that comprehensive waste collection with segregated collection and managed landfilling improves the results of the GHG balance. However, in itself managed landfilling results initially in increased GHG debits, because the specific GHG emissions from managed landfilling are higher than the emissions assumed for shallow unmanaged landfilling in Egypt (according to IPCC lower methane formation potential). For this reason, segregated waste collection is essential to improving the GHG balance, because the source-segregated organic matter can be composted and is not directly landfilled. Mixed-waste composting is not an acceptable solution, because the resulting compost usually contains contaminants.

The future scenarios also show that GHG emissions can be reduced not only by improved waste collection by also by improvements to treatment technology (especially biogas production). Depending on the assumed composition of the waste, this may result in a net credit.

To improve the data situation – which would ultimately result in a more reliable estimate of GHG reduction potentials and an improved basis for making decisions on waste planning and intervention measures – more accurate knowledge of the composition of the waste generated throughout Egypt is desirable.

The results show that it will in future be necessary to have a waste management law that sets out binding national targets and strategies for the management of MSW and controls their implementation. In particular, responsibilities for the collection and treatment of waste and objectives and measures for country-wide segregated waste collection should be enforced. The informal sector should be taken into account and where possible included in implementation of the measures.

8 Estimating CO_2 abatement costs

CO_2 abatement costs indicate the specific costs of reducing GHG emissions. They are calculated from the total costs of the measures involved (less the proceeds from the production of secondary raw materials or energy[54]) divided by the quantity of avoided greenhouse gases. This section provides a rough calculation of the CO_2 abatement costs of the various waste management measures considered likely to be useful in Egypt and India.

8.1 Costs and revenues of waste management measures

The costs of waste management measures are taken in simplified form from (Pfaff-Simoneit 2012) for developing countries and emerging economies; they vary in line with gross domestic product (GDP) (Table 56). The specific costs used here are "full costs", which cover all costs including capital costs for a particular period in relation to the quantity handled during that period. They are thus significantly influenced by the level of capacity utilisation at the processing plants: the higher the proportion of fixed costs, the faster specific costs rise as capacity utilisation falls.

Table 56: Specific full costs of waste management measures as a function of GDP (Pfaff-Simoneit 2012)

GDP (EUR / cap / year)	< 2,000	2,000 – 4,000	4,000 – 6,000	25,000 – 30,000
Process	**(EUR/t)**	**(EUR/t)**	**(EUR/t)**	**(EUR/t)**
Collection and transport	30-40	35-45	35-45	
Separation of dry recyclables	20-30	25-35	35-45	60-70
Separation of fractions of high calorific value	15-25	20-30	25-35	50-60
Composting or organic waste	20-30	20-30	20-30	35-50
Intensive rotting / anaerobic digestion of organic waste	50-60	50-60	50-60	70-90
Simple MBT*	15-25	20-30	20-30	35-50
MBT – intensive rotting and anaerobic digestion*	40-50	40-50	45-55	75-90
MBS / MPS*	40-50	40-50	45-55	65-80
RDF-CHP*	60-80	60-80	65-85	90-120
Thermal waste treatment*	70-90	70-90	75-95	110-140
Managed landfill	10-20	12-22	15-25	40-60

*without costs of residue disposal

On the basis of the costs of the various recycling processes in industrialised countries, (Pfaff-Simoneit 2012) estimated the costs of these processes under the economic conditions prevailing in developing countries and emerging economies. The present study used the relevant means for calculation purposes.

[54] The proceeds of GHG certificates are not included here!

The revenue achievable from recycling was also taken from (Pfaff-Simoneit 2012). The calculations are based on the revenue assumed for a baseline scenario (Table 57).

Table 57: Revenue from waste treatment processes (Pfaff-Simoneit 2012)

Process	Revenue
Recyclables, source-segregated	70 €/t
Recyclables, sorted	50 €/t
Compost, digestate from organic waste	10 €/t
RDF (depending on calorific value)	0 €/MWh
Electricity/heat	25 €/MWh

8.2 CO_2 abatement costs - Egypt

For the first future scenario (SC 1) in Egypt, it is assumed that the following improvements are made to the status quo (see Section 7.4):

- All waste is collected and treated, although a distinction is made between formal and informal collection. Only formally collected waste is included in the calculation of abatement costs: no costs are calculated for informal collection. The formally collected quantity increases by 8.2 Mt.
- An additional 8.9 Mt of biogenic waste is composted.
- An additional 0.3 Mt of waste is fed to animals.
- The recycling volume increases by 0.6 Mt.
- Waste is no longer deposited in unmanaged landfills. The quantity of waste sent to managed landfill increases by 7 Mt.
- Waste is no longer burned in the open.

In the second future scenario (SC 2) for Egypt, the following additional improvements to the status quo are considered:

- All waste is collected formally. The formally collected quantity increases by 12.1 Mt.
- An additional 4.9 Mt of biogenic waste is composted.
- An additional 0.3 Mt of waste is fed to animals.
- The remaining waste (13.8 Mt) is treated by means of simple MBT.

Feeding, unmanaged landfill and open burning incur neither costs nor revenue. For these recycling methods it is only emissions that go into the calculation. For the RDF output from MBT it is assumed that incinerators already exist and that no additional capacity needs to be built. Egypt has a GDP of around US$ 3,314 (approx. €2,469) per person (World Bank 2013). In terms of the costs of waste management processes it thus falls into the category *"GDP 2,000-4,000 €/cap/a"*. Table 58 shows the CO_2 abatement costs for the total waste management system in Egypt (see also Section 11.4).

Table 58: Summary of CO_2 abatement costs – Egypt

	Baseline	SC 1	Diff. from baseline	SC 2	Diff. from baseline
Total waste, in Mt	21.1	21.1	0	21.1	0
Total costs, in million €	363	944	+581	1,491	+1,128
Total emissions, in 1,000 t CO_2-eq	14,498	7,835	-6,667	-988	-15,486
Costs per t waste, in €/t	**17**	**44**		**71**	
Abatement costs, in €/t CO_2-eq			**87**		**73**

For Egypt the estimate yields abatement costs that are lower for consistent implementation of SC 2 than they are in SC 1. This is a result of the significant CO_2 savings, which in SC 1 are only half the size of those in SC 2. The total costs per tonne of waste increase by a factor of around four, from 17 €/t in the baseline situation to 71 €/t in SC 2.

8.3 CO_2 abatement costs – India

For the medium future scenario to 2030 in India, it is assumed that the following improvements are made to the status quo (see Section 6.4):

1. All waste is collected (informally and formally). The formally collected quantity increases by 4 Mt.
2. Waste is no longer deposited in unmanaged landfills. Instead, managed landfills are set up in which 19 Mt of waste is deposited.
3. The quantity of waste treated by simple MBT increases by 17 Mt.
4. No open landfill fires.

The ideal future scenario to 2030 in India includes the following measures:

5. All waste is collected (informally and formally). The formally collected quantity increases by 4 Mt.
6. No waste is landfilled.
7. 50% of formally collected waste goes to MBS and 50% to MSWI (19 Mt in each case).
8. No open landfill fires.

Informal collection, unmanaged landfill and open burning in landfills incur neither costs nor revenue. For these recycling methods it is only emissions that go into the calculation. For the RDF output from MBT facilities it is assumed that incinerators already exist and that no additional capacity needs to be built. India has a GDP of around US$1,499 per person (approx. €1,117) (World Bank 2013). In terms of the costs of waste management processes it thus falls into the category “GDP <2,000 €/cap/a”. Table 59 shows the CO2 abatement costs for the total waste management system in India (see also Section 11.4).

Table 59: Summary of CO_2 abatement costs - India

	Baseline	"medium"	Diff. from baseline	"ideal"	Diff. from baseline
Total waste, in Mt	42	42	0	42	0
Total costs, in million €	1,212	1,976	+764	3,200	+1,988
Total emissions, in 1,000 t CO2-eq	9,242	6,361	-2,881	-6,021	-15,263
Costs per t waste, in €/t	**29**	**47**		**76**	
Abatement costs, in €/t CO_2-eq			**265**		**130**

The increase in costs per tonne of waste is roughly similar to the values calculated for Egypt. However, in relation to this the CO_2 savings achieved are significantly lower, which results in significantly higher CO_2 abatement costs. This can be attributed to the relevant boundary conditions of unmanaged dumping. In India this dumping is always shallow, while in Egypt only half is shallow. The shift to managed landfill thus results in higher additional emissions in India than it does in Egypt.

9 Opportunities to use emission certificates or other climate funds

9.1 Introduction

Projects in the waste sector are eligible for financing through the international carbon market, provided that they contribute to GHG mitigation. The size of the payment stream depends on the volume of GHG emissions saved by the project. Other environmental effects (such as protection of groundwater), while desirable in principle, are not eligible for financing through the carbon market.

To date the only functioning market for climate change mitigation projects in developing countries and emerging economies is the Clean Development Mechanism (CDM). However, this provides numerous opportunities for the development of climate change mitigation projects in the waste sector.

CDM methods have so far focused mainly on methane reduction, for example in landfills or in connection with sewage treatment. In addition, the use of waste biomass for energy generation can be registered as a CDM project. Schemes that involve recycling have so far played only a minor role (see corresponding calculation methods in Section 9.3). The introduction of Programmes of Activities (PoAs) under the CDM has made it possible to combine a number of climate change mitigation activities in a single programme. This enables the CDM to address larger areas within a sector, especially since the requirements relating to the approval and registration of measures are significantly simpler than the CDM requirements for individual projects. A number of PoAs in the areas of landfill gas, biogas and waste biomass are already in the course of registration or validation.[55] There is also a composting PoA.[56]

Another option is to develop such projects as Nationally Appropriate Mitigation Actions (NAMAs) or as part of other market-based mechanisms that are currently being negotiated as an aspect of the international climate change mitigation process.

NAMAs are mitigation measures in developing countries that are planned under the umbrella of a national government initiative. They may be directed at individual sectors or apply across a number of sectors. NAMAs can be supported by technology, financing and capacity-building; they are aimed at achieving a reduction in emissions relative to “business as usual” emissions in 2020.[57] For NAMAs – in contrast to the CDM – there is no established market mechanism with fixed rules via which financing can be ensured. However, developing countries can enter details of NAMAs for which support is needed in the NAMA register,[58] specifying the type, scope and cost of this support. There is, though, no entitlement to support. The NAMA register

[55] https://cdm.unfccc.int/ProgrammeOfActivities/index.html

[56] AeroPod Composting and Co-composting Programme in Malaysia:
https://cdm.unfccc.int/ProgrammeOfActivities/poa_db/HUYOG75D29NF8AT43BXK1LJQ6MPV0W/view

[57] http://unfccc.int/focus/mitigation/items/7172.php

[58] NAMA Registry: http://unfccc.int/cooperation_support/nama/items/7476.php

already contains NAMAs in the waste sector.[59] There are a number of initiatives that provide support to NAMAs, such as the EU's Low Emission Capacity Building Programme.[60] Germany and the United Kingdom have so far contributed 120 million euros to the NAMA Facility for the implementation of NAMAs.[61] Of around 50 project outlines so far submitted, 8% involve NAMAs in the waste/sanitation sector.

Another financing opportunity is the Green Climate Fund, although its funding conditions have not yet been established.[62] The International Climate Initiative[63] of the German Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety (BMUB) also funds climate projects in developing countries and emerging economies and has already funded projects in the waste sector (e.g. in Chile and China).

9.2 The carbon market

The carbon market has been largely shaped by the EU Emissions Trading Scheme (EU ETS), which is the largest source of demand. However, an oversupply of certificates has meant that the price of CO_2 from CDM projects has fallen sharply and is currently well below 0.50 €/CER.[64] In addition, since 2013 CDM certificates from newly registered projects may only be used in the EU ETS if they come from least developed countries (LDCs). Egypt and India are not classed as LDCs,[65] which means that certificates generated by new projects in these countries cannot be sold through the EU ETS.

The general conclusion to be drawn from this is that the attractiveness and cost-effectiveness of CDM projects based on the carbon market is currently severely restricted. It is not at present possible to foresee when this situation is likely to improve significantly. The EU ETS has decided that a certain number of certificates should be temporarily withdrawn from the system, but

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[59] E.g. a landfill gas NAMA in Jordan: http://www4.unfccc.int/sites/nama/_layouts/un/fccc/nama/NamaSeekingSupportForPreparation.aspx?ID=14&viewOnly=1

[60] http://www.lowemissiondevelopment.org/. For example, through this programme Egypt is being helped to formulate NAMAs in the sectors of energy and transport.

[61] http://www.nama-facility.org

[62] http://news.gcfund.org/

[63] http://www.international-climate-initiative.com/de/

[64] Certified Emission Reduction, the emission reduction unit used in the CDM (1 tonne CO_2e). On 23 August 2014 the CER price was 0.17 €/CER (http://www.pointcarbon.com/), which is significantly below the price of EU emission allowances (6.38 €/EUA on 23 August 2014). If CDM and EU ETS certificates are to be fully interchangeable, their prices should converge. In practice, though, there are a number of reasons why the price of CERs is lower than that of EUAs. For example, in terms of absolute quantities the use of CERs in the EU ETS is restricted. In addition, since the start of 2013 certificates from industrial gas projects have been banned in the EU ETS. And in the EU ETS there are restrictions on certificates from non-LDC countries (http://ec.europa.eu/clima/policies/ets/linking/faq_en.htm).

[65] http://www.un.org/esa/policy/devplan/profile/ldc_list.pdf.

because they will be returned to the system at a later date this is unlikely to result in any significant increase in CO_2 prices. If further ambitious climate targets are adopted at the climate conference in Paris in 2015 and by the EU, this could cause prices to rise but it does not at present appear likely to happen. Despite this, new CDM projects – some of them financed by the voluntary market – are being validated and registered.

For NAMAs there is no established market outside the NAMA register (see above).

9.3 CDM methodologies

In connection with the CDM there are a number of methodologies in the waste sector that can be directly used in the development of a climate change mitigation project. In principle it is also possible to develop a new methodology, but this is usually both time-consuming and expensive.

The methodologies listed in Table 60 relate to both small-scale (AMS) and large-scale projects (AM, ACM). The boundary between small-scale and large-scale projects is based on the total CO_2 saving or the installed capacity (e.g. for energy components).

Table 60: CDM methodologies in the waste sector

Sectoral scope	Renewable energy	Energy Efficiency	GHG destruction	GHG emission avoidance
13 Waste handling and disposal	ACM0022	AMS-III.AJ. AMS-III.BA.	AM0073 ACM0001 ACM0010 ACM0014 AMS-III.G. AMS-III.H. AMS-III.AX.	AM0057 AM0080 AM0083 AM0093 ACM0022 AMS-III.E. AMS-III.F. AMS-III.I. AMS-III.Y. AMS-III.AF. AMS-III.AO.

Source: CDM Methodology Booklet, Version November 2012, https://cdm.unfccc.int/methodologies/documentation/index.html

The small-scale projects involve the following project types:

- AMS-III.E: Avoidance of methane production from decay of biomass through controlled combustion, gasification or mechanical/thermal treatment
- AMS-III.F: Avoidance of methane emissions through composting
- AMS-III.G: Landfill methane recovery
- AMS-III.H: Methane recovery in wastewater treatment
- AMS-III.I: Avoidance of methane production in wastewater treatment through replacement of anaerobic systems by aerobic systems
- AMS-III.Y: Methane avoidance through separation of solids from wastewater or manure treatment systems

- AMS-III.AF: Avoidance of methane emissions through excavating and composting of partially decayed municipal solid waste (MSW)
- AMS-III.AJ: Recovery and recycling of materials from solid wastes
- AMS-III.AO: Methane recovery through controlled anaerobic digestion
- AMS-III.AX: Methane oxidation layer (MOL) for solid waste disposal sites
- AMS-III.BA: Recovery and recycling of materials from E-waste

The large-scale projects fall into the following categories:

- ACM0001: Flaring or use of landfill gas
- ACM0010: Consolidated baseline methodology for GHG emission reductions from manure management systems
- ACM0014: Treatment of wastewater
- ACM0022: Alternative waste treatment processes
- AM0057: Avoided emissions from biomass wastes through use as feed stock in pulp and paper, cardboard, fibreboard or bio-oil production
- AM0073: GHG emission reductions through multi-site manure collection and treatment in a central plant
- AM0080: Mitigation of greenhouse gases emissions with treatment of wastewater in aerobic wastewater treatment plants
- AM0083: Avoidance of landfill gas emissions by in-situ aeration of landfills
- AM0093: Avoidance of landfill gas emissions by passive aeration of landfills

Details of the methodologies and of projects that have already been registered can be downloaded from the CDM website:

- Methodologies: https://cdm.unfccc.int/methodologies/index.html
- Projects: https://cdm.unfccc.int/Projects/projsearch.html

9.4 Conclusions for India and Egypt

The above comments show that India and Egypt currently have no opportunity to participate in the regular CDM market (not classed as LCDs). In any case, the present low market prices would not make any relevant contribution to the costs of waste management in India and Egypt.

The various voluntary and as yet informal mechanisms, in particular NAMAs, have no exclusion criteria. The chances of obtaining funding via these measures are difficult to assess in advance: they depend on how convincingly the schemes are presented and on the contributions to climate change mitigation that the schemes can achieve.

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11 Annex

11.1 Derivation of harmonised emission factors for dry recyclables

In line with the recommendations that emerged from the methodology workshop on 18 June 2012 (see footnote 5), it was agreed that this study would where possible use **harmonised emission factors** (default values) for substituted processes(substitution processes and/or credits). The environmental impacts associated with substitution processes usually have a major influence on the result and are in addition beset by data uncertainties. For example, it was reported for the USA that it is very difficult to obtain reliable data on paper production.[66] In all areas the information that is available is sometimes limited; the decision on what primary processes are substituted by secondary products must then be based on assumptions. For example, there is usually no statistical data on the actual use of secondary granulate from plastic waste. For Germany a reliable estimate of the substitution potential of various plastic fractions is possible as a result of numerous studies and discussions with recyclers. Even for other EU countries the availability of such information is limited and for the majority of other OECD countries there is no corresponding information that we are aware of. The OECD study (OECD 2012) used the emission factors from (Prognos et al. 2008) for the EU in its evaluation of the recovery of dry recyclables. The data situation is even more difficult for developing countries and emerging economies, where details of waste volumes and management methods are often based on estimates. For Egypt and India it was only possible to obtain qualitative statements on the uses of secondary granulate in connection with plastic recycling. For purposes of the study, assumptions were therefore made on the basis of these statements.

In the face of these data uncertainties and the major influence of the data on the outcome of substitution processes, harmonised emission factors have the advantage that they enable the inventories of the various different countries to be compared. The use of default values is particularly important for the inventory of the OECD countries, as it was not possible to carry out any more comprehensive country-specific research within the scope of the project.

To obtain the harmonised emission factors, relevant figures were inspected and compared. Life cycle assessments of the waste management sector have been conducted involving various countries. In addition, there are various calculation tools for assessing emissions in the waste management sector that contain emission factors or calculation values. However, not all these tools are publicly available and reports of studies – insofar as they have been published – do not always contain comprehensive documentation of the underlying data used in the assessment.

The first inspection and comparison of data from studies was carried out for the methodology workshop held on 18 June 2012. This involved evaluating four studies of particular relevance to this project, with a particular emphasis on defining substitution processes. These four studies were the OECD study (OECD 2012), the EEA study (EEA 2011), the precursor study to this project (Öko-Institut/IFEU 2010) and a study by Prognos et al. (2008) for which the IFEU Heidelberg provided specific emissions data. These studies were supplemented by data from other

[66] A further problem is that transparency is sometimes limited, because e.g. the WARM tool includes an allowance for carbon sequestration in connection with paper recycling.

available sources. Tables 61 and 62 summarise the relevant figures. The minus signs against net emission factors indicate net credits (debits < credits).

The figures used in (OECD 2012) are not shown in the tables, because as already mentioned that study used the IFEU emission factors from (Prognos et al. 2008). The tables also include emission factors from calculation tools developed at IFEU (IFEU 2011, 2010), from the USEPA study (2006) and from various European institutions, which are combined in the calculation tool for the epe protocol (epe 2008, 2010). The tables also show the net emission factor ranges identified in the ISWA (2009) White Paper.

Table 61: Comparison of studies of emission factors for metal, paper & cardboard and glass recycling in kg CO_2-eq/t waste

Recyclable	(Fraunhofer 2008) for 2007	(Öko-Institut/IFEU 2010) for 2006	(Prognos et al. 2008) for 2004	(IFEU 2011, 2010) Tools	ETC/ SCP for (EEA 2011)	(BIR 2008) international	(AEA Technology 2001) for EU25	(US EPA 2006)	(ADEME 2007) for FR	(Senter Novem 2008) for NL	(CE Delft 2007) for NL	(ISWA 2009) White Paper
Iron	-856	-945	-1,000	-2,025	Metals -3,220 Debit 890 Credit 4110	-970	-1,487	-1,970	-1,600			-2,000
Debit	682	338		22		700						
Credit	1,538	1,284		2,047		1,670						
Aluminium	-9,872	-9,307	-11,100			-3,540	-9,047	-14,960	-7,100			-10,000
Debit	730	406	700			290						
Credit	10,602	9,713	11,800			3,830						
Copper	-3,522		-1,180			-810		-5,420	-1,130			
Debit	1,978		1,690			440						
Credit	5,500		2,870			1,250						
Paper/ cardboard	-94	-674 (-732)	-820		-564	-0,3	-600	-3,900	-370	-2	-1,296	-600 to -2,500
Debit	76	209	180		116	1,4						
Credit	171	883	1,000		680	1,7						
Glass	-170	-465	-180	-480	-159		-253	-310	-460	-324	-321	-500
Debit	30	33	20	20	21							
Credit	200	498	200	500	180							
Textiles			-2,818		-1,728	-3,169				-3,432	-2,919	
Debit			32		232							
Credit			2,850		1,960							

It is frequently the case that the studies do not report the emission factors for the debits (direct emissions) and credits (avoided emissions) on which the net emission factors are based: this makes interpretation difficult. The discussion and derivation of standard emission factors for the various waste fractions is described below.

Metals

In the present study metals are divided only into ferrous (Fe) and non-ferrous (non-Fe) metals, since information is not normally available for any other fractions. Past experience in Germany shows that the non-Fe metal entering the waste stream is predominantly aluminium. As in the previous study, non-Fe metals are therefore assessed using the values for aluminium.

The summary shows that the net values for iron range from around -850 to -2,000 kg CO_2-eq/t scrap iron; for aluminium the range is from around -3,500 to -15,000 kg CO_2-eq/t aluminium scrap. The differences are probably largely due to variations in the assumed degree of purity of the metal. This cannot be explored in all cases. However, it is stated for the USEPA figures that they apply to cans. Because aluminium cans have a high level of purity, they can be recycled in an aluminium smelting facility. Similar considerations apply to the recycling of scrap iron. For example, tin cans are used only in oxygen steelmaking, while scrap iron is normally recycled in an electric arc furnace. The assumptions made (e.g. whether pig iron or steel is substituted) can affect the results of the analysis.

This project uses the emission factors from the previous study (Öko-Institut/IFEU 2010) as standard emission factors for metal recycling. These emission factors are based on data in the Ecoinvent database. The values lie roughly in the middle part of the ranges detailed above; the net value for iron is close to the value quoted by the Federal Association of German Steel Recycling and Disposal Companies (BDSV) for (Prognos et al. 2008).

The standard net emission factors for metal recycling are therefore:

Iron:	-945 kg CO_2-eq/t scrap $\text{iron}_{\text{input to steelworks}}$
Non-Fe metals	-9,307 kg CO_2-eq/t non-Fe $\text{scrap}_{\text{input to metal smelting/pyrolysis}}$

The values involve only the debits and credits from the recycling process; they relate to the quantities of metal sent to the steelworks or to the aluminium smelter / pyrolysis.

Paper and cardboard

In connection with the recycling of paper and cardboard it was agreed that the wood saving[67] considered for the first time in the previous project (Öko-Institut/IFEU 2010) would be included only in a sensitivity analysis because the market situation and the demand for wood, and hence the pressure on the wood supply, cannot be reliably estimated. In addition, in relation to the recycling of paper and cardboard it is not immediately clear what the substitution potential is or what substitution process should be selected. Paper, card and cardboard packaging are made from mechanical or chemical pulp, either in an integrated paper production process or using imported pulp that is then processed.

According to (FAO 2010), most (>90%) of the pulp manufactured worldwide is chemical pulp, and of this chemical pulp the majority (around 95%) is bleached sulphate pulp. Small quantities of pulp (< 1%) are also produced from straw, bagasse and bamboo. Of the mechanical pump, the majority (around 80%) is thermo-mechanical pulp (TMP, at temperatures of up to 140°C).

Mechanical pulp is produced from wood with a yield of 90%; it contains a high percentage of lignin. The lignin makes the pulp more suitable for the manufacture of cardboard boxes, because it provides additional rigidity. However, it also causes yellowing of the paper, which means that mechanical pulp is used only for paper that has a short life, such as newsprint and

[67] In Table 61 the figures in brackets show the result including wood saving in the baseline case "Use of saved wood for energy generation in Sweden".

magazine paper. Mechanical pulp is simpler and usually cheaper to produce than chemical pulp, but a great deal of electricity is needed for the grinding involved (1500-3500 kWh/t wood). As a result, newsprint and cardboard packaging is often produced from low-cost recycled paper fibre instead of mechanical pulp. Within Europe, it is only in Scandinavia that TMP is still made into newsprint.

Chemical pulp is usually also made from wood: in global terms the use of annual plants such as straw, bagasse and bamboo is small (see above). The wood is first turned into wood chips and then broken down chemically, usually with sodium sulphate (sulphate pulp, see above). At temperatures of up to 170°C hemicelluloses and lignin are dissolved in the alkaline process and removed. Unbleached chemical pulp still contains 2-3% lignin; around 50% of the cellulose is removed. The residue – the liquor from the separation process – is usually treated as waste. In modern chemical pulp plants the chemicals are recovered from the waste liquor by vaporising it and combusting the resulting black liquor in a low-oxygen environment. In subsequent stages the chemicals are recovered from the smelt and returned to the process. The energy produced as a result of the combustion supplies the process energy needed to manufacture the chemical pulp. Modern plants produce a surplus of electricity. In integrated paper production the excess steam can be used for paper drying.

The assessment of paper recycling raises the following questions:

1. In what proportions are mechanical pulp and chemical pulp substituted?
2. Is paper from integrated paper production substituted?
3. If it is, are the facilities involved modern ones that use the liquor for energy?

Different substitution processes are used in life cycle assessments depending on the answers to these questions. In Table 61 the net values for paper and cardboard recycling range from −0.3 to −3,900 kg CO_2-eq/t paper and cardboard waste. The values at the lower end of the range probably reflect the assumption that modern integrated paper production processes are involved that use both waste paper and primary fibre and in consequence generate virtually no GHG emissions either from recycling or from primary processes. The high value in (USEPA 2006) is at the top of the range of net emission factors based on paper type and origin, which range from −2,930 for telephone directories to −3,900 kg CO_2-eq/t paper & cardboard for mixed cardboard packaging and mixed waste paper. The figures are taken from (epe 2010).[68] It is stated in (epe 2010) that the figures include an allowance for carbon sequestration ("forest carbon sequestration") but the size of this allowance is not given. For the other emission factors listed in the table it can be assumed or taken as given that the emission factors relate to substitution at fibre level. Differences arise here depending on the percentages of mechanical pulp and chemical pulp that are substituted. For example, the figure from (Fraunhofer 2008) in (Fraunhofer 2009) was changed to a net emission factor of −440 kg CO_2-eq/t paper & cardboard in which only the substitution of sulphate and sulphite chemical pulp is taken into account. The figures in (Öko-Institut/IFEU 2010) and (Prognos et al. 2008) differ in terms of credits mainly in that (Prognos et al. 2008) assume an equally distributed 50:50 substitution of mechanical pulp and chemical pulp while the previous study (Öko-Institut/IFEU 2010) is based on the market mix in Germany (43:57, see below). On the debit side the small difference is

[68] In which figures are converted into metric tons

explained mainly by the fact that transport emissions are taken into account in (Öko-Institut/IFEU 2010) while (Prognos et al. 2008) adopt a simplified approach that excludes them.

For the present study steps were taken to explore whether and how well the above questions can be answered for the countries being considered. For both India and Egypt it was established that integrated paper production takes place. However, in both cases both virgin fibre and recycled paper fibre are used. This means that it is not possible to determine clearly which environmental impacts should be assigned to the recycled paper fibre and which to the new fibre. In addition, it is not possible to establish the extent to which modern facilities are in use in either India or Egypt. It is, though, known that coal is used to generate energy for paper production in India. Information for India is available via the Indian Paper Manufacturers Association (IPMA), whose members account for more than one-third of the country's paper production. It is stated on the IPMA's website that there have been paper mills in India for a long time, so that technology standards range from very old to very modern.[69]

In view of the data uncertainties (fossil energy use) and the difficulties of clear allocation it was agreed that for paper recycling standard values would be used for paper manufacture and for the production of mechanical pulp and chemical pulp. As in the previous project, the substitution is therefore applied generally at fibre level, which results in greater transparency and more readily interpretable results. This method also neutralises the effect on the result where the focus is solely on the global warming potential of the eventuality of being able to use renewable energy (regenerative and CO_2 emissions therefore classed as climate-neutral) in integrated paper production, and it is better at taking account of the advantages of paper recycling, e.g. through land-saving.

The question of the percentages of mechanical pulp and chemical pulp that are substituted has already been discussed in the previous study. In general it can be said that the question of substitution cannot be reduced to the concrete area of use of the recycled paper. Despite the fact that relatively large percentages of recycled paper fibre are used to make cardboard packaging and newsprint, this does not mean that if there is a shortage of recycled paper primary material is used for these purposes. Instead it should then be assumed that recycled paper fibre is withdrawn from the production of copy paper etc. with the result that more primary material is used in this area. In this respect the paper market needs to be considered as a whole. For the most important segments the relevant percentages of mechanical and/or chemical pulp were estimated.[70] Using the marketing figures this yielded a new fibre input mix for Germany of 57% chemical pulp and 43% mechanical pulp. These marketing percentages were fairly stable over a period of four years.

This procedure could in principle be performed for each country, but it requires considerable effort to be put into research. Furthermore, the data are not available for all countries. The data on the global marketing split are likewise inadequate. (FAO 2010) only lists capacities: they are not broken down into magazine paper and copy paper. It is thus not possible to calculate world market shares from this information. This study therefore adopts a simplified

[69] http://www.ipma.co.in/paper_industry_overview.asp

[70] Newsprint 100% mechanical pulp, copy paper 100% chemical pulp, paper and cardboard packaging 30% mechanical pulp and 70% chemical pulp, magazine paper 50% mechanical pulp and 50% chemical pulp

approach to the standard assessment of paper and cardboard recycling and applies a substitution split of 50% mechanical pulp and 50% chemical pulp generally.

The associated emission factors were calculated in relation to the quantity usually transferred to the paper industry after sorting. Deviating from the previous study, this yields emissions for paper recycling of 167 CO_2-eq/t paper & cardboard$_{\text{input paper factory}}$. This includes the emissions of de-inking (pulper) and emissions from disposal of the processing residues. Conversely, the credit arises for the quantity of fibre produced that substitutes new fibre[71] and for the generation of energy from the processing residues (small amount). The total credit is -960 kg CO_2-eq/t paper & cardboard$_{\text{input paper factory}}$.

This yields a standard net emission factor for paper and cardboard recycling of:
Paper & cardboard: -793 kg CO_2-eq/t paper & cardboard$_{\text{input paper factory}}$
The figure includes only the debits and credits from the recycling process and relates to the quantity of paper and cardboard sent to the paper factory.

Glass

Used glass is usually used directly in the glass-melting process in glassworks. The values listed in Table 61 cover a range of net values from approx. −160 to -500 kg CO_2-eq/t waste glass. The difference between the credit values in (IFEU 2011, 2010) and (Prognos et al. 2008) can be explained by a corrected methodological approach. In older studies the substitution potential was measured using the market mix for glass cullet. A credit was awarded only for the percentage of primary material actually used in the market (approx. 30%). However, offsetting on the basis of market-based substitution potential is contraindicated for life cycle assessments in the waste management sector, because the resulting conclusion is that “the more that is recycled, the lower the allocated credit”. This was emphasised at the methodology workshop.

This project uses the emission factors from the previous study (Öko-Institut/IFEU 2010), which were calculated on the basis of technical substitution potential, as standard emission factors for glass recycling. In a departure from that earlier study, the standard emission factors take account only of the debits and credits that arise from the actual glass-melting process. This results in emissions of zero. The figure used in the previous study includes debits from sorting and transport that are accounted for separately in this study. As a result of the use of glass cullet, the glass-melting process itself leads to a reduction in the energy used in the melting process and to substitution of mineral resources (sand, soda, limestone, feldspar, dolomite) and therefore gives rise only to credits. The resource credit is responsible for 85% of the total credit; around 50% of this is the result of avoided mineral CO_2 emissions. The credit for the glass-melting process is -514 kg CO_2-eq/t glass$_{\text{input glassworks}}$; this is also the net emission factor.

The resulting standard net emission factor for glass recycling is:
Glass: -514 kg CO_2-eq/t glass$_{\text{input glassworks}}$

[71] Yield 94%; subsequently assessed with a technical substitution factor of 0.95 to take account of the fact that secondary fibre is of somewhat lower quality than primary fibre.

The figure includes only the (debits and) credits from the recycling process and relates to the quantity of glass sent to the glassworks.

Textiles

The net emission factors for textile recycling shown in the table range from approx. −700 to −3,400 kg CO_2-eq/t waste textiles. The figure used in (Prognos et al. 2008) is a rough estimate calculated by IFEU which for simplification relates only to the provision of raw material. It is assumed for the purposes of this calculation that 1/3 of waste textiles substitute cotton and that 2/3 substitute polyester, with a life shortened to 50%.

The literature contains figures for textile manufacture – e.g. in a JRC document[72] produced by a project for revising the ecolabel criteria for textiles.[73] Figure 30 of the JRC document shows emissions of roughly 15 to 35 kg CO_2-eq/kg fabric (the lower value is for silk and the upper one for acrylic). For some fibres (e.g. polyester), supplying the fibre accounts for only a small proportion of emissions; for cotton this rises to around 50%. By comparison with the figure in (Prognos et al. 2008) in which only the production of new textile materials is considered (2.85 kg CO_2-eq/kg fibre), these figures in combination with a 50% lifespan would result in significantly higher credits of 7.5 to 15 kg CO_2-eq/kg fibre). Similarly high values for textile production can also be found in the Ecoinvent database. Neither data source, though, quotes details of the underlying energy requirements, which means that it is not possible to verify the plausibility of the data.

It was agreed in general for this study that textile recycling would not be assumed to take place unless there was evidence of source-segregated collection of textiles. Reusable textiles cannot be recovered from collected residual waste. No information on source-segregated collection is available for the USA or the OECD countries. The textile recycling for the USA referred to in the statistical data from USEPA is considered in a sensitivity analysis. Doorstep collection in India represents a special case. In this traditional method of collection of recyclables by the informal sector used clothing is purchased and usually resold directly. This form of textile recycling is taken into account in the Indian inventory. In the absence of reliable data, the value in (Prognos et al. 2008) is used for simplification in the assessment. This value is also used in the calculation tool for developing countries and emerging economies developed by IFEU (IFEU 2011, 2010). In Egypt, textiles in municipal waste are used mainly to produce rag rugs and stuffing materials for mattresses and cushions (EEAA/METAP 2005). It was not possible to identify the extent to which substitution processes should be included in the calculation process for this type of recycling.

The standard net emission factor for the re-use of textiles is:

textiles: −2,818 kg CO_2-eq/t textiles

The figure includes only the debits and credits for the re-use of old clothes collected directly from households.

[72] http://susproc.jrc.ec.europa.eu/textiles/docs/120423%20IMPRO%20Textiles_Publication%20draft%20v1.pdf

[73] http://susproc.jrc.ec.europa.eu/textiles/stakeholders.html

Plastic

Table 62 shows the emission factors for the recycling of plastic and light packaging waste (LPW). Data on plastic waste are reported separately for polyethylene (PE), polypropylene (PP), polyethylene terephthalate (PET), polystyrene (PS) and polyvinyl chloride (PVC). In particular, figures for PE-PP and PET are available in a number of studies.

Table 62: Emission factors for packaging/plastic recycling in kg CO_2-eq/t waste

Recyclable	(Fraunhofer 2008) for 2007	(Öko-Institut/IFEU 2010) for 2006	(Prognos et al. 2008) für 2004	(IFEU 2011) ADM Tool	(IFEU 2010) SWM-GHG-Calc.	ETC/SCP for (EEA 2011)	(AEA Technology 2001) for EU25	(US EPA 2006)	(ADEME 2007) for FR	(Senter Novem 2008) for NL	(CE Delft 2007) for NL	(ISWA 2009) White Paper
PE (PP)	-1,194		-160				-491	-1,530/ -1,860	-2,300	-2617	-1,098	
Debit	493		1,040									
Credit	1,687		1,200									
PET	-2,538		-1,640				-1,761	-1,700	-2,700	-2,573	-1,271	
Debit	470		960									
Credit	3,008		2,600									
PS			-1,700									
Debit			1,100									
Credit			2,800									
PVC			-740									
Debit			790									
Credit			1,530									
Plastic		(-416)	-523	-160	-414	-405		-1,640				0 to +1,000
Debit			1,280	348	1,023	1,315						
Credit			1,803	508	1,437	1,720						
LPW	-778	-443										
Debit	442	971										
Credit	1,220	1,410										

The net values for **PE (PP) recycling** shown in the table range from –160 to –2,600 kg CO_2-eq/t PE(PP) waste. The two values listed under (USEPA 2006) represent HDPE and LDPE; those under (Senter Novem 2008) and (CE Delft 2007) represent HDPE. The high net credits must relate to the recycling of highly pure PE, because according to analysis of the data from PlasticsEurope the values for the manufacture of HDPE and LDPE are of the order of 2,000 kg CO_2-eq/t PE (Table 64). The net emission factors that are significantly lower than this must therefore be the result of impurities. This is the case with the value calculated by IFEU for (Prognos et al. 2008), which includes emissions from the collection, sorting and processing of PE-PP waste (mainly film, bottles) and also takes account of loss of mass through impurities (sorting and processing residues) and moisture (mainly with regard to bottles). The resulting yield of secondary granulate is just under 65%. For this 65% it is furthermore assumed that 50% of it substitutes PE and 50% substitutes PP, in both cases with a substitution factor of 0.7 – i.e. on account of its technical properties the PE-PP secondary granulate can only substitute 70% of primary material, which results in a correspondingly reduced credit. The variance arising as a result of the set or given boundary conditions is also manifested for example in the fact that (Fraunhofer 2009) gives a lower net emission factor for PE (–1,143) and PET (–1,689) than (Fraunhofer 2008). And in a sensitivity analysis for the United Kingdom, IFEU obtains a even lower net emission factor for PE (–90) for (Prognos et al. 2008) caused solely by a higher emission factor for electricity for the UK than for the EU27.

The net values for **PET recycling** shown in Table 62 range roughly from –200 to –2,700 kg CO_2-eq/t PET and thus do not cover as large a range as the values for PE(PP). Here again the values at the top of the range are likely to apply to the recycling of relatively pure PET. The debits for the primary production of PET derived from the data from PlasticsEurope are approximately –3,300 kg CO_2-eq/t PET (Table 64). Here too the lower net emission factor in (Prognos et al. 2008) can be explained by the emissions for collection, sorting and processing. The yield of pure PET after sorting and processing losses is less than 70%. However, it is assumed that the processed PET is of high quality and the technical substitution potential is therefore set at 1, meaning that processed PET can substitute 100% of primary PET.[74]

The only values available for **PS and PVC waste** are those from (Prognos et al. 2008), which arise in a similar way to the values described above for PE and PET. Both types of plastic tend not to occur in the packaging mix. For both the yield of secondary granulate is assumed to be 80% and the technical substitution factor is set at 0.9.

For all the values derived from IFEU for (Prognos et al. 2008) incineration of the sorting and processing residues in an average MSWI (with energy generation) is included.

Table 62 shows both the values according to plastic type and total values for plastic and **light packaging waste (LPW)**. The greatest variance is to be expected for LPW, because this involves the recycling of metal cans and paper and cardboard packaging as well as various types of plastic. Differences arise simply from variations in the composition of the LPW. Nevertheless, the differences in the net emission factors are relatively small. For example, (Fraunhofer 2009) by comparison with (Fraunhofer 2008) reports a net credit of 464 kg CO_2-eq/t LPW, which is even closer to the figure calculated in (Öko-Institut/IFEU 2010).

The net values for **plastic recycling** shown in the table range from approximately –160 to –1,600 kg CO_2-eq/t plastic waste. The value given under (USEPA 2006) is for mixed plastics. The emission factors calculated by IFEU for various projects are based on assumptions about the composition of plastic waste in terms of plastic types. For (Prognos et al. 2008) the distribution is based on internal information from Prognos AG. Plastic waste in the EU is made up mainly of PE-PP with smaller fractions of PET, PS and PVC and a percentage of other plastics. For developing countries and emerging economies it was estimated for the SWM-GHG Calculator for (IFEU 2010) that plastic waste generation is 80% PE and PP, 10% PET (bottles), 5% PS and 5% PVC. Because for simplification no percentage of mixed plastic was taken into account, a comparatively high net credit is obtained as the weighted mean of the composition described above and the emission factors according to plastic type (Prognos et al. 2008). In (IFEU 2011) a

[74] PET drinks bottles form an exception: in this case in Germany the degree of purity is usually achieved only via the PET cycle (closed-loop recycling of PET bottles).

different approach was adopted for India on account of the information available. Here it was assumed that after formal collection only mixed plastics are separated out by mechanical means and processed by means of “plastics to pellatisation” into secondary granulate that is used in flower tubs or similarly thick-walled products. To depict this, the plastic recycling was assessed in the same way as the material recycling of mixed plastics in (Öko-Institut/IFEU 2010). This means that a percentage of wood and concrete is also substituted by secondary granulate (65%, remaining 35% substitution of PO granulate). From the summary in Table 62 and the associated explanations it is clear that very different emission factors for the recycling of plastic waste can be given, depending on the assumptions and boundary conditions. The main influences are assumptions about the composition of the plastic waste by plastic types, assumptions about adherent impurities (quantity and type of disposal), and assumptions about the achievable yield of secondary plastics and their technical substitution potential.

Emission factors for this study

In the light of the above explanations it was agreed that this study, instead of using a single global emission factor, would use three different emission factors for plastic recycling, representing different levels of quality. In principle two different approaches to this were discussed:

1. Definition of a standard mix for plastic types and variation of the three quality categories according to the percentage of mixed plastics or unspecified plastic waste (e.g. high quality 0% mixed plastic, medium quality 60%, low quality 100%)
2. Research or plausible assumption of a mixture of plastic types and differentiation of quality according to the substitution potential (substitution factor and percentages of wood and concrete substitution)

On account of the given data situation, the second approach was chosen. This is partly because in the literature it is easier to find details of the composition of the plastic waste fraction according to plastic types[75] than details of the percentage that arises as mixed plastic. In addition, plastic recycling in Germany is regarded as high quality by comparison with other countries. However, according to (Öko-Institut/HTP 2012) the percentage of mixed plastic in all the separated plastic waste in Germany is 75%; it is thus not particularly suitable for further categorisation into “medium” and “low”.

The quality of the recycling of plastic types depends very strongly on the level of success achieved in separating plastic waste (bottles, cups, films, etc.) into segregated plastic types. The higher the degree of purity, the higher the quality of the use to which the secondary granulate arising from processing can be put in manufacturing. In the LCA this is taken into account mathematically via the substitution factor (SF) or the amount that can only be used as a substitute for wood or concrete. In Germany SFs of at least 0.8 are now achieved, and in some cases even an SF of 1 (e.g. in PET recycling). It is only in the case of mixed plastics that the quality of the secondary granulate is usually not good enough to substitute primary plastic. According to (Öko-Institut/HTP 2012), 68% of the granulate produced from mixed plastic is only used to substitute wood or concrete. For the remaining 32%, which is used to substitute

[75] E.g. in (USEPA 2013a) see Figure 20 and information provided by Prognos AG for the previous study.

primary plastic (polyethylene), an SF of 0.8 was calculated. In the light of this the following division into three quality categories was defined:

"high": SF = 1 for plastic types, SF = 0.9 for mixed plastics with 100% PE substitution

"medium" SF = 0.7 for plastic types, SF = 0.8 for mixed plastics with 32% PE substitution, otherwise wood and concrete substitution

"low" no plastic types, only mixed plastics as for "medium"

The classification of mixed plastics under "medium" corresponds to the situation for Germany described above that was identified in (Öko-Institut/HTP 2012). The "low" recycling quality is defined by the fact that it is not possible to segregate the plastic waste according to plastic type and thus only a percentage of primary plastic products – other mainly wood and concrete products – are substituted.

Results from (Öko-Institut/HTP 2012) were used to calculate the emission factors. Table 63 shows the mass balances for the processing of plastic types and mixed plastic. The processing debits are determined mainly by the electricity requirement; as an exception they were calculated here with a relatively high degree of accuracy using the relevant country-specific emission factors for electricity generation. According to (Öko-Institut/HTP 2012) the electricity requirement is 510 kWh/t input for the processing of plastic types and 450 kWh/t input for mixed plastic. For the OECD and USA balances the values were converted to granulate output using the yields shown in Table 63. In accordance with the statistical reporting system, the processing residues are included under the "incinerated" waste (see Section 4.2). For India and Egypt it was assumed that the processing residues are landfilled.[76]

Table 63: Mass balance of plastic processing by plastic types

	PE/PP/PS	Mixed plastic	PET/PVC
Processing residue	20%	20%	15%
Water	8%	15%	10%
Granulate yield	72%	21%	75%
Substitution of wood/concrete		44%	
Source:	(Öko-Institut/IFEU 2012)		assumption

The yields of mixed plastic shown in Table 63 correspond to the above-mentioned use of 32% of the secondary granulate as a substitute for PE, 34% as a substitute for wood and 34% as a substitute for concrete. For substitution of wood and concrete the varying life and density of the materials needs to be taken into account. In the inventories the thickness of wood was set at 0.75 kg/m³, and the density of concrete at 2.6 kg/m³. For concrete it was assumed for simplification that the concrete substitute has the same life in products. For wood it is assumed that the secondary materials have a 2.5 times longer life. Concrete substitute is therefore assigned a factor of 2.6 and wood a factor of 1.875.

[76] Without greenhouse gas emissions, because no organic waste is involved.

The emission factors for the credit effects by plastic type and for substitution of wood and concrete are shown in Table 64. The values for plastics are values from PlasticsEurope for primary plastics as evaluated by IFEU. The credits are for the yield of secondary granulate from the processing of the plastic. The result is also weighted with the SF for the assigned quality level.

Table 64: Emission factors for plastic granulate by plastic type and per t wood/concrete substitute

	EF in kg CO_2-eq/t granulate
Polypropylene	-1,998
Polyethylene	-2,050
Polystyrene	-3,416
PET	-3,321
PVC	-1,897
Wood substitute	-41
Concrete substitute	-265

11.2 Tables relating to the OECD balance

11.2.1 Recycling rates in the individual OECD member states

Table 65: Recycling rates in the OECD member states by waste fractions

Region	Country	Food waste	Garden waste	Paper/ card-board	Plastic	Glass	Fe metals	Non-Fe metals	Textiles
America	**USA** (USEPA 2013a)	4%	57%	66%	8%	28%	33%	21%[1]	15%
	Canada (Assumed to be as USA)								
	Mexico (INECC 2012)		23%	13%	7%	7%	4%	0,4%	
	Chile[2] *as Mexico in (Öko-Institut/ ifeu 2010)*		*2%*	*2%*	*3%*	*1%*			
	Chile (own calc.)		7%	13%	25%	24%			
Japan, South Korea and Pacific	**Australia** (OECD 2012)	10%	41%	56%	11%	40%	21%	12%	20%
	Japan (EnvGo 2010)	26%		86%	53%		89%	93%	
	South Korea (ENG 2007)	92%		79%		73%	61%		
Europe, Turkey and Israel	***EU27***[2,3] *(Öko-Institut/ ifeu 2010)*	*6%*	*7%*	*8%*	*2%*	*3%*	*3%*	*1%*	
	EU 27 (own calc.)	31%	36%	33%	21%	29%	63%	85%	38%
	Switzerland (Bafu 2010)	*17%*[2]		69%	81%	94%	84%	91%	50%
	Norway (OECD 2012)	37%	37%	54%	29%	61%	64%	55%	27%
	Iceland (OECD 2012)	37%	37%	54%	29%	61%	64%	55%	27%
	Turkey (back-calculated from OECD 2013b)	1%[4]	-	-	-	-	-	-	-
	Israel[2] *(old siva 2008)*	*6%*	*4%*	*4%*	*0%*	*1%*	*2%*		
	Israel (own calc.)	15%	0%[5]	17%	4%	17%	74%	7%	0%[5]

1) aluminium only
2) as a percentage of the total volume of waste treated
3) the data for the EU 27 were used here
4) incl. garden waste
5) Israel does not report any garden waste or textiles

11.2.2 Effective gas collection efficiencies – EU27

Table 66: Gas collection efficiencies of the EU-OECD countries and weighted means with and without 50% cap

	Landfilled quantity in 2012 (Eurostat 2014a)	Effective gas collection efficiency in 2010 (NIR 2012f)	Effective gas collection efficiency in 2010 with 50% cap
Austria	150	14%	14%
Belgium	59	53%	50%
Czech Republic	1,828	16%	16%
Germany	205	45%	45%
Denmark	94	17%	17%
Estonia	129	12%	12%
Greece	4,507	42%	42%
Spain	13,725	20%	20%
Finland	901	35%	35%
France	9,937	not reported*	50%
Hungary	2,608	2%	2%
Ireland	1,027	76%	50%
Italy	12,808	47%	47%
Luxembourg	62	14%	14%
Netherlands	140	15%	15%
Poland	7,158	17%	17%
Portugal	2,593	21%	21%
Sweden	33	26%	26%
Slovenia	315	37%	37%
Slovakia	1,297	4%	4%
UK	10,944	72%	50%
weighted effective gas collection efficiency			**34.6%**

*Neither (NIR 2012f) nor the European NIR 2011 or 2013 contains details for France; the French NIR (CCNUCC 2012, Partie 1) cites no gas collection efficiency; from (EEA 2011) it is known that between 2000 and 2007 France cited gas collection efficiencies above 50% - for calculation purposes the 50% cap was therefore applied.

11.2.3 Mass flows and emission factors for incineration in the future scenarios

Table 67: Mass flows – future scenarios

		Recycling (1,000 t)	Composting (1,000 t)	Anaerobic digestion (1,000 t)	Landfill (1,000 t)	Incineration without energy (1,000 t)	Incineration with energy (1,000 t)	Residual-waste composting (1,000 t)	Total* (1,000 t)
America	BAU	68,945	22,033		174,668	216	25,083	564	291,508
	medium	88,756	45,284		91,745		62,410		288,195
	ideal	113,602	35,469	33,065			102,941		285,077
Europe, Turkey and Israel	BAU	66,929	34,106		99,095	9,024	51,973	2,766	263,893
	medium	74,343	56,390		54,921		76,020		261,674
	ideal	92,089	36,822	39,102			91,952		259,964
Japan, South Korea and Pacific	BAU	26,638	249		14,536	3,201	40,714	0	85,338
	medium	29,154	11,247		8,058		36,438		84,898
	ideal	31,840	11,865	10,381			29,458		83,544
OECD total	BAU	162,512	56,388		288,299	12,441	117,770	3,330	640,740
	medium	192,253	112,921		154,725		174,869		634,768
	ideal	237,531	84,155	82,549			224,351		628,585

*The differences in the total quantities arise from losses during MBT and MBS (moisture loss, biodegradation) and in the inert materials to landfill

Table 68: Specific results for incineration in the medium future scenario

	MSWI (kg CO_2-eq/t waste)	RDF-CHP (kg CO_2-eq/t waste)	Co-incineration (kg CO_2-eq/t waste)	MBT operation (kg CO_2-eq/t waste)	Incineration total weighted (kg CO_2-eq/t waste)
Canada	-97	168	-763	8	-111
USA	-400	-333	-763	3	-411
Mexico	-259	-164	-763	5	-274
Chile	-244	-133	-763	5	-259
EU (OECD)	-244	-86	-763	6	-257
Switzerland	-11	370	-763	11	-23
Norway	-12	368	-763	11	-24
Iceland	-12	368	-763	11	-24
Turkey	-294	-182	-763	5	-307
Israel	-346	-285	-763	4	-359
Australia	-394	-595	-763	1	-412
New Zealand	-217	-218	-763	5	-238
Japan	-198	-179	-763	5	-199
South Korea	-198	-179	-763	5	-220

Table 69: Specific results for incineration in the ideal future scenario

	MSWI (kg CO_2-eq/t waste)	RDF-CHP (kg CO_2-eq/t waste)	Co-incineration (kg CO_2-eq/t waste)	MBS operation (kg CO_2-eq/t waste)	Incineration total weighted (kg CO_2-eq/t waste)
Canada	-107	-8	-953	24	-141
USA	-400	-516	-953	69	-420
Mexico	-278	-345	-953	54	-305
Chile	-262	-313	-953	51	-289
EU (OECD)	-232	-265	-953	47	-261
Switzerland	-3	198	-953	7	-41
Norway	-4	196	-953	7	-42
Iceland	-4	196	-953	7	-42
Turkey	-280	-363	-953	55	-308
Israel	-332	-467	-953	64	-357
Australia	-409	-782	-953	92	-437
New Zealand	-218	-399	-953	59	-253
Japan	-198	-360	-953	55	-234
South Korea	-198	-360	-953	55	-234

Table 70: Specific results for anaerobic digestion in the ideal scenario

	Food waste (1,000 t)	Electricity credit [t CO_2-eq]	Heat credit (t CO_2-eq)	Total credit [t CO_2-eq]	Total debit [t CO_2-eq]	Specific net result [kg CO_2-eq/t_{waste}]
Canada	3,500	-136,081	-212,630	-348,711	448,961	29
USA	22,894	-3,005,575	-1,390,690	-4,396,265	2,936,396	-64
Mexico	5,699	-570,996	-346,176	-917,173	730,940	-33
Chile	972	-91,724	-59,042	-150,766	124,665	-27
EU (OECD)	31,099	-2,666,617	-1,889,108	-4,555,725	3,988,789	-18
Switzerland	566	-820	-34,379	-35,199	72,591	66
Norway	198	-364	-12,008	-12,372	25,355	66
Iceland	10	-18	-597	-615	1,261	66
Turkey	5,955	-616,092	-361,732	-977,824	763,786	-36
Israel	765	-93,577	-46,441	-140,018	98,060	-55
Australia	2,309	-414,848	-140,244	-555,092	296,121	-112
New Zealand	248	-27,294	-15,067	-42,361	31,813	-43
Japan	6,021	-619,471	-365,745	-985,216	772,259	-35
South Korea	1,804	-185,563	-109,559	-295,122	231,330	-35

11.2.4 Regional breakdown in (OECD 2012)

Figure 67: Regional breakdown in (OECD 2012)

Table 10. Regional Breakdown of OECD Member Countries for Analysis

Region	Regional Group	Countries
North America	North America	Canada, Mexico, United States
OECD Europe	High-Recycling OECD Europe	Austria, Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, Netherlands, Norway, Sweden, Switzerland, United Kingdom
	Low-Recycling OECD Europe	Czech Republic, Greece, Hungary, Poland, Portugal, Slovak Republic, Spain
OECD Pacific/Asia	OECD Pacific	Australia, New Zealand
	OECD Asia	Japan, South Korea

11.3 Key information and findings from the workshop in India

(Extracts from the IFEU presentation at the closing working, IFAT Munich, 8 May 2015)

Figure 68: Background information on the workshop in India

„Waste and Climate Change“ Workshop in India

- Organised by the GIZ, supported and cooperation with UBA and IFEU
- Objective:
  - facilitate a discussion between key players in SWM, to discuss challenges and opportunities for India
  - gather feedback on data, regional differencies, informal sector, political issues on municipal level, experiences
- Participants:
  Representatives from Ministry of Environment & Forests (MoEF), Ministry of Urban Development (MoUD), Ministry of New & Renewable Energy (MNRE), Municipal Officials, Waste Management Companies, NGOs
- Concerns and situation in India were subject to a panel discussion, also engaging the audience

Figure 69: Key concerns / suggestions of stakeholders

Concerns of different stakeholders

- Data situation is difficult; not enough quality data to enable planning; measurement/monitoring system is needed and overview of exisiting data
- Alternative waste treatment options often failed due to missing or not fitting data (waste quantity, heating value too low, organic waste content lower than expected), and due to technical problems, missing infrastructure or miscalculations
- Good practise low-tech and low-cost options seem to be more accepted
- Regional differencies require different technologies - a „one size fits all“ approach will not work for all of India
- More capacity is needed at the municipal level, which could be financed by PPP. Most of current funding is directed to the private sector
- There is a need for targets that are set and strived for, also accepted on the federal states level, and in synergy between regulators and implementers of waste management rules (different ministeries), otherwise the situation will not progress
- But it has to be recognized that alternatives to the status quo are in direct competition with waste picker employment; need to be formerly integrated

11.4 Calculation of CO_2 abatement costs

Table 71: Cost calculation – Egypt, future scenario SC 1

	Baseline			SC-1		
	Total (Mt)	**Costs (million €)**	**Revenue (million €)**	**Total (Mt)**	**Costs (million €)**	**Revenue (million €)**
Collection, formal	9.0	361		17.2	689	-
Simple MBT/composting (in SC-1)	0.8	20	8	10.6	265	106
Feeding to animals	1.2			1.5		
Recycling	0.9	27	44	1.5	46	76
Managed landfill	0.5	8		7.4	127	
Unmanaged landfill	16.6			0		
Open burning	1.1			0		
Total	**21.1**	**416**	**52**	**21.1**	**1,127**	**182**

Table 72: Cost calculation – Egypt, future scenario SC 2

	Baseline			SC-2		
	Total (Mt)	**Costs (million €)**	**Revenue (million €)**	**Total (Mt)**	**Costs (million €)**	**Revenue (million €)**
Collection, formal	9.0	361		21.1	844	-
Simple MBT/composting (in SC-2)	0.8	20	8	5.7	144	57
Feeding to animals	1.2			1.5		
Recycling	0.9	27	44			
Simple MBT				13.1	346	
RDF-CHP		8		0.9		
from MBT in rec.	0			0.9		47
Anaerobic digestion (biogas)	0.5			5.3	316	144
Managed landfill				5.7	90	
Unmanaged landfill	16.6			0		
Open burning	1.1			0		
Total	**21.1**	**416**	**52**	**21.1**	**1,740**	**248**

Table 73: Cost calculation – India, future scenario to 2030, medium

	Baseline			2030 medium		
	Total (Mt)	Costs (million €)	Revenue (million €)	Total (Mt)	Costs (million €)	Revenue (million €)
Collection, formal	34	1,178		38	1,320	
Open burning						
Unmanaged landfill	4					
Informal doorstep collection	4			4		
Unmanaged landfill	29					
open fires landfill	3					
Managed landfill				19	282	
Simple MBT	2	34		19	377	
Metals from MBT	0.01		0.26	0.06		3
MBS						
MSWI						
informal sector	0.340			0.340		
Total	**42**	1,212	0.26	**42**	1,979	3

Table 74: Cost calculation – India, future scenario to 2030, ideal

	Baseline			2030 ideal		
	Total (Mt)	Costs (million €)	Revenue (million €)	Total (Mt)	Costs (million €)	Revenue (million €)
Collection	34	1,178		38	1,320	
Open burning						
Unmanaged landfill	4					
Informal doorstep collection	4			4		
Unmanaged landfill	29					
open fires landfill	3					
Managed landfill						
Simple MBT	2	34	0.26			
MBS				19	847	
Metals from MBT						3
MSWI				19	1,506	471
informal sector						
Total	**42**	**1,212**	**0.255**	**42**	**3,674**	**474**