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Best Environmental Management Practice for the Waste Management Sector

Learning from frontrunners

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Title: Best Environmental Management Practice for the Waste Management Sector

<u>Abstract</u>
The way communities generate and manage their waste plays an absolutely key role in their ability to use resources efficiently. While making the European economy more resource-efficient and circular requires a large spectrum of actions, a huge opportunity for saving resources lies in improving waste management at the local level in Europe.
On the basis of an in-depth analysis of the actions implemented by frontrunner organisations in the waste management sector, this report describes a set of best practices with significant potential for broad uptake. They are called Best Environmental Management Practices (BEMPs) and aim to help local authorities in charge of waste management and waste management companies move towards a circular economy.
The BEMPs, identified in close cooperation with a technical working group comprising experts from the sector, cover the areas of waste management which determine the overall waste management performance the most: setting a waste management strategy, promoting waste prevention, establishing an efficient waste collection that supports reuse and recycling, and stimulating waste preparation for reuse and product reuse. Certain areas of waste treatment are also covered. The BEMPs address mainly the management of municipal solid waste, but also of construction and demolition waste and healthcare waste.
Additionally, the report provides a set of environmental performance indicators that organisations can use to assess their waste management performance and monitor progress as well as benchmarks of excellence that give an indication of the levels achieved by best performers.
The report presents a wide range of information (environmental benefits, economics, case studies, references, etc.) for each of the best practices and aims to provide inspiration and guidance to organisations of the sector. In addition, the report will be the technical basis for the development of an EMAS (EU Eco-Management and Audit Scheme) Sectoral Reference Document on Best Environmental Management Practice for the Waste Management Sector according to Article 46 of Regulation (EC) No 1221/2009 (EMAS Regulation).

Contents

ACKNOWLEDGMENTS

This report was prepared by the European Commission's Joint Research Centre in the framework of supporting the development of an EMAS Sectoral Reference Document for the Waste Management Sector[1]. This report is based on different preparatory and complementary studies carried out by BZL Kommunikation und Projektsteuerung GmbH (Germany), E3 Environmental Consultants Ltd (UK), Ambiente Italia (Italy), Association of Cities and Regions for sustainable Resource management – ACR+ (Belgium) and Bipro GmbH (Germany).

A technical working group, comprising a broad spectrum of experts in waste management, supported the development of the document by providing input and feedback.

[1] Further information on the development of the EMAS Sectoral Reference Documents is available at: http://susproc.jrc.ec.europa.eu/activities/emas/documents/DevelopmentSRD.pdf

EXECUTIVE SUMMARY

The management of waste at local level plays a key role in the ability of communities to use resources efficiently and make progress towards achieving a more circular economy. Many waste authorities and waste management companies are interested in improving their waste management performance, for instance by promoting waste prevention and reaching higher levels of reuse and recycling. This report describes best practices (called best environmental management practices (BEMPs)) that can provide them with inspiration and practical tips based on actions and techniques that have been implemented by frontrunner organisations and proven successful.

BEMPs were identified by the European Commission's Joint Research Centre, in close cooperation with a technical working group of experts and stakeholders from the waste management sector.

Scope

This document addresses two types of organisations: waste management companies (public and private), including companies implementing producer responsibility schemes, and waste authorities (public administrations in charge of waste management, mainly at local level). It does not cover the activities of organisations that generate waste and do not belong to the waste management sector (i.e. most organisations).

It describes best practices for the waste management phases and activities with the greatest circular economy potential:

- establishing a waste management strategy;
- fostering waste prevention;
- promoting the reuse of products and preparation of waste for reuse;
- waste treatment, limited to operations enabling material recycling.

In the area of waste treatment, the scope is limited to waste treatment operations not covered in the Best Available Techniques Reference Document (BREF) for Waste Treatment and to facilities performing treatments outside the scope of the Industrial Emissions Directive[2] (e.g. sorting facilities whose aim is to recycle plastics).

It deals with three waste streams:

- municipal solid waste (MSW): household waste and waste from other sources, such as retail, administration, education, health services, accommodation and food services, and other services and activities, which is similar in nature and composition to waste from households;
- construction and demolition waste (CDW);
- healthcare waste (HCW).

[2] Directive 2010/75/EU of the European Parliament and of the Council on industrial emissions.

Industrial waste and commercial waste not included in MSW are not covered in this document.

Structure of the report

Chapter	Brief description of the content
1. General information and scope of the document	Chapter 1 sets the context with facts and figures about the waste management sector in the EU and explains the definition of the scope of the report.
2. Common environmental performance indicators for municipal solid waste	Chapter 2 describes a set of environmental performance indicators that can be used to describe the overall performance of a municipal waste management system as well as a number of associated benchmarks of excellence.
3. Cross-cutting BEMPs	Chapters 3 to 6 are the core of the report: they describe the identified best practices (called Best Environmental Management Practices (BEMPs)), together with the associated indicators and benchmarks of excellence. Chapter 3 deals with cross-cutting best practices that apply to all the waste streams covered in this document, from setting a waste strategy, to the use of economic instruments and to finding additional best practices in other EU reference documents.
4. BEMPs for MSW	Chapter 4 presents how waste authorities and waste management companies can best manage MSW, including the design of the strategy, waste prevention, product reuse and preparation of waste for reuse, waste collection and waste treatment operations. The chapter also includes a BEMP addressing producer responsibility organisations.
5. BEMPs for CDW	Chapter 5 focuses on the activities of waste authorities and waste management companies directly or indirectly responsible for the management of CDW. The main areas addressed are CDW management plans, avoidance of PCB contamination of CDW, processing of CDW and waste plasterboard and management of removed waste asbestos.
6. BEMPs for HCW	Chapter 6 presents how waste authorities and waste management companies can best deal with the management of HCW. The main areas covered are the optimisation of HCW segregation and the adoption of alternative treatments for HCW.
7. Conclusions	Chapter 7 provides a summary of the main outcomes of the report: (i) common environmental performance indicators for municipal solid waste management and the corresponding benchmarks of excellence, (ii) best environmental management practices and the associated BEMP-specific indicators and benchmarks of excellence.

Content of the report

Cross-cutting BEMPs (applicable to MSW, CDW and HCW)

Effective and efficient waste management needs a comprehensive strategy that includes all waste streams under the responsibility and/or control of the local authority or waste management company concerned. A successful strategy is based on, among others, the assessment of current and future trends in the size and composition of waste streams, consideration of environmental attitudes of residents and appropriate and solid data monitoring. The waste management strategy needs to prioritise actions according to the waste hierarchy (reduce, reuse, recycle, recover, dispose of) and life-cycle assessment tools can be used to complement the general rules and better shape the most effective solutions.

Most advanced waste management strategies include targets both for the long term (i.e. 10–20 years) and for the short term (i.e. 1–5 years) and a regular review of the strategy (at least every 3 years).

Local authorities and waste management companies can integrate a number of economic instruments (such as taxes and tax modulation, waste pricing, deposit refund schemes) into the strategy in order to drive behavioural change and align economic incentives with the improvement of waste management performance.

BEMPs for MSW - waste strategy

In the specific case of MSW, a successful economic instrument is the pay-as-you-throw model, where the waste fees paid by users are modulated according to the amount of mixed waste delivered to the waste management system. The adoption of pay-as-you-throw can lead to outstanding results in waste management, increasing the amount of fractions collected separately and sent for recycling while reducing mixed waste. It is essential that a pay-as-you-throw system is complemented by a user-friendly and effective collection infrastructure for the separately collected fractions covering the greatest range of waste types possible. The implementation of the waste strategy and of the instruments it uses needs to be supported by advanced waste monitoring with timely data at the level of individual fractions and collection methods.

To realise the potential of the waste management strategy developed, frontrunner waste authorities and waste management companies consider awareness-raising of residents and other economic players very important. In order to effectively encourage waste prevention, reuse and recycling, it is important that messages are tailor-made for well-defined target audiences, and delivered consistently over time through a range of complementary means. Frontrunner organisations found it made sense to invest significant amounts (at least EUR 5 per resident) in awareness-raising.

A specific measure adopted for awareness-raising is the establishment of a network of waste advisers. These are employees or volunteers trained in waste prevention and management who support residents in reducing and correctly separating at source the waste generated in households at the very local level down to even individual buildings or households. Frontrunner waste authorities have put in place one waste adviser per 20 000 inhabitants.

BEMPs for MSW - prevention and reuse

Local authorities and waste management companies can implement waste prevention measures that target households as well as public and private organisations. Ideally, the most suitable prevention measures are identified based on a systematic assessment of waste generation patterns in the territory, prioritisation of the most relevant waste streams in terms of prevention potential (e.g. food waste, furniture) and involvement of relevant stakeholders (e.g. residents, local businesses, social economy organisations).

Local authorities can for example introduce local plastic bag charges or support the setting up of repair shops for products reaching their end-of-life. Another example is the operation of product/material exchange areas in civic amenity sites, where residents and local businesses can leave products (e.g. furniture, household appliances, clothing) that they no longer need but which are still usable. Citizens or social economy organisations can collect them for reuse.

BEMPs for MSW - waste collection

Frontrunner waste authorities and waste management companies complement an effective door-to-door (kerbside) MSW collection system with civic amenity sites where citizens can drop off at least 20 different individual waste fractions for separate collection. It is important that the network of civic amenity sites is well distributed and accessible to residents; mobile collection points also prove very useful.

BEMPs for MSW - extended producer responsibility schemes

In many EU Member States, a large number of recyclable waste streams (e.g. packaging) are managed under extended producer responsibility (EPR) schemes. Producer responsibility organisations can increase separate collection, recycling and reuse rates for the waste collected under the EPR schemes by implementing actions such as competitions among territories and benchmarking of the environmental achievements of different local authorities.

BEMPs for MSW - waste treatment

Achieving outstanding levels of recycling requires not only effective source separation and collection of MSW but also state-of-the-art treatment operations.
For instance, advanced plants sorting co-mingled light packaging, able to separate fibres, metals by type and plastics by polymer and colour, achieve a plant sorting rate of at least 88 %.
Other examples of advanced waste treatment plants analysed are plants processing mixed plastic packaging, mattresses and absorbent hygiene products.

BEMPs for CDW

Local authorities can foster better management of CDW through ambitious construction and demolition waste plans. In these plans they can for example prioritise CDW prevention, establish minimum CDW sorting requirements for large construction sites and set targets for CDW recycling that go beyond EU and national obligations.
Waste authorities and waste management companies can also foster progress on specific areas with large potential environmental benefits such as: processing of waste plasterboard and CDW for recycling, avoidance of PCB contamination of CDW and management of waste asbestos removed by residents.

BEMPs for HCW

The segregation of HCW at the point of waste generation is strongly regulated and the top priority for HCW management is ensuring hygiene and infection control. However, respecting those prerequisites, there is often scope to improve recycling and reduce the environmental impact. For instance, waste management companies can support the improvement of HCW segregation in healthcare institutions by preventing recyclable non-hazardous waste being placed in the hazardous waste bins. Frontrunner waste management companies carry out waste audits in healthcare facilities and play an active role in defining their waste management practices, clear categories of waste to be sorted and precise guidelines. Opportunities for reducing the environmental impacts of segregated HCW also arise from optimising its treatment operations.

Improvement of waste management - the role of indicators and benchmarks

It is very important for waste authorities and waste management companies to regularly assess their waste management performance using meaningful indicators. Such an assessment can improve the understanding of the waste management system and help identify areas for improvement and, thus, the most relevant BEMPs.

For instance, in the case of MSW, waste authorities and waste management companies can calculate capture rates and impurity rates for the different separately collected recyclable fractions. Capture rates indicate what percentage of waste of a certain material ends up in the separate collection for that material out of the total waste generation. Impurity rates indicate the amount of non-target waste contained in the separate collection for that particular waste fraction. These types of indicators are instrumental to understanding which part of the system is performing less well and opening the avenue to investigating why and how to improve. They become even more useful when used in combination with benchmarks of excellence which provide an indication of the levels of performance achieved by frontrunner organisations which can help others estimate their improvement potential. In the case of capture rates, currently achieved outstanding capture rates for waste glass can be higher than 90 %, for waste paper and cardboard higher than 85 % and for waste metals higher than 75 %.

Other important indicators with associated benchmarks for the overall MSW management are the total quantity of waste generated per resident (frontrunner local authorities achieve values lower than 360 kg/capita/year, limited to a number of defined MSW fractions), the amount of waste sent for energy recovery and/or disposal (frontrunners achieve less than 70 kg/capita/year, without considering rejects from sorting/recycling) and the amount of waste sent for disposal (frontrunners achieve less than 10 kg/capita/year).

Policy context

The EU has the objective to foster a sustainable circular economy[3] in which materials are extensively reused and recycled through feedback loops. Without significant improvements of waste management practice at local level it is not possible to achieve a more circular management of waste and to meet the waste management targets set by legislation. This document aims at supporting those actors in the waste management sector committed to improving at local level. It provides them with a source of inspiration and guidance in terms of best practices derived from the actions implemented by frontrunners.

The content of this report has been developed to form the technical basis for the development of a Sectoral Reference Document for the Waste Management Sector according to Article 46 of Regulation (EC) No. 1221/2009[4] on the EU Eco-Management and Audit Scheme (EMAS). EMAS is a management tool for companies and other organisations to evaluate, report and improve their environmental performance. In

[3] See the European Commission communication on "Closing the loop - an EU action plan for the circular economy": http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52015DC0614

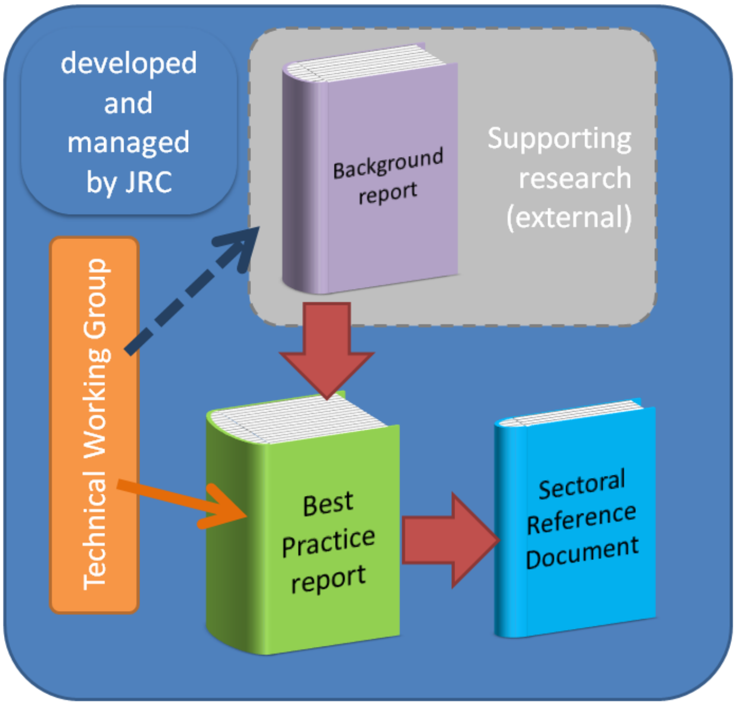
[4] The full text of Regulation (EC) No. 1221/2009 is available at: http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32009R1221.

order to support the efforts of organisations embarking on continuous environmental performance improvement, the EMAS Regulation includes a provision requesting the European Commission to produce Sectoral Reference Documents to provide information and guidance on BEMPs. These are being developed for 11 priority sectors, including the waste management sector. Once an EMAS Sectoral Reference Document for the Waste Management Sector is adopted, EMAS-registered organisations will take into account its content as laid out in the EMAS Regulation. However, both this report and the EMAS Sectoral Reference Document can be used, on a voluntary basis, by any organisation, whether EMAS-registered or not.

PREFACE

This **Best Practice Report**[5] provides an overview of techniques that are **Best Environmental Management Practices (BEMPs)** in the waste management sector. The document was developed by the European Commission's Joint Research Centre (JRC) on the basis of desk research, interviews with experts, site visits and in close cooperation with a Technical Working Group (TWG) comprising experts from the sector. This document is based on a preparatory external study carried out by BZL Kommunikation und Projektsteuerung GmbH (Germany) and E3 Environmental Consultants Ltd (UK), whose findings are presented in a Background Report[6].

This Best Practice Report provides the basis for the development of the EMAS Sectoral Reference Document (SRD) for the Waste Management Sector (Figure I). The structured process for the development of this Best Practice Report is outlined in the guidelines on the "Development of the EMAS Sectoral Reference Documents on Best Environmental Management Practice" (European Commission, 2014), which are available online[7].



Source: JRC

Figure I: The Best Practice Report in the overall development of the Sectoral Reference Document (SRD)

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[5] This report is part of a series of 'Best Practice Reports' published by the European Commission's Joint Research Centre covering a number of sectors for which the Commission is developing Sectoral Reference Documents on Best Environmental Management Practice. More information on the overall work and copies of the 'Best Practice Reports' available so far can be found at: http://susproc.jrc.ec.europa.eu/activities/emas/

[6] The background report produced by BZL Kommunikation und Projektsteuerung GmbH and E3 Environmental Consultants Ltd on which this report is based is available online at: http://susproc.jrc.ec.europa.eu/activities/emas/documents/WasteManagementBackgroundReport.pdf

[7] The methodology for the development of the EMAS Sectoral Reference Documents is available online at: http://susproc.jrc.ec.europa.eu/activities/emas/documents/DevelopmentSRD.pdf

EMAS (the EU Eco-Management and Audit Scheme) is a management tool for companies and other organisations to evaluate, report and improve their environmental performance. To support this aim and according to the provisions of Article 46 of the EMAS Regulation (EC No. 1221/2009), the European Commission is producing SRDs to provide information and guidance on BEMPs in several priority sectors, including the EEE manufacturing sector.

Nevertheless, the guidance on BEMP is not only for EMAS-registered companies, but is rather intended to be a useful reference document for any relevant company that wishes to improve its environmental performance or any actor involved in promoting best environmental performance.

BEMPs encompass techniques, measures or actions that can be taken to minimise environmental impacts. These can include technologies (such as more efficient machinery) and/or organisational practices (such as staff training).

An important aspect of the BEMPs proposed in this document is that they are proven and practical, i.e.:

- they have been implemented at full scale by several companies (or by at least one company if replicable/applicable for others);
- they are technically feasible and economically viable.

In other words, BEMPs are demonstrated practices that have the potential to be taken up on a wide scale in the waste management sector, and that are expected to result in exceptional environmental performance compared to current mainstream practices.

A standard structure is used to outline the information concerning each BEMP, as shown in Table a.

Table a: Information gathered for each BEMP

Category	**Type of information included**
Description	Brief technical description of the BEMP including some background and details on how it is implemented.
Achieved environmental benefits	Main potential environmental *benefits* to be gained through implementing the BEMP.
Appropriate environmental indicators	Indicators and/or metrics used to monitor the implementation of the BEMP and its environmental benefits.
Cross-media effects	Potential *negative* impacts on other environmental pressures arising as side effects of implementing the BEMP.
Operational data	Operational data that can help understand the implementation of a BEMP, including any issues experienced. This includes actual and plant-specific performance data where possible.
Applicability	Indication of the type of plants or processes in which the technique may or may not be applied, as well as constraints to

	implementation in certain cases.
Economics	Information on costs (investment and operating) and any possible savings (e.g. reduced raw material or energy consumption, waste charges, etc.).
Driving force for implementation	Factors that have driven or stimulated the implementation of the technique to date.
Reference organisations	Examples of organisations that have successfully implemented the BEMP.
Reference literature	Literature or other reference material cited in the information for each BEMP.

Sector-specific Environmental Performance Indicators and Benchmarks of Excellence are also derived from the BEMPs. These aim to provide organisations with guidance on appropriate metrics and levels of ambition when implementing the BEMPs described.

- <u>Environmental Performance Indicators</u> represent the metrics that are employed by organisations in the sector to monitor either the implementation of the BEMPs described or, when possible, their environmental performance directly.
- <u>Benchmarks of Excellence</u> represent the highest environmental standards that have been achieved by companies implementing each related BEMP. These aim to allow all actors in the sector to understand the potential for environmental improvement at the process level. Benchmarks of excellence are not targets for all organisations to reach but rather a measure of what it is possible to achieve (under stated conditions) that companies can use to set priorities for action in the framework of continuous improvement of environmental performance.

The sector-specific Environmental Performance Indicators and Benchmarks of Excellence presented in this report were agreed by a technical working group, comprising a broad spectrum of experts in the waste management sector, at the end of its interaction with the JRC.

Role and purpose of this document

This document is intended to support the environmental improvement efforts of all organisations dealing with waste management by providing guidance on best practices (see Section 1.2). Organisations and companies from this sector can use this document to identify the most relevant areas for action and find detailed information on best practices to address the main environmental aspects, as well as organisation-level environmental indicators and related benchmarks of excellence to track sustainability improvements.

In addition, this Best Practice Report provides the technical basis for the development of the EMAS SRD for the waste management sector according to Article 46 of the EMAS Regulation[8].

How to use this document

This document is not conceived to be read from beginning to end, but as a working tool for professionals willing to improve the environmental performance of their organisation and who seek reliable and proven information in order to do so.

Different parts of the document will be of interest and will apply to different professionals and at different stages.

The best way to start using this document is by reading the short section below about its structure to understand the content of the different chapters and, in particular, the areas for which BEMPs have been described and how these BEMPs have been grouped.

Then, Chapter 1 would be a good starting point for readers looking for a general understanding of the sector and its environmental aspects.

Those looking for an overview of the BEMPs described in the document could start from Chapter 7 (Conclusions) and in particular with Table 7.1 and Table 7.2 outlining all the common environmental performance indicators and benchmarks of excellence as well as the BEMPs together with the related specific environmental performance indicators and benchmarks of excellence, i.e. the exemplary performance level that can be reached in each area.

For readers looking for information on how to improve their environmental performance in a specific area, it is recommended to start directly at the concrete description of the BEMPs on that topic, which can be easily found through the table of contents (at the very beginning of the document).

Structure

After this Preface section, which gives an overview of the framework within which this document was developed, Chapter 1 presents the scope of the document and some general facts and figures of the waste management sector in the EU context. Chapter 2 defines the common environmental performance indicators for waste management systems while Chapter 3 presents in detail the best environmental management practices dealing with cross-cutting issues of waste management, independently from the type of waste (i.e. municipal solid waste, construction and demolition waste and healthcare waste). Chapter 4 presents specific best environmental management practices for municipal solid waste, from the development of the management strategy to measures and techniques for the best waste prevention, collection and treatment. Moreover, a specific best environmental management practice in Chapter 4 deals with extended producer responsibility schemes. Chapter 5 presents best environmental management practices for construction and demolition waste, while Chapter 6 focuses on the best segregation and treatment of healthcare waste.

[8] When published, the EMAS SRD for the waste management sector will be available online at: http://susproc.jrc.ec.europa.eu/activities/emas/waste_mgmt.html

Finally, Chapter 7 summarises the main outcomes of the document: BEMPs, environmental performance indicators and corresponding benchmark of excellence.

Table b: Summary of the structure of the document

Chapter	Topics and BEMPs
Chapter 1	Scope of the document – waste streams and waste management actors. General facts and figures of the waste management sector.
Chapter 2	Common environmental performance indicators for waste management systems.
Chapter 3	Cross-cutting BEMPs: - integrated waste management strategies; - life-cycle assessment of waste management options; - economic instruments; - link to other relevant reference documents.

Chapter	Topics and BEMPs

Chapter 4	Best environmental management practices for municipal solid waste (MSW):
	- Strategy BEMPs:
	  ○ cost benchmarking;
	  ○ advanced waste monitoring;
	  ○ pay-as-you-throw;
	  ○ performance-based waste management contracting;
	  ○ awareness-raising;
	  ○ establishment of a network of waste advisers;
	  ○ home and community composting.
	- BEMPs on waste prevention and reuse:
	  ○ local waste prevention programmes;
	  ○ schemes fostering the reuse of products and the preparation for reuse of waste.
	- BEMPs for waste collection:
	  ○ waste collection strategy;
	  ○ inter-municipal cooperation among small municipalities;
	  ○ civic amenity sites;
	  ○ logistics optimisation for waste collection;
	  ○ low-emission vehicles.
	- BEMPs for extended producer responsibility schemes:
	  ○ best use of incentives by producer responsibility organisations (PROs).
	- BEMPs on waste treatment:
	  ○ sorting of co-mingled light packaging waste to maximise recycling yields for high-quality output;
	  ○ sorting of collected mixed plastics to maximise recycling yields for high-quality output;
	  ○ treatment of mattresses for improved recycling of materials;
	  ○ treatment of absorbent hygiene products for improved recycling of materials.

Chapter 5	Best environmental management practices for construction and demolition waste: - Integrated construction and demolition waste plans; - Avoidance of polychlorinated biphenyl (PCB) contamination of construction and demolition waste; - Local schemes for proper management of waste asbestos removed by residents; - Processing waste plasterboard to foster recycling; - Processing CDW for the production of recycled aggregates.

Chapter	Topics and BEMPs

Chapter 6	Best environmental management practices for healthcare waste:
	- BEMPs for healthcare waste segregation:
	  ○ encouragement of healthcare waste segregation at healthcare facilities;
	  ○ healthcare waste collection for residents.
	- BEMPs for the treatment of healthcare waste:
	  ○ alternative treatments for healthcare waste.

Chapter 7	Conclusions: BEMPs, environmental performance indicators and benchmarks of excellence.

Best practices presented in this document can also be mapped according to the waste management strategy they address and the waste stream they refer to, as shown in the following figure.

	Waste management strategy	Waste prevention and reuse	Waste collection	EPR schemes	Waste treatment
Cross-cutting	- Integrated waste management strategy - LCA of waste management options - Economic instruments				- Link to other relevant reference documents
Municipal solid waste	- Waste monitoring, performance-based contracting - PAYT, cost benchmarking - Home and community composting - Awareness-raising and waste advisers	- Local waste prevention programmes - Schemes for product reuse and waste preparation for reuse	- Waste collection strategy - Logistics optimisation for waste collection and low-emission vehicles - Civic amenity sites - Inter-municipal cooperation	- Best use of incentives by PROs	- Sorting co-mingled and mixed plastic waste - Treatment of mattresses - Treatment of absorbent hygiene products
Construction and demolition waste	- Integrated construction and demolition waste plans - Avoidance of PCB contamination of CDW - Management of waste asbestos				- Processing of CDW for production of RCA - Processing of waste plasterboard for recycling
Healthcare waste	- Encouragement of healthcare waste segregation at healthcare facilities - Healthcare waste collection for residents				- Selection of alternative treatments for healthcare waste

1. General information about the waste management sector, its environmental relevance and EMAS implementation in the sector

1.1. General information about the waste management sector

Waste management is an integrated part of our economy which is characterised by huge mass streams. The most important parts of waste management are illustrated in more detail by means of consumer waste from food and drink products in Figure 1-1.

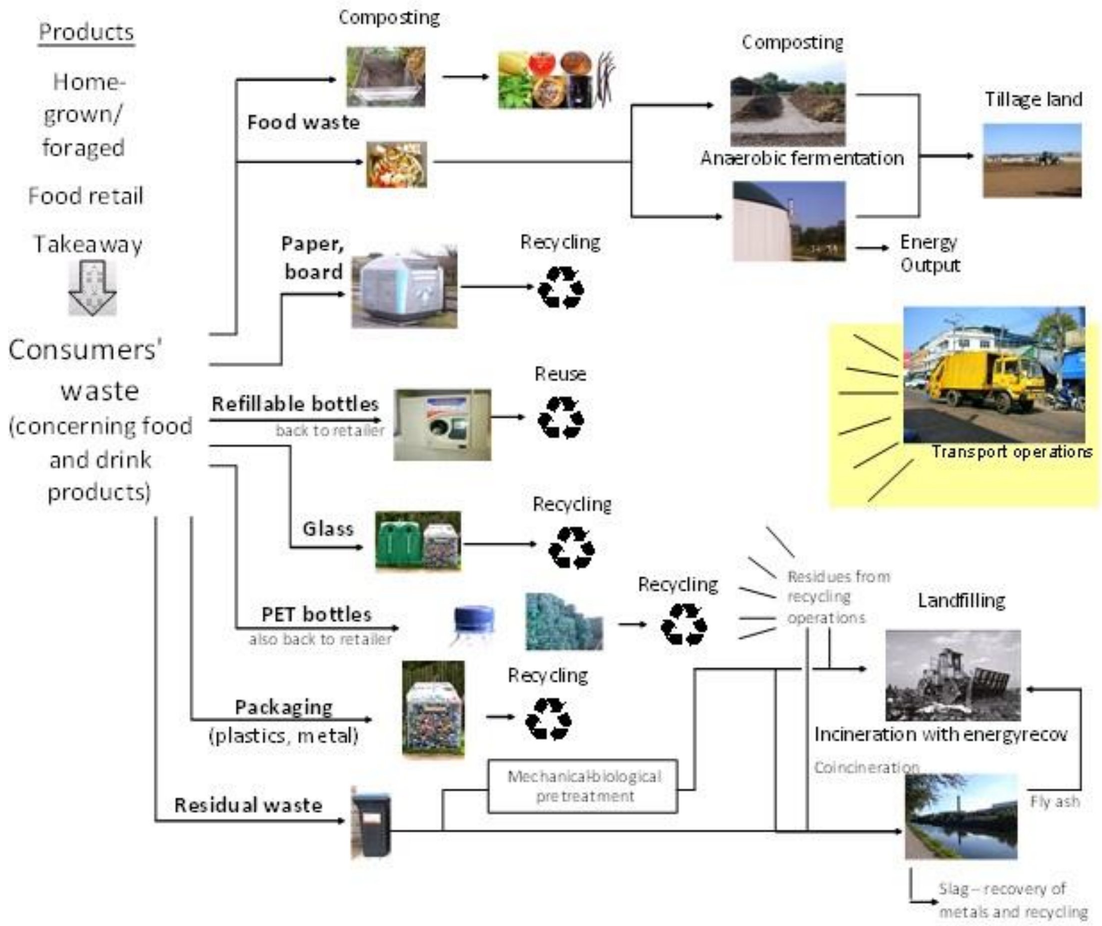


Figure 1-1. Reuse, recovery, recycling and disposal of consumer waste including the associated transport activities

On average, each EU citizen consumes 16 tonnes[9] of materials annually, of which 6 tonnes are wasted, according to the Roadmap to a Resource-Efficient Europe (EC, 2011). Total waste generation in the EU-28 in 2010 was over 2.5 billion tonnes, with

[9] 1 tonne is a non-SI metric unit of mass equal to 1000 kilograms and is thus equivalent to one megagram (Mg).

the largest share, 34 %, coming from the construction sector (Figure 1-2). In total, 4 % of the waste generated is estimated as hazardous.

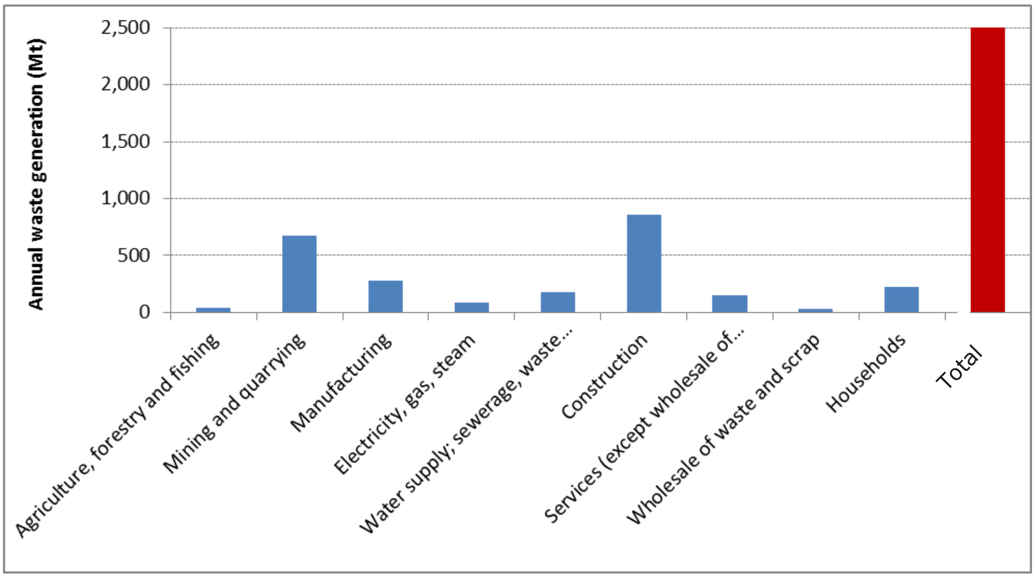


Figure 1-2. Waste generated by NACE sectors across the EU-28 in 2010 in Mt (million tonnes) - Source Eurostat, 2014

Germany, France and the UK together account for more than 39 % of the total waste generated in Europe (Eurostat, 2014) (see Figure 1-3).

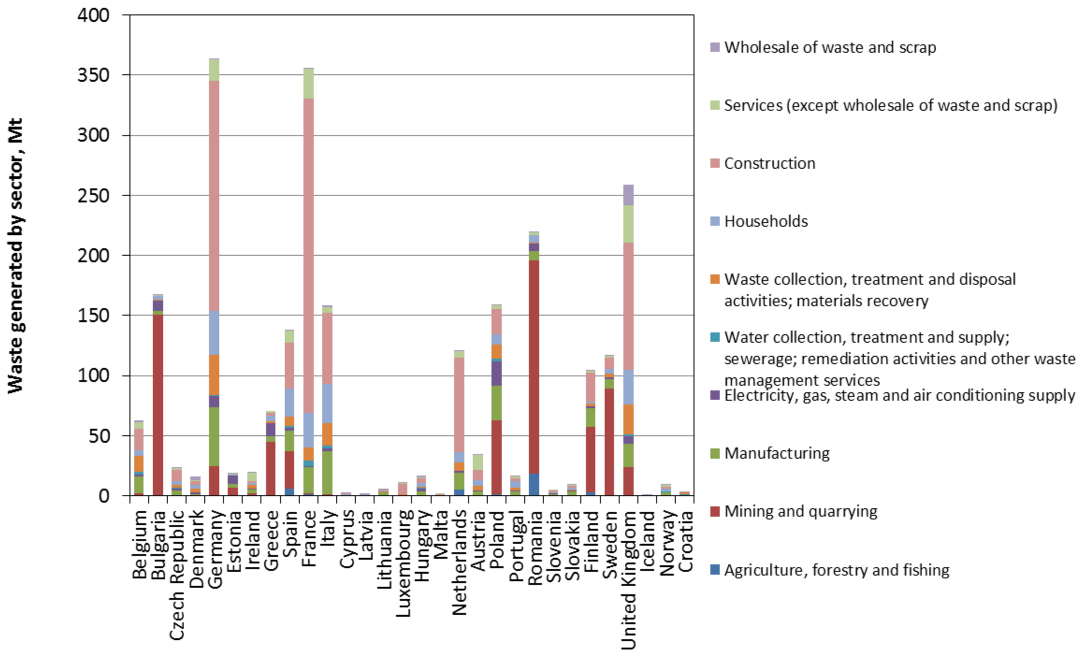
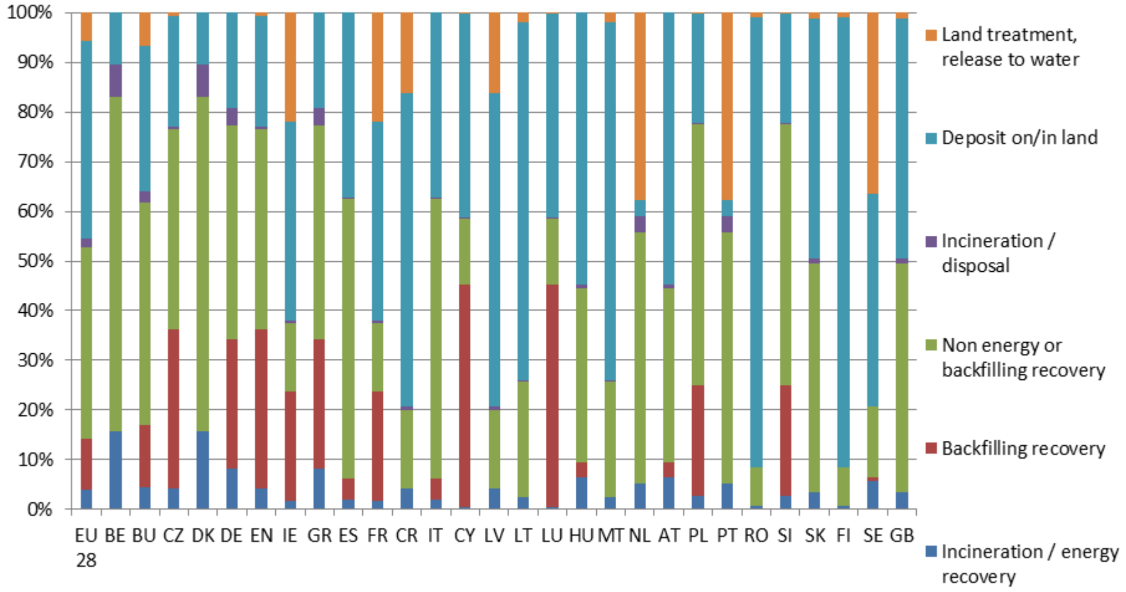


Figure 1-3. Waste generated by NACE sector in European countries in 2010 in Mt - Source Eurostat, 2014

Although the generation of waste has been stable in Europe in recent years, the main reason for this is assumed to be the decrease in consumption provoked by the economic crisis.

Waste management systems in the EU Member States differ significantly, varying from zero to 90 % disposal of untreated waste in landfills (Figure 1-4).

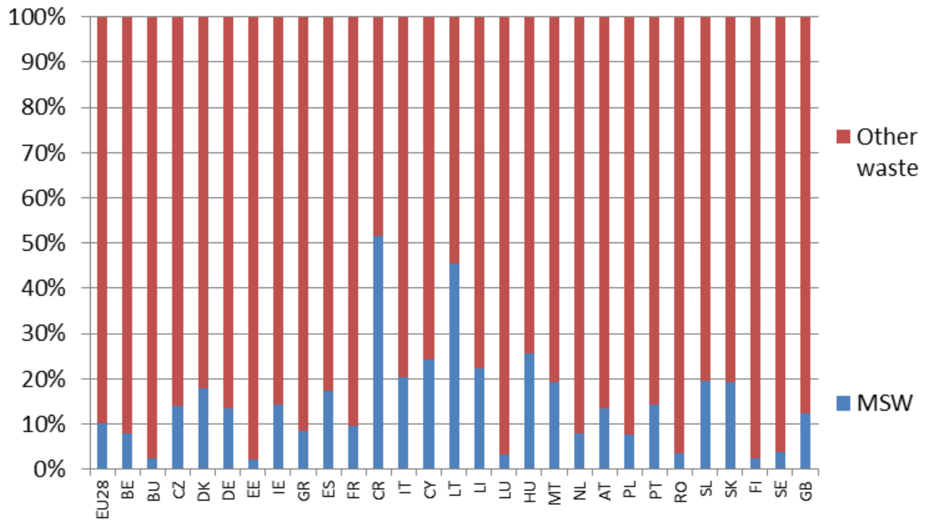


Source: Eurostat (2014)

Figure 1-4. Percentages of total waste undergoing different treatment or disposal options across the EU-28 in 2010[10]

Municipal solid waste (MSW) represents around 10 % of the total waste generated in the EU-28 by mass (Figure 1-5), i.e. household waste and similar commercial, industrial and institutional waste (EC, 2014), and it includes a wide range of fractions including organic materials, plastics, paper and metals. Households generate 60 % to 90 % of MSW, although there are wide variations among the methodologies used to produce waste statistics across EU Member States. The statistical value is mainly affected by how household-type waste from commerce, industry and institutions is considered.

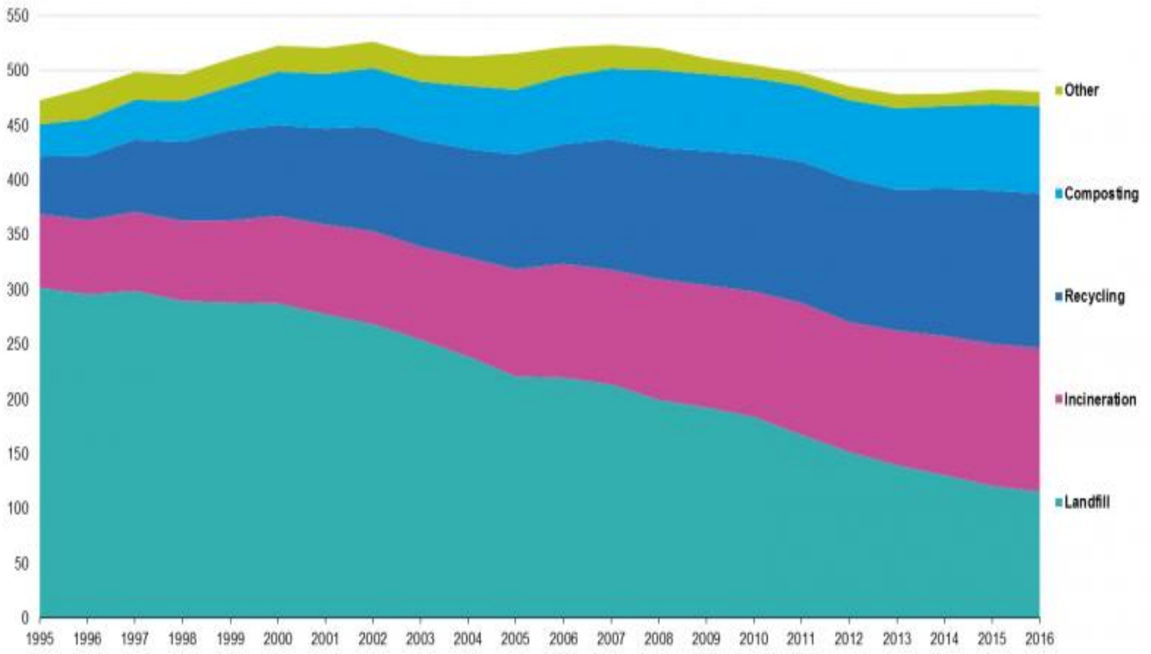
[10] As seen in the original publication, some of the country abbreviations are not standard.



Source: Eurostat (2014)

Figure 1-5. Percentage of total waste categorised as municipal solid waste (MSW) across the EU-28[10]

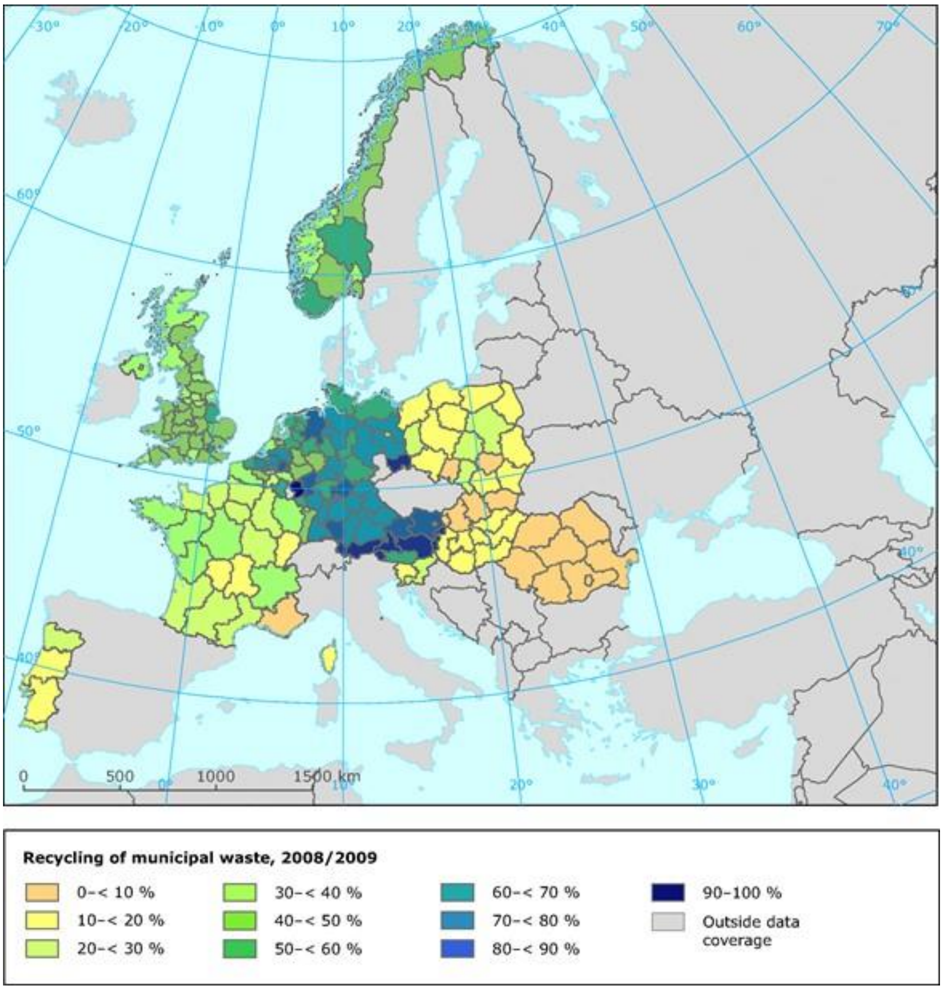
On average, each EU citizen generated 480 kg of municipal waste in 2016, down from 520 kg in 2000 (Eurostat, 2018). On average (see Figure 1-6), only a limited share (45 %) of the municipal waste generated is recycled or composted, with the rest being landfilled (24 %) or incinerated (27 %) (Eurostat, 2018).



Source: Eurostat (2018)

Figure 1-6. Share of municipal waste undergoing different treatment or disposal options across the EU-28 from 1995 to 2016

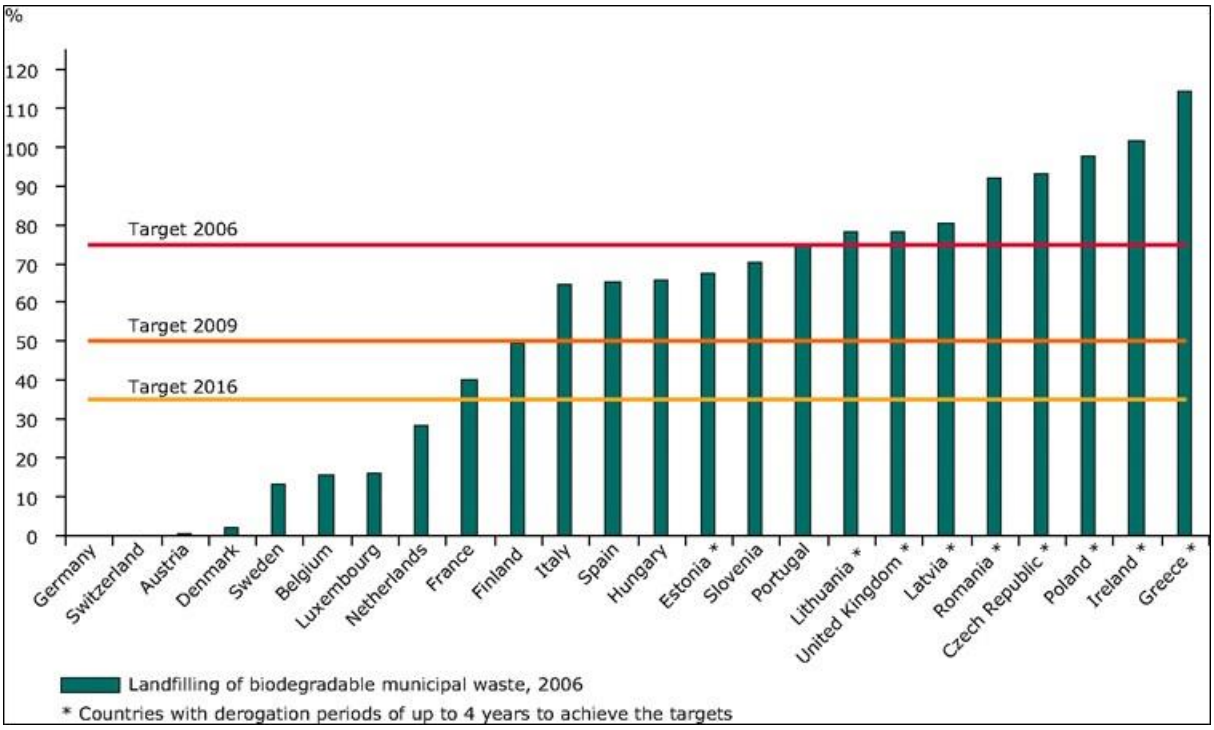
The European Environment Agency reported (EEA, 2013a) that, whilst 11 Member States have already met, or are on track to meet, the Waste Framework Directive's target for 50 % of MSW to be recycled by 2020, the majority of Member States will have to make unprecedented progress in increasing recycling rates (some examples of differences in recycling rates in the EU are presented in the figure below for the 2008-2009 period) in order to meet this target.



Source: EEA (2013a)

Figure 1-7. Recycling rates for municipal solid waste across local authorities in selected EU Member States, 2008-2009

Similarly, many Member States need to make rapid progress if they are to meet targets established in the Landfill Directive to reduce landfilling rates for the particularly polluting biodegradable municipal waste fraction (Figure 1-8). Whilst meeting these targets is ultimately the responsibility of national and local government, private companies, including small and medium enterprises, are also heavily involved in delivering waste management and recycling services.



Source: EEA (2012)

Figure 1-8. Biodegradable municipal waste landfilled in 2006 (% of biodegradable municipal waste generated in 1995), compared to targets of the European Landfill Directive

In order to improve waste management, actions are prioritised following the "waste hierarchy" (Figure 1-9).

Figure 1-9. Waste hierarchy according to the Waste Framework Directive (2008/98/EC)

1.1.1. Waste policy

Global demand for food, feed and fibre in aggregate is expected to increase by 70 % by 2050. However, finite resources are becoming increasingly scarce and expensive to extract, whilst renewable resources are often harvested at unsustainable rates. Raw material extraction, processing, transport and disposal are associated with environmental burdens such as climate change, air pollution and water pollution. 60 % of the world's major ecosystems are degraded or are used unsustainably, and on current trends two planet Earths would be required to support global economic activity by 2050.

Our economic system is based on huge mass streams, as shown in Figure 1-10.

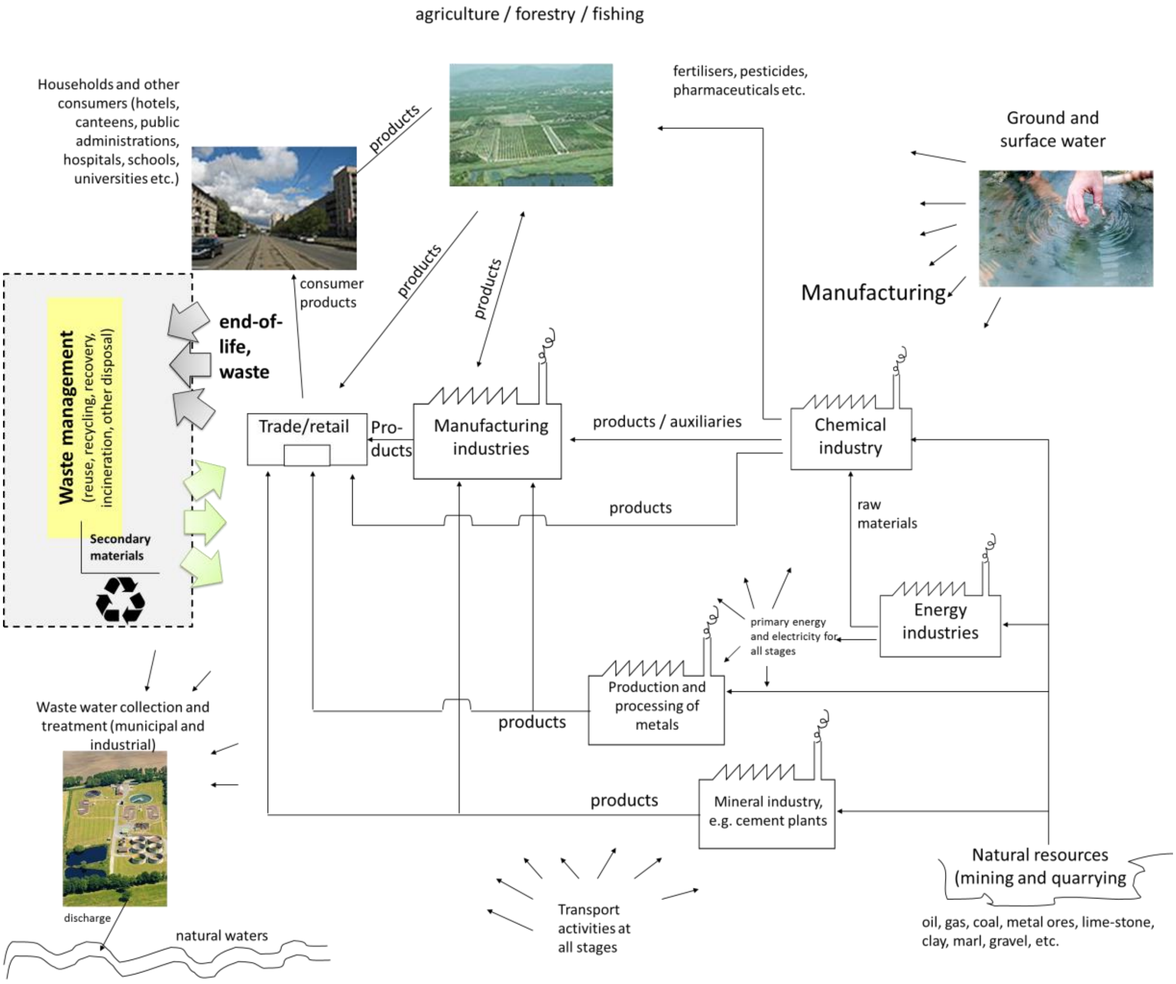
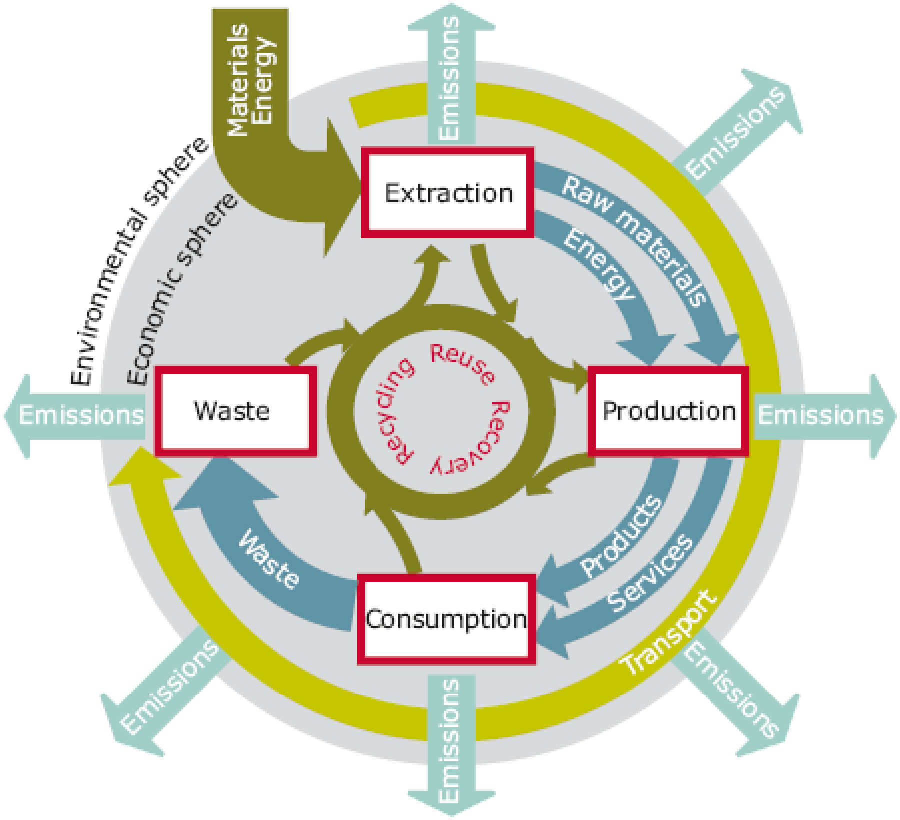


Figure 1-10. Basic illustrative scheme for the mass streams of our current economic system

The European Commission has a long-term objective to foster a sustainable circular economy (i.e. the European Commission communication on closing the loop - an EU action plan for the circular economy[11]) in which materials are extensively reused and recycled through feedback loops that both support and directly generate economic activity (Figure 1-11). This objective is integral to achieving long-term economic stability, prosperity and a high quality of life for European citizens.

[11] The EU action plan for the circular economy adopted in 2015 is available at: http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52015DC0614



Source: EEA (2010)

Figure 1-11. A conceptual representation of raw material and energy flows, services and transport in the European economy.

Efficient waste management, in particular waste prevention, reuse and recycling, is a critical component of a resource-efficient economy. Relevant EU Directives underpinning national regulations include:

- Directive 2012/19/EU on waste electrical and electronic equipment (recast);
- Directive 2011/65/EU on the restriction of the use of certain hazardous substances in electrical and electronic equipment (recast);
- Directive 2010/75/EU on industrial emissions (integrated pollution prevention and control) (recast);
- Directive 2006/21/EC on mining waste;
- Directive 2006/66/EC on batteries and accumulators and waste batteries and accumulators;
- Directive 2005/20/EC amending Directive 94/62/EC on packaging and packaging waste;
- Regulation 1774/2002 laying down health rules concerning animal by-products not intended for human consumption;
- Directive 2000/76/EC on waste incineration;
- Directive 2000/53/EC on end-of-life vehicles;
- Directive 99/31/EC on landfill of waste;
- Directive 91/676/EC concerning the protection of waters against pollution caused by nitrates from agricultural sources;
- Directive 75/439/EEC regarding disposal of waste oils.

European policy instruments relevant to waste avoidance and management include:

- Integrated Product Policy (COM(2003) 302);
- Sustainable Consumption and Production and Sustainable Industrial Policy (SCP/SIP) Action Plan (COM(2008) 0397);
- The EU Ecolabel scheme (Regulation (EC) No 66/2010);
- The Eco-design Directive (Directive 2009/125/EC);
- Green Public Procurement guidelines and procurement directives (COM(2008) 400, Directive 2004/17/EC, Directive 2004/18/EC);
- Eco-Management and Audit Scheme (Regulation (EC) 1221/2009);
- The Green Action Plan for SMEs 2014 – 2020 (COM(2014) 440);
- Closing the loop - An EU action plan for the Circular Economy (COM(2015) 0614 final).

1.1.2. Structure of the sector

The activities covered by best environmental management practices in this report, according to the "statistical classification of economic activities in the European Community" known as NACE from its French name "*Nomenclature statistique des activités économiques dans la Communauté européenne*" (Eurostat, 2008), are those shown in Table 1-1. The waste management sector is defined under NACE codes 38 and 39 (collection, treatment, recovery, disposal and trade of waste). From the perspective of the environmental performance of the waste management sector, not only waste management companies but also waste authorities (public administrations in charge of managing wastes from their citizens, policies and regulations) are considered to be within the boundaries of the sector, because the consequences of the decisions made at public administration level are key to determining the sector's performance.

Table 1-1. Main NACE code activities covered by integrated waste management activities

NACE Rev. 2 Main Category	Division	Group	Class
E – WATER SUPPLY, SEWERAGE, WASTE MANAGEMENT AND REMEDIATION	**38** Waste collection, treatment and disposal activities, materials recovery	**38.1** Waste collection	**38.11** Collection of non-hazardous waste
			38.12 Collection of hazardous waste
		38.2 Waste treatment and disposal	**38.21** Treatment and disposal of non-hazardous waste
			38.22 Treatment and disposal of hazardous waste
		38.3 Materials recovery	**38.31** Dismantling of wrecks
			38.32 Recovery of sorted materials
	39. Remediation activities and other waste management services	**39.0** Remediation activities and other waste management services	**39.00** Remediation activities and other waste management services
G — WHOLESALE AND RETAIL TRADE, REPAIR OF MOTOR VEHICLES AND MOTORCYCLES	**46**. Wholesale trade, except of motor vehicles and motorcycles	**46.7** Other specialised wholesale	**46.77** Wholesale of waste and scrap
O — PUBLIC ADMINISTRATION AND DEFENCE, COMPULSORY SOCIAL SECURITY	**84**. Public administration and defence, compulsory social security	**84.1** Administration of the State and the economic and social policy of the community	**84.12** Regulation of the activities of providing health care, education, cultural services and other social services, excluding social security

Waste management is mainly undertaken by micro companies of less than 10 employees, usually specialised in collection and materials recovery. Indeed, from a total of 44 424 companies in NACE division 38 (according to Eurostat), 77 % are micro companies and 99.7 % are SMEs (less than 250 employees). Besides the proportion of companies, it is also important to note the existence of big players in Europe, which currently manage more than 40 % of MSW in Europe. There is no data on the number and size of waste authorities, which are often waste departments in municipalities or other local authorities. However, many of the SMEs reported below are public companies.

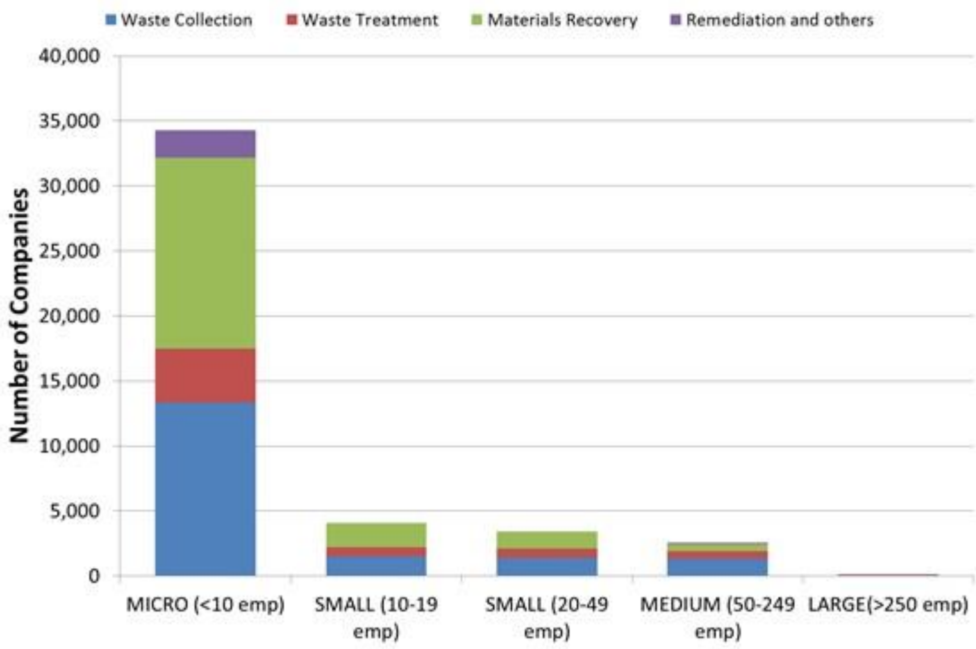


Figure 1-12. Number of companies in Europe (EU-28) per waste subsector and size (Data from Eurostat, sbs_na_ind_r2)

The structure per country is extremely heterogeneous regarding the size and the number of companies (Figure 1-13), which indicates a very different approach, not only at national level, but also at regional and local levels.

The number of organisations affects the replicability of any best practice. However, in terms of turnover, the waste management sector is dominated by medium and large companies (Figure 1-14). The turnover of the whole waste collection subsector (including all types of wastes) sums EUR 50 000 million, with waste treatment accounting for around EUR 35 000 million, and materials recovery EUR 62 000 million. The value added (approximately the gross income after taxes and subsidies) of these three main subsectors of waste management in Europe is shown in Figure 1-15. In this case, the highest value is observed for the waste collection subsector and, again, the values are heavily dominated by large and medium companies. The material recovery subsector, however, is dominated by smaller companies.

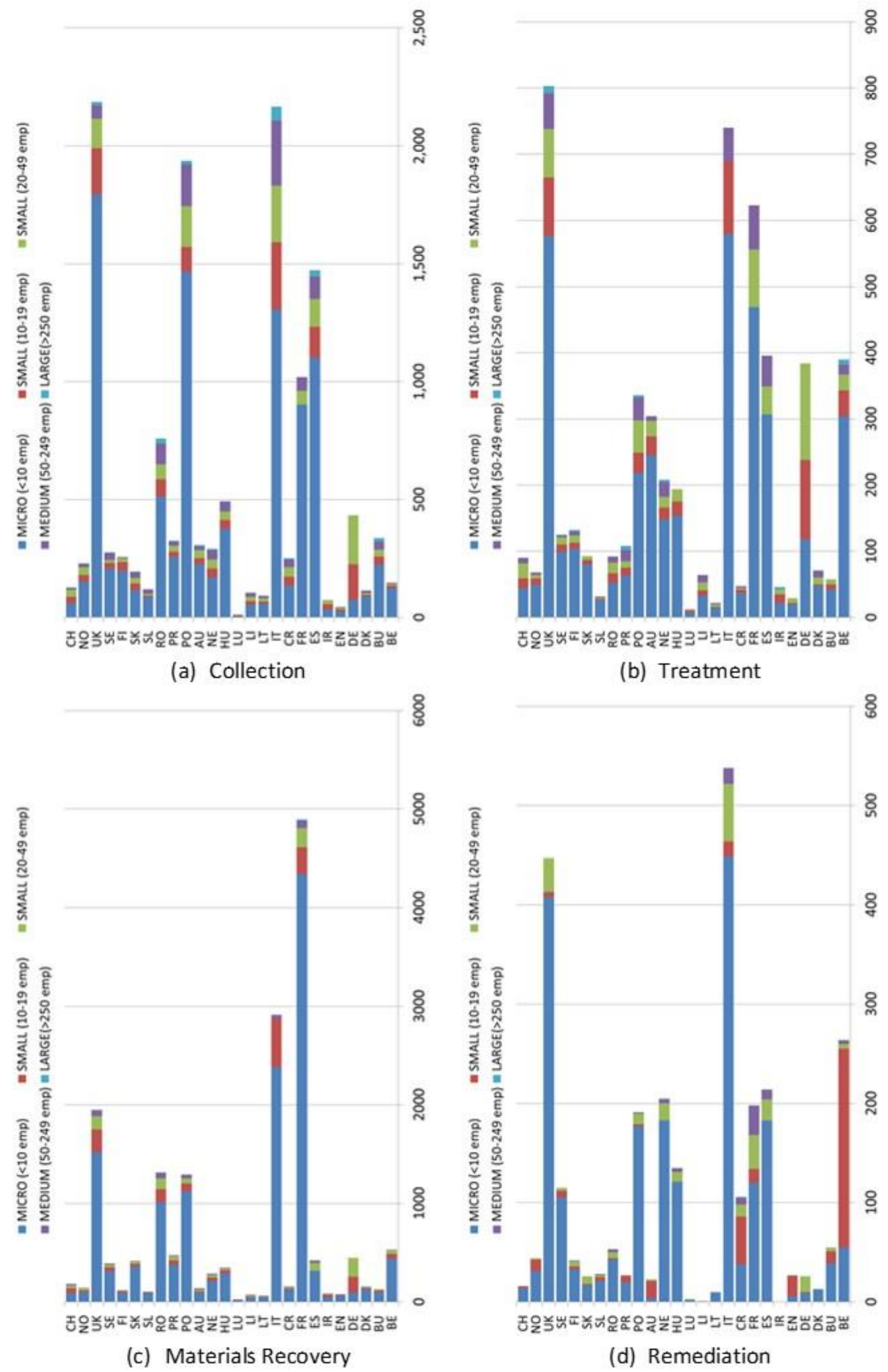


Figure 1-13. Number of companies per country and size for a) waste collection, b) waste treatment, c) materials recovery and d) remediation (Data from Eurostat, sbs_na_ind_r2)

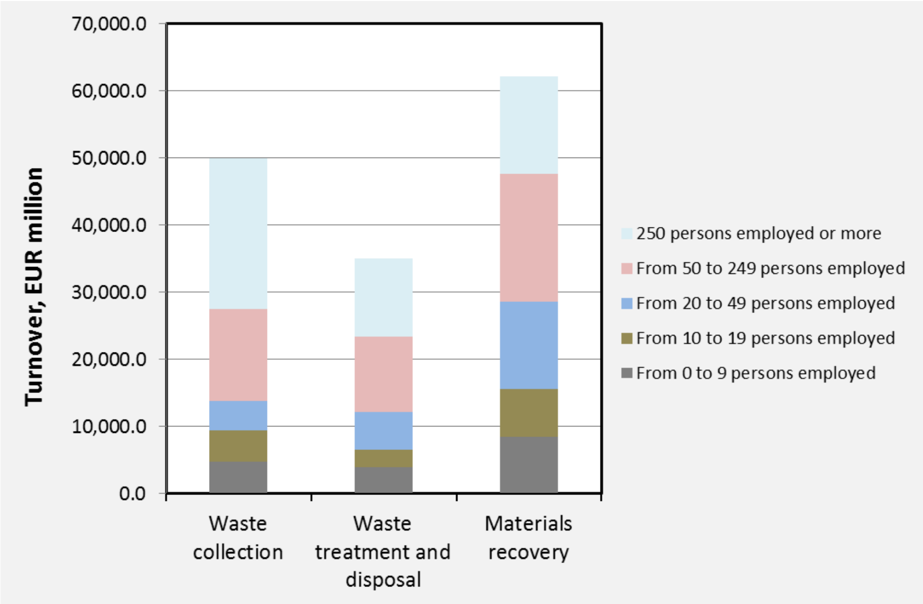


Figure 1-14. Turnover per waste subsector and size of company (remediation excluded) (data from Eurostat, sbs_na_ind_r2, 2013)

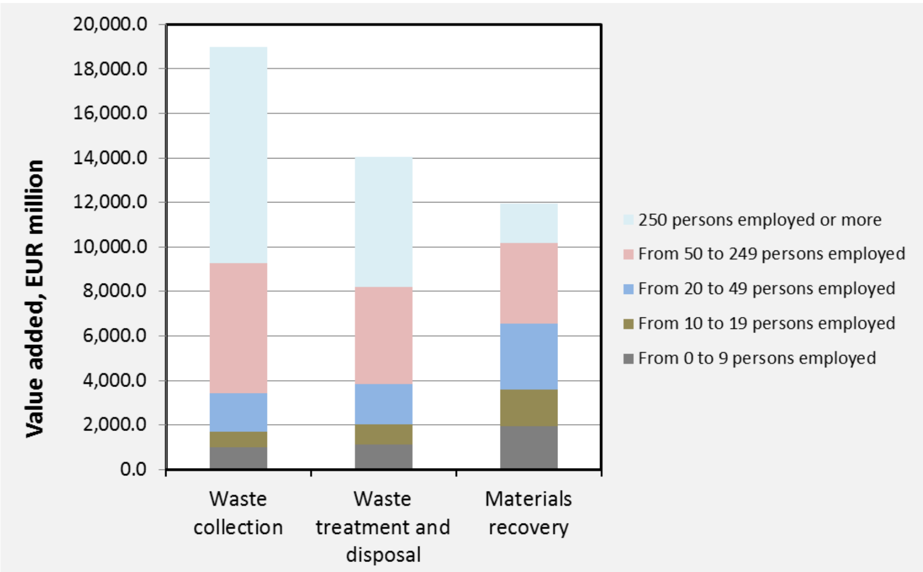


Figure 1-15. Value added per waste subsector and size of company (remediation excluded) (Data from Eurostat, sbs_na_ind_r2, 2013)

The number of persons employed per subsector and size of company is shown in Figure 1-16. In total, 900 000 people are reportedly employed by the sector, but this number could be 20 % to 30 % higher due to different statistical approaches (Hall and Nguyen, 2012).

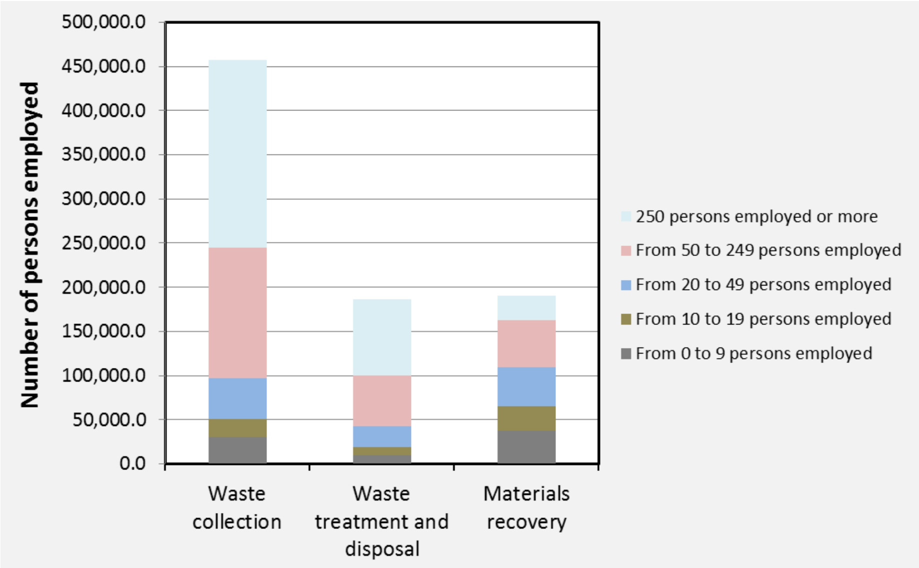


Figure 1-16. Persons employed by the waste sector in Europe (Data from Eurostat, sbs_na_ind_r2, 2013)

There is an evident high labour intensity in the waste collection subsector, while waste treatment or materials recovery have a similar, smaller, number of employees. Most of the employment in waste collection and waste treatment is in the hands of bigger companies, while materials recovery is still dominated by smaller companies.

The apparent productivity, i.e. the value added per person employed, varies with the labour intensity and the size of the company (Figure 1-17).

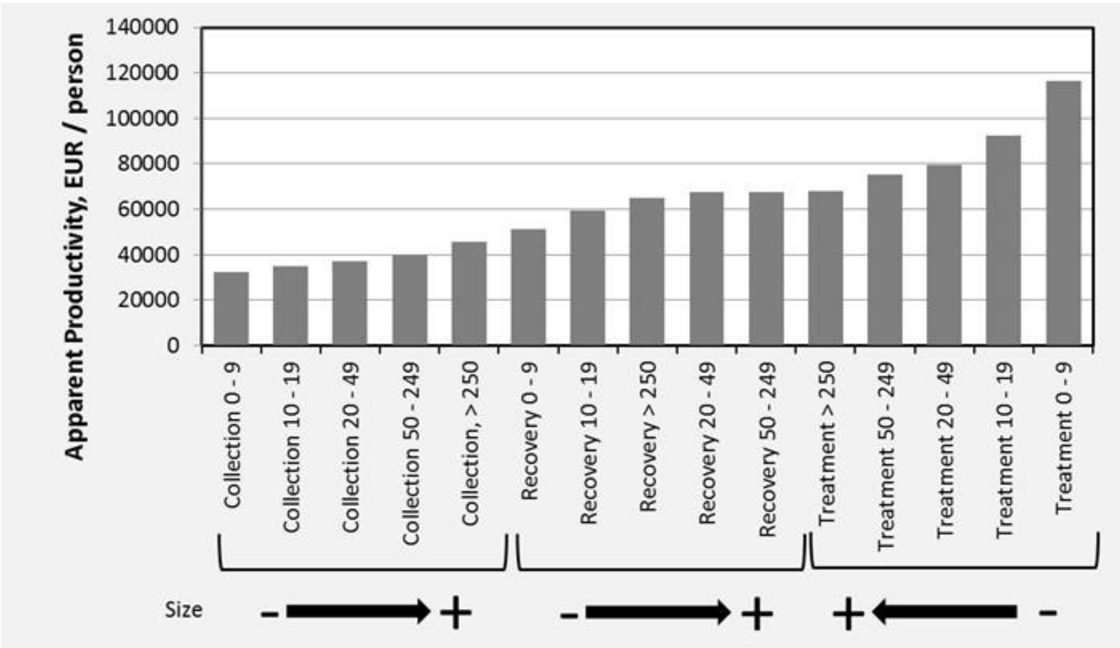


Figure 1-17. Apparent productivity of the waste sector in Europe (Data from Eurostat, sbs_na_ind_r2, 2013)

Treatment has higher productivity, probably due to the existence of larger facilities, with a lower unitary cost of treatment and higher throughput per employee. Results also show this effect on the economy of scale, although the data may reflect the low labour intensity of landfills compared to other treatment and disposal facilities. On the other hand, collection of waste has an apparently lower productivity, as its labour intensity is higher and its performance is relatively limited by transport capacities and fuel costs. Large companies perform better, but a lower productivity compared to other sectors is observed. Materials recovery productivity does not vary much with the size of the company and its value lies between treatment and collection.

The influence of the economic performance on the environmental performance is not negligible. The resources of smaller companies for the implementation of environmentally friendly practices are somewhat limited and their investment capacities are probably low for those with lower productivity. A higher number of employees requires more awareness, training, and better management structures than organisations with fewer employees but with the same waste flow. Bigger companies have highly standardised procedures, so best practice implementation would be more efficient. Smaller companies belonging to bigger groups will run the environmental policy of the matrix company, but independent, smaller organisations will require other incentives. Also, the public or private character of the organisation has a strong influence on the decision-making processes: private companies in the waste management sector are service providers and will implement practices mainly driven by client policy (e.g. the public waste authority or the consortium managing an extended producer responsibility scheme).

Large companies play a considerable role in the European waste management sector. The turnover of the 16 biggest private organisations in waste management accounts for 40 % of the total revenue of the sector, mainly in treatment and collection (Hall, 2007). There are countries where these differences could be even higher. The Public Services International Research Unit (PSIRU) calculated (Hall, 2007) the national concentration of waste management companies in 2006 (Table 1-2). Although the data are outdated, the order of magnitude can still be considered correct and the actual current values may even be higher, as the remunicipalisation of services has had little impact on the European waste management sector.

Table 1-2. Concentration by country in 2006: market share of largest three operators (Hall, 2007)

Country	Market share of largest 3 operators (%)
Spain	57
France	47
Netherlands	44
Belgium	41
Germany	38
UK	23

1.2. Scope of the document

This brief introduction outlines the proposed scope and priorities of the document.

1.2.1. Target group

- Waste management companies (public and private), including companies implementing producer responsibility schemes.
- Waste authorities (public administrations in charge of waste management, mainly at local level).

The document does not cover organisations which generate waste and do not belong to the waste management sector (i.e. most organisations). In fact, these other organisations are addressed in the SRDs for their respective sectors.

1.2.2. Waste management phases

Best environmental practices in several areas of waste management are already set out in European legislation and other European reference documents, such as:

- the Best Available Techniques Reference Documents (**BREFs**) for **waste incineration** and **waste treatment** developed under the IPPC (Industrial Pollution Prevention and Control) Directive and then the IED (Industrial Emission Directive)[12];
- the EU **Landfill Directive** (99/31/EC) which aims to prevent and reduce negative effects on the environment from the landfilling of waste;
- end-of-waste criteria[13] (developed under the **Waste Framework Directive**) which specify when certain waste ceases to be considered waste and obtains the status of a product (or a secondary raw material).

This document covers the phases and activities where best environmental practices are not already set out by other existing EU legislation and reference documents. More specifically, the document covers the following phases:

- Establishing a **waste management strategy** (i.e. which options are best for each waste stream under which conditions; which kind of collection; how many fractions; which treatments; which final disposal; etc.).
- **Waste prevention** (i.e. reducing the amount of waste generated or diverting reusable products away from waste streams and into reuse streams, for instance reducing the food waste generated at household level thanks to information campaigns and courses; measures aimed at influencing consumers to ask for more environmentally friendly products and less packaging; etc.).
- **Waste collection** (vehicles used, choice of routes, schedule of the collection, etc.).

[12] The Industrial Emissions Directive, IED (2010/75), determines rules on integrated prevention and control of pollution arising from industrial activities. It also lays down rules designed to prevent or, where that is not practicable, to reduce emissions to air, water and land and to prevent the generation of waste, in order to achieve a high level of protection of the environment as a whole. Best Available Techniques Reference Documents (BREF) are drawn up at sectoral level to determine best available techniques and to limit imbalances in the Union as regards the level of emissions from industrial activities.

[13] End-of-waste criteria were introduced by Article 6 of the Waste Framework Directive of December 2008. The objective of end-of-waste criteria is to remove the administrative burdens of waste legislation for safe and high-quality waste materials, thereby facilitating recycling. The objective is achieved by requiring high material quality of recyclables, promoting product standardisation and quality assurance, and improving harmonisation and legal certainty in the recyclable material markets.

- **Waste and product reuse** (e.g. schemes promoting repairing and reselling of end-of-life electronic equipment and furniture).
- **Waste treatment facilities** not covered in the Waste Treatment BREF such as facilities performing treatments outside the scope of the IED (e.g. sorting facilities with the aim of recycling plastics).

For other phases (i.e. other waste treatment and disposal facilities, recycling and recovery operations), this report references and briefly presents, in Section 3.3.4, other useful documents for the identification of relevant best practices.

The figure below illustrates the waste management phases in relation to this report: in green the ones covered, in yellow the one partially covered and in red the ones not explicitly covered but for which reference is made to other suitable documents.

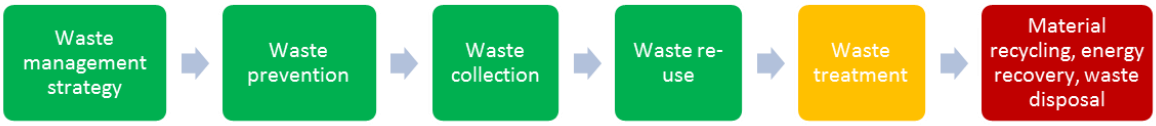


Figure 1-18. Waste management activities covered in the scope of this document

Detailed description of the waste management activities covered

In general, the activities of organisations belonging to NACE code 38.11 (waste collection) will be included in the scope:

- collection of non-hazardous solid waste within a local area, such as the collection of wastes from households and business activities by means of refuse bins, wheeled bins, containers, including mixed recoverable materials; collection of non-hazardous solid waste includes also construction and demolition waste, debris and the operations of transfer facilities;
- collection of recyclable materials;
- collection of refuse in litter bins in public places.

The collection of hazardous wastes (code 38.12), in principle, is included if the hazardous waste falls under the main focus of this document (i.e. municipal solid waste, construction and demolition waste, and healthcare waste). Nuclear waste is outside the scope of the activities to be covered. Collection of biohazardous and healthcare waste, used batteries, used oil from small garages, etc. is within the scope of activities to be considered.

Treatment and disposal of non-hazardous waste (code 38.21) is not covered completely in the document: operation of landfills is excluded, as are the disposal through incineration with or without energy recovery and the production of substitute fuels (RDF, SRF or biogas) at least at the scales covered by the IED BREFs. The same applies for the treatment and disposal of hazardous waste (code 38.22). These activities may thus only be covered from a management perspective (e.g. choice of the type of treatment).

The processing of waste and its conversion into secondary raw materials is classified as code 38.3 (materials recovery). This NACE code includes material recovery from sorted materials and from the dismantling of wrecks (cars, ships, computers, etc.) only if the final purpose is to obtain secondary materials but not to obtain re-sell parts or spares[14]. Under the scope of this report, material recovery activities are considered if they are (i) performed by a waste manager, public or private, and (ii) are excluded from the BREFs waste-related best available techniques. Waste processing by companies not belonging to the waste management sector is only considered if required as part of integrated management strategies.

Not all the activities under NACE code 39 (remediation) are considered. Remediation activities for soils, asbestos, lead-containing paints and other toxic materials, e.g. from construction waste management activities, may be included in the scope of the document.

NACE code 46.77 includes the wholesale of metal and non-metal waste and scrap for other waste treatment or recovery operations. The importance of this activity lies in the environmental performance of waste trading activities and their impact on the environmental performance of the waste (or end-of-waste) material supply chain (e.g. transportation and movement of traded waste considerably reduces the carbon reduction achievable by its use in manufacturing processes from an LCA perspective).

This report covers the activities under code 84.12 of the NACE classification on "health care, education, cultural services and other social services, excluding social security", where "administration of waste collection and disposal operations" are included (Eurostat, 2008). Indeed, many strategic decisions, and planning and development activities are designed and managed, or at least strongly influenced, by public administrations. As for the implementation (waste collection and treatment), this is sometimes carried out by the public administrations (directly or through public companies) but frequently outsourced. In Finland, for instance, almost all collections are carried out by private companies, but waste treatment is managed by public administration. In Spain, most of the waste is collected and treated by private contractors. In Germany, 60 % of waste collection is performed by public companies. These choices depend on several factors, but studies (Bel et al., 2010) have shown that there is no evidence that private waste services are cheaper. In fact, cooperation in rural areas between municipalities or different levels of government has been shown to deliver a better economic and environmental performance than private schemes (Bel and Mur, 2009). In recent years, the waste management sector is also subject to a *re-municipalisation* effect, i.e. the public administration *insources* waste management, ending the contract with the private service provider (Halmer and Hauenschild, 2014). This has mainly happened in France, the United Kingdom and, especially, in Germany and Austria. The driving force is often public opinion and the willingness to reduce the waste management costs and associated fees for the citizens, but, in some case studies, it has also been caused by the poor environmental performance of private schemes. Also, the public sector tends to take control of waste management schemes when new policies, treatment and processes are required, e.g. to increase the production of secondary materials. As the service remains profitable,

[14] According to the NACE definitions, if the waste is used as an input of a manufacturing process, the use of this waste is considered to belong to the manufacturing code (section C of the NACE list).

revenues in municipalities revert to the citizens in the form of increased social services. On the other hand, EU institutions are also giving more importance to Private Public Partnerships (PPPs) (Hall and Nguyen, 2012).

1.2.3. Waste streams

The waste streams covered in this report are as follows:

- Municipal solid waste (MSW): household waste and waste from other sources, such as retail, administration, education, health services, accommodation and food services, and other services and activities, which is similar in nature and composition to waste from household. This fraction includes organic, plastic, metal, paper, glass, bulky items, batteries, exhaust oils/lubricants, light bulbs, etc.
- Construction and demolition waste (CDW).
- Healthcare waste (HCW).

These streams were chosen because of their relevance (not only in terms of quantity but also geographical coverage) and the high replicability of best practices concerning them. CDW and HCW are included especially because they are not specifically addressed in other European best practice reference documents.

Industrial waste and commercial waste not assimilated to household waste are not targeted in this document as they are better addressed in the specific document(s) tailored to the specific sector where the waste is generated (e.g. end-of-use vehicles are addressed in the document on car manufacturing[15]).

Detailed description of the waste streams covered

Table 1-3 shows the waste streams covered in the document: construction and demolition waste (CDW), municipal solid waste (MSW) and healthcare waste (HCW). These were chosen because they are waste fractions with a high environmental impact (MSW), or with high volumes (CDW), or with a significant environmental impact and not specifically addressed in other environmental initiatives of the European Commission (HCW).

In the table, those with an asterisk (*) are considered hazardous and therefore best environmental management practice for these fractions may require further specific consideration if regulated by regional or national legislation, or are outside the scope if they fall under the IED scope.

[15] For further information see: http://susproc.jrc.ec.europa.eu/activities/emas/car.html

Table 1-3. Categories of waste to be considered under the European List of Wastes (EC, 2014)

Chapter	Subchapter	Category
17 CONSTRUCTION AND DEMOLITION WASTES (INCLUDING EXCAVATED SOIL FROM CONTAMINATED SITES)	17 01 concrete, bricks, tiles and ceramics	17 01 01 concrete 17 01 02 bricks 17 01 03 tiles and ceramics 17 01 06* mixtures of, or separate fractions of concrete, bricks, tiles and ceramics containing hazardous substances 17 01 07 mixtures of concrete, bricks, tiles and ceramics other than those mentioned in 17 01 06
	17 02 wood, glass and plastic	17 02 01 wood 17 02 02 glass 17 02 03 plastic 17 02 04* glass, plastic and wood containing or contaminated with hazardous substances
	17 03 bituminous mixtures, coal tar and tarred products	17 03 01* bituminous mixtures containing coal tar 17 03 02 bituminous mixtures other than those mentioned in 17 03 01 17 03 03* coal tar and tarred products
	17 04 metals (including their alloys)	17 04 01 copper, bronze, brass 17 04 02 aluminium 17 04 03 lead 17 04 04 zinc 17 04 05 iron and steel 17 04 06 tin 17 04 07 mixed metals 17 04 09* metal waste contaminated with hazardous substances 17 04 10* cables containing oil, coal tar and other hazardous substances 17 04 11 cables other than those mentioned in 17 04 10
	17 05 soil (including excavated soil from contaminated sites), stones and dredging spoil	17 05 03* soil and stones containing hazardous substances 17 05 04 soil and stones other than those mentioned in 17 05 03 17 05 05* dredging spoil containing hazardous substances 17 05 06 dredging spoil other than those mentioned in 17 05 05 17 05 07* track ballast containing hazardous substances 17 05 08 track ballast other than those mentioned in 17 05 07
	17 06 insulation materials and asbestos-containing construction materials	17 06 01* insulation materials containing asbestos 17 06 03* other insulation materials consisting of or containing hazardous substances 17 06 04 insulation materials other than those

Table 1-3. Categories of waste to be considered under the European List of Wastes (EC, 2014)

Chapter	Subchapter	Category
		mentioned in 17 06 01 and 17 06 03 17 06 05* construction materials containing asbestos
	17 08 gypsum-based construction material	17 08 01* gypsum-based construction materials contaminated with hazardous substances 17 08 02 gypsum-based construction materials other than those mentioned in 17 08 01
	17 09 other construction and demolition wastes	17 09 01* construction and demolition wastes containing mercury 17 09 02* construction and demolition wastes containing PCB (for example PCB-containing sealants, PCB-containing resin-based floorings, PCB-containing sealed glazing units, PCB-containing capacitors) 17 09 03* other construction and demolition wastes (including mixed wastes) containing hazardous substances 17 09 04 mixed construction and demolition wastes other than those mentioned in 17 09 01, 17 09 02 and 17 09 03
18 WASTES FROM HUMAN OR ANIMAL HEALTH CARE AND/OR RELATED RESEARCH (except kitchen and restaurant wastes not arising from immediate health care)	18 01 wastes from natal care, diagnosis, treatment or prevention of disease in humans	18 01 01 sharps (except 18 01 03) 18 01 02 body parts and organs including blood bags and blood preserves (except 18 01 03) 18 01 03* wastes whose collection and disposal is subject to special requirements in order to prevent infection 18 01 04 wastes whose collection and disposal is not subject to special requirements in order to prevent infection (for example dressings, plaster casts, linen, disposable clothing, diapers) 18 01 06* chemicals consisting of or containing hazardous substances 18 01 07 chemicals other than those mentioned in 18 01 06 18 01 08* cytotoxic and cytostatic medicines 18 01 09 medicines other than those mentioned in 18 01 08 18 01 10* amalgam waste from dental care
	18 02 wastes from research, diagnosis, treatment or prevention of disease involving animals	18 02 01 sharps (except 18 02 02) 18 02 02* wastes whose collection and disposal is subject to special requirements in order to prevent infection 18 02 03 wastes whose collection and disposal is not subject to special requirements in order to prevent infection 18 02 05* chemicals consisting of or containing hazardous substances 18 02 06 chemicals other than those mentioned in 18 02 05 18 02 07* cytotoxic and cytostatic medicines 18 02 08 medicines other than those mentioned in

Table 1-3. Categories of waste to be considered under the European List of Wastes (EC, 2014)

Chapter	Subchapter	Category
		18 02 07
20 MUNICIPAL WASTES (HOUSEHOLD WASTE AND SIMILAR COMMERCIAL, INDUSTRIAL AND INSTITUTIONAL WASTES) INCLUDING SEPARATELY COLLECTED FRACTIONS	20 01 separately collected fractions (except 15 01)	20 01 01 paper and cardboard 20 01 02 glass 20 01 08 biodegradable kitchen and canteen waste 20 01 10 clothes 20 01 11 textiles 20 01 13* solvents 20 01 14* acids 20 01 15* alkalines 20 01 17* photochemicals 20 01 19* pesticides 20 01 21* fluorescent tubes and other mercury-containing waste 20 01 23* discarded equipment containing chlorofluorocarbons 20 01 25 edible oil and fat 20 01 26* oil and fat other than those mentioned in 20 01 25 20 01 27* paint, inks, adhesives and resins containing hazardous substances 20 01 28 paint, inks, adhesives and resins other than those mentioned in 20 01 27 20 01 29* detergents containing hazardous substances 20 01 30 detergents other than those mentioned in 20 01 29 20 01 31* cytotoxic and cytostatic medicines 20 01 32 medicines other than those mentioned in 20 01 31 20 01 33* batteries and accumulators included in 16 06 01, 16 06 02 or 16 06 03 and unsorted batteries and accumulators containing these batteries 20 01 34 batteries and accumulators other than those mentioned in 20 01 33 20 01 35* discarded electrical and electronic equipment other than those mentioned in 20 01 21 and 20 01 23 containing hazardous components (*) 20 01 36 discarded electrical and electronic equipment other than those mentioned in 20 01 21, 20 01 23 and 20 01 35 20 01 37* wood containing hazardous substances 20 01 38 wood other than that mentioned in 20 01 37 20 01 39 plastics 20 01 40 metals 20 01 41 wastes from chimney sweeping 20 01 99 other fractions not otherwise specified
	20 02 garden and park wastes (including cemetery waste)	20 02 01 biodegradable waste 20 02 02 soil and stones 20 02 03 other non-biodegradable wastes

Table 1-3. Categories of waste to be considered under the European List of Wastes (EC, 2014)

Chapter	Subchapter	Category
	20 03 other municipal wastes	20 03 01 mixed municipal waste 20 03 02 waste from markets 20 03 03 street-cleaning residues 20 03 04 septic tank sludge 20 03 06 waste from sewage cleaning 20 03 07 bulky waste 20 03 99 municipal wastes not otherwise specified

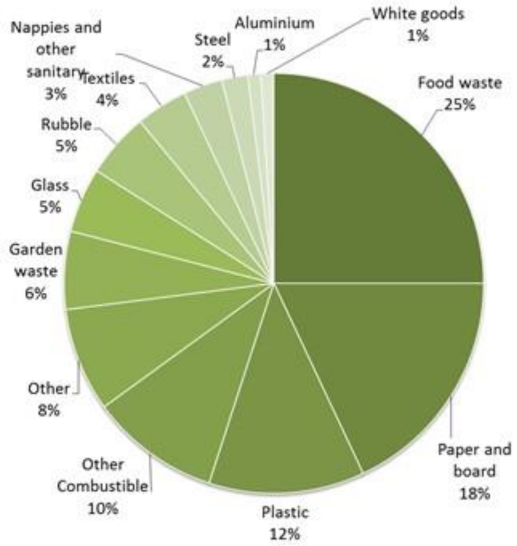
(*) Hazardous components from electrical and electronic equipment may include accumulators and batteries mentioned in 16 06 and marked as hazardous, mercury switches, glass from cathode ray tubes and other activated glass, etc.

Municipal solid waste

According to Eurostat (2012), **municipal solid waste** (MSW) is waste "*mainly produced by households, though similar wastes from sources such as commerce, offices and public institutions are included. This municipal waste consists of waste collected by or on behalf of municipal authorities and disposed of through the waste management system"*. This definition is used mainly for reporting purposes under the Waste Framework Directive or the Landfill Directive. MSW is thus the waste generated from households as well as other waste which, because of its nature or composition, is similar to waste from households and is collected and treated together with waste from households. In terms of weight, only 10 % of the total amount of waste can be considered MSW. Its special consideration in all waste regulations and policies comes from its highly political character due to its complexity, its composition, dispersed generation and the obvious link to the consumption patterns of communities. From 60 % to 90 % of total MSW comes from households, and the rest from commercial activities with a similar waste composition to households (e.g. offices, administration services, schools).

However, in 2013 the European Environment Agency (EEA) found that European countries have very different approaches to the definition and quantification of these wastes, which even poses a challenge to the study of different waste prevention and diversion policies (EEA, 2013b). One example is how to take into account gardening waste or bulky waste. More importantly, packaging waste seems to be accounted for in very heterogeneous ways in Europe. While some countries include all packaging from municipal waste in the municipal waste category, some of them separate out the packaging waste considered in the producer responsibility schemes. The same happens for waste under other producer responsibility schemes, such as WEEE (Waste from Electrical and Electronic Equipment) or batteries.

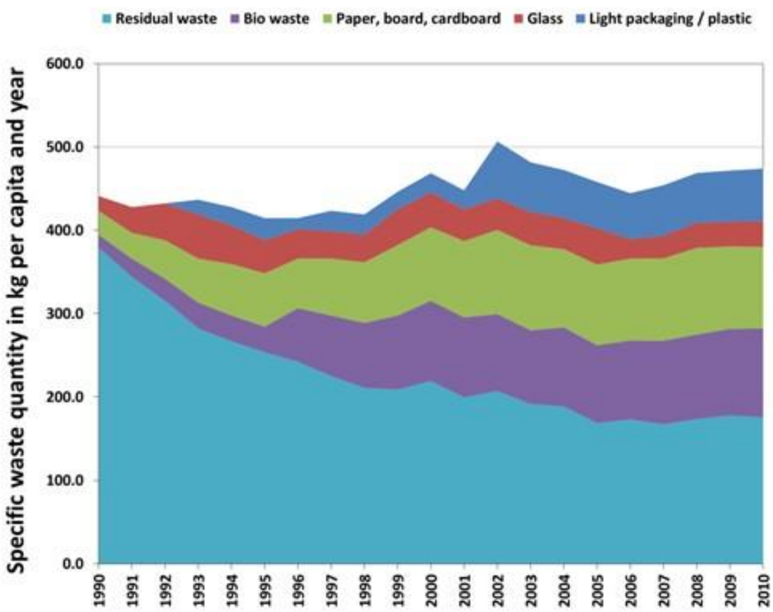
As this document focuses on environmental management practice, the most appropriate definition is according to the "nature" or "compositional" characteristics of the waste. The typical qualitative composition of municipal waste (Figure 1-19) is used to classify materials and practices described in this document.



Source: Zero Waste Europe, 2015

Figure 1-19. Sample composition of municipal solid waste in Europe

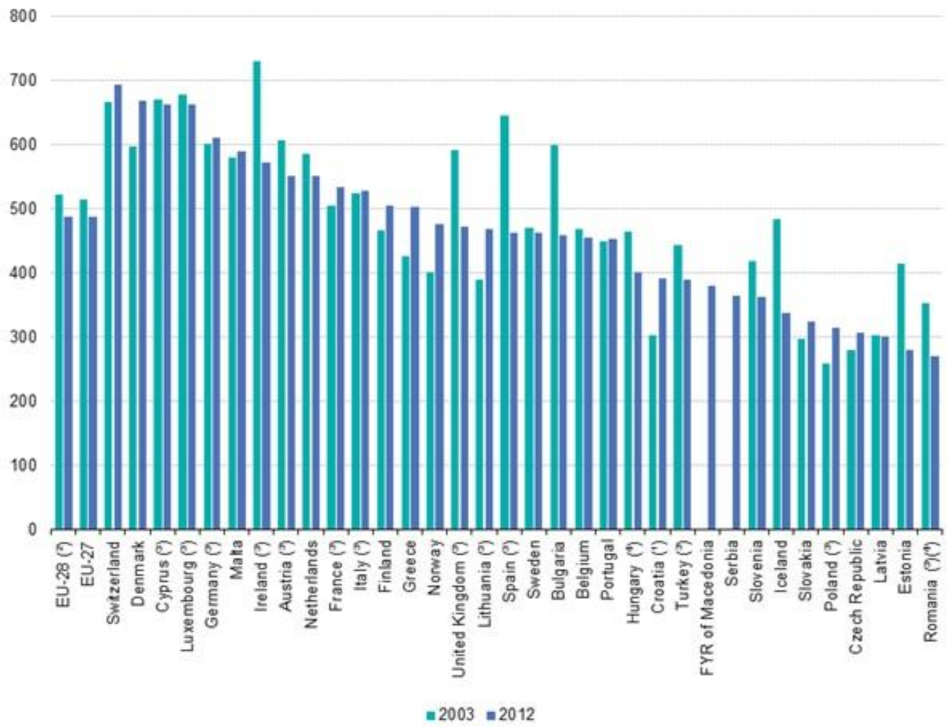
During the last 25 years, a huge change has taken place in the way municipal waste is managed. Many countries (see, for instance, data for Germany in Figure 1-20) have reduced the production of unsorted residual waste, thanks to the separate collection of recyclable fractions, such as paper, glass and plastics. Also, organic waste collection schemes have been introduced, aimed both at recovering nutrients from organic waste and avoiding the emissions from landfilling. During the last 10 years, the relative proportions of these fractions have not changed considerably.



Source: Eurostat, 2014

Figure 1-20. Development of the quantities of certain waste fractions in Germany from 1990 to 2010

Figure 1-21 shows the change in total MSW generation per capita in European countries between 2003 and 2012. In several countries, this has decreased.



Source: Eurostat (2014)

(1) No data for 2002; 2004 data instead. (2) No data for 2003; 2007 data instead. (3) 2012 data estimates. (4) 2003 data estimates.

Figure 1-21. Municipal waste generated by country in 2003 and 2012 in kg per capita and year, and sorted by 2012

The current historical statistical data only allows the classification of waste treatments under four categories: landfill, incineration (also called "waste-to-energy", WtE, when incineration includes energy recovery), recycling and composting. Eurostat includes the category "others" in order to compensate the mass balance caused by statistical methodologies (e.g. how Member States consider the input to mechanical and biological treatment, MBT, plants has a significant influence in countries like Germany, the UK or Italy). Looking at incineration statistics from 1995 until the introduction of the WFD and the application of the energy efficiency criterion in 2010, it is not possible to differentiate between incineration plants with energy recovery and plants without energy recovery. The same happens with composting, which includes any biological treatment, composting and fermentation. Figure 1-22 shows the development of these different waste treatment categories in Europe since 1995 (data from Eurostat). In 1995, 63 % of MSW was landfilled, but this amount decreased to 34 % in 2012 (around 164 kg per capita per year). However, the total amount of waste generated increased until the year 2007. The decrease in the per capita generation of MSW in the years 2010-2012 is explained as a consequence of the economic crisis and its impact on consumption and not because of the success of waste prevention policies.

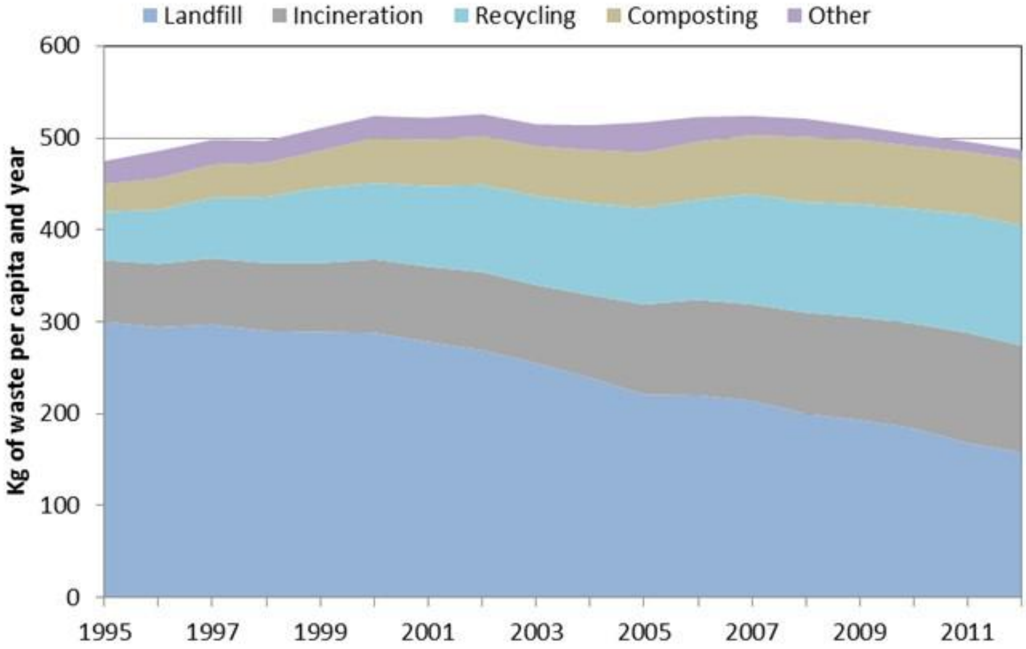


Figure 1-22. Municipal waste by type of treatment, EU-27 in kg per capita and year[16] (Data from Eurostat, 2014)

Waste management strategies at national level are oriented to divert waste from landfill as a consequence of the ambitious objectives of the Landfill Directive. There are countries where priority is given to recycling, while others are implementing incineration. The existence of national regulations also has a strong effect on the share of different waste treatment/disposal options. For instance, in the Netherlands, Sweden and Denmark, landfilling any combustible waste is banned, and Belgium, Austria and Germany have banned the landfilling of any untreated waste. As a consequence, these countries do not landfill any municipal waste (see Table 1-4).

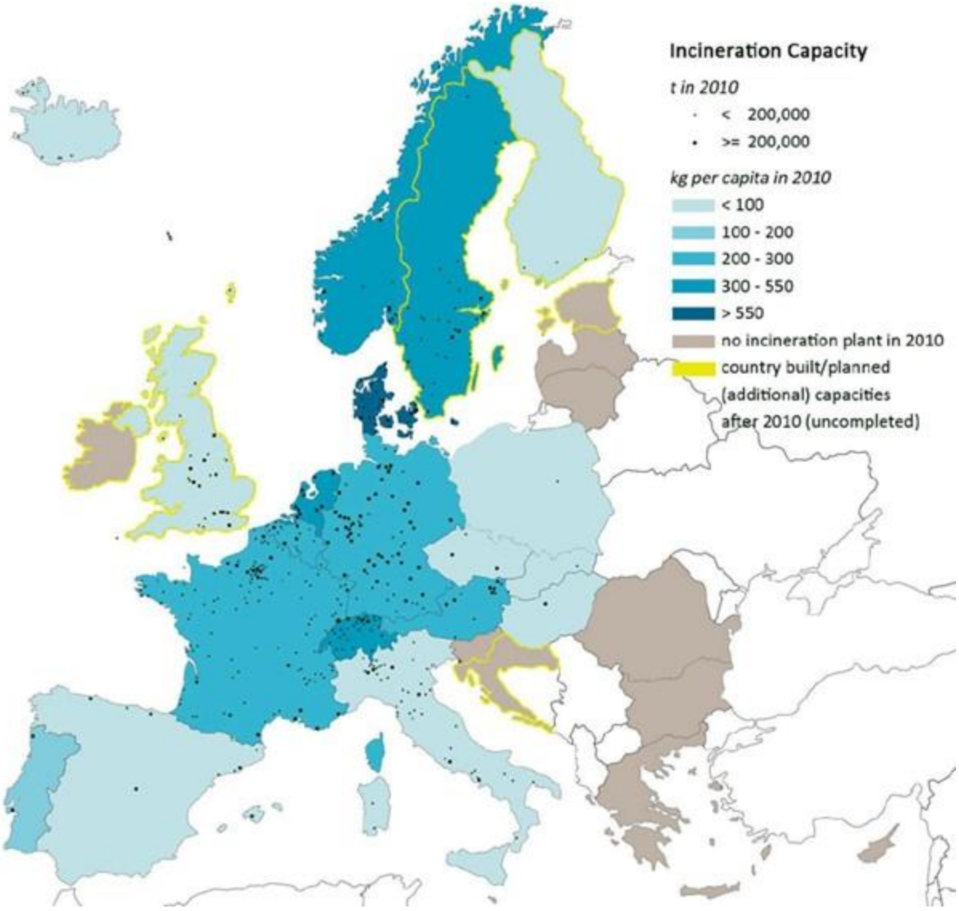
[16] As Croatia has only been a Member State of the European Union since 1 July 2013 it is therefore not included here.

Table 1-4. Landfill bans in Member States (Adapted from Stengler, 2014)

Member state	Disposal [%]	WtE [%]	Recycling / Composting [%]	Ban on landfilling
Netherlands	1	38	60	Since 1995 for 35 types of waste
Denmark	3	54	43	Since 1997 for biologically degradable waste
Sweden	1	51	48	Since 2002 for separated combustible waste Since 2005 for organic waste
Belgium	1	42	56	Since 2004 in Wallonia for household waste, sludge, bottom ash, waste with a high content of biodegradables Since 2006 in Flanders for combustible household waste and industrial / commercial waste (exceptions possible until 2015) Since 2007 throughout Belgium for untreated waste, including biodegradable municipal waste
Austria	3	35	62	Since 2004 for biodegradable municipal waste Since 2008 for waste with > 5 % TOC. Exception: Mechanically and biologically treated waste with a net calorific value ≤ 6.6 $MJ/kg_{d.m.}$ (and TOC < 8 %)
Germany	1	37	62	Since 1.6.2005 for untreated municipal waste

The large differences among European countries are a result of the implementation time of waste policies. Those countries with the lowest landfilling rates are those with a historically-long political aim and investment schemes, while the others show a similar evolution but just delayed. The geographical disparities in Europe are quite evident and reflect the level of economic development and the level of investment in environmental policies, as well as the different historical approaches in waste management. Wilts and von Gries recently published an ETC/SCP[17] Working Paper in which they analysed the capacities for municipal waste management in Europe (Wilts and von Gries, 2014). Most European countries have an incineration capacity of less than a quarter of their municipal solid waste generation, but in some specific regions there is a certain overcapacity, which is increasing imports and creating a barrier for recycling through the so-called vacuum cleaner effect, especially for commercial waste. The current incineration plants and the incineration capacities of European countries are shown in Figure 1-23.

[17] European Topic Centre on Sustainable Consumption and Production



Source: Wilts and von Gries (2014)

Figure 1-23. Incineration capacity and incinerators in Europe

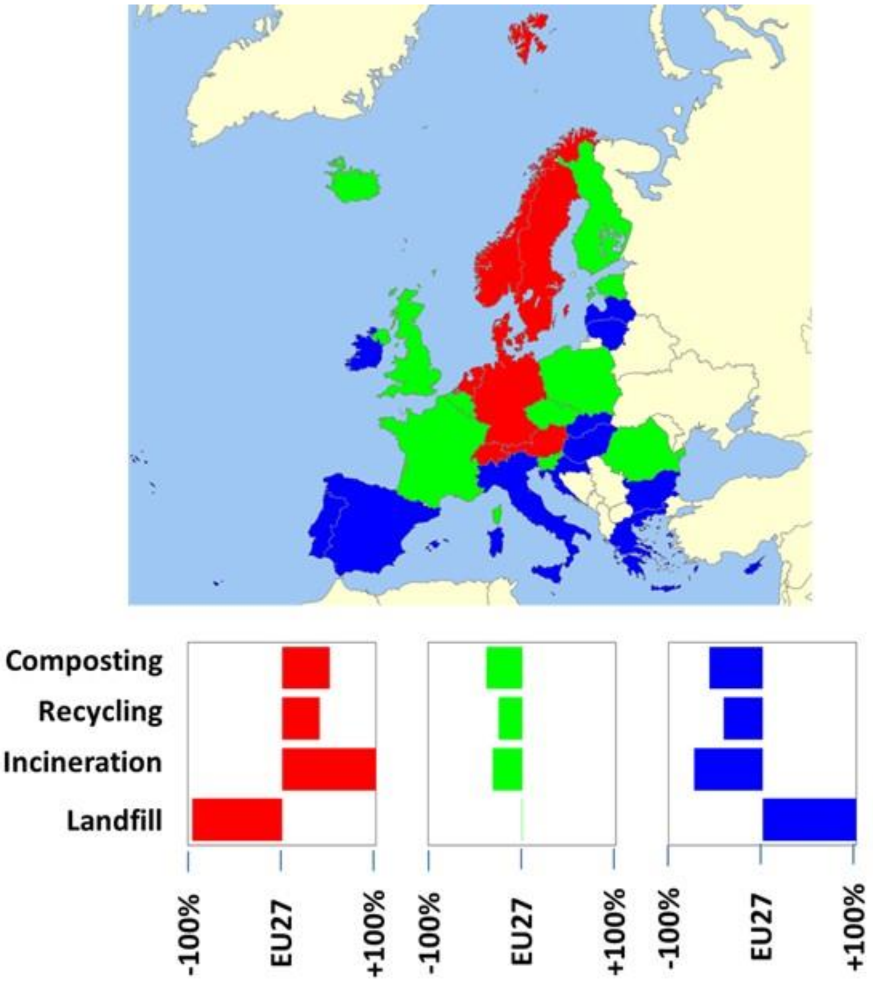
Source: **Eurostat, 2013**

Figure 1-24 shows the geographical distribution of different waste treatment strategies. Red represents the countries where almost no untreated waste is landfilled and where incineration, materials recycling and composting are more developed than the European average. Green represents the countries with the same average as the EU-27 average (around 34 % of total waste), where some improvement can still be achieved in other treatments. Countries with very high landfill rates and still lacking incineration, recycling or composting capacity are represented in blue. The data are taken from the last statistical survey done by Eurostat for 2012 and the countries are grouped by their landfilling rate (ordered from smallest to largest)[18]. Source: **Eurostat, 2013**

Figure 1-24 shows how the treatment strategy differs across Europe. This chart also reveals where the best practices are most likely to be found. Countries like Germany, Denmark, Netherlands, Sweden, etc. have applied a zero landfill policy very

[18] In this analysis, the composition of the groups is different to the clusters designed by Eurostat to analyse the data.

successfully during the last 10 to 20 years. Others, with very similar policies, have applied them with less intensity, as in the case of France, or the investment has been relatively delayed, as in the UK.



Source: Eurostat, 2013

Figure 1-24. Geographical distribution of waste treatment practices, compared to the EU-27 average. Colour classification highlights waste management differences per capita in EU.

Most of the investment of national waste strategies has been directed to better waste treatments, e.g. by avoiding waste landfilling and increasing material recovery. However, the application of better treatment technologies is not intended (primarily) to reduce the total amount of waste generated. Also, it can be observed that those countries with outstanding performances in waste treatment compared to the European average are those with an on average higher municipal waste generation. This can be seen in Figure 1-25, where the generation of waste is represented along with the rate of landfilling. The red line is the moving average of waste generated per capita yearly, showing the average of the previous six data points, i.e. the six previous country MSW generation values per capita. The maximum corresponds to around 550 kg per habitant and year, due to the average of countries with reduced landfilling practices, reaching a minimum for those with a much higher landfilling rate (390 kg/year per capita). This effect has also been acknowledged by Eurostat in its data, although it recognises that data inconsistency and data management can have an influence on this result. However, the general trend is confirmed over the years and is due to the higher waste generation in countries with higher consumption patterns.

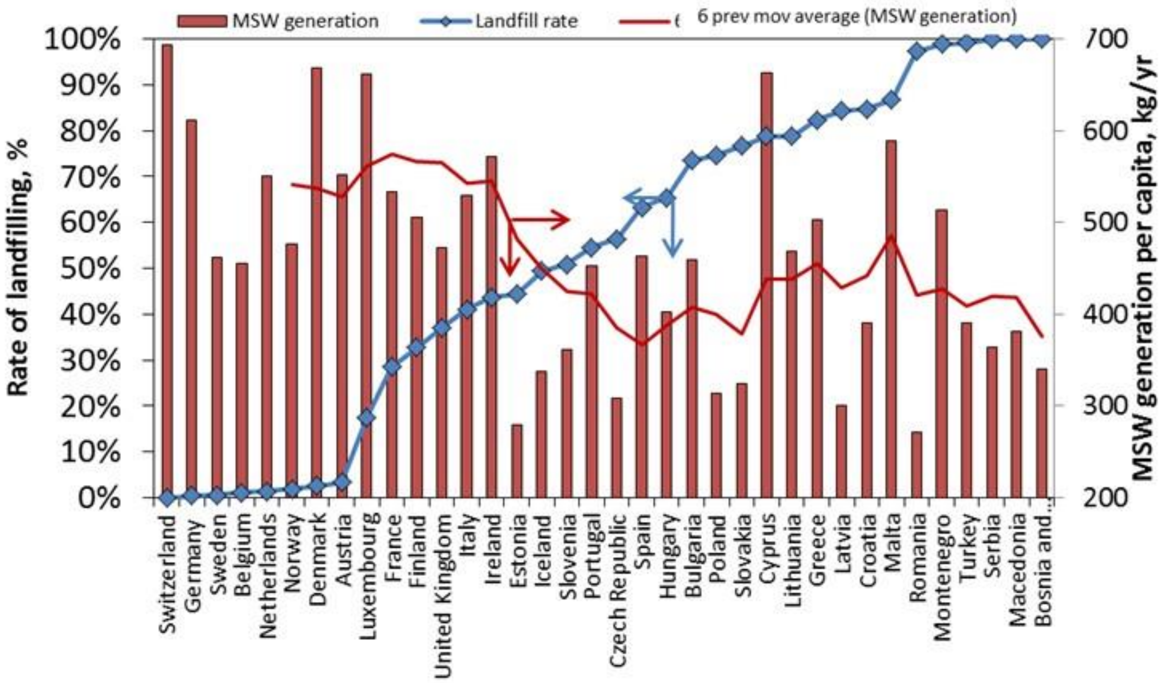


Figure 1-25. Rate of landfilling and MSW generation in 2012 for European countries. The red line plots the average of the six previous values of MSW generation (moving average) (Data from Eurostat, 2013)

Packaging waste, one of the main components of MSW, is covered by the European Directive on packaging and packaging waste (94/62/EC). For these fractions, very specific objectives have been set (see Table 1-5). In general, except for some exemptions, the recovery and recycling targets have been achieved. The packaging waste collected separately amounts to 159 kg per capita per year and has been kept constant in the last decade. In total, 63.5 % of packaging waste was recycled in 2011 and 77.3 % was recovered (including recycling plus incineration with energy recovery) (Eurostat, 2013).

Table 1-5. Second stage recovery and recycling targets of the Packaging and Packaging Waste Directive and years in which targets must be achieved

Country (EU-27)	**Recovery**	**Recycling**					
	Target: 60 %	Overall target: 55-80 %	Glass: 60 %	Paper and board: 60 %	Metals: 50 %	Plastics: 22.5 %	Wood: 15 %
Belgium, Denmark, Germany, Spain, France, Italy, Luxembourg, Netherlands, Austria, Finland, Sweden, UK	2008						
Greece, Ireland, Portugal	2011						
Czech Republic, Estonia, Cyprus, Lithuania, Hungary, Slovenia, Slovakia	2012						
Malta	2013						
Poland	2014						
Latvia	2015						
Bulgaria	2014	2014	2013	2008	2008	2013	2008
Romania	2013	2013	2013	2008	2008	2013	2011

However, these objectives do not take into account reuse practices as defined by the WFD. For instance, wood pallets are the main component of wood packaging waste. Current practices with wood pallets include a high rate of reuse through deposit schemes with the industry. A similar situation can be found for reusable glass bottles, which are not taken into account as recycling or reuse. This may be the main reason for disparities in glass recycling in Nordic countries (Eurostat, 2014).

Construction and demolition waste

Construction and demolition waste (CDW) is a very broad definition for all the waste generated by the construction, maintenance, demolition and selective deconstruction of buildings and civil works. Its nature varies and depends on the construction project that generates the waste. For instance, road construction creates a huge amount of excavated material, usually inert, that can be considered waste if it needs to be disposed of, but contractors tend to reuse these materials as fillings in the same or other road construction, reducing the waste treatment fee and the resources consumed. The heterogeneity of construction activities, along with different consumption patterns, makes it almost impossible to define a typical composition in this regard. For that reason, in the context of this work, construction and demolition waste is considered as any waste generated in the activities of companies belonging to the construction sector (NACE divisions 41, 42 and 43) and included in category 17 of the European List of Wastes (see Table 1-3), comprising mainly concrete, ceramic and bituminous waste. Other fractions fall into the scope of commercial waste in MSW management (e.g. packaging), or other schemes (take-back system for wood pallets, recycling for metals, etc.).

In total, approximately 800 million tonnes of construction and demolition waste were recorded for the year 2012 in Europe according to Eurostat, which is 34 % of the total waste generated. However, the majority of this waste is inert excavated soil, which has almost no impact on the environment. Around 50 million tonnes of actual construction and demolition waste were generated in 2010 at European construction sites (new construction, demolition or refurbishment). Depending on the nature of the construction project, concrete waste makes up around 40 % to 85 % of the total waste generated on site (Rimoldi, 2010). "Clean" concrete waste is rerely reusable and its recycling produces a downgraded product, aggregates, as recovery of initial constituents is not feasible. Recycled concrete aggregates, RCA, are usable for the so-called unbound applications (e.g. road sub-base fillings) or as secondary materials in the manufacture of new concrete.

Concrete is the most used material in the world. Its success relies on three key factors: durability, affordability and the availability of raw materials. In that sense, the low cost of extracted natural aggregates is one of the main drawbacks for the uptake of secondary materials, as extracted resources would have similar costs to recycled aggregates. Also, there is no scarcity of raw materials and the economical relevance of the total cost of aggregates in the final product is quite low. The environmental impact of natural and recycled aggregates, e.g. in terms of greenhouse gas emissions, is highly dependent on the transport. These factors contribute to a very different scenario for CDW compared to other wastes, and require different driving forces (i.e. regulation, taxation, etc.) for best practice implementation.

A popular myth regarding the application of recycled aggregates in concrete is that these aggregates have a much lower performance than natural aggregates. It is proven that, given a proper waste separation, the quality of certain fractions of recycled concrete aggregates, RCA, can substitute 100 % natural aggregates. Even, in some cases, for structural applications, a 20-30 % replacement can be done without any impact on performance.

Europe consumes around 3 billion tonnes of aggregates (European Concrete Platform, 2007). In the UK, 25 % of the aggregates market came from secondary sources or recycled materials in 2007 (The Concrete Centre, 2009) and there are no technical barriers for the recycling of CDW. Aggregates from masonry and ceramic wastes, even mixed with concrete, are less applicable, but their volume is certainly smaller and many applications have succeeded. Several showcases around Europe showed more than 95 % CDW recycling (European Commission, 2012) and simplified the market barriers to (i) availability, (ii) economics and (iii) acceptability. The profit margin on recycled aggregates also depends on the location of the source, which has to be closer than other quarries, and the tax schemes for landfill and natural aggregates extraction (UEPG, 2006). Denmark and the Netherlands have been very successful in promoting the recycling of CDW.

CDW generation is linked to the construction activity and the amount of waste per unit of built, demolished or refurbished area is often used as an indicator and easily benchmarked against different types of structures, construction techniques and traditional practices. For instance, precast and prefabricated structures generate less waste, as the manufacturing process is less wasteful and designs are specific for each building. At the same time, the expected amount of CDW and its composition are very different if timber or reinforced concrete structures are used. Mália et al. (2013) calculated the range of CDW generation for different types of building projects and structures (Table 1-6 and Table 1-7).

Table 1-6. CDW generation rates per waste type and activity, in kg/m²

Waste	New residential construction		New non-residential construction		Residential demolition		Non-residential demolition	
	Timber structure	**Reinforced concrete**	**Timber structure**	**Reinforced concrete**	**Timber structure**	**Reinforced concrete**	**Un-defined**	**Rein-forced concrete**
17 01 01 Concrete	0.3 – 1.9	17.8 – 32.9		18.3 – 40.1	137 – 300	492 – 840		401 – 768
17 01 02 Bricks	0.5 – 0.8	19.2 – 58.6		15.6 – 54.3	84 – 90	170 – 486	176 – 438	
17 01 03 Tiles	-	1.7 – 3.2		0.4-3.2	-	10.6 – 17.6	16 – 27	
17 02 01 Timber	0 – 2	2.5 - 6.4	4.7 – 10.7	1.7 – 5.4	70-275	12 – 58		20 – 159
17 02 02 Glass	0.0 – 0.3		0.0 – 0.8		0.4 – 2.6		0.2 – 4.4	
17 02 03 Plastics	0.1 – 0.8		0.3 – 1.9		0.4 – 5.6		0.4 – 6.1	
17 03 02 Bituminous mixtures	0.4 – 2.6		0.7 - 6.6		1.0 – 1.4		1.0 – 1.4	
17 04 07 Metal mixtures	0.1 – 0.9	0.9 – 3.9	0.2 – 2.9	1.0 – 7.2	4.8 – 22.5	9.8 – 28.4	3.4 – 55.0	25.4-53.0
17 06 04 Insulation Materials	0.1 – 1.2		0.1 – 1.5		0.1 – 2.2		0.1 – 2.2	
17 08 02 Gypsum-based	2.4 – 7.2	3.7 – 7.6	0.5 – 3.4	10.8 – 81.3	10.9 – 105.4	10.8 – 64.3	10.8 – 81.3	10.8 – 75.7
17 09 03 CDW containing hazardous substances	0.02 – 0.33		0.01 – 0.74		0.4 – 0.6		0.2 – 0.6	
Total	**10 – 39**	**44 - 115**		**48 - 135**	**195 – 725**	**805 – 1,371**	**600 – 1,750**	**742 – 1,637**

Source: Mália et al. (2013)

Table 1-7. (Continues from Table 1-6) CDW generation rates per waste type and activity, in kg/m²

Waste	Residential refurbishment	Non-residential refurbishment
17 01 01 Concrete	18.9 – 45.9	18.9 – 191.2
17 01 02 Bricks	63.3 – 319.5	11.2 – 62.0
17 01 03 Tiles	1.1 – 12.6	0.2 – 16.9
17 02 01 Timber	2.0 – 37.9	23 – 42.6
17 02 02 Glass	0.2 – 1.4	0.3 – 0.9
17 02 03 Plastics	0.6 – 1.3	1.9 – 2.6
17 03 02 Bituminous mixtures	12	8 -12
17 04 07 Metal mixtures	0.4 – 6.8	0.2 – 16.4
17 06 04 Insulation Materials	0.1 – 0.6	0.1 – 0.6
17 08 02 Gypsum-based	2.4 – 23.5	2.3 -22.9
17 09 03 CDW containing hazardous substances	0.03 – 0.05	0.03 – 0.05
Total	**28 – 397**	**20 - 326**

The main waste fraction is made up of concrete (more than 50 % in most cases) and masonry. Gypsum-based materials, timber and metal are also of relevance in the final mass of wastes. The mineral fraction of construction waste constitutes category 12.1 of the European Regulation on waste management statistics. In 2012, Member States reported the treatment of this fraction as shown in Figure 1-26.

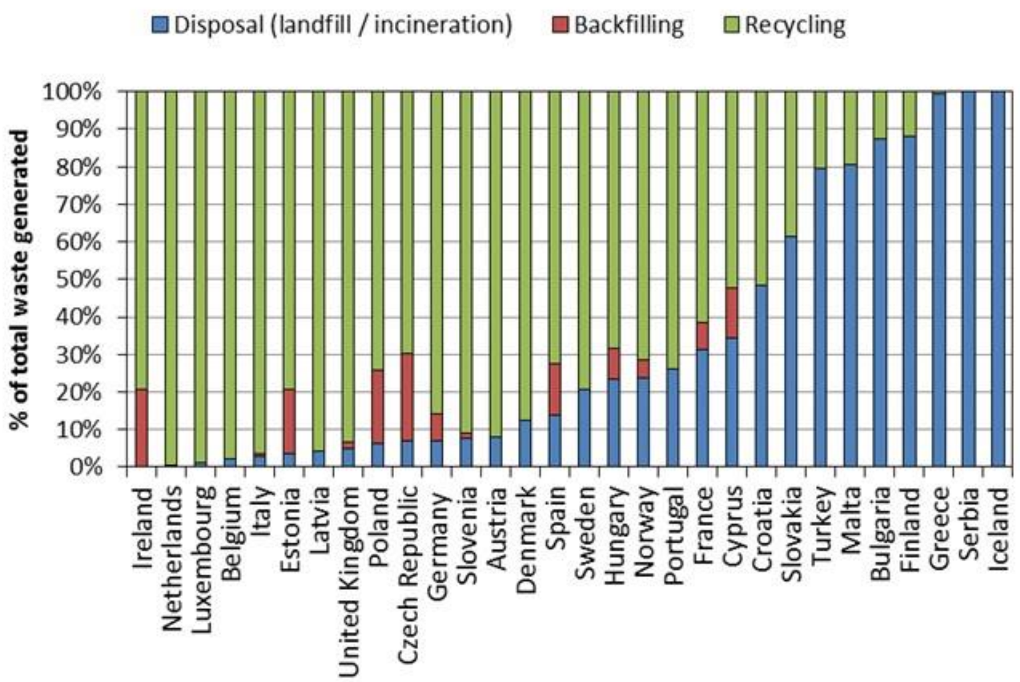


Figure 1-26. Construction and demolition waste mineral fraction treatment in 2012 (Data from Eurostat, env_wasgen, 2013)

As observed, many countries have already achieved the objective of 70 % recycling for this waste fraction. The total mass flow of recovered waste accounts for more than 80 % of the total waste generation. However, the different methodologies observed for municipal solid wastes in the previous section also apply to these results. The existence of illegal dumping and the different management approaches among countries are also relevant: while there are countries with high recycling rates, the market uptake of recycled materials is very low. Large storage areas of treatment plants have been converted into temporary landfills (EC, 2012).

Healthcare waste

Healthcare waste refers to waste generated in the operation of health services for humans and animals: diagnosis, treatment and immunisation of humans and animals, as well as in scientific research, biological production, and testing. A large part of healthcare waste is considered hazardous, because it may contain toxic materials and/or pathogenic agents that require special handling. Other waste fractions generated by the facilities of health institutions will be considered according to their nature or composition (e.g. waste electrical and electronic equipment or MSW-like waste).

Due to the difficulty to report exclusively waste generated only by medical activities, statistical data usually includes any waste that arises from healthcare activity and focuses on:

- infectious waste:
    - anatomical;
    - sharps;
    - blood;
    - pharmaceutical;
    - radioactive materials;
- offensive/hygienic waste;
- MSW-like waste.

This waste is commonly generated by hospitals from the public or the private sector, nursing homes, doctors' surgeries, dentists, pharmacists and veterinary clinics. Other smaller generators would include public parks, first aid and washrooms in public areas and retail or hospitality premises. The non-hazardous fraction of the waste varies from 40 % to 60 % of the total waste, but the MSW-like waste cannot be determined with accuracy due to the different approach in segregation.

Hazardous waste has to be disposed of safely. The Health Technical Memorandum 07-01 (Department of Health, 2007) of the UK government defines "a rendered safe [treatment] is an accepted method or process that has been applied which:

a. demonstrates the ability to reduce the number of infectious organisms present in the waste to a level at which no additional precautions are needed to protect workers or the public against infection from the waste,

b. destroys anatomical waste such that it is no longer generally recognisable,

c. renders all clinical waste (including any equipment and sharps) unusable and unrecognisable as clinical waste,

d. destroys the component chemicals of chemical or medicinal and medicinally-contaminated waste".

Suitable treatments for healthcare waste are divided into high-temperature processes and alternative treatments:

- High-temperature treatments:
    - Incineration: a primary combustion chamber operating at 800 – 1 000 °C and a second chamber operating at 850 – 1 100 °C.
    - Pyrolysis: involves thermo-chemical cleavage of waste at 545 – 1 000 °C without oxygen.
    - Plasma: the waste is treated at temperatures of 1 300 – 1 700 °C and converted to a glass-like material.
    - Gasification: the materials decompose in the presence of a substoichiometric amount of oxygen for combustion. The process is energetically self-sustained.
- Alternative treatments (usually referred to as non-combustion treatments) reduce or eliminate the hazardous component of the waste. Examples of these are as follows:
    - Heat treatment, intended to sterilise the infectious material: autoclaves, steam augur, dry heat treatment, microwave or radiofrequency sterilisation, etc.
    - Chemical treatment: uses chemical substances to sterilise the infectious materials: e.g. hypochlorite, chlorine dioxide, peracetic acid.

The suitability of each treatment to each HCW stream is shown in Table 1-8.

Table 1-8. Treatment type per healthcare waste stream

Waste	Code	Treatment
Clinical (chemicals)	18-01-03	High-temperature
Clinical (swabs, soiled dressings, gloves, etc.)	18-01-03	Alternative
Sharps	18-01-01	High-temperature
Anatomical	18-01-02	High-temperature
Offensive (e.g. diapers)	18-01-04	Alternative
Cytotoxic and cytostatic	18-01-08	High-temperature (> 1 000 °C)
Medicines	18-01-09	High-temperature

Source: Tudor et al. (2009)

For non-hazardous waste (clinical or non-clinical), segregation at source can increase the fraction recovered. Current practices in the UK indicate that most of the recyclable waste is not well sorted and is fed to the high-temperature incinerators as a support fuel to improve the efficiency.

Eurostat in 2014 reported the data shown in Figure 1-27 for the year 2012. The level of reporting of Member States for HCW seems heterogeneous and the quantities per capita are not comparable. The generation of waste and treated waste do not match. In total, for the countries reported in Figure 1-27, about 2.7 million tonnes of waste were generated, while 1.4 million tonnes were reported as treated. The difference probably is due to different methods for the quantification of the MSW-like waste.

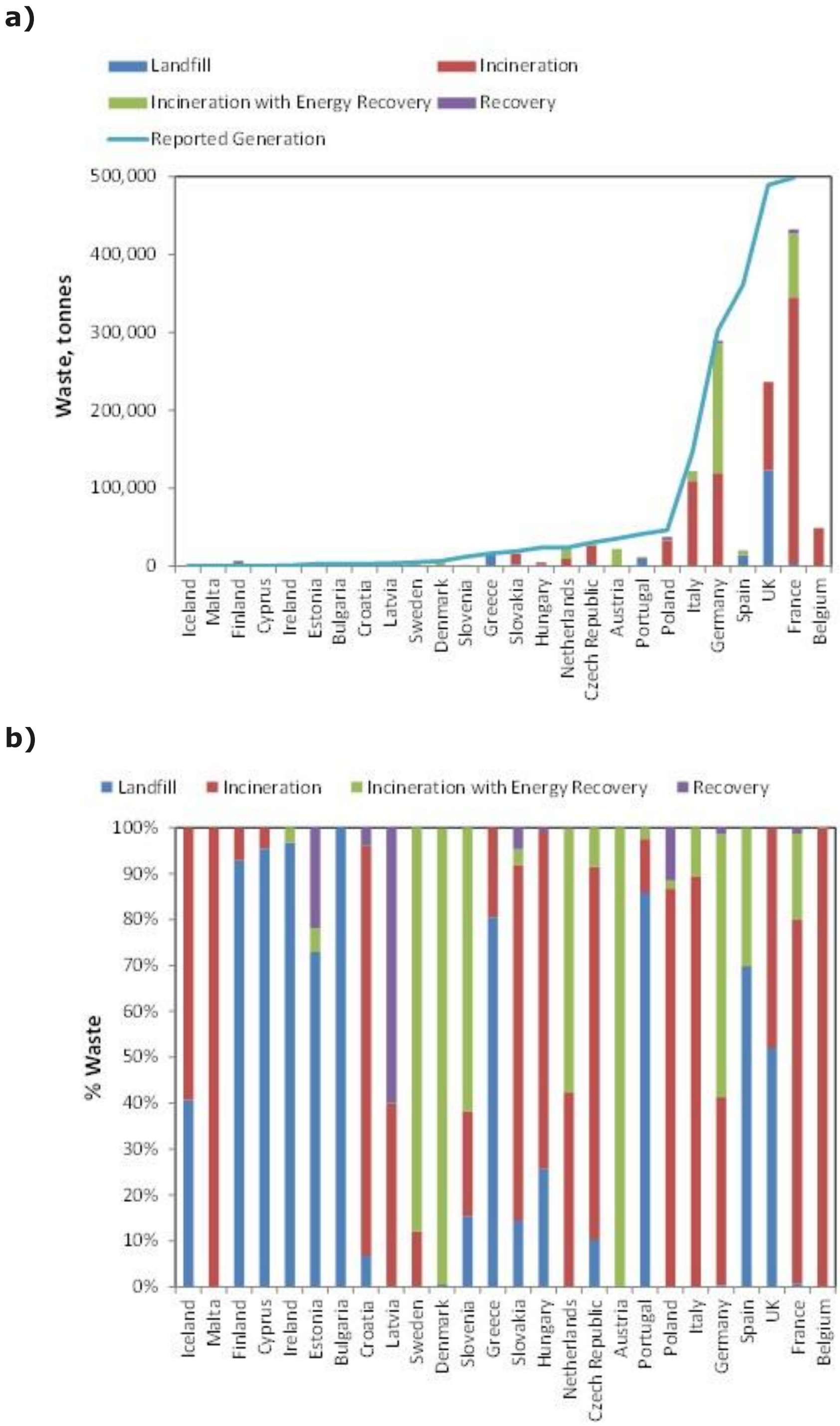


Figure 1-27. Healthcare waste generation and treatment in Europe (a) in tonnes and (b) as a percentage of the total. (Data from Eurostat, env_wasgen, 2013)

The World Health Organisation (2014) estimates that a total of 85 % of HCW generated in a hospital is non-hazardous and, with some exemptions, could be managed under other schemes (e.g. for MSW). Sengupta (1990) published a survey of more than 230 healthcare facilities in Florida, United States, developing several indicators for different healthcare facilities (Table 1-9).

Table 1-9. Survey results of HCW generation in Florida, United States

Healthcare facility	Total HCW generation	Infectious waste generation
Metropolitan general hospitals	10.7 kg/occupied bed/day	2.79 kg/occupied bed/day
Rural general hospitals	6.40 kg/occupied bed/day	2.03 kg/occupied bed/day
Psychiatric and other hospitals	1.83 kg/occupied bed/day	0.043 kg/occupied bed/day
Nursing homes	0.90 kg/occupied bed/day	0.038 kg/occupied bed/day
Laboratories	7.7 kg/day	1.9 kg/day
Doctor’s office (group practice, urban)	1.78 kg/physician-day	0.67 kg/physician-day
Doctor’s office (individual, urban)	1.98 kg/physician-day	0.23 kg/physician-day
Doctor’s office (rural)	0.93 kg/physician-day	0.077 kg/physician-day
Dentist’s office (group practice)	1.75 kg/dentist-day	0.13 kg/dentist-day
Dentist’s office (individual)	1.10 kg/dentist-day	0.17 kg/dentist-day
Dentist’s office (rural)	1.69 kg/dentist-day	0.12 kg/dentist-day
Veterinarian (group practice, metropolitan)	4.5 kg/veterinarian-day	0.66 kg/veterinarian-day
Veterinarian (individual, metropolitan)	0.65 kg/veterinarian-day	0.097 kg/veterinarian-day

Source: Sengupta (1990), as cited by WHO (2014)

In Europe, there are several national EPR schemes attending to healthcare waste for old or unused medicines (Austria, Belgium, Finland, France, Portugal, Sweden, Spain, Hungary, Slovenia, Estonia) managing around 240 000 tonnes of healthcare waste (Monier et al., 2014). The treatment usually consists of separation and recovery of the packaging material and the incineration of the medicine, which in some cases can be considered hazardous.

1.3. Main environmental aspects and environmental relevance of the waste management sector

Waste disposal leads to direct environmental impacts, such as land occupation, resource depletion, amplification of global warming due to methane and other greenhouse gas emissions, eutrophication and ecotoxicity in waters from leachate in the case of landfilling, or resource depletion, and acidification and ecotoxicity effects from emissions to air in the case of incineration. Direct emissions from waste management represent a significant but comparatively small share of European climate change, acidifying, eutrophying and toxic emissions, as summarised in the sections below, although toxicity effects can be locally important.

However, resource depletion is linked with highly significant indirect environmental impacts associated with resource extraction and processing to compensate for materials removed from circulation in the economy. Full implementation of the waste management hierarchy, including waste prevention and reuse wherever possible, can avoid considerable environmental impacts when assessed from a life-cycle perspective – considering direct and indirect effects.

Table 1-10 summarises the main environmental aspects and impacts linked with some of the primary activities undertaken and services provided by the waste management sector. As per the EMAS Regulation, "environmental aspect" refers to an element of an organisation's activities, products or services that has or can have an impact on the environment. "Environmental credits" refer to avoided material extraction or energy generation in the wider economy associated with particular actions or services. These may be accounted for using an expanded boundary life-cycle assessment (LCA) approach.

Although, from a material resource-efficient perspective, disposal options such as landfill and incineration do not represent best practice for separately collected recyclables and mixed MSW, it is important to quantify the impacts associated with such disposal operations, in order to quantify the environmental benefits realised through the adoption of best practices. Both EMAS and the 2015-revised ISO 14001 standard require life-cycle environmental impacts to be considered. The revised ISO 14001 also places an emphasis on the "risk" associated with environmental aspects.

Table 1-10. Main activities in the waste management sector, and associated environmental aspects, pressures, credits and risks

Service or activity	Main environmental aspects	Main environmental impacts	Main environmental credits	Main environmental risks
Administration	- Office energy consumption (heating, lighting, ICT, equipment) - Paper use and printing - Generation of municipal waste for disposal - Transport of staff - Printing emissions	- Fossil resource depletion - Finite resource depletion - Climate change (GHG emissions) - Air pollution (indoor and outdoor) - Traffic	- See recycling credits	- Long-term employee health effects of office environment (minor risk)
Waste collection	- Collection (truck) operations - Infrastructure construction and maintenance - Equipment production	- Climate change (GHG emissions) - Air pollution - Fossil resource depletion - Traffic - Finite resource depletion	- See recycling credits	- Employee safety risks associated with collection operations - Reputational risk via visible impacts - Operational efficiency risks of changes - Costs of repair and upgrade
Waste separation/ treatment	- Operational energy consumption (electricity, natural gas) - Residual waste generation - Infrastructure construction and maintenance - Equipment production - Disposal of non-reusable or recyclable materials	- Climate change (GHG emissions) - Air pollution - Fossil resource depletion - Traffic - Finite resource depletion	- See recycling credits	- Employee safety risks (heavy machinery) - Operational efficiency risks of changes - Cost of infrastructure & machinery repair and upgrade
Material transport	- Transport operations - Infrastructure construction and maintenance - Equipment production	- Climate change (GHG emissions) - Air pollution - Fossil resource depletion - Traffic - Finite resource depletion		- Employee safety risks - Reputational risk via visible impacts
Equipment/ component /	- Collection and transport operations - Heating and lighting of distribution	- Climate change (GHG emissions) - Air pollution	- Avoided abiotic resource use - Avoided fossil energy use	- Employee safety risks (heavy machinery) - Operational efficiency risks

Service or activity	Main environmental aspects	Main environmental impacts	Main environmental credits	Main environmental risks
material reuse	centres - Disposal of non-reused fraction	- Fossil resource depletion - Traffic	- Avoided waste disposal	of changes - Cost of infrastructure & machinery repair and upgrade
Composting (organic recycling)	- Machinery operations - Emissions from biological processes - Transport of compost - Field application - Fertiliser replacement - Soil carbon sequestration	- Climate change (GHG emissions) - Air pollution - Water pollution (nutrient leaching) - Fossil resource depletion	- Avoided fertiliser manufacture and application - Avoided GHG emissions - Avoided waste disposal	- Employee safety risks (heavy machinery) - Respiratory effects of aerosols in local population - Reputational damage from local noise / odour / air quality issues
Anaerobic digestion (organic recycling)	- Machinery operations - Water consumption - Infrastructure construction and maintenance - Equipment production - Fugitive emissions - Transport of digestate - Digestate application emissions - Fertiliser replacement - Soil carbon sequestration	- Climate change (GHG emissions) - Air pollution - Water stress - Water pollution (nutrient leaching) - Finite resource depletion	- Avoided fossil energy use - Avoided fertiliser manufacture and application - Avoided GHG emissions - Avoided waste disposal	- Employee safety (fatalities from explosion or hydrogen sulphide poisoning) - Major clean-up costs and reputational damage from digestate leakage (water pollution) - Cost of infrastructure & machinery repair and upgrade - Reputational damage from local noise / odour / air quality issues
Equipment disassembly	- Machinery operations - Infrastructure construction and maintenance - Leakage of hazardous substances - Equipment production - Residual material for disposal - Transport of materials - Disposal of non-recycled components	- Climate change (GHG emissions) - Air pollution - Human toxicity and ecotoxicity impacts - Fossil resource depletion - Traffic - Finite resource depletion - Disposal impacts	- See recycling credits	- Employee safety risks (heavy machinery) - Operational efficiency risks of changes - Cost of infrastructure & machinery repair and upgrade
Inorganic fraction	- Machinery operations	- Climate change (GHG emissions)	- Avoided abiotic resource use	- Employee safety risks

Service or activity	Main environmental aspects	Main environmental impacts	Main environmental credits	Main environmental risks
recycling	- Energy consumption - Infrastructure construction and maintenance - Equipment production - Transport of materials - Raw material substitution	- Air pollution - Fossil resource depletion - Traffic - Finite resource depletion	- Avoided fossil energy use - Avoided waste disposal	(heavy machinery) - Operational efficiency risks of changes - Cost of infrastructure & machinery repair and upgrade
Landfill	- Infrastructure construction and maintenance - Machinery operations - Decomposition of organic material - Nutrient leachate - Heavy metal and organic leachate - Sequestered nutrients - Sequestered resources - Energy recovery	- Climate change (GHG emissions) - Air pollutant emissions - Leachate to waters (eutrophication and ecotoxicity) - Pathogen release - Abiotic resource depletion - Fossil resource depletion - Land occupation	- Avoided fossil energy use (where biogas energy recovery implemented)	- Risk of water pollution (leaching) - Risk of problematic odours - Employee safety (heavy machinery and explosion risk of biogas) - Major clean-up costs and reputational damage from leaching (water pollution) - Reputational damage of pursuing outdated disposal method - Cost of infrastructure & machinery repair and upgrade - Reputational damage from local noise / odour / air quality issues
Incineration (includes biomass combustion)	- Infrastructure construction and maintenance - Handling operations - Fossil fuel requirements - Combustion process - Energy recovery - Ash/slag disposal (landfill)	- Climate change (GHG emissions) - Air pollution - GHG emissions - Abiotic resource depletion - Fossil resource depletion - Human toxicity and ecotoxicity	- Avoided fossil energy use (where energy recovery implemented) - Sanitation of the waste (disease prevention) - Avoided abiotic resource use (where metal recovery implemented)	- Employee safety (heavy machinery and explosion risk of biogas) - Cost of infrastructure & machinery repair and upgrade - Reputational damage from local noise / odour / air quality issues
Illegal dumping	- Littering - Hazardous substance leakage to air	- Land occupation - Climate change (GHG	- None	- Major clean-up costs borne by municipality

Service or activity	Main environmental aspects	Main environmental impacts	Main environmental credits	Main environmental risks
	and water	emissions) - Water pollution (leachates) - Ecotoxicity		- Reputational damage for local authority in relation to poor enforcement of the law

1.3.1. Direct environmental impacts

Climate change

Direct greenhouse gas (GHG) emissions from waste management across the EU-28 declined from 185 126 000 tonnes CO_2e in 2002 to 140 803 000 tonnes of CO_2e in 2012 (Eurostat, 2014). Waste management represents 3 % of total GHG emissions in the EU-28. Methane (CH_4) and nitrous oxide (N_2O) make important contributions to these CO_2 equivalent emissions.

Figure 1-28 displays direct GHG emissions arising from waste management across the EU-28 in 2011. National waste management sectors in Germany, Spain, France, Italy, Poland and the UK each emit considerably more than 10 Mt CO_2e/year, largely reflecting the large population shares in these Member States. Waste management accounts for a comparatively very high share (about 10 %) of national GHG emissions in Portugal and Cyprus, and a comparatively high share (about 5 %) of GHG emissions in Bulgaria, Latvia, Lithuania, Hungary, Romania, Slovakia and Greece.

Emissions of methane (CH_4) from landfill account for a large share of GHG emissions from waste management. Data on the quantity of MSW landfilled per capita across municipalities and countries are presented in Figure 1-29. Although the data are incomplete, it can be seen that countries with high rates of landfilling tend to have comparatively high shares of GHG emissions from waste management. This is a consequence of the high global warming potential (GWP) of 25 for methane and of 298 for nitrous oxide compared to 1 for CO_2 (IPCC, 2007), which is the main emission after thermal treatment of waste.

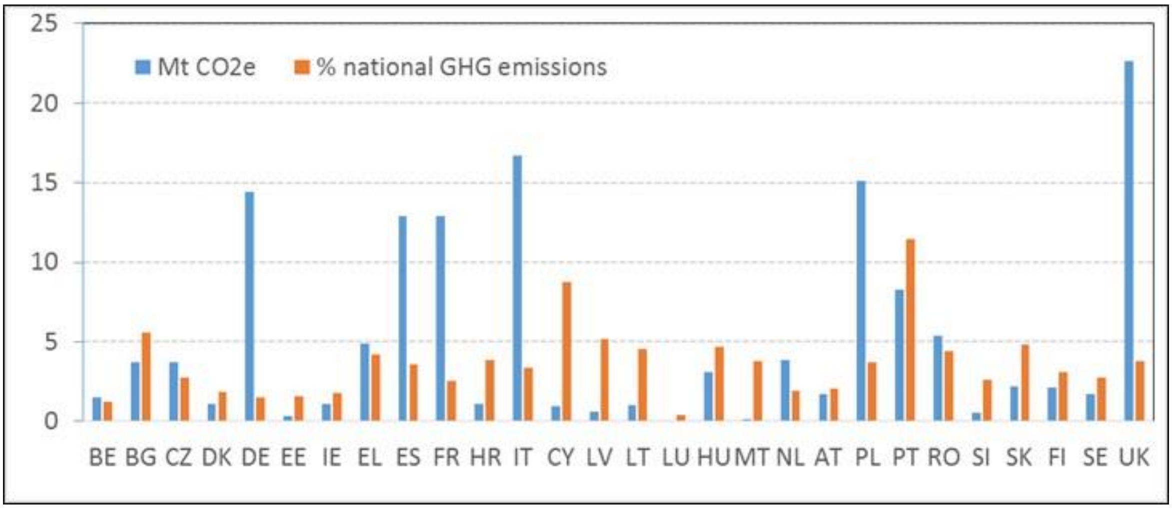
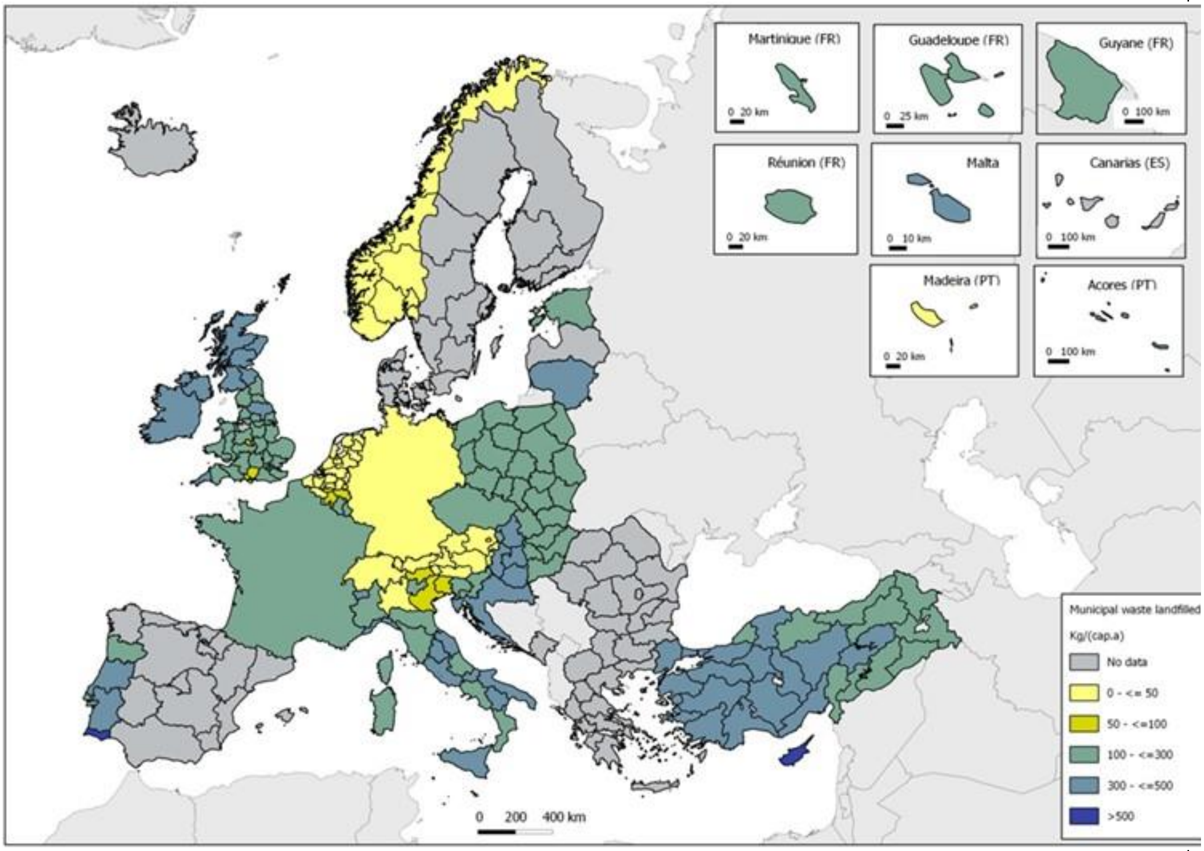


Figure 1-28. GHG emissions arising from waste management across the EU-28 in 2011 (blue), and the share of national emissions they represent (orange) - Source Eurostat, 2014)

It should be noted that statistics reported above on GHG emissions from waste management relate only to direct emissions from a limited range of activities, such as landfilling, classified as "waste management" under UNFCCC national GHG reporting guidelines. These statistics exclude many activities and some important sources associated with waste management, including waste collection and transport emissions, electricity consumption for waste handling and processing, and emissions arising from field application of composts and digestates. They also exclude the emissions associated with replacement of materials lost from the economy through disposal (see next section).



Source: Eurostat (2014)

Figure 1-29. Quantity of municipal solid waste landfilled per capita across European municipalities and countries

Air pollution

The waste sector across the EU-28 was responsible for 95 370 tonnes (3 %) of ammonia emissions (NH_3) in 2011, and 77 220 tonnes (1 %) of non-methane volatile organic compound (NMVOC) emissions in 2011. The waste sector accounts for only a minor share of NO_x and SO_x emissions (Eurostat, 2014).

Figure 1-30 displays ammonia emissions by country across EU Member States. Waste sectors in Spain, Romania and the UK are the largest emitters. As described below in relation to composting and anaerobic digestion, ammonia emissions arising from organic waste residues may arise in, and thus be attributed to, other sectors, in particular agriculture (Eurostat, 2014).

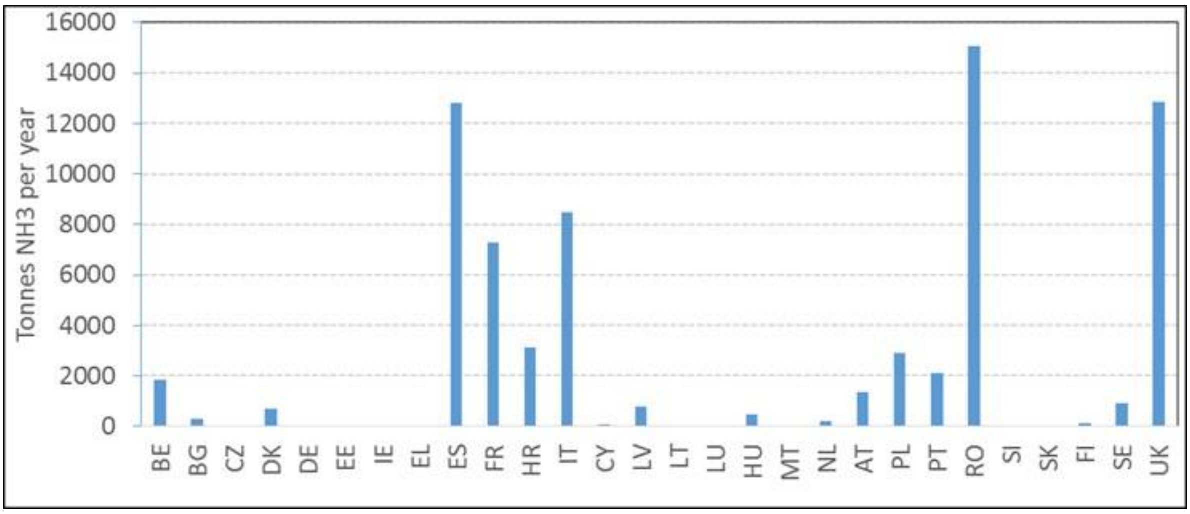


Figure 1-30. Ammonia emissions arising from waste management across the EU-28 in 2013 - Source Eurostat, 2014

Toxic emissions

Toxic emissions comprise a large suite of compounds emitted from a wide array of processes and sectors, including diffuse emissions. Therefore they are not well captured in emissions inventories. Quantities of hazardous waste generated per capita across EU Member States (Figure 1-31) may provide an indication of the risk of toxic emissions arising from waste management across Europe. Differences in accounting or definition may lie behind the wide variation in reported quantities of hazardous waste generated per capita. The manner in which these wastes are handled is likely to be more important in determining toxicity effects than the quantities generated.

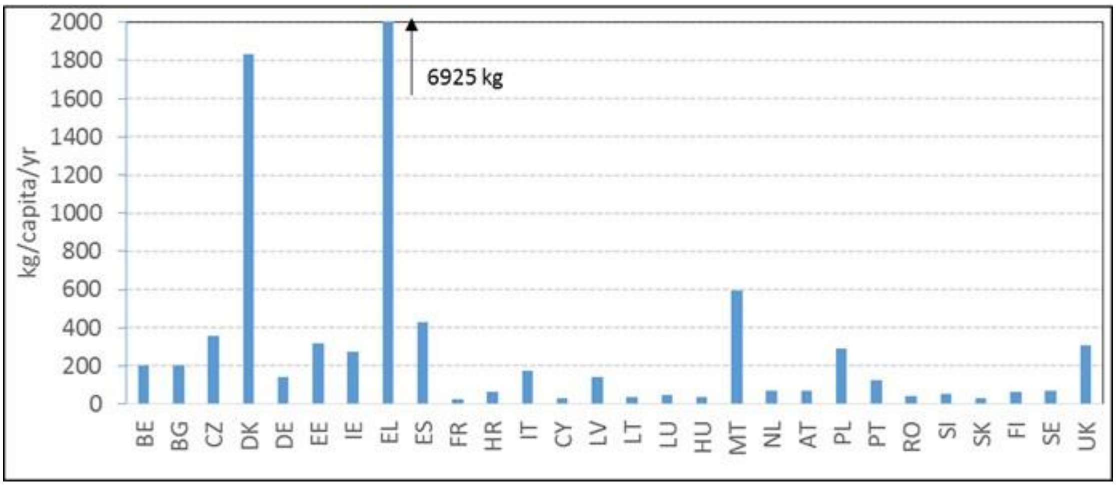


Figure 1-31. Hazardous waste generation across EU Member States in 2012 - Source Eurostat, 2014

Some important emissions with respect to ecotoxicity are reported for large industrial waste management facilities in the E-PRTR database (EEA, 2015), listed in Table 1-11.

Table 1-11. Key emissions related to toxicity and ozone depletion from large (IED-licensed) industrial waste facilities in 2012, reported in the E-PRTR database

Substance	Emission to air (kg)	Emission to water (kg)	Substance	Emission to air (kg)	Emission to water (kg)
As	1 070	29 037	Carbon monoxide	54 399 000	
Cd	1 573	13 400	Chlorofluorocarbons (CFCs)	18 300	
Cr	2 430	77 571	Dioxins and furans (Teq = Toxicity equivalents)	2.09	0.133
Cu	3 020	187 397	PCBs	6.69	87.8
Hg	1 330	3 250	PM10	3 295 000	
Ni	2 130	162 151			
Pb	1 970	85 004			
Zn	12 100	1 066 000			

Further information on landfill and incineration emissions is given in the dedicated sections below.

Recently, construction and demolition waste (CDW) has been linked with potentially toxic effects. CDW is not entirely inert. An important fraction (around 1–5 % in weight) of waste generated in demolition can be considered hazardous (asbestos, PCB-containing waste, paints, etc.). The case of PCB has recently become quite important in the management of CDW. PCB-containing sealants were banned in the 1970s but their use was frequent in the 1960s. Nowadays, demolition of buildings from this time has produced an alarming increase of leachable PCB in disposed CDW. Recent studies have shown how the PCB content of cement, concretes and CDW has increased from undetectable concentrations up to average concentrations of 17 μg/kg (± 84 %) in samples from the Danish construction industry (Butera et al., 2014).

Litter and illegal dumping

One direct consequence of poor waste management is litter accumulation on land and in oceans. In addition to visual impact, such litter can represent a danger to wildlife through strangulation and toxicity effects (Figure 1-32). Drinks cans holders and plastic bags are a particular threat to wildlife, including birds and turtles. Plastics are persistent in the environment, but degrade following exposure to sunlight, mechanical abrasion and plasticiser migration, creating tiny fragments that may be ingested by fauna, including fish. In addition, plastics adsorb toxins, and thus represent a pathway for various toxic compounds into the food chain.

Source: https://en.wikipedia.org/wiki/Marine_debris

Source: BZL GmbH (2014)

Figure 1-32. A dead albatross that had ingested various plastic flotsam, and a coastal village in Indonesia

Plastic pollution of oceans is a problem receiving increasing attention, though it is difficult to accurately quantify. A recent study estimated that a minimum of 5.25 trillion particles with a combined weight of nearly 270 000 tonnes are floating in the world's oceans (Eriksen et al., 2014). The authors of that study classified plastic pieces into micro-plastic (< 4.75 mm) and meso- and macro-plastic (> 4.75 mm), and proposed various mechanisms of micro-plastic loss from the sea surface that include entering into the food chain and sinking to the ocean floor. They concluded that although their conservative estimate of plastic fragments in the world's oceans represents just 0.1 % of annual plastic production, it could be associated with significant ecological and human toxicity effects.

A significant though poorly quantified share of environmental burdens associated with waste disposal arise from illegal dumping that bypasses regulatory controls on waste handling and emissions. This can be a particular problem for waste oils and white goods for example, which can leak harmful compounds into the environment. Insulation materials and refrigerants can leak ozone-depleting substances and substances with high GWPs to the atmosphere. For example, a domestic refrigerator containing 0.5 kg of HFC-134a (CH_3CHF_2) could contribute 1 900 kg CO_2e to the atmosphere via refrigerant leakage following improper disposal (Defra, 2012). This is equivalent to its electricity-related CO_2e emissions arising over eight years of operation. Older appliances contain more damaging refrigerants.

Pathogens and hazardous substances

A significant amount of healthcare waste is hazardous as it contains pathogenic agents. Inappropriate management of healthcare waste causes odour, proliferation of insects and adverse local effects due to the disposal of hazardous pharmaceuticals. A high percentage of healthcare waste is generally deposited in landfills or treated in inadequate incinerators, releasing a significant amount of dioxins, furans, HCl, and heavy metals (Insa et al., 2010). Waste disposal in landfill, or relatively low-temperature incineration as well as improper design and operation of biological treatment plants, can lead to the release of potentially pathogenic biological agents into the environment, posing risks for human health (Zeschmar-Lahl, 2004).

1.3.2. Indirect environmental impacts

Removal of resource streams from the economy via waste disposal (landfill or incineration) generates additional demand for raw materials. The extraction and processing of raw materials represents a large share of environmental impacts attributable to EU consumption (Tukker et al., 2006). Many of these impacts may arise outside the EU. Tukker et al. (2013) presented some conclusions from the EXIOPOL Input-Output database for European consumption:

- Land use embodied in Europe’s imports is higher than the domestic land use in Europe.
- Water use embodied in Europe’s imports equates to 70–90 % of Europe’s domestic use.
- The used and unused material extractions embodied in Europe’s imports represent around 40–50 % of the used and unused material extractions within Europe.
- The net energy use embodied in imports and exports are in the same order of magnitude. Imports of embodied energy are around 20 % of the total energy use for final European consumption.

Figure 1-33 displays the domestic material consumption (DMC) per capita across EU Member States. National DMC is the annual quantity of raw materials extracted from the domestic territory, plus all physical imports minus all physical exports (Eurostat, 2014). It provides an indication of the net quantity of resources consumed within an economy. Estonia, Finland and Ireland stand out as having a particularly high DMC per capita, all above 25 tonnes per year. Reuse and recycling of materials can significantly reduce the DMC.

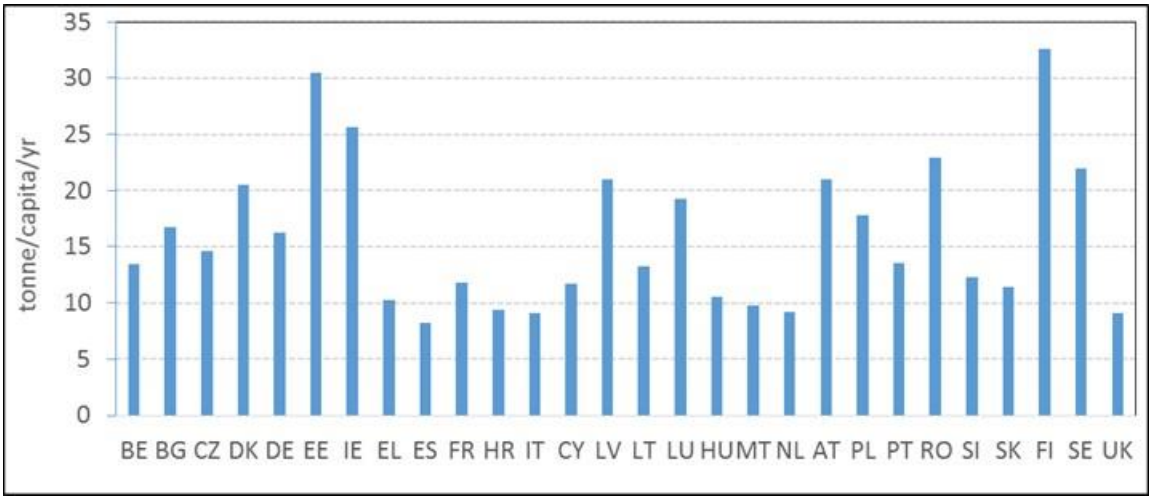


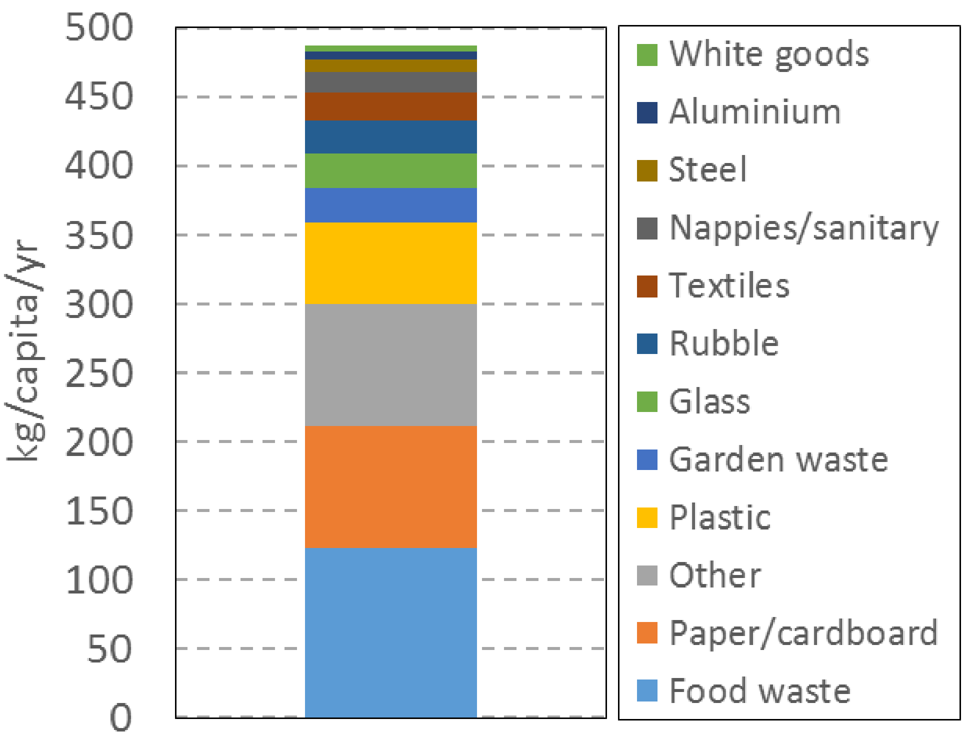
Figure 1-33. Domestic material consumption (DMC) per capita across the EU-28 in 2012 - Source Eurostat, 2014

Table 1-12 below summarises some of the major environmental burdens, expressed as environmental impact potentials used in LCAs, arising from the extraction and primary processing of a selection of major raw materials. These burdens can be avoided through waste prevention, including reuse and recycling.

Table 1-12. Environmental burdens per kg produced (global average) for a selection of raw materials, derived from data in Ecoinvent v.3.0

Raw material	Global warming potential (kg CO_2e)	Eutrophication potential (kg PO_4e)	Acidification potential (kg SO_2e)	Fossil resource depletion potential (MJe)	Human toxicity (kg 1.4-DCBe)
Steel	2.32	0.0035	0.0095	26.8	0.975
Aluminium (cast alloy)	3.18	0.0080	0.025	39.7	4.86
White packaging glass	1.15	0.0013	0.0096	15.4	0.628
Paper pulp	1.27	0.0037	0.0067	19.1	0.49
PET granules	3.08	0.0034	0.0152	72.2	0.921
PVC bulk	2.2	0.0012	0.0065	49	0.237
Cotton (knit)	22.8	0.040	0.139	267	5.99

Figure 1-34 presents the quantities of different materials sent for disposal or reuse by an average EU citizen over the course of one year. On average, each EU citizen generates over 490 kg of MSW per year, comprising 123 kg of food waste, 89 kg of paper/cardboard and 59 kg of plastic alongside an assortment of other fractions including textiles, glass and metals.

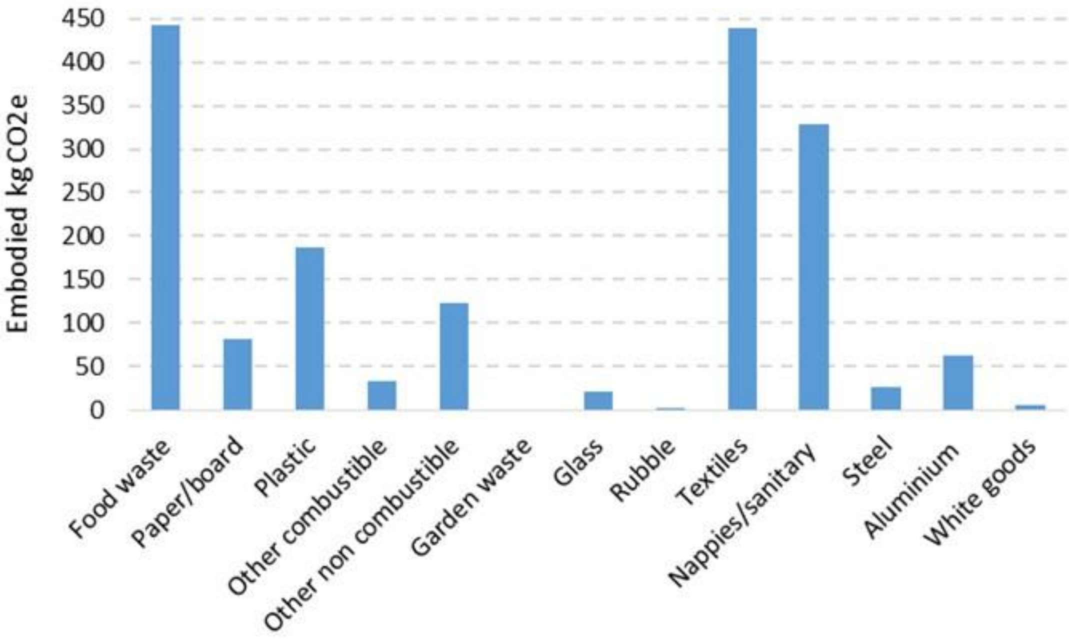


Source: Derived from data in Eurostat (2014).

Figure 1-34. Typical composition of MSW in the EU, expressed as mass of different fractions generated per capita per year, including fractions before separate collection

Based on the average quantities of MSW fractions generated per capita across the EU-28 (Eurostat, 2014), and GHG emissions associated with the production of dominant materials within those fractions (Defra, 2014), the GHG emissions embodied in MSW can be estimated. For an average EU citizen, these emissions amount to 1 755 kg CO_2e/year, approximately 20 % of an average EU citizen's annual carbon footprint calculated from emissions occurring within the EU (excluding "imported" emissions referred to by Tukker et al., 2013). The profile of embodied GHG emissions within MSW differs from the mass composition, reflecting a particularly high carbon intensity for textiles (Defra, 2014). Food waste, textiles and nappies/sanitary products make the largest contributions, followed by plastics (Figure 1-34). Extrapolating the above per capita emissions up to the EU-28 population of over 507 million people (Eurostat, 2014) indicates that emissions embodied in MSW amount to over 890 Mt CO_2e/yr. Overall indirect emissions associated with waste management will be greater than 20 % of EU total direct GHG emissions when other non-MSW fractions are accounted for. This compares with the 3 % of EU GHG emissions directly attributed to waste management activities (Eurostat, 2014), and emphasises the importance of addressing waste prevention, reuse and recycling in order to effectively reduce the environmental burden of waste (management).

Although insufficient data are available to perform the same calculations for all major environmental burdens embodied in MSW fractions, it is likely that contributions to some environmental burdens at the EU level could be even higher than for GHG emissions. For example, food waste is an important component of MSW. The United Nations Food and Agricultural Organisation (FAO) estimated that 30–50 % of the food produced annually at the global level is wasted, amounting to between 1.2 billion and 2 billion tonnes of waste (FAO, 2011). An Environmental Impact of Products (EIPRO) study found that food and drink production accounted for almost 30 % of GHG emissions arising from EU consumption, but almost 60 % of eutrophying emissions (Tukker et al., 2006).



Source: Derived from MSW data in Eurostat (2014); embodied GHG emission data from Defra (2014).

Figure 1-35. Greenhouse gas emissions embodied across different waste fractions in the annual MSW generated by an average European citizen

A typical household will throw away hundreds of euros worth of food every year, much of which could be avoided by better meal planning, appropriate food storage and careful checking of food labels (WRAP, 2015b). WRAP (2013) estimated that the GHG emissions linked to avoidable food and drink waste from UK households accounted for approximately 17 million tonnes of CO_2 equivalent per year (approximately 250 kg CO_2e per capita per year). According to the same source (WRAP, 2013), the land that is required to produce this amount of food and drink is estimated at approximately 19 000 km^2 (or equivalent to approximately 0.03 ha per capita per year).

Waste prevention

Waste prevention has a major role to play in reducing the overall environmental burden arising from consumption within the EU. The environmental benefits that can be achieved from waste prevention are referred to throughout this document. Below two brief examples are listed.

One example of a largely avoidable waste stream, and the associated upstream raw material extraction, processing and transport impacts, is plastic used to manufacture water bottles. An estimated 2.7 million tonnes of plastic is used to bottle water globally each year, and 25 % of bottled water is exported across national boundaries

(EEA, 2010). In addition to environmental impacts arising from production and disposal of the plastic (e.g. non-renewable resource depletion), transportation of bottled water incurs environmental impacts via energy consumption, GHG emissions, emissions to air and congestion, compared with minor impacts arising from the piped transport of drinking water from treatment works to consumers' taps. Whilst tap water is served automatically alongside food and drinks in some European countries, sometimes in other countries eateries are not legally required to provide tap water on request. In France, it has been required by a decree of the General Directorate for Competition, Consumption and Fraud (Direction générale de la concurrence, de la consommation et de la répression des frauds, DGCCRF) since 1967 that, besides bread and condiments, a carafe of water accompanies the meal and the guest cannot be charged for this separately (Die Zeit, 2013).

Waste prevention is particularly important for the voluminous CDW fraction. Construction, demolition and excavation waste is the most important fraction of waste in terms of weight and the second in volume due to the relatively higher density of the mineral waste of CDW. The average composition of CDW shows that most of the waste is concrete, ceramics and masonry (up to 85 %). This fraction is frequently labelled “inert”, as it is characterised by a lack of chemical reactivity at ambient conditions. However, the main environmental impacts generated by CDW are quite relevant due to its volume and weight. The impact of management and logistics of CDW is shown in Table 1-13.

Table 1-13. Life-cycle environmental burdens for one tonne of construction and demolition waste treated according to different methods

Treatment	Global warming potential (kg/CO_2e)	Primary energy (MJ)	Land use, PDF* (m^2/year)
Collection	6	100	0.15
Landfill	15	300	0.80
Recycling	2.5	45	0.18
*Potentially Disappeared Fraction, Ecoindicator 99 method.			

Source: Blengini and Garbarino (2010)

One of the most important impacts of CDW disposal is the fraction of natural aggregates not substituted by easy and succesful measures, and the large impact of landfill operations. In the Netherlands, the recycling rate of CDW is around 95 %. However, this fraction can only satisfy 18 % of the total natural materials demand of the construction industry in the country, which still needs to import natural aggregates.

All environmental aspects in the CDW chain are influenced by design decisions at the start of the construction value chain. “Designing out” waste is a term in use for CDW, and refers to designing and planning commercially available techniques to avoid the generation of waste. The most popular way of designing out wastes is the use of prefabricated modules or modern methods of construction. With this approach, more than 80 % of total CDW can be avoided. For instance, the construction of a new residential building where the structure is prefabricated would save around 80–100 kg of waste per 100 m^2 floor area. Therefore, all environmental burdens (land use, energy consumption, GHG emissions, hazardous substances, etc.) of the CDW life cycle are highly dependent on prevention techniques.

1.4. Environmental impacts of key activities within the waste management sector

The environmental performance of specific activities and services delivered within the waste management sector are evaluated and presented in more detail in subsequent chapters of this report, applying an expanded boundary LCA approach to include impacts associated with recycling operations and avoided resource extraction. Below are some summaries of the key environmental impacts arising for the most environmentally significant waste management operations.

1.4.1. Collection and transport

Prospective wastes often have to be transported considerable distances from the point of use/disposal to reuse or treatment locations. From a life-cycle perspective, transportation of waste may give rise to significant GHG and NO_x emissions, and result in significant fossil resource depletion and traffic (i.e. circulation of trucks). The relative importance of these emissions will vary by waste type, management option and transport distance, and will be quantified for some examples in subsequent chapters. The principle environmental impacts associated with transport include:

- fossil resource depletion;
- global warming potential;
- acidification;
- photochemical ozone formation;
- human toxicity.

Also, traffic congestion, noise and potentially odours are important nuisances that could be taken into consideration in waste management strategies.

Municipal waste collection from residential areas can lead to significant emissions owing to inefficient start-stop driving of large waste collection trucks. As a consequence, separate collection of waste fractions may lead to higher transport burdens compared with non-separated MSW collection. Fruergaard and Astrup (2011) estimate a diesel consumption of 7.2 litres per tonne of organic waste collected for anaerobic digestion, compared with 3.3 litres per tonne for incineration in more widespread incineration plants with energy recovery in Denmark. However, from a life-cycle perspective, the GWP effect of this extra transport amounts to approximately 12 kg CO_2e per tonne of waste, which is minor compared with the life-cycle impacts of organic waste recycling when an expanded boundary LCA approach is taken. This transport GWP impact is also low compared with GWP impacts avoided through material recycling.

1.4.2. Landfill

Landfill and incineration have long been established as the most common treatment options for unsorted MSW or residual waste, and are associated with various environmental impacts that can be minimised through good design (specified in the Waste Treatments BREF: JRC, 2006), but more importantly through measures to minimise waste sent to landfill or incineration, as described in this report.

Landfill is being reduced under EU and national policies, with targets for diminishing shares of waste going to landfill over the coming years. For example, the UK target for 2015 is a 65 % reduction in the quantity of waste going to landfill compared with 1995. Therefore, landfill is becoming less relevant as a "baseline" against which to evaluate best management practices. However, integrated waste management strategies and other best practice techniques described in this document can accelerate the move away from landfill in those countries where it is still practised. And the environmental impacts of existing landfills will continue to manifest themselves for decades to come. Therefore, it remains relevant to consider the environmental impacts of landfill in this document.

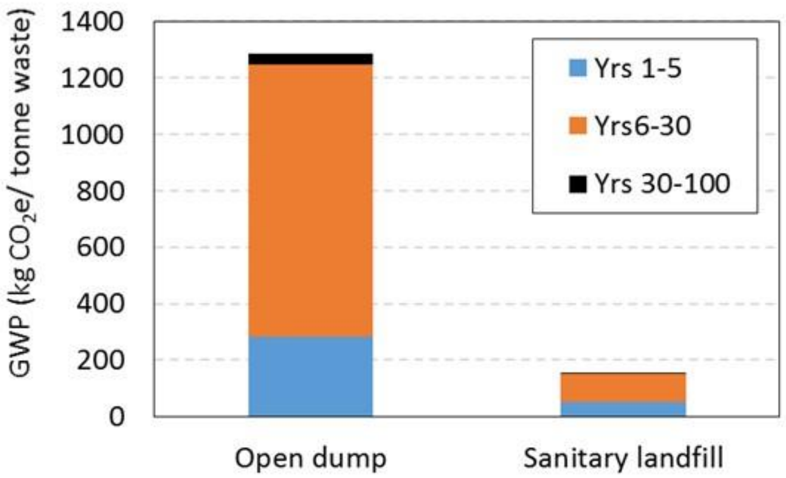
Table 1-14 summarises the main environmental impacts associated with landfilling. The overall environmental impact of landfilling varies considerably depending on the landfill design and management and the type of material being landfilled. The worst impacts arise from poorly lined, open dumps with disposal of unsorted MSW (including organic materials, various metals and chemical product residues). The landfills that have the lowest environmental impacts are those that are equipped with impermeable lining and caps, where most landfill gas is captured and combusted to generate electricity, or landfills containing primarily inert materials. For every tonne of MSW (fresh weight) entering a typical landfill, approximately 120 m^3 of biogas is produced, containing 60 % methane (CH_4) with a global warming potential (GWP) of 25 x CO_2e (Obersteiner et al., 2007) (Figure 1-36). One tonne of MSW deposited in an open dump can generate up to 1 285 kg CO_2e, though in a well-managed landfill this can be reduced to 158 kg CO_2e. If MSW undergoes mechanical and biological treatment (MBT) prior to landfill, landfill gas production can be reduced by approximately 95 % (JRC, 2006).

Table 1-14. Main environmental impacts arising from landfill (with energy recovery) of mixed waste[19]

Environmental aspects	Main environmental impacts
Infrastructure construction and maintenance	– Abiotic resource depletion – Fossil resource depletion – Land occupation – Landscape appearance and loss of amenity value – Biodiversity displacement
Machinery operations	– Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation
Sequestered resources	– Abiotic resource depletion
Landfill gas leakage	– Global warming (CH_4) – Acidification and eutrophication (NH_3 and NO_x) – Photochemical ozone formation (VOCs and NO_x)

[19] Mixed waste includes all waste that is not source separated by the users (e.g. households) of the waste management system. Sometimes mixed waste is referred as residual waste, however, for clarification, in this document only the term mixed waste is used.

	– Odour nuisance
Landfill gas capture and energy recovery	– Avoided fossil fuel combustion burdens – Acidification – Photochemical ozone formation
Leachate generation	– Eutrophication – Ecotoxicity – Waste water treatment plant burdens



Source: Derived from data in Obersteiner et al. (2007).

Figure 1-36. Methane emissions per tonne of MSW over the lifetime of an open dump and a sanitary landfill, expressed in terms of global warming contribution (as kg CO_2e/t)

Damgaard et al. (2011) found that the most important environmental impact categories for landfill were GWP, human toxicity via soil contamination, and stratospheric ozone depletion, displaying normalised person equivalent (PE) burdens per tonne of MSW of up to 0.154, 0.07 and 0.04, respectively. Normalised acidification, human toxicity and ecotoxicity in water, nutrient enrichment, photochemical oxidation and human toxicity via air burdens were also considerably lower. Those authors also found that the GWP burden of landfill could become negative, down to almost –0.07 PE, when landfill gas was used to replace fossil energy.

A wide range of compounds is emitted to air and water from landfills, including volatile organic compounds and heavy metals. However, the relative contribution of landfills to the overall emissions of these compounds is typically small.

1.4.3. Incineration

Table 1-15 summarises the main environmental impacts associated with different aspects of incineration.

Table 1-15. Main environmental impacts arising from incineration (with energy recovery) of mixed waste

Environmental aspects	Main environmental impacts
Infrastructure construction and maintenance	– Abiotic resource depletion – Fossil resource depletion – Land occupation
Machinery operations	– Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation
Incinerated resources	– Abiotic resource depletion
Combustion	– Global warming – Acidification (NO_x and SO_x) – Photochemical ozone formation (VOCs and NO_x) – Human toxicity (particulate matter, dioxins, furans, PCBs)
Energy recovery	– Avoided fossil fuel combustion burdens – Destruction of pathogens (avoided health burden)
Ash/slag production	– Abiotic resource depletion – Ecotoxicity – Landfill burdens

The Waste Incineration Directive (2000/76/EC), superseded by the Industrial Emissions Directive (2010/75/EU), set emission limit values for incineration plants to limit harmful emissions, including:

- sulphur dioxide (SO_2);
- nitrogen oxide and nitrogen dioxide (NO and NO_2);
- hydrogen chloride (HCl);
- hydrogen fluoride (HF);
- gaseous and vaporous organic substances, as total organic carbon (TOC);
- carbon monoxide (CO);
- dust;
- heavy metals;
- polychlorinated dibenzo-p-dioxins and -furans (PCDD/F).

Consequently, waste incineration in dedicated plants with IED permits involves application of pollution abatement techniques such as combustion temperatures exceeding 850 °C and selective catalytic reduction, and accounts for a minimal share of EU emissions to air, as indicated in Section 1.1.1, above. Nonetheless, from a life-cycle perspective, Cherubini et al. (2009) demonstrate that incineration leads to comparatively high acidification burdens and dioxin emissions compared with landfill and recycling options. They also note that there is a significant residual landfill requirement for bottom ash and fly ash that may contain relatively high concentrations of heavy metals. Bottom ash can represent 20–30 % of the weight, and 10 % of the volume, of input MSW, and may be used in construction, for road construction, etc. (Defra, 2013). Pollution control residues including fly ash, reagents and waste water can represent 2–6 % of the weight of input waste, and can contribute towards toxicity effects depending on their management. Metals representing 2–5 % by weight of input materials may be recovered from bottom ash and resmelted.

In terms of GWP, incineration with energy recovery can perform well in comparison with landfill, and even with recycling for paper and plastic fractions in some circumstances of high energy recovery efficiency and comparatively shorter transport distances (Merrild et al., 2012). However, the energy recovery efficiency of incineration plants varies considerably, especially depending on whether heat output is utilised directly or only to generate electricity. In the former case (e.g. heat used for district heating), thermal efficiencies of up to 90 % are achievable. In the latter case, thermal efficiencies range from 14 % to 27 %, reflecting the relatively low calorific value of some waste inputs and the necessary pollution abatement interventions (Defra, 2014).

Waste may also be casually incinerated (including illegally) on domestic or commercial premises, or may be incinerated in large combustion boilers in place of coal, e.g. in cement plants (Galvez-Martos and Schoenberger, 2014).

1.4.4. Organic waste recycling

Organic waste gives rise to large environmental impacts when landfilled or composted owing to CH_4 and NH_3 emissions and energy requirements, although these may be somewhat offset by the use of landfill gas to generate electricity and by the fertiliser replacement and soil improver (humus) properties of compost. Composting can also give rise to N_2O emissions and nutrient leaching. Capturing the net environmental effects of waste management options, to include the multitude of indirect effects, requires an expanded boundary LCA approach, and ideally a consequential LCA approach. This is demonstrated in the simplified example in Figure 1-37. In reality, a wider range of counterfactual fates may apply to waste that is collected for centralised composting or anaerobic digestion, and in some cases the marginal effects of removing this waste stream from other processes may be non-linear. For example, removing wet organic waste from incineration waste streams can improve the efficiency of energy recovery from the residual combusted waste (ICU, 2014). Therefore, in order to obtain representative results, consequential LCA modelling of waste management options can require large quantities of data on a wide range of affected processes, as will be demonstrated in subsequent chapters of this report.

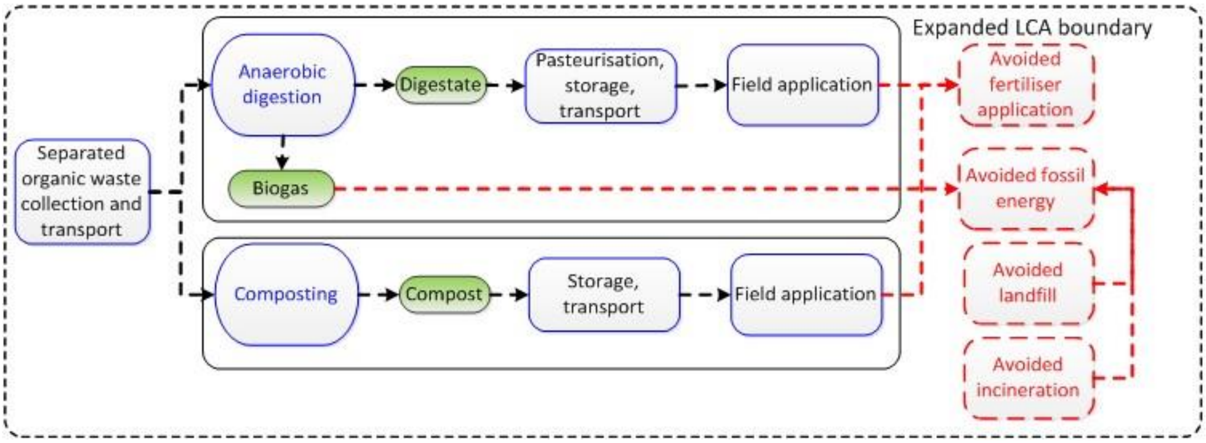


Figure 1-37. Major stages and processes affecting the life-cycle balance of organic waste going to anaerobic digestion or composting, in a simplified scenario that assumes counterfactual landfill or incineration is avoided

Table 1-16 summarises the main environmental burdens associated with different aspects of organic waste recycling, principally anaerobic digestion (AD) and composting, but also energy recovery via combustion (green waste).

Table 1-16. Main environmental impacts arising from organic waste recycling

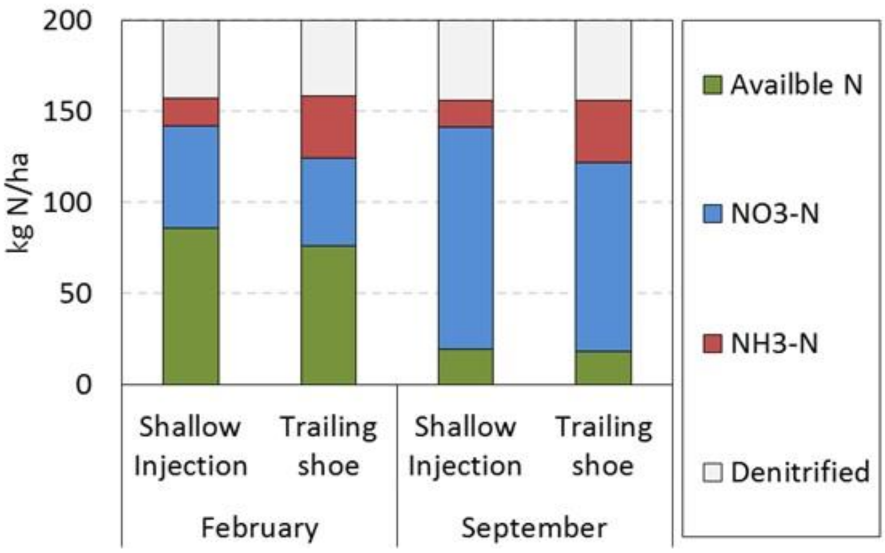
Environmental aspects	Main environmental impacts
Separated organic waste collection	– Fossil resource depletion – Traffic congestion and noise – Odour nuisance – Pest nuisance
Infrastructure construction and maintenance	– Abiotic resource depletion – Fossil resource depletion – Land occupation
Machinery operations	– Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation
Biogas leakage (composting and anaerobic digestion)	– Global warming (CH_4) – Acidification and eutrophication (NH_3)
Digestate and compost storage and application	– Acidification and eutrophication (NH_3, NO_3, PO_4) – Fossil resource depletion – Global warming potential (diesel CO_2 plus soil N_2O) – Avoided fertiliser manufacture and application burdens – Avoided global warming potential (soil carbon sequestration)
Energy recovery (biogas or biomass combustion)	– Acidification (NO_x and SO_x) – Photochemical ozone formation (VOCs and NO_x) – Human toxicity (particulates and polycyclic aromatic hydrocarbons) – Avoided fossil fuel combustion burdens
Extracted inorganic materials and	– Landfill burdens

combustion ash	

Anaerobic digestion (AD) can be an efficient option to recycle nutrients and recover energy from organic wastes, although the overall environmental balance is highly dependent on factors such as fugitive emission rates of CH_4 and NH_3 from primary and secondary fermenters, and digestate storage and application methods. Emissions may be high from small plants. Larger centralised AD plants can be more efficient, but may send digestate to landfill because transport costs to agricultural fields are high and demand for digestate is low, despite its significant fertiliser value.

Transport of organic waste fractions, compost and digestate can give rise to significant transport-related impacts, although these are typically small compared with waste disposal impacts. Transport distances are always constrained by economic factors before they dominate the environmental footprint of organic waste management options.

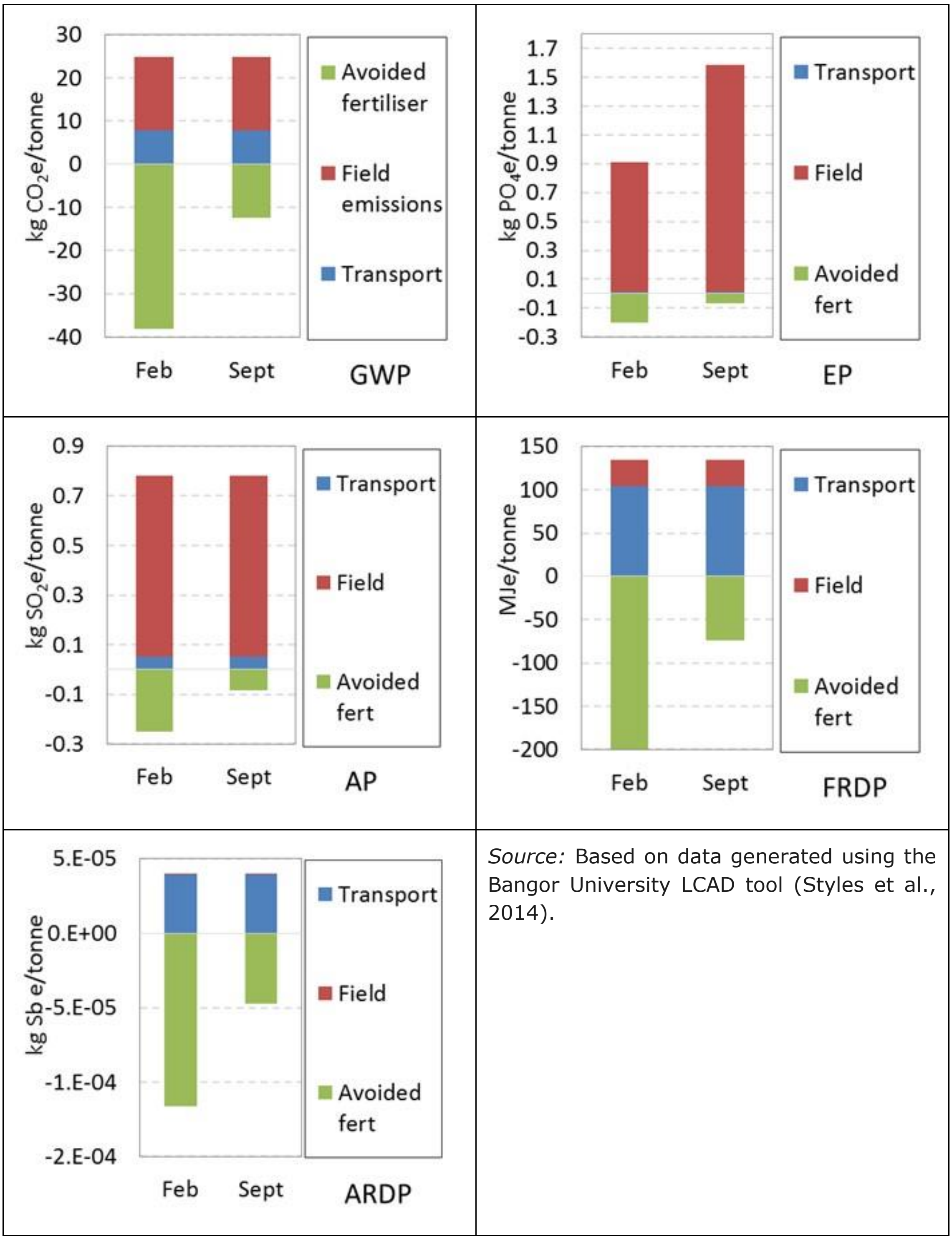
Digestate application to land as a bio-fertiliser is a hotspot for eutrophication and acidification impacts in the AD life cycle, and can sometime results in these impacts exceeding those of otherwise less efficient organic waste treatment options. Figure 1-38 shows the fate of nitrogen (N) applied to arable land in food-waste digestate. The application technique, but especially the timing of spreading, has a significant influence on losses to air (NH_3, denitrified N_2 and N_2O) and water (NO_3), the environment, and the fertiliser replacement value.



Source: Data from MANNER NPK (Nicholson et al., 2013)

Figure 1-38. Fate of nitrogen applied to arable land in food-waste digestate, at a rate of 40 t/ha, using shallow injection and trailing hose techniques in February and September, calculated using the MANNER NPK tool

Consequently, the environmental balance of digestate application varies considerably, as shown in Figure 1-39. Whilst application of digestate always results in higher net eutrophication and acidification burdens compared with avoided fertiliser manufacture and application, it can result in net GWP and fossil resource depletion reductions if spread in spring. However, autumn application increases the impacts in net GWP and fossil resource depletion.



Source: Based on data generated using the Bangor University LCAD tool (Styles et al., 2014).

Figure 1-39. Environmental balance for one tonne of food-waste digestate applied in February and September by shallow injection, across five impact categories (global warming potential, eutrophication potential, acidification potential, fossil resource depletion potential and abiotic resource depletion potential)

Table 1-17 compares environmental impacts arising from sanitised landfilling (typical UK landfill with 70 % CH_4 capture), composting and anaerobic digestion of organic waste. These impacts reflect the avoided marginal grid (natural gas combined cycle turbine) electricity generation for landfill and anaerobic digestion, and the avoided fertiliser manufacture and application for composting and anaerobic digestion. Overall, anaerobic digestion exhibits the best environmental performance, although it leads to slightly higher eutrophication and acidification impacts than sanitised landfill. Composting requires significant energy inputs and gives rise to NH_3 emissions, whilst having a low short-term fertiliser replacement value (Styles et al., 2014). However, as noted below, long-term soil organic carbon accumulation and nutrient release from composts could lead to a better long-term performance.

Table 1-17. Life-cycle environmental burdens (system expansion approach) for one tonne of food waste (26 % dry matter) treated according to different methods

Treatment	Global warming potential (kg CO_2e)	Eutrophication potential (kg PO_4e)	Acidification potential (kg SO_2e)	Fossil resource depletion potential (MJe)
Sanitised landfill (70 % CH_4 capture and energy recovery)	517	0.14	0.42	-1 563
Compost (use as soil improver)	170	0.83	1.81	500
Anaerobic digestion (electricity generation and digestate used as fertiliser)	-95	0.50	0.59	-2 788

Source: Styles et al. (2014)

In a report to the German Federal Agency for Environmental protection, Knappe et al. (2012) recommend that organic waste is treated anaerobically where possible, or alternatively composted, in order to achieve maximum resource efficiency. They noted significant benefits for soil humus and phosphorus recycling arising from composting and digestion, compared with landfill or incineration disposal. Soil humus accumulation leads to improved soil fertility, lower irrigation requirements and reduced erosion, effects often neglected in LCA studies based on short-term responses.

1.4.5. Waste sorting and product disassembly

Waste sorting may occur at the point of generation or in a dedicated sorting plant. In the latter case, burdens associated with collection may be reduced, but significant quantities of energy (usually electricity but in some MBTs also natural gas for drying (ICU, 2011)) are required to power the operations. Disassembly operations lead to similar burdens through electricity demand. In addition, disassembly operations must be carefully controlled to minimise leakage of hazardous compounds, such as refrigerants, used lubricating oils, PCBs, etc. (Table 1-18).

Table 1-18. Main environmental impacts arising from waste sorting and product disassembly

Environmental aspects	Main environmental impacts
Separated waste collection	– Fossil resource depletion – Traffic congestion and noise
Infrastructure construction and maintenance	– Abiotic resource depletion – Fossil resource depletion – Land occupation
Machinery operations	– Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation
Hazardous substance leakage	– Global warming (e.g. refrigerants and insulation gases) – Human toxicity and ecotoxicity (used oils, heavy metals, PCBs, etc.)
Material recovery	– Avoided resource depletion – Avoided raw material processing burdens
Material recycling	– Recycling burdens
Rejected materials	– Landfill or incineration burdens

Waste sorting and product disassembly are essential steps in material recycling. Impacts incurred by these processes must be balanced against the impacts incurred by disposal options for non-sorted waste streams, primarily landfill and incineration.

Table 1-19. GHG emissions arising from the transport, treatment and disposal of different waste fractions across alternative management options

	Reuse	Open loop*	Closed loop**	Combustion	Composting	Landfill
	kg CO_2e/tonne waste					
Mineral oil			21	21		0
Tyres	21	21	21			0
Wood	67	21	21	21	21	851
Glass		21	21	21		26
Clothing	21		21	21		552
MSW	21	21	21	21		290
Food and drink			21	21	6	570
Garden waste			21	21	6	213
Waste electronics		21	21	17		
Aluminium			21	21		21
Steel			21	31		21
Plastics		21	21	21	34	
Paper and board			21	21	21	553

*Primary products recycled back into different secondary products.

**Products recycled back into the same product.

Source: Data from Defra (2014)

Table 1-19 summarises GHG emissions across alternative management options of different waste fractions. These data were generated by Defra (2014) according to International GHG Protocol guidelines for company GHG reporting (WRI, 2004, 2011).

Landfill emissions are calculated using a "gate-to-grave" scope whilst recycling and energy recovery emissions cover only transport to the reclamation facility – including separated collection and transport. Subsequent emissions are attributed to recycled products (next section) or generated energy.

1.4.6. Material recycling

As with organic material recycling and waste sorting/disassembly activity impacts above, material recycling impacts must be considered against avoided raw material extraction and processing impacts (Table 1-20).

Table 1-20. Main environmental impacts arising from material recycling

Environmental aspects	Main environmental impacts
Waste collection/separation	– Waste sorting and disassembly impacts
Infrastructure construction and maintenance	– Abiotic resource depletion – Fossil resource depletion – Land occupation
Machinery operations	– Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation
Material cleaning	– Water stress (consumption) – Abiotic resource depletion (chemicals) – Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation – Ecotoxicity (discharges to water)
Material recovery	– Avoided resource depletion (credit) – Avoided raw material processing (credit)
Rejected materials	– Waste disposal impacts

Recycling is usually associated with lower environmental impacts than virgin production for most materials, especially metals with a high level of embodied energy (Table 1-21). For example, recycled aluminium gives rise to energy and air pollution impacts 75–90 % lower than virgin aluminium, and avoids most of the resource depletion associated with aluminium ore extraction. Recycled glass is associated with life-cycle energy requirements 20–30 % lower than virgin glass. Nonetheless, recycling processes can be energy-intensive and give rise to various environmental impacts, whilst separated waste collection is energy-intensive and can give rise to additional traffic, air pollution and noise. Dinkel (2008) reported that 37 % of the life-cycle environmental impact of recycled PET plastic arises from logistics activities, and 63 % from production processes, but that recycling PET results in lower life-cycle environmental impacts than incineration with waste heat recovery.

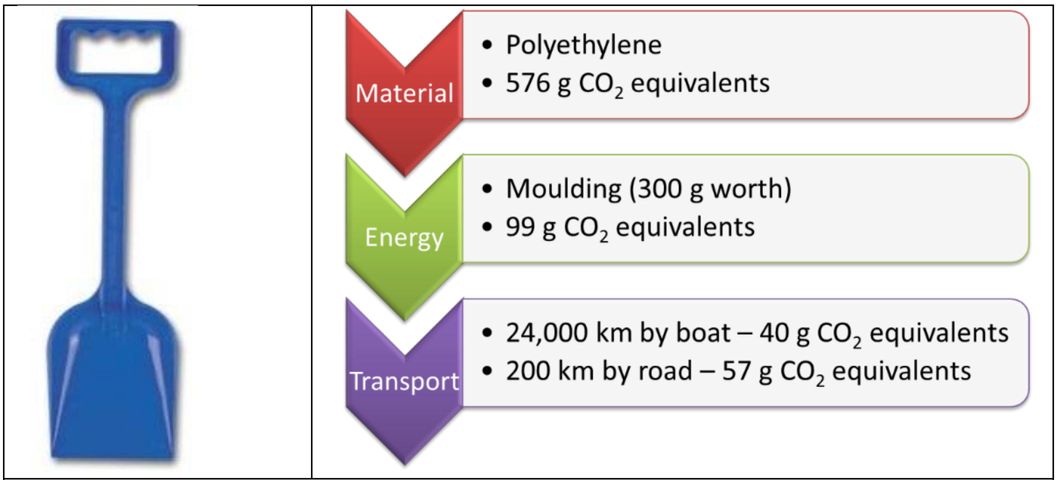
Table 1-21. GHG emissions avoided per tonne of different types of waste avoided or recycled

	Glass	Board	Wrapping paper	Dense plastic	Plastic film	Metals

Avoided	kg CO_2e/t	920	1 600	1 510	3 320	2 630	12 000
Recycled		390	1 080	990	1 200	1 080	3 300

Source: WRAP (2011), Ecoinvent (2014).

The effect of recycling compared with landfilling or incineration is illustrated with the following example of a plastic spade's carbon footprint.



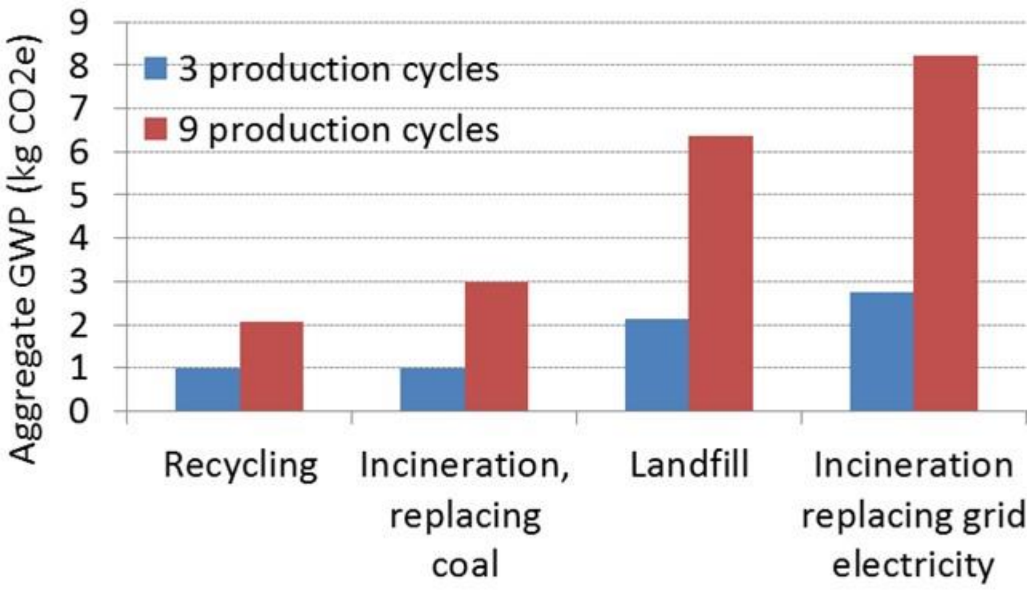
Source: Based on own elaboration and Ecoinvent (2014).

Figure 1-40. Greenhouse gas emissions from the manufacture and transport of a polyethylene spade manufactured in China

Landfilling, incineration or recycling of the polyethylene plastic give rise to GHG emissions of 0.03 kg, 0.90 kg and 0.10 kg CO_2e, respectively. However, the life-cycle effects of these different options depend upon:

- the number of times plastic is recycled;
- fossil energy carriers replaced (if any) with incineration energy recovery.

Figure 1-41 presents the life-cycle global warming potential (GWP) results of a few scenarios, considering closed-loop recycling, over three and nine cycles, alongside spade manufacture from virgin polyethylene three or nine times followed by landfill or incineration. Considering three recycling loops, recycling is on a par with the most efficient energy recovery scenario in which plastic directly substitutes coal through co-incineration, in terms of GWP. However, considering nine recycling loops, recycling achieves the lowest carbon footprint of all the options considered by some margin.



Source: Derived from data in Schanssema (2007), Plastics Europe (2008), Ecoinvent (2014).

Figure 1-41. Life-cycle GWP burden for three and nine production cycles of a polyethylene spade assuming recycling, landfilling, or incineration with energy recovery replacing coal directly, or replacing grid electricity in the UK

A somewhat surprising and initially counter-intuitive result displayed in Figure 1-41 is the poor performance of incineration with electricity generation, with a higher GWP impact than landfill. This reflects the fact that the release of fossil carbon into the atmosphere from plastic combustion can be higher, per kWh of electricity generated, in a low-conversion-efficiency incineration plant than in a dedicated fossil fuel power station. Thus, burying the plastic in a landfill can actually lead to a lower net carbon emission to the atmosphere. However, landfilling also has a wide range of other environmental impacts that must be considered alongside these GWP results. The key message is that, in order to achieve a significant environmental advantage from WtE plants, such plants should use as much of the combustion heat produced as possible to replace fossil energy carriers, via dedicated heating systems, co-incineration, or combined heat and power generation. Then, the GWP balance of plastic incineration with energy recovery can be comparable to the GWP balance of recycling (e.g. "incineration replacing coal" in Figure 1-41), although as the number of recycling loops increases, the comparative efficiency of recycling continues to improve beyond all other options.

1.4.7. Product reuse

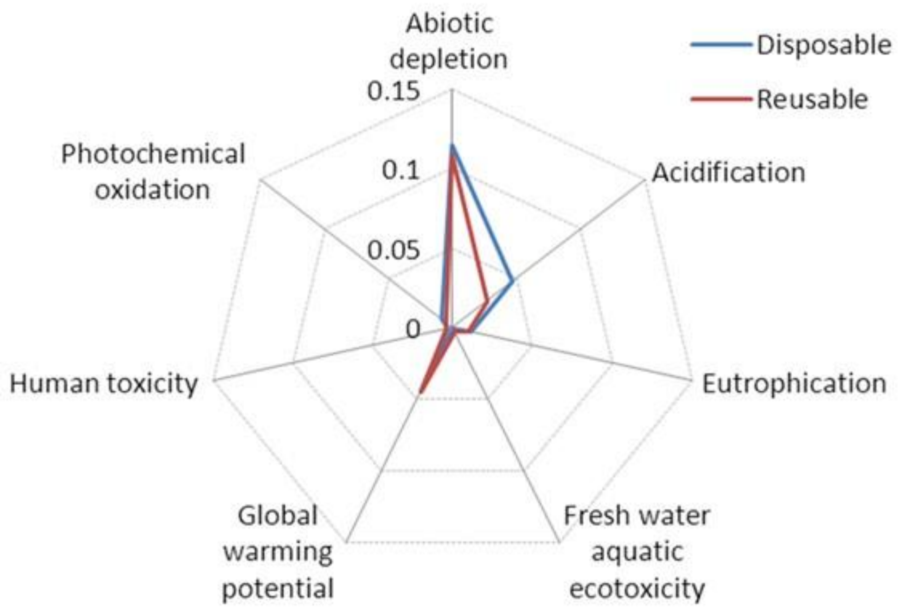
Waste management organisations can play an important role in encouraging and facilitating product reuse, diverting potential waste away from their own operations. Such diversion, if managed appropriately and associated with effective preparation for reuse, can make a significant contribution to waste prevention – avoiding the considerable administrative burdens associated with the preparation and classification of "waste" for use.

In general, the environmental balance of product reuse is simpler to estimate than the environmental balance of recycling, and may often be approximated to avoided production impacts (Table 1-22).

Table 1-22. Main environmental impacts arising from product reuse

Environmental aspects	Main environmental impacts
Collection and transport	– Fossil resource depletion – Traffic congestion and noise
Product cleaning (energy and cleaning products)	– Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation – Ecotoxicity (discharges to water)
Avoided production	– Avoided resource depletion (credit) – Avoided raw material processing (credit) – Avoided manufacturing and transport burdens (credit)

In some cases, reuse of products may incur significant environmental impacts that can be complex to analyse and compare against avoided impacts. The overall environmental balance may be highly sensitive to context-specific factors, as demonstrated for the following example for reusable nappies. The UK Environment Agency compiled a report in 2008 looking at the environmental balance of disposable and reusable nappies, considering the average UK landfill/incineration mix for disposable nappies and average UK wash temperatures, loads, share of tumble-dried washing, etc., for reusable nappies. The results indicated only a marginal advantage for reusable nappies owing to the high energy demand for washing and drying (Figure 1-42), but it was noted that results were highly sensitive to factors such as the grid electricity mix and the type of drying. Efficient washing and drying of reusable nappies in commercial laundries, necessitating a collection service, can lead to significant environmental benefits. Similarly, in countries with a lower environmental impact for electricity generation (carbon footprint of 0.49 kg CO_2e/kWh in the UK in 2008: Defra, 2014), the environmental advantages of reusable nappies will be considerably higher. Their relative performance will also improve over time as the energy efficiency of domestic equipment and grid electricity generation improves, highlighting the need to produce forward-looking LCA scenarios in order to inform strategic decisions regarding resource efficiency.



Source: Derived from Environment Agency (2008)

Figure 1-42. Environmental profile of disposable and reusable nappies according to a UK study

1.5. EMAS implementation in the waste sector

In Europe, there are 383 companies within the waste management sector with an EMS registered in EMAS, which include 942 sites, according to the EMAS register (EMAS, 2015)[20]. This value represents less than 1 % of the total sector (around 45 000 organisations in NACE divisions 38 and 39)[21]. These companies are mainly classified as SMEs, although many of them may belong to bigger companies (see Figure 1-43a). The proportions of waste management activities are equally represented in the EMAS register (see Figure 1-43b), i.e. collection, treatment and recovery, with a very low proportion of remediation companies.

[20] The figures represent only valid registrations and does not include historical or withdrawal values. Any error in the values shown has to be understood as an error in the published data of the EMAS register.

[21] For public administration implementation of EMAS, please, refer to the Best Environmental Management Practice Technical Report (http://susproc.jrc.ec.europa.eu/activities/emas/public_admin.html)

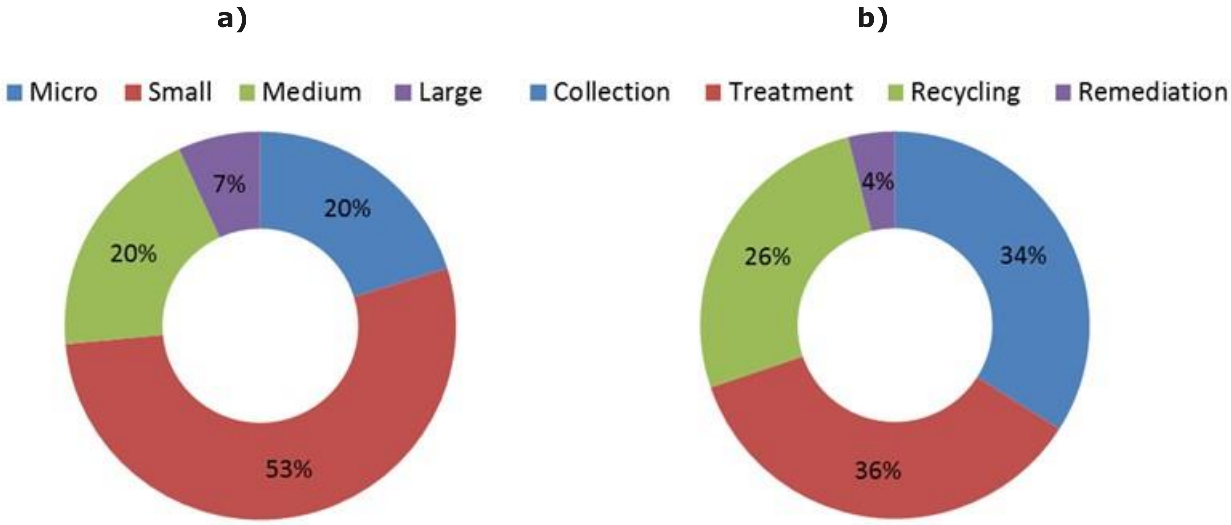


Figure 1-43. Percentage of EMAS-registered companies in Europe per site (a) and per registered activity (b)

Table 1-23 presents the number of the EMAS-registered sites and companies in the different European countries; the same table shows that more than half of the companies of the EMAS-registered sites are Italian SMEs.

Table 1-23. Number of EMAS-registered sites and companies per European country

Country	Number of sites	Number of companies
Austria	367	33
Belgium	27	8
Bulgaria	2	1
Cyprus	2	2
Czech Rep	4	2
Germany	30	21
Denmark	105	18
Spain	81	61
France	2	2
Greece	14	9
Hungary	2	2
Italy	247	194
Lithuania	2	1
Norway	10	10
Poland	19	11
Portugal	25	5

Romania	1	1
United Kingdom	2	2

Austria has registered 367 sites for 33 companies; most of the sites belong to three large organisations, the environmental department of the city of Vienna, with 164 sites (probably many administration sites included in this figure), AVE (in 2014: rebranding as Energie AG Oberösterreich Umwelt Service GmbH; 30 sites), and Upper Austria's O.Ö. Landes-Abfallverwertungsunternehmen AG (130 sites).

Every company registered in EMAS may cover more than one waste management activity, so it is not possible to accurately estimate the potential impact of EMAS on the different waste management activities. For instance, a company registers its waste collection activities for non-hazardous and hazardous waste and also any recovery activity that it may undertake. Therefore, Table 1-24 shows the number of registrations covering each activity per European country, but the sum of these values is much higher than the real number of registrations.

Table 1-24. Number of EMAS registrations covering main waste activities per country

Organisation country	38.11 Collection of non-hazardous waste	38.12 Collection of hazardous waste	38.21 Treatment and disposal of non-haz. waste	38.22 Treatment and disposal of hazardous waste	38.31 Dismantling of wrecks	38.32 Recovery of sorted materials	39.00 Remediation activities
Austria	17	6	12	8	6	12	0
Belgium	5	7	8	7	6	7	0
Bulgaria	1	0	0	0	1	1	0
Cyprus	2	2	2	2	1	1	0
Czech Rep	1	1	1	1	1	1	1
Germany	11	11	13	14	15	16	1
Denmark	10	7	10	8	9	10	0
Spain	29	13	9	9	9	9	2
France	0	0	0	0	0	2	0
Greece	6	2	6	4	4	10	0
Hungary	0	1	2	2	1	0	1
Italy	88	89	122	91	35	73	31
Lithuania	1	0	0	0	0	0	0
Norway	3	0	0	0	0	7	0
Poland	6	3	3	2	1	8	1
Portugal	1	0	2	0	2	1	0
Romania	0	0	1	0	0	1	0
United Kingdom	1	0	1	1	0	1	0

The last ISO survey for ISO 14001 (parental standard of EMAS) shows a large increase in the last few years in the number of recycling sector companies implementing ISO-certified environmental management systems, e.g. from 100 in 1998 to more than 3 300 in 2013 (ISO Survey, 2013).

In any case, the number of EMAS-registered organisations in the waste sector is very low, compared to the total number of waste management organisations operating in the EU in this sector. This does not neglect the fact that EMAS is a great help for companies or public administrations to set higher standards of environmental performance. Within this understanding, this report on Best Environmental Management Practice for the Waste Management Sector not only addresses organisations implementing EMAS or ISO 14001, but also the activities of all European waste sector companies and waste authorities wishing to improve their environmental performance.

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2. Common environmental performance indicators for municipal solid waste

For waste authorities responsible for municipal solid waste management it is important to assess and understand the performance of their waste management system as a whole and to identify areas for improvement, where BEMPs presented later in the document can be applied. This chapter supports waste authorities (or waste management companies) in charge of waste management at local level in conducting an assessment of their municipal solid waste management performance, independently of whether they have already been doing so or not. This chapter of the document defines a number of environmental performance indicators, describes what their features and limitations are, where to start monitoring and how monitoring can be improved.

By presenting several indicators which assess the actual environmental performance of the waste management system, this chapter is intended to enable good decision making by local waste authorities in order to improve their environmental performance. In fact, thanks to solid and clear results, actions to modify the municipal solid waste management strategy can be implemented.

2.1. Overall considerations

This chapter defines common environmental performance indicators that enable local waste authorities to monitor and assess the performance of their municipal solid waste management. Additionally, the use of environmental performance indicators can also enable the comparison, with caution, of results against the benchmarks of excellence presented below and the environmental performance of frontrunners.

The environmental performance indicators presented below cover the most relevant phases of municipal solid waste management (e.g. the overall MSW generation and the treatment operations) and its waste streams (e.g. the performance of separate collection of specific fractions).

Overall, the effectiveness of changes to the municipal solid waste management system on the territory of a local authority can be assessed by monitoring the evolution of the environmental performance indicators. Taking decisions for the adaptation of the municipal solid waste management based on the results obtained from the environmental performance indicators may result in environmental improvements and/or economic benefits for the local authority. Further information on the aims and benefits of waste monitoring is provided in the next section (Section 2.2).

Selection and overview of indicators

Table 2-1 lists all the environmental performance indicators described in this chapter. Section 2.4 presents both environmental performance indicators suitable for all common municipal waste streams while Section 2.5 introduces additional indicators that cover specific waste streams for which a local waste authority may have an established collection system, e.g. textiles.

Table 2-1. Overview of common environmental performance indicators described in Sections 2.4 and 2.5

Common environmental performance indicators
General MSW indicators
MSW generation
Amount of mixed MSW collected
MSW sent to energy recovery and/or disposal
MSW sent to disposal
Waste stream specific indicators
Capture rate of a specific waste stream
Impurity rate of a specific waste stream
Biowaste in mixed waste
Additional waste stream specific indicators (see section 2.5)
Collection scheme for glass bottles
Amount of textiles separately collected
Textiles in mixed waste
Capture rate for textiles

Benchmarks of excellence and their use

Benchmarks of excellence (BoE) represent the highest levels of waste management performance that have been achieved by frontrunner waste authorities and waste management companies.

The benchmarks of excellence allow waste authorities and waste management companies to understand the potential for improvement of their waste management performance, but are not targets for all organisations to reach. They are rather a measure of what it is possible to achieve (in certain cases, as demonstrated by the achievements of frontrunner organisations) that others can use to develop their understanding.

In this chapter there are eight benchmarks of excellence which cover different areas of municipal waste management, all the way from generation to disposal and each represents the highest waste management performance level achieved in the specific dimension they address. In order for waste authorities and waste management companies to develop an adequate understanding of their waste management performance and potential for improvement, it is fundamental to assess their own performance against the whole set of benchmarks and not only against individual benchmarks seen in isolation. The development of such understanding of the waste management performance will also allow the identification of the areas where taking actions may yield to significant environmental benefits.

2.2. Aims and objectives

Monitoring the waste management system is an activity which is normally carried out by all local waste authorities (directly or by the waste management company in charge of waste management), because of the need to comply with national/local waste legislation, with the aim of assessing the waste flows managed and ensuring the waste management service to residents. However, waste monitoring can be carried out to different levels of detail, e.g. from just quantifying the total MSW collected annually to assessing the quality of the separately collected waste streams.

When waste monitoring is performed in detail, systematically and regularly, it can be an important source of information for decision making, supporting the development and improvement of an efficient and effective waste management system. In fact, areas needing improvement can be identified by using a number of indicators that allow the assessment of the environmental performance of different waste management phases.

It is key that municipal solid waste management monitoring enables (i) the identification and understanding of the municipal solid waste management status quo, (ii) the monitoring of improvements and (iii) the development and assessment of activities and actions. This chapter of the report presents a number of environmental performance indicators that allow the conversion of raw monitoring data into a meaningful result able to support sound decision making and strategic planning. Once the monitoring of the waste management system has helped the identification of the waste management phases in need of improvement, other sections of this report and the BEMPs can be of use, as specified in the summary table introducing each indicator in Section 2.4. Best environmental management practices suitable for different steps of the waste management system are cross-referenced in each introductory table, with the objective of improving the environmental performance of the relevant phase of the waste management system.

In the event that the environmental performance indicators are used to compare the environmental performance of a local waste authority with the benchmarks of excellence and the performance of frontrunners, this should be carried out with caution, since there are certain limitations (e.g. waste definitions, boundaries of the municipal solid waste management system, level of economic activity, consumption patterns).

In summary, municipal solid waste management performance monitoring is important for several reasons:

- waste monitoring allows compliance with local/national legislation such as assessing quantitative targets, e.g. targets set in waste management strategies at national/regional/municipal level;
- waste management phases can be evaluated through monitoring, e.g. the status of the implementation, the effectiveness and success of different instruments but also challenges such as malfunctions or areas with poor performances can be identified;
- it can support the definition of new waste strategies or targets and, more broadly, the decision-making process.

Figure 2-1 illustrates how environmental performance indicators (EPIs) support the continuous improvement of waste management.

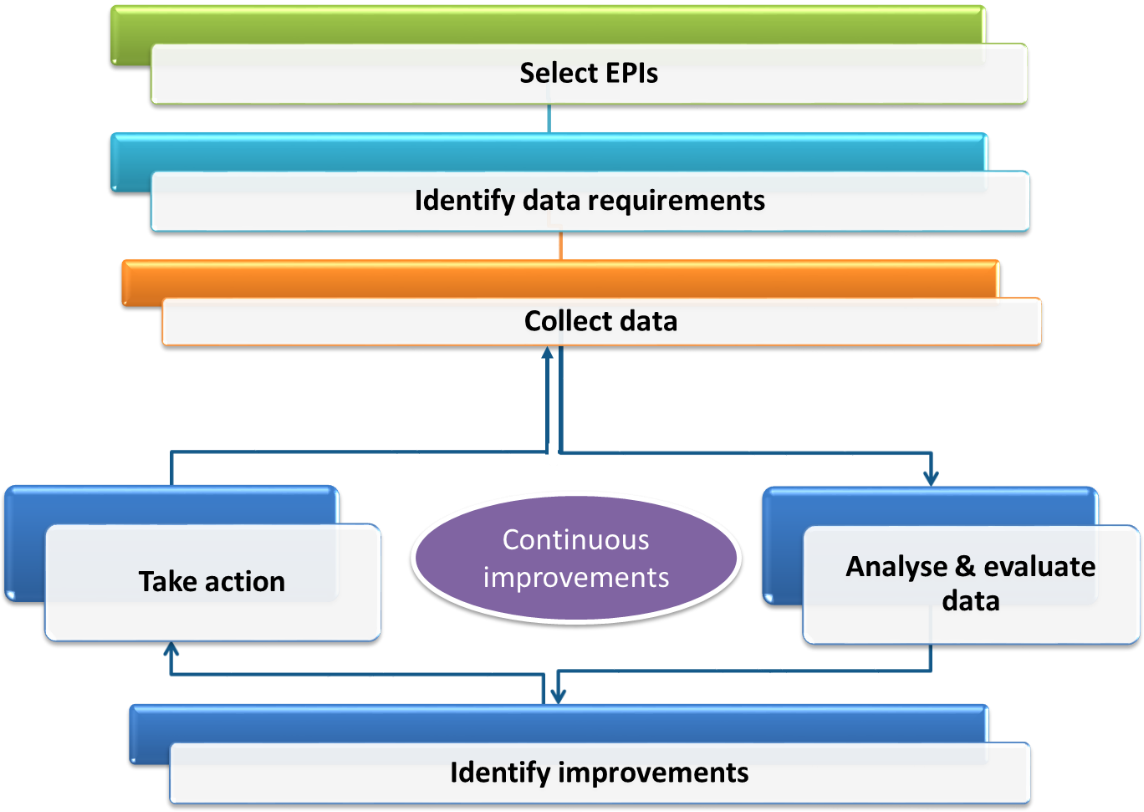

Figure 2-1. Monitoring and evaluation of waste management performance with indicators adapted from (Waste and Resources Action Programme, WRAP, 2010)

In general, various phases, activities and actors are involved in municipal solid waste management and different aspects of the waste management performance can be measured, such as:

- quantities generated;
- quantities collected;
- sorted fractions;
- treatment, recovery and disposal.

Municipal solid waste management performance is influenced by two main categories of factors: a) waste strategy and instruments and b) external factors, for which it is difficult to quantify their influence. These factors should be taken into account when interpreting the results of the environmental performance indicators and some examples of such factors are presented in Table 2-2 and Table 2-3. Further details on the factors listed below and their interpretation with respect to the indicators are given in Section 2.4.

Table 2-2. Some examples of waste strategies and instruments influencing municipal solid waste management performance

Waste strategy and instruments
Collection and treatment operations and equipment
Legal framework
Economic instruments

Costs and incomes
Communication activities

Table 2-3. Some examples of external factors influencing municipal solid waste management performance

External factors
Level of economic activity
Consumption patterns
Rural/urban area and population density
Type of housing
Weather
Tourism

2.3. How to get started with a municipal solid waste management performance assessment

This section gives an overview to local waste authorities and waste management companies in order to support them in implementing a performance assessment of the municipal solid waste management system. It includes information on the establishment of a municipal solid waste management flow diagram for all waste fractions, general information on data for the monitoring, etc. Furthermore, the objective of this section is to create a common understanding of key aspects of municipal solid waste management and its monitoring, as well as how the monitoring in a local waste authority can be established, if it has not yet been performed (the following section provides an explanation of the steps for starting monitoring and assessing municipal solid waste management). Topics covered below range from establishing the monitoring system to verifying the efficiency of the implemented measures. The list is not exhaustive and is given solely as an indication; it may be adapted by local waste authorities based on their individual municipal solid waste management system and needs. Later in the report, in Section 4.3.2, additional information on actions that can be implemented in order to adopt an advanced waste monitoring system are presented and can be considered in addition to the information presented below.

1. <u>Development of an understanding of municipal solid waste management</u>

As explained in the previous paragraphs, it is important to understand the features of the municipal solid waste management system, the influencing factors, the main phases and the actors and organisations involved. Local waste authorities and waste management companies should have an overview of all steps/operations that their municipal solid waste management system comprises as well as stakeholders involved, including the roles they play. Therefore, it is necessary to know which collection systems are in place, who is responsible for collecting every specific waste stream, which waste treatment operations are performed and by whom, etc.

One possibility is to develop a MSW flow diagram summarising the entire route of the MSW from the collection until its final treatment. As municipal solid waste management systems vary significantly across local authorities, this report proposes an example of how such a flow chart could hypothetically look in Figure 2-2.

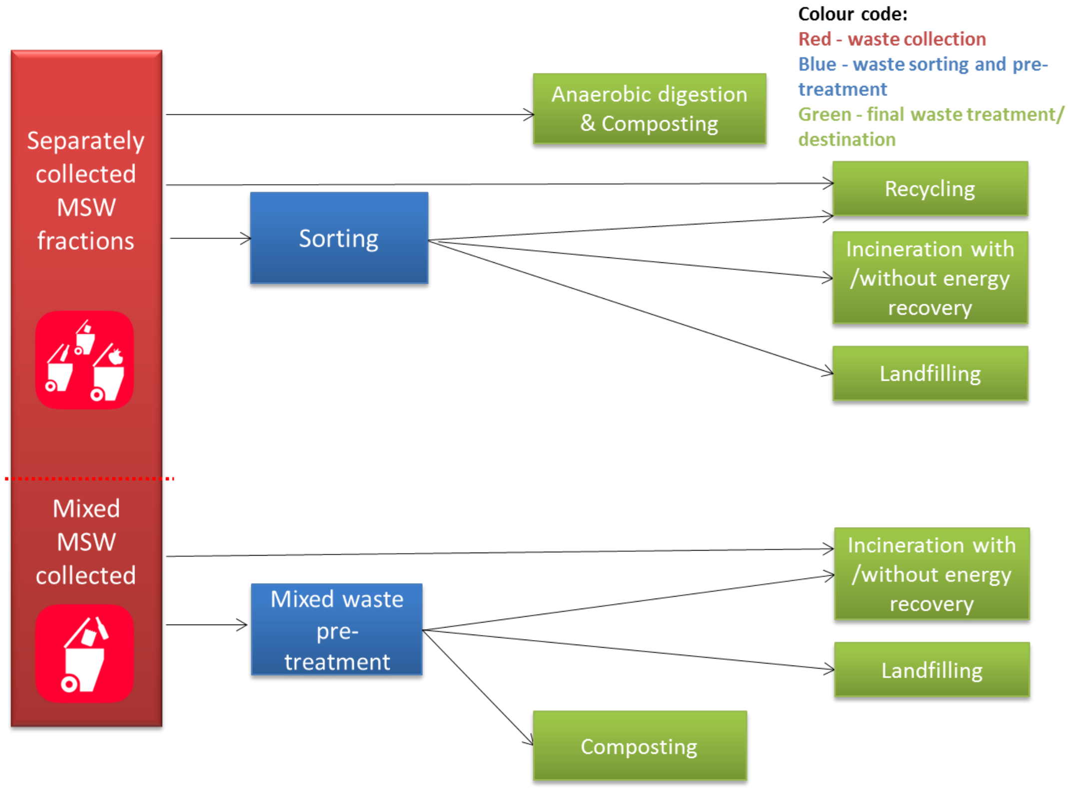

Figure 2-2. Example of a municipal solid waste management flow chart

The example flow chart includes the routes of MSW collected and sent directly to its final destination (e.g. recycling, energy recovery) or first to sorting/pretreatment. Waste streams covered by this flow chart are: separately collected materials such as paper and cardboard, plastic, glass, metal, and biowaste, either via a material-specific collection or a co-mingled collection, and mixed waste.

The collection or treatment of a certain waste fraction can be conducted within an EPR scheme. If this is the case in a local authority, it should also be included within the flow chart. Figure 2-2 represents the separately collected MSW fractions as a single stream and not as specific waste streams, e.g. glass or plastic. However, when drawing this type of flow chart, at local level, each separately collected waste fraction should be represented together with its waste treatment operations and disposal. A real-case example of a well-developed waste flow diagram is reported in the following figure, where all fractions collected in the department of Lot (France), their destination and also the main quantities are reported.

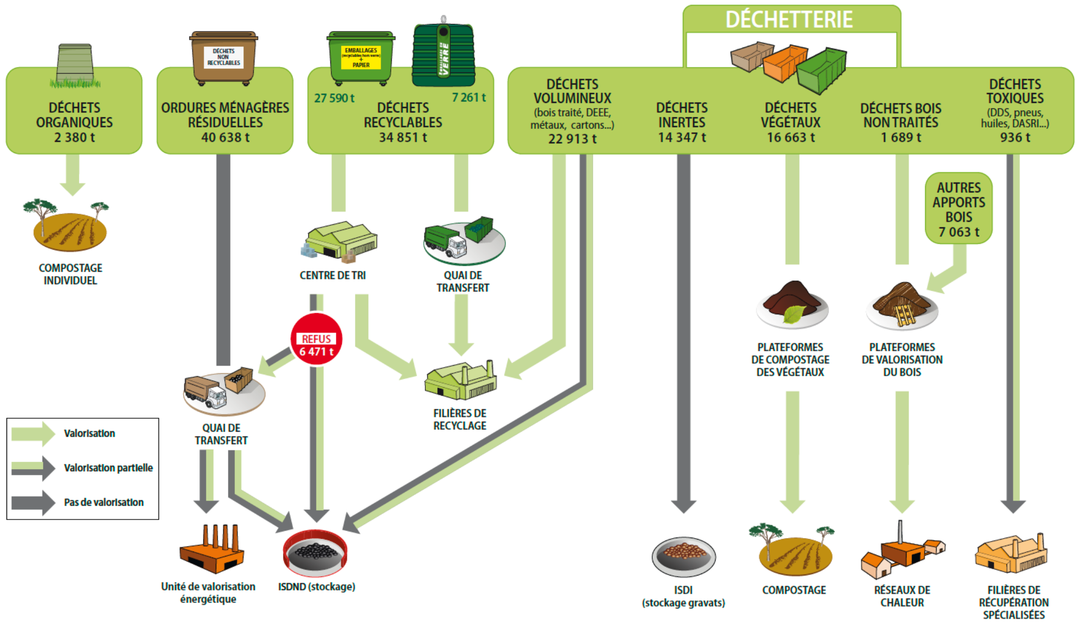

Figure 2-3. Waste flow diagram from the department of Lot (France) (SIDED, 2016)

In Figure 2-2 the red phase represents MSW collection: it includes the primary waste streams collected as mixed waste or separately collected waste streams. The collected waste can then be sent to the blue phase, sorting of recyclables and pretreatment (e.g. MBT) of mixed waste. Afterwards, the green phase corresponds to the final treatment and destination of the waste collected which could be recycling, anaerobic digestion/composting, incineration (with or without energy recovery), and final landfilling.

Please note that the following summary of the main MSW management phases reported in the hypothetical case of Figure 2-2 is not exhaustive and it focuses on the relevant parameters for the application of the environmental performance indicators presented in this chapter.

Collection - Red phase

Total MSW consists of mixed waste collected in the mixed waste bin and all materials collected separately. MSW also includes WEEE, bulky waste, street sweepings, etc. depending on the MSW definition applied. The total mixed waste mostly includes waste collected in the mixed waste bin or/and via another collection system, e.g. door-to-door, civic amenity site, bring points. The separately collected MSW fraction is the sum of materials collected by a separate collection scheme; this can include the following materials: paper and cardboard, glass, biowaste, plastic, metal, textiles, batteries, and cooking oils, etc. In general, separately collected MSW can include any type of material for which a separate collection scheme is established, independently of the type of collection.

Sorting and pretreatment - Blue phase

Separately collected waste streams can go to a sorting plant. The main objective is to prepare the material for recycling by sorting out materials that are contaminated or not in the target of the specific separately collected waste stream. Rejects/refuse obtained from sorting can be sent to waste incineration with or without energy recovery and landfilling. If sorting of separately collected waste is in place, it is mainly due to the conditions and technologies available at local level for separately collected fractions.

Mixed waste collected separately can instead be pretreated in a sorting/M(B)T plant before being sent to energy recovery and/or disposal. The pretreatment of mixed waste can generate fractions for the production of compost or digestate and rejects/refuse which can be sent to incineration and landfill. In some cases, some materials can also be recovered from MBT for recycling, e.g. ferrous and non-ferrous metals.

Final waste destination and treatment - Green phase

Separately collected waste streams are sent directly, or after sorting, to recycling. Recycling allows the recovery of materials (e.g. glass, paper, plastics) to be used as feedstock for production processes. Recycling processes also generate a certain amount of refuse/rejects which can be sent either to energy recovery or disposal.

Separately collected biowaste is instead usually sent directly to anaerobic digestion for the generation of energy and/or composting for the production of soil amendment for example.

Mixed waste can instead be sent directly to energy recovery and/or disposal. The difference between incineration with energy recovery and without energy recovery is that incineration with energy recovery can be regarded as a recovery operation (R1) according to the Waste Framework Directive (WFD)[22] whilst incineration without energy recovery is defined as a disposal operation. After the incineration process, the remaining incineration slag/ashes are disposed of in landfills. As mentioned above, if mixed waste is instead pretreated, rejects/refuse from the process can be sent to incineration and landfills.

2. Formulation of goals for the monitoring process

Before starting or improving a monitoring process, the local waste authority should clarify the aim of the monitoring, i.e. what questions shall be answered and what the expected outcomes are. This ensures that all relevant aspects can be considered and no redundant information is gathered. This step can be guided, for example, by questions such as: For which waste streams / collection systems / treatment operations is a monitoring process established? Which additional waste streams / collection systems / treatment operations need to be measured and assessed? What strategic goals are defined, e.g. reduction of waste generation, increase of separate collection shares, reduction of landfilling? What must be complied with?

3. Listing of all data requirements

From the knowledge of the municipal solid waste management system from step 1 and the goals defined in step 2, the data needed can be identified. The outcome

[22] EUR-Lex: http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0098

should be a list or set of data that is required by the local authority to monitor the defined goals.

Besides knowing how much waste is collected and treated, it is important to know how much waste is actually generated per specific waste stream. It is noteworthy that, especially for the amounts collected by established separate collection schemes, waste collected by the municipal solid waste management system does not reflect how much waste is actually generated but only what is collected within the territory of the local authority. Local waste authorities have the possibility to investigate the actual generation of a waste stream by conducting a composition analysis[23] of the mixed waste collected, e.g. from mixed waste bins. More information on the composition analysis can be found in Section 2.4.

One example of listing data requirements based on a defined goal could be that a local waste authority has the priority to increase the separate collection rate for glass. Therefore, specific data requirements can be derived; it is useful for the local waste authority to have data for the following aspects:

- how much glass is currently collected by the established separate collection system (consider all established systems, e.g. bring points, door-to-door);
- how much mixed waste is collected;
- the composition of the mixed waste (in percentages).

On the basis of these data it is possible to investigate in a latter step how much glass is not collected by the separate collection system and ends up in mixed waste.

4. Identification of data sources and gathering of data

After the assessment of the data requirements, local waste authorities need to investigate how the data can be gathered. Data from waste management operations that are in the hands of the local waste authority should be readily available. However, many municipalities have other actors (e.g. private operators) that are to some extent responsible for the waste collection and/or the treatment operations. To obtain a complete picture of municipal solid waste management in the territory, data from those other actors is also needed.

It is therefore important for local waste authorities, when contracting other actors for MSW collection or treatment, to include a data accessibility requirement in the contract. In general, it is useful for a local waste authority to have access to all data on waste collection and treatment at regular intervals (e.g. monthly, quarterly, yearly).

All the necessary data from step 3 above can then be gathered from the identified sources. It is important for local waste authorities to assess specific details of the data, e.g. is it for MSW or household waste only or do the data cover all collection or treatment systems, etc.? Such assessments are imperative for the proper interpretation of results of each environmental performance indicator.

[23] Sometimes also referred to as morphological or sorting analysis

5. Definition of data gaps

Step 5 is about defining data limitations of the monitoring and assessment process; it can be conducted in parallel to the steps below. When calculating and interpreting the results of the indicators for municipal solid waste management, it is useful for local waste authorities to keep in mind what data are missing and where uncertainties lie. More specific information on the impacts of missing data for the interpretation of individual indicators is provided in Section 2.4.

6. Calculation

This step relates to the actual calculation of the environmental performance indicators from the data available to the local waste authority. Concrete paths for the calculation are provided with each indicator in Section 2.4. Some aspects that require attention are, for example, using the population equivalent for the quantity-based indicators in highly touristic areas (see Section 2.4.1) and considering all different collection systems when calculating indicators for the individual fractions.

7. Interpretation

Once the environmental performance indicators have been calculated by the local waste authority, the results need to be interpreted. This can be done by taking into account the assessments made in steps 4 and 5; proposals for the interpretation of the results of the environmental performance indicators are presented for each indicator in Section 2.4. Furthermore, each indicator's result depends on overall limitations that are also addressed in Section 2.4, for each common environmental performance indicator; however, they can be summarised as follows:

- data collected by other actors (e.g. private operators) might not be available;
- if deposit/return schemes are established at national level, e.g. via an EPR scheme, for some waste fractions, data for those fractions might not be available at local level or might not be in the hands of the local waste authority;
- data availability per single waste stream, especially when there is a co-mingled collection which prevents the disaggregation of figures per waste stream;
- composition analyses of mixed waste are not available/conducted;
- limited comparability of indicators due to the different definitions of MSW;
- the extent to which commercial waste is included in MSW may be unclear and the proportion of household waste in MSW is calculated on the basis of estimations;
- external factors influence waste data, e.g. econometric factors (household sizes, household expenditures, GDP, etc.) or number of tourists and commuters in a local territory.

8. Identification of measures and activities to implement

The results of the environmental performance indicators may show, for example, that strategic goals from national, regional or local waste management plans are not met and thus local waste authorities can aim at improving their performance in a certain area. In these cases, it is important to identify suitable measures or activities to adopt from the results of the calculations. How such measures can be defined by local waste authorities is highly dependent on the current waste management system and on the goals encountered in the territory. To this aim, this report provides helpful support to local waste authorities to improve in specific areas of municipal solid waste management (see Chapters 3 and 4), after assessing the performance of the municipal solid waste management system.

9. Continuous monitoring

If local waste authorities wish to monitor changes over time, to conclude on the efficiency of activities and/or measures implemented, it is useful for the monitoring/collection of data to become a continuous process – e.g. weekly recording of waste quantities collected and regular processing of data. In certain cases, e.g. for a composition analysis, it may also be worth gathering quarterly data to identify seasonal changes (e.g. more biowaste during summer).

2.4. Common environmental performance indicators

The following pages describe each of the environmental performance indicators divided into two categories:

- the first category comprises indicators for the overall MSW; they look, for example, at the total amount of waste generated or its final treatment;
- in the second category, indicators specific to different waste streams are described.

Presentation of environmental performance indicators

Each environmental performance indicator is described using the same format: A table provides an overview of the main information, such as the name, the phase of waste management assessed by the indicator, a short description, the calculation method, the relevant BEMPs which can help in improving the result of the indicator and some examples.

A distinction is made between quantity-based indicators and performance ratios.

1) **Quantity-based indicators** reflect an amount of waste divided by a normalising factor, such as population and year, population equivalent (eq_{pop}) and year, area and year, etc. They provide a good overview of the overall waste management performance in absolute terms as well as insights into how the municipal solid waste management develops over time. However, this type of indicator has limitations in terms of its interpretation because it is highly dependent on external factors that are not related to waste management, such as: GDP, consumption patterns, rural/urban location, weather, tourism, illegal dumping, etc. Thus, when interpreting the results of quantity-based indicators, it is important to take the relevant external factors into account.
2) **Performance ratios** are defined as percentages and are especially useful for specific waste fractions. These indicators are more complex because they put absolute terms in relation to each other, e.g. capture rates. This provides a good overview of the actual municipal solid waste management performance of a local waste authority for a waste management phase (collection, treatment, etc.) or a specific waste fraction (glass, paper, etc.). Unlike quantity-based indicators, performance ratios calculated for specific waste fractions can give indications of which specific improvements are needed in the municipal solid waste management strategy. The performance ratio indicators also have

limitations, such as data availability issues or their dependence on external factors such as the presence of EPR systems for certain types of waste, the collection infrastructure in place, etc.

In the next sections, after the overview table of each indicator, a short text provides further information about its calculation method. In order to calculate the environmental performance indicators, several types of information are required. Figure 2-2 provides an overview of such types of information within a hypothetic flow chart summarising the entire route of the MSW.

Types of information required to calculate the indicators

1. Composition analysis of the mixed waste

For the calculation of most of the performance ratio environmental indicators it will be necessary to have conducted a composition analysis of the mixed waste. This reveals which materials, i.e. paper and cardboard, plastics, metals, glass, etc. have not been sorted out into the specific separately collected fractions at source. Therefore it allows a better understanding of the effectiveness and efficiency of the collection. For guidance on how to conduct such an analysis, the following examples may be of use:

- Scottish Environmental Protection Agency (SEPA): Guidance on the Methodology for Waste Composition Analysis - For local waste authorities commissioning waste composition analysis of municipal waste, in English (http://www.zerowastescotland.org.uk/sites/default/files/WCAMethodology_Jun15.pdf)
- Nordtest method (Nordic countries): Solid waste, municipal sampling and characterisation, in English (http://www.nordtest.info/images/documents/nt-methods/environment/NT%20envir%20001_Solid%20waste,%20municipal_Sampling%20and%20characterisation_Nordtest%20Method.pdf)
- Waste & Resources Action Programme (WRAP): Monitoring and evaluation guidance - Chapter 7: monitoring capture rates, in English (http://www.wrap.org.uk/content/monitoring-and-evaluation-guidance-chapter-7-monitoring-capture-rates)
- Edjabou, M. et al., Department of Environmental Engineering, Technical University of Denmark (2016): Food waste from Danish households: Generation and composition, in English (http://www.sciencedirect.com/science/article/pii/S0956053X16301167?via%3Dihub)
- Edjabou, M. et al., Department of Environmental Engineering, Technical University of Denmark (2014): Municipal solid waste composition: Sampling methodology, statistical analyses, and case study evaluation, in English (http://www.sciencedirect.com/science/article/pii/S0956053X14005261)
- Da Graça Madeira Martinho, M. et al., New University of Lisbon (2008): New guidelines for characterization of municipal solid waste: the Portuguese case (http://journals.sagepub.com/doi/abs/10.1177/0734242X08094624?url_ver=Z39.88-2003&rfr_id=ori%3Arid%3Acrossref.org&rfr_dat=cr_pub%3Dpubmed&)
- Inter-municipal Waste Management of Greater Porto (LIPOR): Waste composition analysis methodology in Portugal, in English (see Annex 8.2)

- French regional waste observatory (ORDIF): Données de caracterisations locales des ordures ménagères residuelles (omr) en Île-de-France, in French (http://www.ordif.com/sites/ordif/files/document/publication/rapport-caracterisations-vd.pdf)
- French Environment and Energy Management Agency (ADEME): Guide méthodologique pour la caractérisation des flux de déchets encombrants collectés dans les déchèteries et l'expérimentation du démantèlement d'objets, in French (http://www.sinoe.org/thematiques/consult/ss-theme/36)

For comparability of data, it can be valuable to refer to the method applied for the analysis when reporting the results. The composition analysis of mixed waste requires the sampling and analysis of the composition of the waste for the given territory of a local waste authority.
The calculations of some indicators (e.g. capture rates) require a composition analysis of mixed waste. Composition analyses tend to be expensive and a stringent approach and methodology are required to ensure that results are comparable over the years. It is important to define the reference methodology to be applied. Also, factors such as the season and the type of households covered by the analysis can be taken into account when conducting the analysis. For example, during summer the share of biowaste in the mixed waste tends to be higher than in winter and the sorting performance of recyclable materials tends to be higher in independent households compared to apartment buildings which are more anonymous.

2. <u>Municipal solid waste or household waste</u>

Another main aspect is to specify whether MSW or only household waste is considered when calculating the environmental performance indicators. MSW encompasses household waste and similar commercial wastes as well as waste from street cleaning, etc.. It is therefore important to be consistent, as far as possible, in the use of MSW or household waste for the calculation of environmental performance indicators. Moreover, results of calculations should describe what is included in the MSW or household waste – i.e. are textiles, waste electric and electronic equipment (WEEE), batteries and accumulators, waste oils, etc. taken into account?

3. <u>Residents and tourists/commuters within the territory</u>

The non-resident population (e.g. tourists, commuters) in areas where their presence is relevant throughout the year or during specific seasons needs to be taken into account for the calculation of reliable indicators for monitoring and improving municipal solid waste management (Saladié, 2016). However, many other factors (e.g. seasonality, consumption patterns, type of economy, climate) influence municipal solid waste management (Saladié, 2016), and this report does not introduce any parameter/factor able to model and quantify numerically their influence on the calculation of the environmental performance indicators. Anyway, the indicators presented below warn the reader when they are influenced by some of the aforementioned parameters; these factors, in fact, need to be taken into account when analysing the results obtained.
Regarding the presence of tourists, instead, it is possible to introduce a factor which determines an equivalent number of residents over the whole year.

Therefore, in areas where the presence of tourists is relevant, it is recommended not to take into account only the number of inhabitants/residents living within the territory that the local waste authority is responsible for, because this does not necessarily represent the real number of people producing waste (please note that the presence of tourists affects only the calculation of the quantity-based indicators presented below). As an explanatory example, municipalities with a high number of tourists generate a significantly higher amount of MSW in total and per capita (if not considering their presence) than municipalities without such a flow of people. Within the scope of the environmental performance indicators, it is proposed to consider these additional waste producers, in territories where their presence is significant, by calculating the population equivalent and using this value (instead of number of residents) for the calculation of the quantity-based indicators. The population equivalent (eq_{pop}) can be calculated by taking into account the tourist guest nights which is usually an easily available number (e.g. from the local tourism office or the specific tourism department of the municipality) in areas with a relevant tourist presence:

$$eq_{pop} = residents + \frac{tourist\ guest\ nights}{365}$$

2.4.1. Indicators for the overall municipal solid waste management system

This section introduces the environmental performance indicators selected for the assessment of the overall municipal solid waste management. This means that the environmental performance indicators are not related to a specific waste stream, e.g. paper, but to the total municipal solid waste. Four **quantity-based** indicators have been selected for the assessment of the overall municipal solid waste management system.

Please note that all of these indicators are calculated for **municipal solid waste**, but for local waste authorities or waste management companies with separate waste statistics for MSW and household waste (i.e. quantifying household-like commercial waste and household waste separately), the indicators presented below can also be calculated only for **household waste**. It is then important, for internal reference or when publicly reporting waste statistics, to specify whether the data presented refers to MSW (including a share of commercial waste) or household waste only. Ideally, given that the fraction of MSW generated from businesses varies to a large extent from municipality to municipality (e.g. depending on the type and size of businesses whose waste is accounted for as MSW) and that this parameter is not a measure of the success of the waste management system in place, calculating these indicators for household waste would be more helpful. However, for the vast majority of local waste authorities and waste management companies, waste data monitoring is carried out only for MSW, with no differentiation between household and household-like commercial waste. In the territory of some local waste authorities, household and household-like commercial waste are collected and quantified separately which makes the data for each easily available. In a few cases, household waste is estimated from the MSW statistics; however, this operation needs to be transparent and well documented or it may not be meaningful.

MSW generation

Overview of indicator	
Phase of waste management	Waste generation and prevention
Name of indicator	MSW generation; calculated for MSW or, if data available, just for household waste
Type of indicator	Quantity-based indicator [kg/capita/year]
Explanation	The indicator describes the amount of total MSW generated within the territory administered by a local waste authority per year, in relation to the resident population. Waste monitoring is key in order to regularly record waste quantities for each different waste stream collected separately by all the different collection systems available in the territory (e.g. door-to-door, civic amenity sites, street bins). This indicator is useful for assessing overall waste generation trends as well as the results of any effort to promote waste prevention.
Calculation method	$Annual\ MSW\ generation = \frac{kg\ of\ municipal\ solid\ waste}{number\ of\ residents}$ The "number of residents" can be substituted by the "population equivalent" where tourist presence is relevant. Similarly, "municipal solid waste" can be substituted by "household waste" if data for household waste generation are available. MSW definition differs substantially across the EU (e.g. small quantities of construction and demolition waste may or may not be included). When a waste authority uses this indicator for comparison of its own results over time, the definition of MSW (i.e. the waste fractions included) just need to be consistent in the timeframe considered. Meanwhile, for comparison of the total MSW generation with an absolute reference value, such as the benchmark of excellence given in this document, this indicator can be calculated including only the fractions for which data are included in the reference value. In the case of the benchmark of excellence given in this document, reliable data was only available for the following waste fractions: organic/biowaste, paper and cardboard, glass, plastics, metals, bulky, WEEE and mixed waste. Therefore, only these waste fractions should be added up when comparing one local waste authority's performance with the second option of the benchmark. As far as possible, all figures used should refer to the same year.

Overview of indicator	
Relevant BEMPs	The relevant BEMPs for improving the environmental performance of the waste management system assessed with this indicator are: - integrated waste management strategies (Section 3.3.1); - pay-as-you-throw (Section 4.3.3); - awareness-raising (Section 4.3.5); - establishment of a network of waste advisers (Section 4.3.6); - home and community composting (Section 4.3.7); - local waste prevention programmes (Section 4.4.1); - schemes fostering the reuse of products and waste (Section 4.4.2).
Example calculation	The city of Ljubljana (SI) has 313 708 residents while the total amount of MSW generated yearly (2016) is 115 532 tonnes. Therefore, the annual MSW generation is 368 kg/capita. When the annual MSW generated is instead calculated only including the waste fractions of organic/biowaste, paper and cardboard, glass, plastics, metals, bulky, WEEE and mixed waste, the result is about 314 kg/capita. The city of Bristol (UK) has 449 300 residents and the total amount of MSW generated yearly (2016) is 171 698 tonnes. Therefore, the annual MSW generation is 382 kg/capita. When the annual MSW generated is instead calculated only including the waste fractions of organic/biowaste, paper and cardboard, glass, plastics, metals, bulky, WEEE and mixed waste, the result is about 358 kg/capita. Val di Non (IT) is a rural area in the north of Italy. The number of residents is 39 420 while the population equivalent, since the presence of tourists is relevant, is 43 081. The total MSW generated yearly (2016) is 17 697 tonnes and therefore the annual MSW generation is 411 kg/capita, calculated taking into account the population equivalent and not the number of residents. When the annual MSW generated is instead calculated only including the waste fractions of organic/biowaste, paper and cardboard, glass, plastics, metals, bulky, WEEE and mixed waste, the result is about 338 kg/capita. References: Petek I., 2017; Anthony S., 2017; Coletti D., 2017
Benchmark of excellence	The annual generation of MSW in the territory administered or managed (collected by all the different waste collection systems

Overview of indicator	
	available in the area) is: - lower than 75 % of the national average of municipal waste generation[24], using the national definition of municipal waste of their own country; **<u>or</u>** - lower than 360 kg/capita, if calculated only for the following waste fractions[25]: (i) organic/biowaste (e.g. green cuttings, food, kitchen waste), (ii) co-mingled packaging, (iii) paper and cardboard, (iv) glass, (v) plastics, (vi) metals, (vii) bulky, (viii) WEEE, and (ix) mixed waste.

Further explanation

When calculating the total amount of municipal solid waste generated, it is important to include the waste collected through all the different collection systems (e.g. door-to-door collection, kerbside collection, bring points, civic amenity sites, and deposit refund systems) by municipal services as well as by private companies operating on behalf of the local waste authority or an EPR scheme (this might be particularly important for WEEE as well as packaging waste depending on local conditions).

Two major waste streams need to be included in the calculation of the total MSW:

- All separately collected materials such as glass, plastic, metal, or biowaste but also WEEE, bulky waste, waste textiles, waste oils, etc. if covered by a separate collection scheme (independently of the type) and if included in the MSW definition applicable in the territory administered by a local waste authority. This waste fraction is referred to in Figure 2-4 by number 3.
- All mixed waste from the mixed waste bin; this fraction is illustrated by number 2 in Figure 2-4.

[24] As reported by national authorities or by the statistical office of the European Union (Eurostat)
[25] The following fractions have been selected because they are commonly monitored in the EU by local waste authorities and waste management companies and they are generally the most relevant fractions (by weight) in MSW.

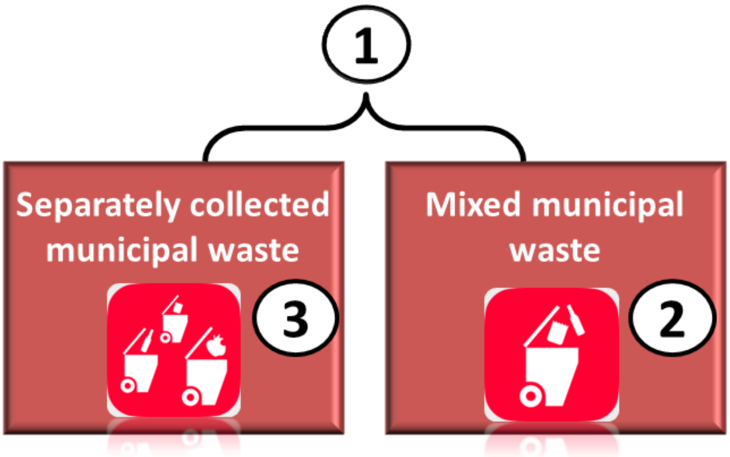

Figure 2-4. Data requirements to calculate the "Total MSW/household waste generation" indicator

It is very important to specify, both for internal reference and in public reporting, which waste streams were included in the calculation, e.g. whether WEEE, bulky waste, street sweeping waste, etc. were accounted for.

Data needs and potential sources of data

The data needed for the calculation of this indicator is the amount of all different waste streams collected within the territory administered by a local authority via the different collection systems in a specific year. If part or all of the waste collection is carried out by private companies, the local waste authority can obtain the data from the external contractors in order to have a comprehensive and reliable dataset. If some data are unavailable, this needs to be clearly acknowledged as a note to the calculation, including the estimation of its significance.

For the calculation of this indicator, it is fundamental to take into account all the MSW generated in the local area (including for example street sweepings) thanks to the full coverage of the territory by a waste collection system and detailed waste monitoring and accounting.

It needs to be stressed that figures on waste generation can be affected by waste collected by the informal sector (intercepting waste outside the official waste collection channel) and also by illegal dumping and combustion of dry waste (e.g. paper, plastic, wood waste), which is a practice still present in many areas, especially in rural settings. It is therefore fundamental that, together with the data needed for this indicator, the waste authority or waste management company also investigates whether these practices are significant in the local area considered and if so, possibly, estimates reliable figures for the amount of MSW generated but not accounted for in the indicators because it is not collected by the formal municipal waste collection system. Such a task may be complex and should be carried out with a robust methodology and involving the relevant stakeholders (e.g. local NGOs) that may have useful information and data.

Finally, the number of residents is required for the calculation and it is preferable that it refers to the same year as the waste statistics. As mentioned previously, in areas with a relevant tourist presence, the population equivalent can be calculated instead.

Advantages and/or disadvantages of the indicator

The indicator gives an important general overview of MSW and how much of such waste is generated in a certain year; its calculation enables a better understanding of the situation and improved decision-making. Over time, it can provide insights into how the waste generation changes from year to year.

In general, data for the calculation of this indicator is easily accessible to local waste authorities because the data on the MSW generation is the most commonly collected data for municipal solid waste management. However, there might be some limitations for MSW when some waste streams, e.g. packaging or textiles, are collected by private companies. As an example, some countries have established an EPR scheme for packaging waste for which the collection and treatment is implemented by private companies. There are cases where the data per territory administered by a local waste authority is not (made) available because the companies are required to report amounts only at the national/regional level but not per local waste authority. Thus, the total MSW generation might not be reflected in its totality by this environmental performance indicator. When this is the case, it needs to be clearly presented alongside the figures for the indicator and, if possible, an estimation of the missing quantities (e.g. based on the regional figures, if available) can be used to complement the indicator.

Interpretation of result

It is important to put the value calculated for this indicator into context.

The calculated MSW generation is highly dependent on external factors that are not related to waste management at all, such as rural/urban location, consumption patterns, weather, significant presence of daily commuters, GDP in the territory administered by a local authority or the fact that waste generation tends to be lower when the economy is stagnant or in recession.

Additionally, in areas where there is no detailed waste monitoring or where a part of the waste generated is not collected by the formal municipal waste collection system, figures on MSW generation calculated according to this indicator could underestimate the real situation. This is the case, for example, in areas where illegal dumping or burning of some waste fractions is significant, or where relevant quantities of waste are picked up by the informal sector rather than delivered to the official waste collection channels. In these cases, the indicator can be correctly interpreted only if looked at alongside a reliable estimate of the amount of waste collected by the informal sector, illegally dumped or burned by citizens or businesses (i.e. all MSW not monitored or not collected by the formal municipal waste collection system).

In addition, MSW generation does not provide an indication of how the waste management system works in terms of performance. For a more comprehensive understanding, the results for this indicator should be analysed together with the results for the other indicators proposed in this chapter.

MSW generation across municipalities will also vary significantly depending on the types of waste streams that are included in the calculation, e.g. if heavier fractions such as bulky waste are excluded, this will significantly lower the quantity per capita.

The indicator can however provide an overview of the status of waste prevention or if waste prevention measures have been successful when it is measured over time.

Anyhow, a direct link between waste prevention measures, e.g. an information campaign addressed to residents, and a decrease in MSW generation cannot be made as the decrease may also depend on changes in the economic factors, as described above, which need to be analysed as part of the interpretation.

Overall, it is very important to consider the context when interpreting the results of quantity-based indicators, such as average income of the population, population density, type of local economy, urban/rural environment, and other socio-economic factors. This is important when analysing changes in the indicators over time but even more so if comparing data from different local waste authorities, as variations may be due to socio-economic factors. A decrease in the generation over time, which is not linked to a decrease in population or economic downturn, could be an indication that the local waste authority has been successful with its waste prevention strategy. An increase or constant figures may indicate a need to take action (see Section 4.4 on waste prevention).

Amount of mixed MSW collected

Overview of indicator	
Phase of waste management	Waste generation, waste collection
Name of indicator	Amount of mixed MSW collected; calculated for MSW or, if data available, just for household waste
Type of indicator	Quantity-based indicator [kg/capita/year]
Explanation	The indicator describes the amount of mixed MSW collected per capita per year. Its calculation takes into account the waste collected as non-source separated mixed waste. Mixed MSW contains all waste fractions for which no separate container or other collection system is available. In systems where most of the waste is segregated at source and collected separately, this is often referred to as "residual waste". The calculation of the indicator amount of mixed MSW collected can be integrated by adding the amount of separately collected fractions that cannot be recycled (i.e. rejects from sorting/recycling plants), provided that the local waste authority (or the waste management company) is aware of these quantities. The amounts of rejects from sorting/recycling can be based on actual data (from sorting/recycling plants) or reliable estimations based on the amount of misthrows found in the separately collected fractions. Similarly, in the event that mixed waste is pretreated (e.g. in an MBT plant) and the local waste authority (or the waste management company) is aware of the dry recyclables that are sorted out from mixed waste and sent for

Overview of indicator	
	recycling, the quantity of dry recyclables can be subtracted from the amount of mixed MSW collected.
Calculation method	$\text{Amount of mixed MSW collected} = \frac{\text{mixed waste } (+\text{rejects from sorting/recycling} - \text{dry recyclables from MBT})}{\text{number of residents}}$ The number of residents can be substituted by the population equivalent where tourist presence is relevant. As far as possible, all figures used should refer to the same reference year.
Relevant BEMPs	The relevant BEMPs for improving the environmental performance of the waste management system assessed with this indicator are: - integrated waste management strategies (Section 3.3.1); - pay-as-you-throw (Section 4.3.3); - awareness-raising (Section 4.3.5); - establishment of a network of waste advisers (Section 4.3.6); - home and community composting (Section 4.3.7); - local waste prevention programmes (Section 4.4.1); - schemes fostering the reuse of products and waste (Section 4.4.2).
Example calculation	The resident population in the department of Lot in France is 180 553. The annual (2015) amount of mixed MSW collected is 47 882 tonnes. Therefore the amount of mixed MSW collected is 265 kg/capita. However, the department knows that, out of the separately collected waste, which is sent for recycling, there are rejects from the sorting and recycling plants of 4 111 tonnes per year. In order for this indicator to be more meaningful, the department may thus calculate it including those rejects together with the mixed MSW collected. This would correspond to an annual amount of mixed MSW collected of 287 kg/capita. References: (Lot Tourisme, 2017) and (Syndicat Départemental pour l'Élimination des Déchets Ménagers et Assimilés, SYDED, 2015)
Benchmark of excellence	N/A

Further explanation

The total amount of mixed MSW is the amount of waste that is collected through the mixed waste collection system, e.g. the "black bin" where citizens put all the waste that is not source separated. In systems with advanced separate collection, only fractions that are unrecyclable are collected as mixed waste. In these cases, mixed waste is sometimes referred to as "residual waste"; however, in this document only the term mixed waste is used.

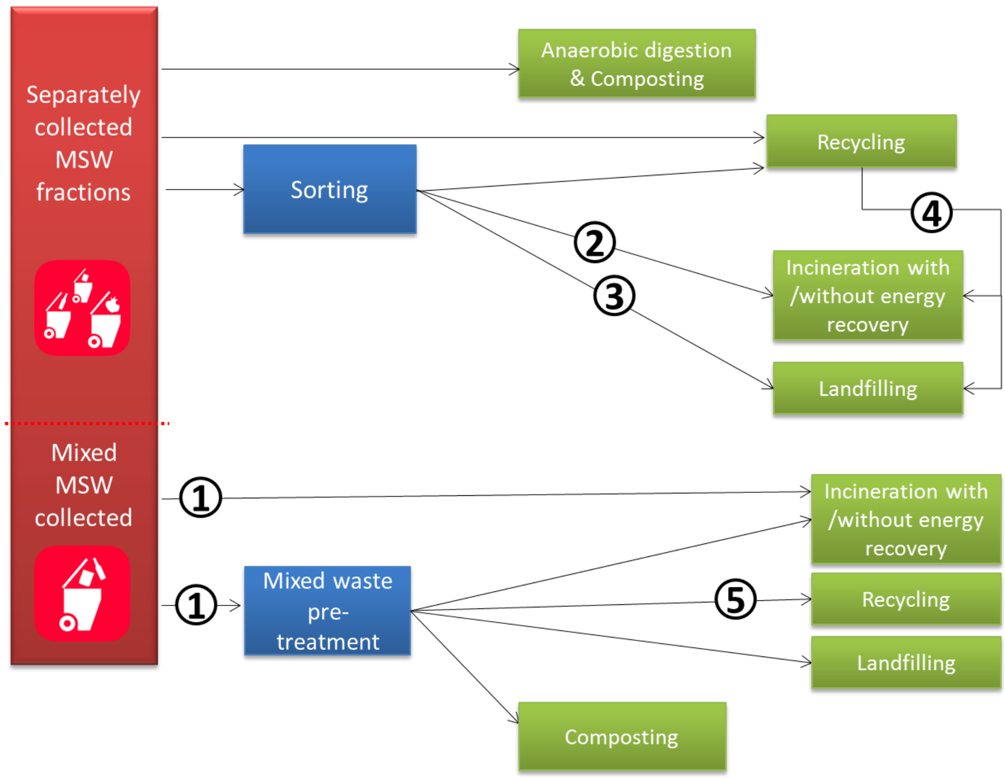

Figure 2-5. Data requirements to calculate the "Amount of mixed MSW collected" indicator

The amount of mixed MSW collected corresponds to number 1 in Figure 2-5 above, where a general model of the waste flows in a municipality is reported. Please note that, in order to calculate this environmental performance indicator, a specific waste flow model of the local waste management system needs to be available (see Section 2.3). It is noteworthy that mixed MSW collected not only includes the mixed waste fraction collected by door-to-door or street bin systems but also mixed waste that is collected (if any) at civic amenity sites. If the amount of waste collected from illegal temporary or permanent dumping sites is known, this can also be added to the total amount of mixed waste collected, as well as waste from street sweeping and street litter bins.

The calculation of the indicator amount of mixed MSW collected can be integrated, if known, by adding the waste rejected from sorting and recycling (numbers 2, 3, 4) of the separately collected fractions.

The amount of mixed MSW collected represented by number 1 in Figure 2-5 is either sent directly to recovery (e.g. incineration with energy recovery) or disposal operations or sent to pretreatment, e.g. in an MBT plant. This distinction depends on the waste management system in place in each local waste authority. If mixed waste is pretreated, the amount (if known) of any recyclable that is separated from mixed waste and sent for recycling (number 5) could be subtracted from the reported amount of mixed waste collected.

Data needs and potential sources of data

The data needed for the calculation are the amount of mixed MSW collected within the territory administered by a local waste authority in a specific year including all collection schemes for mixed waste and, where relevant, the amount of waste that is found to be disposed of by illegal dumping, if available.

If the waste collected is delivered for sorting/recycling/treatment to an independent actor (such as a private recycling company), knowing the amount of rejects from sorting/recycling and the amount of recyclables removed from mixed waste in M(B)T plants and sent for recycling requires data from the treatment operator. Local waste authorities can experience difficulties in obtaining data from other actors (e.g. private operators) due to the restricted access to information. This can depend on the unwillingness of other actors to share this information or on the fact that the required data are not collected by the other actors as they receive waste for treatment from several sources and do not investigate rejects or recyclables individually from the waste coming from each specific territory. The waste authority or waste management company can try, thanks to the introduction of a specific clause in the contract specifications, to obtain the data. It is also important to think about all the waste treatment routes in order not to miss any potentially relevant data.

Furthermore, the number of residents is required for the calculation and it should preferably refer to the same year as the waste statistics. As mentioned previously, in areas with a relevant tourist presence, the calculated population equivalent can be used instead.

Advantages and/or disadvantages of the indicator

The environmental performance indicator for the amount of mixed MSW collected provides an insight into the total amount of MSW which generally undergoes a lower treatment option in the waste hierarchy compared to separately collected fractions. This environmental performance indicator can be compared to the previous one (total MSW generation) in order to get a better insight into the performance of the collection systems, i.e. how much of the waste generated is not collected separately.

Over time, this indicator can provide an overview of how the amount of mixed waste has changed; a decrease in the amount of mixed waste collected (either because of waste prevention or because of more waste being collected separately) is considered an improvement in MSW management.

Interpretation of result

This indicator allows the interpretation of how much of the MSW collected is mixed waste (i.e. not separately collected). This is very relevant because mixed waste generally undergoes a lower treatment operation, in the waste hierarchy, compared to separately collected fractions that are sent for recycling.

The indicator can provide an overview of the performance of separate waste collection systems. If the separate collection systems (including deposit refund schemes and extended producer responsibility schemes) work well and include most waste fractions, the share of mixed waste in the total waste generation should be very low. Meanwhile, high quantities of mixed waste might indicate that the separate collection systems in place do not work well. A composition analysis can then provide an indication of which waste stream is the most problematic. For instance, if there is a high amount of glass in the mixed waste, glass bring points may not be sufficiently used. A possible cause may be that they are too far away from many households (thus leading them to dispose of it in the mixed waste).

However, this indicator also has several limitations. The indicator is highly dependent on external factors that are not related to waste management at all, such as rural/urban location, consumption patterns, weather, significant presence of daily commuters, GDP in the territory administered by a local authority, or the fact that waste generation tends to be lower when the economy is not performing well. Additionally, the numbers across municipalities will vary significantly depending on the types of collection systems established, e.g. if there is no separate biowaste collection as this fraction is usually almost 50 % of the mixed waste (when no separate collection is established). Thus, if interpreting the results of this indicator across municipalities, it is important to take into account all external factors.

An increase in the mixed MSW collection over time, which is not linked to population or economic growth, could be an indication for the local waste authority that the separate waste collection strategy needs improvement. However, it may also indicate that more waste generated in the territory is being delivered to the official waste management collection system.

Finally, this indicator only assesses certain elements of the performance of the MSW management system. For a more comprehensive understanding, the results for this indicator should be analysed together with the results for the other indicators proposed in this chapter.

MSW sent to energy recovery and/or disposal

Overview of indicator	
Phase of waste managemen t	Waste treatment
Name of indicator	Waste sent to energy recovery and/or disposal; calculated for MSW or, if data available, just for household waste
Type of indicator	Quantity-based indicator [kg/capita/year]
Explanation	The indicator measures the annual amount of MSW that is treated by either incineration with energy recovery and/or disposal operations, such as landfilling or incineration without energy recovery. If this information is not available as such (e.g. in the case of waste authorities or waste management companies not managing the whole process), it can be calculated as follows. The fate of the mixed waste collected is taken into account: if mixed waste is directly sent to energy recovery and/or disposal, the quantity can be directly used for the calculation. In the event that the mixed waste is pretreated (e.g. in an MBT plant), the local waste authority (or waste management company) includes in the calculation of the indicator the actual quantities of waste that, after the pretreatment, are sent to energy

Overview of indicator	
	recovery and/or disposal. Similarly, it is important that the local waste authority (or the waste management company) also takes into account in the calculation of the indicator the amount of rejects from the sorting/recycling of the separately collected fractions that are not recycled but sent to energy recovery and/or disposal. The amounts of rejects from sorting/recycling can be based on actual data (from sorting/recycling plants) or estimations based on the amount of misthrows found in the separately collected fractions. In the event that the local waste authority (or waste management company) cannot fully calculate the indicator, considering all its factors, it can report only the amount of mixed waste sent to energy recovery and/or disposal, acknowledging that the indicator is partially calculated. In such cases, it is important to clearly state the elements that are not included in the calculation (e.g. rejects from separately collected fractions sent to energy recovery and disposal). Moreover, appropriate measures to obtain reliable data for the full calculation of the indicator can be put in place to improve the usefulness of this indicator.
Calculation method	$MSW\ sent\ to\ energy\ recovery\ and/or\ disposal$ $= \frac{Mixed\ waste_{to\ energy\ recovery/disposal} + Rejects_{to\ energy\ recovery/disposal}}{number\ of\ residents}$ The number of residents can be substituted by the population equivalent where tourist presence is relevant. As far as possible, all figures used should refer to the same reference year. When a waste authority or waste management company uses this indicator for comparison of its own results over time, the factors included in the calculation (i.e. only mixed waste sent to energy recovery and/or disposal or also the rejects from sorting/recycling sent to energy recovery and/or disposal) need to be consistent in the timeframe considered. This indicator can also be used for comparing the amount of waste sent to energy recovery and/or disposal with an absolute reference value, such as the benchmark of excellence reported in this document. Given the fact that (local) waste authorities and waste management companies do not usually have access to the data on the amounts of rejects from sorting and/or recycling operations of the fractions collected to be recycled, the benchmark of excellence associated with this environmental performance indicator includes only the amount of collected mixed MSW sent to energy recovery and/or disposal. In order to obtain a complete picture from the calculation of this indicator, however, it is important that waste authorities and waste

Overview of indicator	
	management companies collect complete information on all the flows of waste sent to energy recovery and/or disposal.
Relevant BEMPs	The relevant BEMPs for improving the environmental performance of the waste management system assessed with this indicator are: - integrated waste management strategies (Section 3.3.1); - pay-as-you-throw (Section 4.3.3); - awareness-raising (Section 4.3.5); - establishment of a network of waste advisers (Section 4.3.6); - home and community composting (Section 4.3.7); - local waste prevention programmes (Section 4.4.1); - schemes fostering the reuse of products and waste (Section 4.4.2).
Example calculation	In the city of Treviso (IT), the resident population is 83 950. All the MSW collected as mixed waste is sent to energy recovery and/or disposal and in 2016 the amount was 5 851 tonnes. The local waste authority, however, is not aware of the amount of rejects from sorting/recycling of the separate waste fractions, since separately collected waste fractions are treated in plants which serve a broader area not limited only to the city of Treviso. Therefore, the waste sent to energy recovery and/or disposal indicator can be calculated partially, only taking into account the amount of mixed waste collected and sent to energy recovery and/or disposal. The value obtained for the year 2016 is 69.7 kg/capita. In the county of Aschaffenburg (DE), the resident population (2016) is 173 585. All the MSW collected as mixed waste is sent to energy recovery and/or disposal and in 2016 the amount was 11 852 tonnes. The local waste authority, however, is not aware of the amount of rejects from sorting/recycling of the separate waste fractions, since separately collected waste fractions are treated in plants where waste from different locations is processed. Therefore, the waste sent to energy recovery and/or disposal indicator can be calculated partially, only taking into account the amount of mixed waste collected and sent to energy recovery and/or disposal. The value obtained for the year 2016 is 68.3 kg/capita. Val di Non (IT) is a rural area in the north of Italy. The number of residents is 39 420 while the population equivalent, since the presence of tourists is relevant, is 43 081. All the MSW collected as mixed waste is sent to energy recovery and/or disposal and in 2016 the amount was 3 009 tonnes. The local waste authority, however, is not aware of the amount of rejects from sorting/recycling of the separate waste fractions, since separately collected waste fractions are treated in plants where waste from different locations is processed. Therefore, the waste sent to energy recovery and/or disposal indicator can be calculated partially, only taking into account

Overview of indicator	
	the amount of mixed waste collected and sent to energy recovery and/or disposal. The value obtained for the year 2016 is 68.3 kg/capita, calculated taking into account the population equivalent and not the number of residents. <u>References</u>: Mattiello M., 2017; Morlok J., 2017; Coletti D., 2017
Benchmark of excellence	The annual amount of collected mixed MSW sent to energy recovery and/or disposal is: - lower than 15 %[26] of the national average of municipal waste generation[27]; <u>**or**</u> - lower than 70 kg/capita.

Further explanation

Following the waste hierarchy, prevention, preparation for reuse and recycling are to be prioritised over energy recovery and disposal (i.e. landfilling and incineration without energy recovery). This indicator evidences the amount of MSW that is directed to waste incineration with and without energy recovery and is disposed, leading to valuable materials potentially leaving the material-cycle. Incineration with energy recovery refers to treatment option R 1 according to the Waste Framework Directive (WFD) and disposal operations are subsumed as D operations e.g. incineration without energy recovery and landfilling. The full list of treatment operations referred to in the WFD is available in Annex 8.1 of this document.

This indicator applies to all MSW generated within the territory administered by a local waste authority collected by any type of scheme (door-to-door, bring points, civic amenity sites etc.) and for all streams including, where relevant, WEEE, bulky waste, street sweeping waste, litter bins and more depending on the MSW definition.

[26] Please note that the formulation 'the annual amount of collected mixed MSW sent to energy recovery and/or disposal is lower than 15%...' does not necessarily mean that 85% of municipal waste is separately collected for reuse and recycling. For municipalities that generate less municipal waste than the national average in their country, 15% of the national average would correspond to a higher share (e.g. 20–30%) of their own municipal waste generation.

[27] As reported by National Authorities or by the statistical office of the European Union (Eurostat).

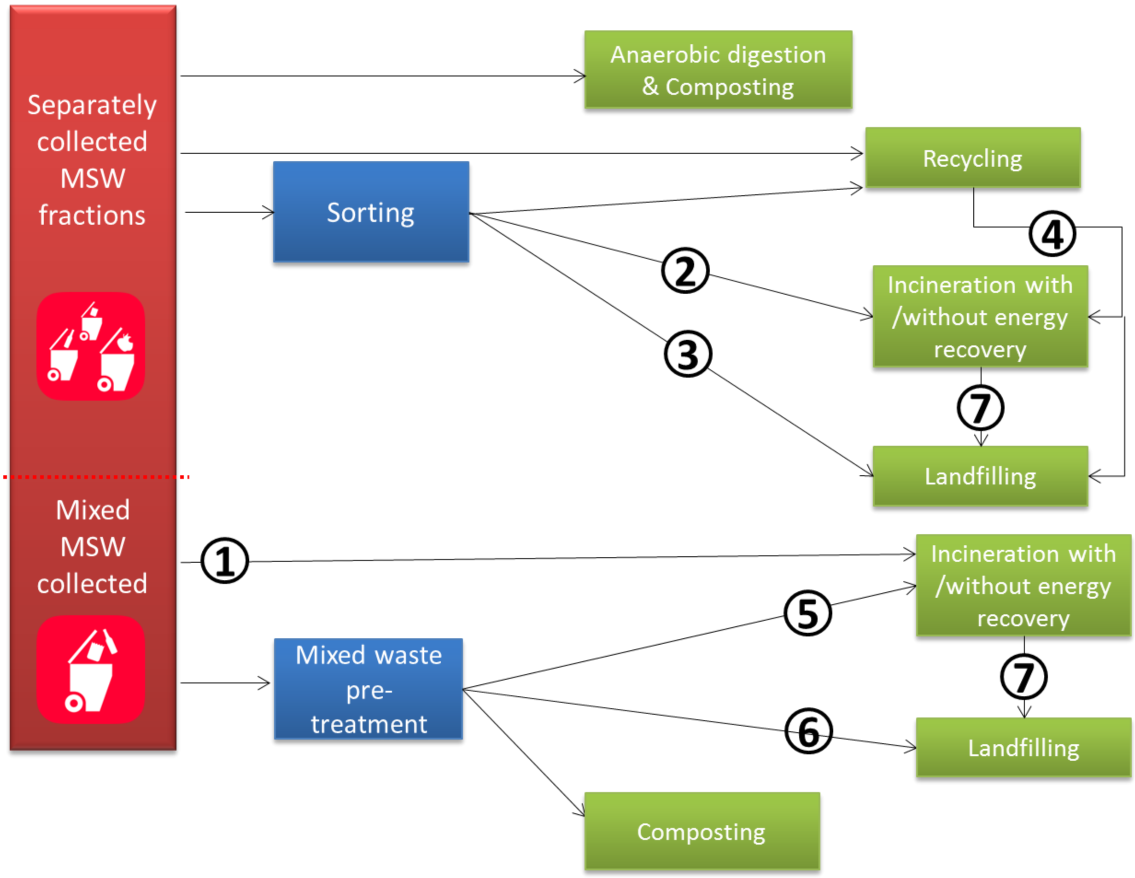

Figure 2-6. Data requirements to calculate the "MSW sent to energy recovery and/or disposal" indicator

If mixed waste is sent directly to energy recovery and/or disposal, the quantity marked with the number 1 in Figure 2-6 can be directly used for the calculation of this indicator. In the event that mixed waste is pretreated (e.g. in an MBT plant) and the local waste authority (or waste management company) is aware of the actual quantities of waste sent to energy recovery and/or disposal (numbers 5 and 6 in Figure 2-6), these can be used for the calculation of the indicator. Similarly, in the event that the local waste authority (or the waste management company) is aware of the amount of rejects from sorting/recycling of separately collected fractions sent to energy recovery and/or disposal (numbers 2, 3 and 4 in Figure 2-6), these quantities could also be added to the indicator.

For number 7, it is important not to count amounts of waste treated twice, e.g. waste incineration results in ashes/slags which are disposed in landfill (number 7), thus this amount, if taken into consideration, cannot be counted in addition to the material that enters the incineration treatment (numbers 5). For number 4 in Figure 2-6, it needs to be included in the calculation of this indicator; however, it is important that the same waste stream is not double counted as the amount of waste recycled.

Data needs and potential sources of data

When the waste authority or waste management company calculating the indicators manages the whole process, it can know the exact amount of waste sent to each energy recovery and disposal plant. Once it ensures that there is no double counting (incineration ashes/slags that are landfilled), the data of inputs to those plants can be used as such.

If this information is not available as such (e.g. because some or all of the waste is given for treatment/sorting/recycling to other parties), the indicator can be calculated as follows.

Several pieces of data are needed to calculate this indicator and might sometimes be difficult to obtain. Altogether, if certain amounts needed for the calculation are not known, e.g. rejects from mixed waste pretreatment going to incineration (number 5 in Figure 2-6), it is good to specify this as a note to the calculation of the indicator. It is important to consider the following:

- The information most commonly available to local authorities / waste management companies for the calculation of this indicator is the amount of mixed waste that is (directly) sent to energy recovery/disposal.
- Data from operators of pretreatment plants, on the amounts of mixed waste sent to energy recovery and/or disposal (data from private plants might not be directly accessible to the local waste authority), is usually difficult to obtain and not representative of the specific waste delivered to the plant from the territory considered.
- Data from sorting/recycling plant operators, if not municipal, for rejects that were sent to incineration and/or landfill are also difficult to obtain. The amounts of rejects from sorting/recycling can be based on actual data or estimations based on the amount of misthrows found in the separately collected fractions.

It is clear that for the comprehensive calculation of this indicator it is important to obtain data, if the relevant operations are not managed in-house, from the waste treatment operators or the contracted MSW collectors. The provision of this data can be included as a clause (e.g. for annual or monthly transmission) in the contract with the other actors such as private operators.

Furthermore, the number of residents is required for the calculation and it would be preferable for it to refer to the same year as the waste statistics. As mentioned previously, in areas with a relevant tourist presence, the population equivalent can be calculated instead.

Advantages and/or disadvantages of the indicator

The indicator provides a good overview of where a local waste authority stands in relation to the two least favourable options of the waste hierarchy: energy recovery and disposal.

Overall, since this is a quantitative indicator, the results might vary significantly across the municipalities: if there is high waste generation in general, the amount for this indicator may also be high. Amounts may depend on external factors that are not directly related to MSW management at all, such as GDP within the territory administered by a local waste authority, economics of recycling vs energy recovery, or the fact that waste generation tends to be lower when the economy is not performing well. In addition, only measuring generation amounts does not provide an indication of how well the system works in terms of performance. Furthermore, the values of this environmental performance indicator across municipalities can vary significantly depending on the types of waste streams that are included in the calculation, i.e. only the quantities of mixed waste sent to energy recovery and disposal or also rejects from sorting/recycling operations.

A barrier to the calculation of this indicator is that currently it is difficult for waste authorities and waste management companies to obtain data from all steps of the waste management chain, especially when treatment operations (e.g. sorting/recycling, MBT) are performed by other actors (e.g. private plant operators). As a consequence, in order to ease the comparison with the calculated indicator, the benchmark of excellence associated with this indicator includes only the amount of collected mixed MSW sent to energy recovery and/or disposal. However, waste authorities and waste management companies need to put in place all actions possible in order to obtain, step by step, a comprehensive overview of all waste streams going to energy recovery and/or disposal.

Interpretation of result

This indicator gives an indication of the amounts of MSW treated with the least favourable options (according to the waste hierarchy), namely disposal and energy recovery operations (treatment option R1 and D according to Annex II to the Waste Framework Directive (WFD), see Annex 8.1).

When interpreting the result of the indicator, it is important to take into consideration the treatment infrastructure available for the MSW. If a local waste authority relies heavily on incineration with energy recovery, the result of the indicator cannot be interpreted in the same way as if the same value for this indicator is calculated by a local waste authority that mostly disposes of waste in landfill (see next indicator). A local waste authority which mainly incinerates with energy recovery has a better environmental performance in terms of municipal solid waste management (according to the waste hierarchy) than a local waste authority which disposes of all quantities, although they might have the same/similar amount calculated under this environmental performance indicator. This aspect will be reflected by the results of the following environmental performance indicator, 'waste sent to disposal'. Therefore, it is useful to consider this aspect when analysing the results.

If monitored over time, this indicator can help the local waste authority assess whether the treatment of MSW has moved up in the waste hierarchy. If the amount of waste sent to energy recovery and/or disposal decreases, the local waste authority performs better in MSW management because more waste was prevented or reused or recycled.

Finally, this indicator only assesses certain elements of the performance of the MSW management system. For a more comprehensive understanding, the results for this indicator should be analysed together with the results for the other indicators proposed in this chapter.

MSW sent to disposal

Overview of indicator	
Phase of waste management	Waste treatment

Overview of indicator	
Name of indicator	MSW sent to disposal; calculated for MSW or, if data available, just for household waste
Type of indicator	Quantity-based indicator [kg/capita/year]
Explanation	The indicator measures the annual amount of MSW that is sent to disposal, such as landfill or incineration, without energy recovery (all disposal operations are defined in Annex I to the WFD, see Annex 8.1). If this information is not available as such (e.g. in the case of waste authorities or waste management companies not managing the whole process), it can be calculated as follows. Firstly, the fate of the MSW collected as mixed waste is taken into account for the calculation: if mixed waste is sent directly to incineration without energy recovery the quantity can be directly used for the calculation. If mixed waste instead undergoes pretreatment (e.g. in an MBT plant), the quantities actually sent to disposal after treatment are needed. Finally, for the calculation of this indicator, it is important to include also the amount of rejects from sorting/recycling of separately collected fractions that are sent to disposal, if known by the local waste authority/waste management company.
Calculation method	$Waste\ sent\ to\ disposal = \frac{Mixed\ waste_{to\ disposal}\ (+Rejects_{to\ disposal})}{number\ of\ residents}$ The number of residents can be substituted by the population equivalent where tourist presence is relevant. As far as possible, all figures used should refer to the same year.
Relevant BEMPs	The relevant BEMPs for improving the environmental performance of the waste management system assessed with this indicator are: - integrated waste management strategies (Section 3.3.1); - pay-as-you-throw (Section 4.3.3); - awareness-raising (Section 4.3.5); - establishment of a network of waste advisers (Section 4.3.6); - home and community composting (Section 4.3.7); - local waste prevention programmes (Section 4.4.1); - schemes fostering the reuse of products and waste (Section 4.4.2).
Example calculation	The resident population in Vienna (AT) is 1 741 246. The annual (2013) amount of MSW sent to disposal is 325 tonnes while the amount of rejects from sorting/recycling of separately collected

Overview of indicator	
	fractions sent to disposal is not known. Therefore, the annual amount of waste sent to disposal per person is 0.2 kg/capita. In the county of Aschaffenburg (DE), the resident population (2016) is 173 585. The annual (2016) amount of mixed waste sent to disposal is 914 tonnes. The amount of rejects from sorting/recycling of separately collected fractions sent to disposal is not known. Therefore, the annual amount of waste sent to disposal per person is 5.3 kg/capita. The city of Ljubljana (SI) has 313 708 residents. The annual (2016) amount of mixed waste sent to disposal is 3 319 tonnes. The amount of rejects from sorting/recycling of separately collected fractions sent to disposal is not known. Therefore the annual amount of waste sent to disposal per person is 10.5 kg/capita. <u>References:</u> BiPRO, Capital Factsheet - Vienna, 2015; City of Vienna, 2017; Morlok J., 2017; Petek I., 2017.
Benchmark of excellence	The annual amount of MSW sent to disposal is: - lower than 2 % of the national average of municipal waste generation[28]; **<u>or</u>** - lower than 10 kg/capita.

Further explanation

Following the principle of the waste hierarchy, prevention, preparation for reuse and recycling are to be prioritised over energy recovery and disposal. This indicator provides a good overview of all MSW that is directed towards the least favourable treatment option in the waste hierarchy (incineration without energy recovery and landfilling). In general, it can be assumed that the municipal solid waste management of a local waste authority is not performing well if a high share of the MSW generated is sent to disposal. A detailed list of disposal operations is reported in Annex 8.1.

This indicator applies to all MSW generated within the territory administered by a local waste authority collected by any type of scheme (door-to-door, bring points, civic amenity sites etc.) and for all streams including, where relevant, WEEE, bulky waste, street cleaning waste etc., depending on the MSW definition.

[28] As reported by National Authorities or by the statistical office of the European Union (Eurostat)

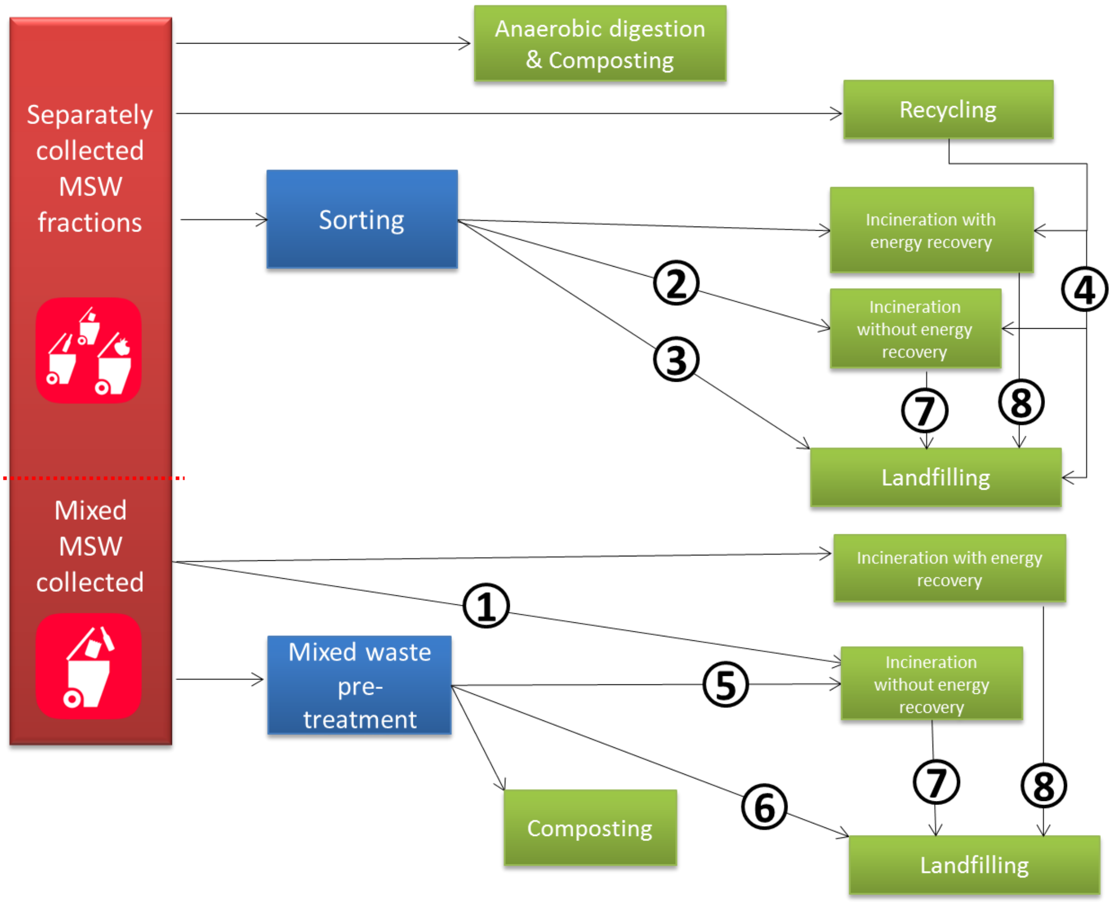

Figure 2-7. Data requirements to calculate the "MSW sent to disposal" indicator

If mixed waste is sent directly to incineration without energy recovery, the quantity marked with the number 1 in Figure 2-7 can be directly used for the calculation of this indicator. In the event that mixed waste is pretreated (e.g. in an MBT plant) and the local waste authority (or waste management company) is aware of the actual quantities of waste sent to disposal (numbers 5 and 6 in Figure 2-6), these can be used for the calculation of the indicator. Similarly, in the event that the local waste authority (or the waste management company) is aware of the amount of rejects from sorting/recycling of separately collected fractions sent to disposal (numbers 2, 3 and 4 in Figure 2-7), these quantities could also be added to the indicator. The calculation of the indicator could also include (if known) the amount of ashes/slags generated from incineration with energy recovery which are sent to landfill (number 8 of Figure 2-7).

For number 7, it is important not to count amounts of waste treated twice, i.e. waste incineration results in ashes/slags which are disposed of in landfill (number 7), thus this amount, if taken into consideration, cannot be counted in addition to the material that enters the incineration without energy recovery treatment (e.g. number 5). For number 4 in Figure 2-7, it needs to be included in the calculation of this indicator; however, it is important that the same waste stream is not double counted as the amount of waste recycled.

Data needs and potential sources of data

When the waste authority or waste management company calculating the indicators manages the whole process, it can know the exact amount of waste sent to each disposal plant. The data of inputs to those plants can be used as such to calculate this indicator.

If this information is not available as such (e.g. because some or all of the waste is given for treatment/sorting/recycling to other parties), the calculation of this indicator can rely on the following data:

- data on the amounts of mixed waste sent to incineration without energy recovery and/or landfill;
- data from operators of pretreatment plants, if not municipal, on the amounts of treated mixed waste sent to incineration without energy recovery and/or landfill;
- data from sorting/recycling plant operators, if not municipal, for rejects that were sent to incineration without energy recovery and/or landfill.

It is clear that for the comprehensive calculation of this indicator it is important to obtain data, if the relevant operations are not managed in-house, from the waste treatment operators or the contracted MSW collectors. The provision of this data can be included as a clause (e.g. for annual or monthly transmission) in the contract with the other actors such as private operators.

Furthermore, the number of residents is required for the calculation and it would be preferable for it to refer to the same year as the waste statistics. As mentioned previously, in areas with a relevant tourist presence, the population equivalent can be calculated instead.

Advantages and/or disadvantages of the indicator

The indicator provides an overview of the least preferable (according to the waste hierarchy) MSW treatment operation and can show improvements in terms of the waste management over time. This indicator can therefore be compared to the waste sent to energy recovery and/or disposal and provide an overview of the changes over time in the waste management, e.g. towards increased energy recovery and away from disposal.

A barrier to the calculation of this indicator is that it can be difficult to obtain data from all steps of the waste management chain, especially when treatment operations (e.g. sorting/recycling, MBT) are performed by other actors such as private plant operators.

Interpretation of result

Comparing the value obtained for this indicator over time can help the waste authority or waste management company to assess if the treatment of MSW has moved up the waste hierarchy. If waste sent to disposal has decreased, waste has either been prevented, reused, recycled or recovered, all of which are preferable options in the waste hierarchy.

Since this is a quantitative indicator, the amounts calculated may depend on external factors that are not directly related to MSW management, such as GDP within the territory administered by a local authority or the fact that waste generation tends to be lower when the economy is not performing well.

Furthermore, the values of this environmental performance indicator across municipalities can vary significantly depending on the types of waste streams that are included in the calculation, i.e. only the quantities of mixed waste sent to disposal or also rejects from recycling operations.

In any case, if a large share of the waste generated within the territory administered by a local authority is sent to disposal, this indicator gives a clear signal of the need for an urgent radical change to the waste management strategy in favour of options higher up in the waste hierarchy. This should also be reflected in the current and future plans for the development of waste treatment plants.

Finally, this indicator only assesses certain elements of the performance of the MSW management system. For a more comprehensive understanding, the results for this indicator should be analysed together with the results for the other indicators proposed in this chapter.

2.4.2. Waste-stream-specific indicators

This section introduces the common environmental performance indicators selected for the assessment of specific collected waste streams in municipal solid waste management, such as paper and cardboard, plastics, metal, glass and biowaste. Three indicators have been selected, of which two are performance ratios and one is quantity-based. The first two indicators (capture rate and impurity rate) can be calculated per individual waste stream, i.e. paper and cardboard, plastics, metal and glass. If a co-mingled collection is established, e.g. for plastics and metals, and the fractions cannot be disaggregated, the ratios can be calculated for the co-mingled waste stream. No capture rate is calculated for biowaste as data for home composting for example are difficult to gather and have a strong influence on the result.

The third, quantity-based, indicator is for biowaste only and reflects its amount in mixed waste.

Capture rate of a specific waste stream

Overview of indicator	
Phase of waste management	Waste collection
Name of indicator	Capture rate of a specific waste stream
Type of indicator	Performance ratio [%]
Explanation	The capture rate is the percentage of the estimated generation of a specific waste fraction that is collected separately. It provides insights into the efficiency (i.e. how efficient in intercepting the recyclables) of a separate collection system. The precondition for the calculation of this indicator is that a composition analysis of the mixed waste has been performed. In

Overview of indicator	
	addition, the amounts collected by each collection system for each material can be compared to the total amount of the same material generated within the territory administered by a local authority. The capture rate can be calculated for the separately collected fractions, e.g.: - plastic; - metal; - paper and cardboard; - glass; - co-mingled packaging; - biowaste[29].
Calculation method	Example: capture rate for glass: $Capture\ rate\ for\ glass = \frac{kg\ of\ glass\ separately\ collected}{kg\ of\ total\ glass\ waste\ generation} * 100\ \%$ Where $Total\ glass\ waste\ generation = kg\ separately\ collected\ glass + kg\ of\ glass\ in\ mixed\ waste$ With $kg\ of\ glass\ in\ mixed\ waste = kg\ of\ total\ mixed\ waste * \%\ of\ glass\ in\ mixed\ waste$ *Where the % of glass in mixed waste is calculated from the composition analysis of the mixed waste.* The calculations for the other waste streams are done accordingly.
Relevant BEMPs	The relevant BEMPs for improving the environmental performance of the waste management system assessed with this indicator are: - integrated waste management strategies (Section 3.3.1); - pay-as-you-throw (Section 4.3.3); - awareness-raising (Section 4.3.5); - establishment of a network of waste advisers (Section 4.3.6); - home and community composting (Section 4.3.7); - waste collection strategy (Section 4.5.1); - inter-municipal cooperation among small municipalities

[29] The interpretation of the result of the capture rate for biowaste needs to be particularly careful taking into account the context (i.e. type of biowaste collection system adopted and presence of home/community composting) of the local area considered.

Overview of indicator	
	(Section 4.5.2); - civic amenity sites (Section 4.5.3); - logistic optimisation for waste collection (Section 4.5.4).
Example calculation	Val di Non is a rural area in the north of Italy. The annual amount (2016) of separately collected glass (all types) is 1 310 tonnes. From the composition analysis of the mixed waste, the share of glass in the mixed waste is about 1.9 % and the total amount of mixed waste collected annually is 3 009 tonnes. Therefore, the annual amount of glass in mixed waste is 57 tonnes and the capture rate is 95.8 %. The county of Aschaffenburg (DE) collects annually (2016) 15 137 tonnes of separately collected paper and cardboard. From the composition analysis of the mixed waste, the share of paper and cardboard in the mixed waste is about 6.65 % and the total amount of mixed waste collected from households annually is 9 650 tonnes. Therefore, the annual amount of paper and cardboard in mixed waste is 642 tonnes and the capture rate is 95.9 %. The county of Aschaffenburg (DE) collects annually (2016) 2 381 tonnes of separately collected metals (any type). From the composition analysis of the mixed waste, the share of metals in the mixed waste is about 3.1 % and the total amount of mixed waste collected from household annually is 9 650 tonnes. Therefore, the annual amount of metals in mixed waste is 299 tonnes and the capture rate is 87 %. The city of Treviso (IT) collects annually (2016) 6 569 tonnes of separately collected co-mingled packaging (glass, plastic, metal). Additionally, in the civic amenity sites, 233 tonnes of separately collected plastics, 375 tonnes of metals and 232 tonnes of glass are also collected. From the composition analysis of the mixed waste, the share of the three materials (glass, plastic and metals) in the mixed waste is 22.1 % and the total amount of mixed waste collected annually is 5 851 tonnes. Therefore, the annual amount of glass, plastic and metals in mixed waste is 1 293 tonnes and the capture rate for the co-mingled materials is calculated by applying the formula below: $Capture\ rate\ for\ co-mingled\ materials = \frac{6569+233+375+232}{6569+233+375+232+1293} =$ 85 %. <u>References</u>: Coletti D., 2017; Morlok J., 2017; Mattiello M., 2017
Benchmark of excellence	- The capture rate for waste glass separately collected as a single fraction (i.e. not in a co-mingled collection system) is higher than 90 %.

Overview of indicator	
	- The capture rate for waste paper and cardboard separately collected as a single fraction (i.e. not in a co-mingled collection system) is higher than 85 %. - The capture rate for waste metals separately collected as a single fraction (i.e. not in a co-mingled collection system) is higher than 75 %. - The capture rate for co-mingled waste packaging is higher than 65 %.

Further explanation

It is noteworthy that the precondition for the calculation of this indicator is that a composition analysis of the mixed waste collected (e.g. door-to-door, kerbside) within the territory administered by a local authority has been performed. The capture rate helps local waste authorities to understand how much of a specific material has been collected separately for recycling out of the total waste generated of that type within the territory administered by a local authority. It needs to be stressed that, unlike what is often presented in waste statistics, the waste generation of a certain material is not identical to what has been collected. On the contrary, there is a large proportion of a given fraction that is not source separated (at household level) before the collection and thus ends up in the mixed waste. For this reason, the tonnage of recyclable materials in the mixed waste can be calculated on the basis of the composition analysis. This amount can then be added to the amount of recyclables collected, resulting in the correct amount of waste generation for a specific stream.

Moreover, the capture rate can be calculated for each material collected separately within the territory administered by a local authority, i.e. paper and cardboard, glass, metal and plastic. Depending on the system in place, this can include co-mingled recyclables or separate metal and plastic waste collections.

The capture rate can be calculated not only for recyclables (e.g. glass, metals, paper and cardboard) but also for biowaste. However, when analysing the results obtained, the strategy adopted for the collection and treatment of biowaste needs to be considered. In fact, if only uncooked food is admitted in the separately collected biowaste, it is clear that there will be a higher presence of biowaste (i.e. cooked food) in the mixed waste, reducing the value of the calculated capture rate. Moreover, if in a local area there are both home/community composting and separate collection of biowaste, the calculation of the capture rate is underestimated, because it is not able to correctly assess the amount of biowaste processed in home/community composting.

The calculation of the capture rate needs to take into consideration the amount of each specific waste stream collected by all collection schemes in place. This means that if glass, as an example, is collected door-to-door/kerbside, at bring points, and at civic amenity sites it is necessary to sum up all these amounts. Furthermore, the collection systems might also be operated by private companies on behalf of the local waste authority, in which case data from those other actors should be collected.

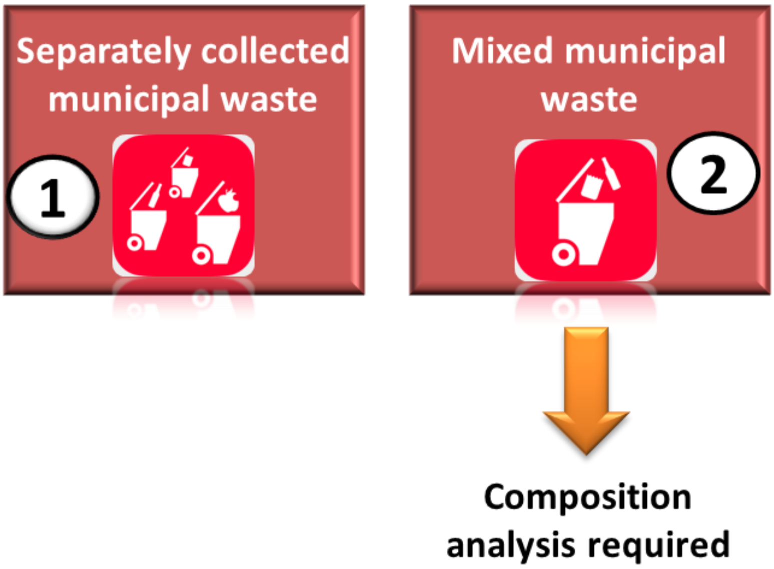

Figure 2-8. Data requirements to calculate the "Capture rate (per specific waste stream)" indicator

The capture rate of a specific separately collected waste stream is calculated by dividing the total amount of that waste stream collected separately (number 1 in Figure 2-8) by the total amount of that waste stream collected separately plus the amount of that waste stream in the mixed waste (number 2 in Figure 2-8), calculated thanks to a composition analysis.

Data needs and potential sources of data

Three figures are required for the calculation of each capture rate:

- amount (kg) of separately collected material;
- composition analysis of the mixed waste;
- amount (kg) of a specific waste stream contained in the mixed waste (determined by a composition analysis).

Please note that data might need to be collected from different operators and different private collection schemes, for instance from door-to-door/kerbside collection, bring points and civic amenity sites. This may mean approaching different actors (e.g. private operators) if the different waste fractions and collection systems are not all managed by one organisation.

Advantages and/or disadvantages of the indicator

The capture rate is a performance ratio that allows a very comprehensive assessment of the actual amount of a recyclable fraction which is captured by the municipal solid waste management system in place. In comparison to quantity-based indicators, this type of indicator expresses the performance of the separate collection system for a specific fraction within the territory administered by a local authority.

Therefore, the indicator is important to obtain information about the efficiency of the separate collection system and about how thoroughly waste is separated at source. It can give indications on where improvements in waste collection strategy and economic instruments could be required. Moreover, it enables comparison on a yearly or even more frequent basis depending on the frequency of data gathering.

Like the other environmental performance indicators, this one also has its disadvantages. As an example, one limitation might be due to EPR systems established at national level, such as bottle deposit refund schemes, for which data at municipal level can mostly not be disaggregated. This will partially hide the actual collection performance because the amounts collected by the EPR system will not appear in the municipal statistics. The amount in the mixed waste will be lower but so too will the amount collected.

Interpretation of result

Unlike the environmental performance indicators presented in Section 2.4.1, the capture rate is a performance ratio, which means that it puts absolute terms in relation to each other, thus avoiding limitations imposed by quantity-based indicators in terms of interpretation. In contrary to quantity based indicators performance ratios allow the assessment of the performance of a local waste authority related to a specific aspect of municipal solid waste management.

The capture rate for a specific waste stream is important to gain a better understanding of the efficiency of the established collection system. As an example, the percentage of MSW collected separately compared to the mixed collection can be very low in one municipality but this does not provide an indication of how well the separate collection system works. Of course, a generally low separate collection rate for MSW is an indication that the collection performance needs to be improved but it is necessary to investigate further to determine for which waste stream collected separately this is most important.

In general, the reasons for low separate collection rates are that there are weaknesses in the collection infrastructure, i.e. bring points or civic amenity sites are too far away, or that the population does not have a proper understanding of the functioning of the system and lacks awareness. Different solutions can be applied by local waste authorities to increase the capture rate:

- Investigate the collection infrastructure and the distance of citizens from collection points.
- Assess the knowledge of the population on the separate collection schemes and awareness-raising.
- Examine the cost structure for waste collection, e.g. how much is the fee for door-to-door mixed waste collection? Is there sufficient incentive to sort the fractions instead of putting them in the mixed waste?

Furthermore, for better interpretation of the results, it is important to provide, as notes to the calculations, information on the types of collection systems applied per respective waste stream within the territory administered by the local authority. Interesting additional information might also be by whom the collection system is established/managed, private or public organisations.

Several additional external factors influence the result of the capture rates per material and it is important to consider them. Important factors are, inter alia, waste scavengers, large generators of a specific material, and retailers and recyclers that have established their own (informal) system.

This is also the case if a deposit refund scheme (DRS) for some types of waste (e.g. glass drink bottles) is in place. If such a system is present and working, the capture rate of the relevant waste stream will increase because there is an additional incentive for the population not to dispose of the specific type of waste in mixed waste. For local waste authorities that have implemented such a DRS, it is important to include this information in the reporting on this environmental performance indicator and to also take it into account in the interpretation of the corresponding capture rates per waste stream.

Impurity rate of a specific waste stream

<table>
<tr><th colspan="2">Overview of indicator</th></tr>
<tr><td>Phase of waste management</td><td>Waste generation, collection and sorting</td></tr>
<tr><td>Name of indicator</td><td>Impurity rate in a specific separately collected waste stream</td></tr>
<tr><td>Type of indicator</td><td>Performance ratio [%]</td></tr>
<tr><td>Explanation</td><td>The impurity rate of a specific waste stream refers to the amount of non-target materials in the separately collected waste stream. This indicator is closely linked to the previous indicator (capture rate) as it monitors the effectiveness (i.e. how effective in selecting the recyclables at home the residents are) of a separate collection. It provides information about the amount of misthrows and materials contained in the separately collected recyclables that cannot be recycled.

The impurity rate can be calculated for the separately collected fractions, e.g.:
- plastic;
- metal;
- paper and cardboard;
- glass;
- co-mingled packaging;
- biowaste

Two indicators may be calculated for biowaste if kitchen waste and garden waste are collected separately:
a) impurity rate in separately collected kitchen waste;
b) impurity rate in separately collected garden waste.</td></tr>
<tr><td>Calculation method</td><td>Example: impurity rate for metal

$Impurity\ rate\ for\ metal = \frac{kg\ of\ non-target\ material\ in\ collected\ metal\ waste}{kg\ of\ separately\ collected\ metal\ waste} * 100\ \%$

The calculations for the other waste streams are done accordingly.</td></tr>
</table>

Overview of indicator	
	Please note that impurity rates for the separately collected fractions may be received directly as percentage values from the sorting operator. In this case, no further calculation is necessary.
Relevant BEMPs	The relevant BEMPs for improving the environmental performance of the waste management system assessed with this indicator are: - integrated waste management strategies (Section 3.3.1); - pay-as-you-throw (Section 4.3.3); - awareness-raising (Section 4.3.5); - establishment of a network of waste advisers (Section 4.3.6); - home and community composting (Section 4.3.7); - waste collection strategy (Section 4.5.1); - inter-municipal cooperation among small municipalities (Section 4.5.2); - civic amenity sites (Section 4.5.3); - logistic optimisation for waste collection (Section 4.5.4).
Example calculation	In the department of Lot, in France, the annual (2015) amount of dry recyclables (paper, plastic and metal) collected is 15 810 tonnes and the amount of impurities detected is 4 097 tonnes. Therefore, the impurity rate for the waste fraction of dry recyclables is 26 %. <u>References:</u> (Lot Tourisme, 2017) and (Syndicat Départemental pour l'Élimination des Déchets Ménagers et Assimilés, SYDED, 2015)
Benchmark of excellence	N/A

Further explanation

It is useful to calculate the impurity rate for each separately collected waste stream within the territory administered by a local authority. Depending on the system in place, this can include co-mingled recyclables or separate metal and plastic waste collections.

This indicator includes the impurities collected in all different collection schemes established for a material; this can include door-to-door collection, bring point collection, collection by an EPR scheme, and collection in civic amenity sites. Before calculating this indicator, it is important to develop an understanding of all the collection and treatment routes that are established for a material in order to collect all relevant data from private actors too.

Data needs and potential sources of data

Data needed for calculating the impurity rate are manifold and might sometimes be difficult to obtain. For each separately collected waste stream, the amount of impurities in the stream needs to be available. This is reflected by the amounts of non-target material determined during sorting at the recycling facility. Please note that the impurity rate does not include material that is excluded from recycling due to particularities of the recycling process or the economics of the plant. This means for example that glass in plastic waste is considered a non-target material but different types of plastics that are contaminated or are too low in quality for the recycling process are not considered non-target material as they have been sorted correctly at source.

Besides the amount of non-target material, the amount of separately collected waste per waste stream is required.

It is important to obtain data from the local treatment operator(s) or the contracted MSW collector(s) if this is not available to the local waste authority. The provision of this data can be included as a clause (e.g. for annual or monthly transmission) in the contract with other actors such as private operators. Furthermore, it is important that data from all the different collection systems (e.g. door-to-door collection, bring points and civic amenity sites) is gathered.

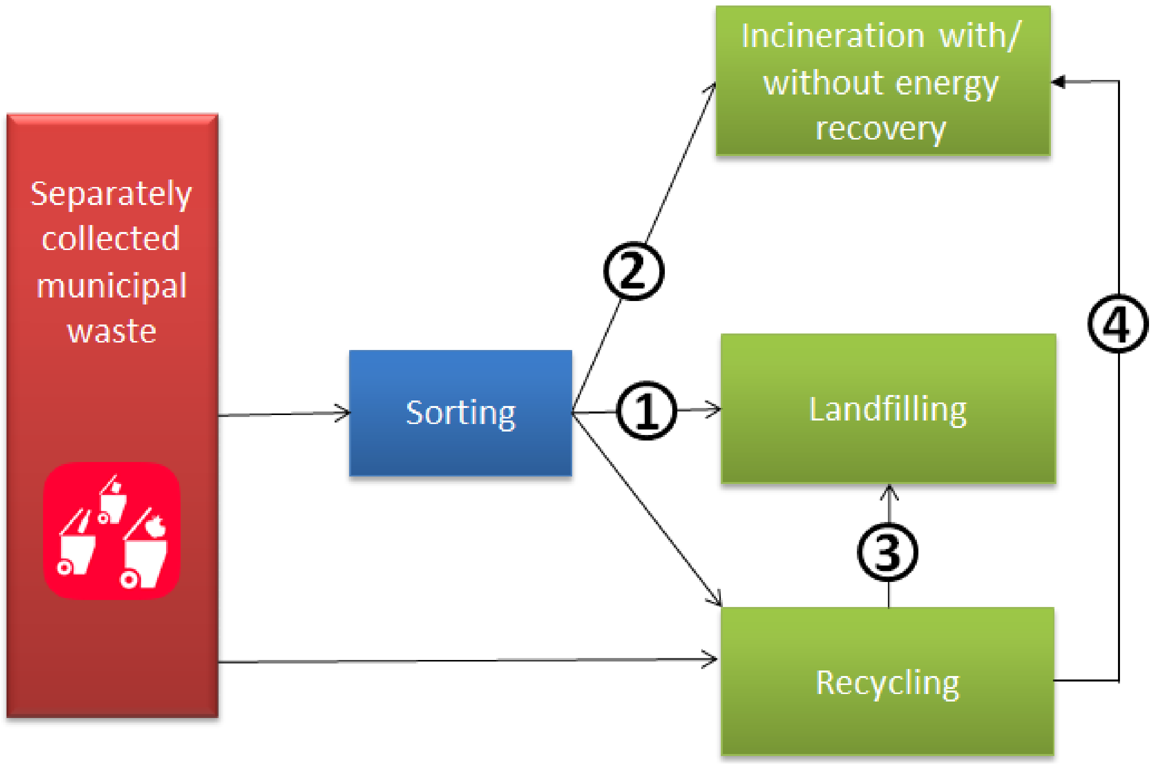

Figure 2-9. Data requirements to calculate the "Impurity rate (of a specific waste fraction)" indicator

Based on the model waste flow described in Figure 2-9, amounts of separately collected waste fractions that cannot be directed to recycling and are thus sent to landfill (number 1) or incineration (number 2) are needed. In addition, further rejects from the recycling plant that are sent to landfill (number 3) or to incineration (number 4) can be added for the calculation of this indicator.

For a local waste authority in the process of calculating this indicator, instead of assessing the quantities for each of the relevant flows mentioned in the previous paragraph, it might be easier to obtain data on the impurity rate from other actors (e.g. private operators), based on an analysis of the quality (amount of misthrows) of the material delivered to the sorting/recycling plant. In fact, for some local waste authorities, information from the recycler on the quality (i.e. share of non-target materials) of the separately collected waste delivered is available as payments are based on this factor. However, one constraint is that data availability might be severely hampered if the collection of a specific material, e.g. glass, is not performed by a public collection service. In this case, it is possible that both the collector and the recycler are private actors and the municipalities can have difficulties obtaining the data on non-target material quantities. This situation can be prevented by establishing contracts with data provision clauses; if this is not possible, then the only possibility is to sample separately collected waste, although this operation is very expensive.

Advantages and disadvantages

The indicator provides information about the quality of sorting at source. The impurity rate is an important indicator to assess the functioning of the sorting at source and may change over time, e.g. when awareness-raising measures are effective.

Interpretation of results

It is important to consider in the interpretation of results that impurity rates for a collection system can be lower for materials where an EPR system, e.g. for plastic bottles, is established as this leads to higher recycling rates. Also, deposit schemes, e.g. for glass bottles, may have a positive impact on impurities in glass waste as the bottles are returned for reuse instead of being disposed of with other glass waste.

Impurities can vary a lot among different fractions in the co-mingled waste collected. For this reason, it is important to strive for precise data on the impurity rates of the different material fractions included in the co-mingled collection, although such data might be difficult to obtain.

For biowaste, the results might also significantly vary depending on which types of biowaste are collected by the system. There are often misunderstandings on the part of residents on what they are allowed to dispose of as biowaste. For this reason, when interpreting the results, it can be important to look at the scheme established and consider whether it collects uncooked biowaste only or uncooked and cooked biowaste and kitchen waste together.

Local waste authorities should take care when data are provided for all rejects and not only for misthrows and non-target material because this may lead to high impurity rates despite proper sorting at source due to specific requirements of the recycling facility, e.g. to generate high-quality recycling products. For this reason, it is important to define and describe the impurities for each material.

When impurity rates are high, potential measures for local waste authorities include awareness-raising campaigns on target materials for each separately collected waste stream or an assessment of the suitability of the collection systems, e.g. are enough bring-points established in a certain area?

Biowaste in mixed waste

Overview of indicator	
Phase of waste management	Waste generation, collection and sorting
Name of indicator	Biowaste in mixed waste
Type of indicator	Quantity-based indicator [kg/capita/year]
Explanation	The indicator describes the annual amount of biowaste included in mixed waste, which is identified by a composition analysis.
Calculation method	$Biowaste\ in\ mixed\ waste = \frac{kg\ of\ mixed\ waste * \%\ of\ biowaste}{number\ of\ residents}$ Where the % of biowaste in mixed waste is calculated from the composition analysis of the mixed waste. The number of residents can be substituted by the population equivalent where tourist presence is relevant. As far as possible, all figures used should refer to the same year.
Relevant BEMPs	The relevant BEMPs for improving the environmental performance of the waste management system assessed with this indicator are: - integrated waste management strategies (Section 3.3.1); - pay-as-you-throw (Section 4.3.3); - awareness-raising (Section 4.3.5); - establishment of a network of waste advisers (Section 4.3.6); - home and community composting (Section 4.3.7); - waste collection strategy (Section 4.5.1); - inter-municipal cooperation among small municipalities (Section 4.5.2); - civic amenity sites (Section 4.5.3); - logistic optimisation for waste collection (Section 4.5.4).
Example calculation	In Schladming (AT) the population in the area is 22 550, while the population equivalent, since it is a touristic area during summer and winter, is 26 557. The annual (2015) amount of mixed waste collected is 1 952 tonnes and the share of biowaste in the mixed waste, obtained from a composition analysis, is 18 %. Therefore, the annual amount of biowaste in the mixed waste is 351 tonnes which means 13.2 kg/capita, calculated taking into account the population equivalent and not the number of residents. Reference: Amt der Steiermärkischen Landesregierung, 2017

Overview of indicator	
Benchmark of excellence	The annual amount of biowaste in mixed waste is lower than 10 kg/capita.

Further explanation

The indicator describes the amount of biowaste included in mixed waste per population equivalent. It is noteworthy that the preconditions for the calculation of this indicator are that biowaste is collected separately as a single fraction in the waste management system considered and a composition analysis of the mixed waste collected within the territory administered by a local waste authority has been performed. When biowaste is collected separately, this indicator allows the assessment of the performance of the system in capturing biowaste. In fact, calculating the actual capture rate for biowaste is very difficult (e.g. the presence of home composting in the territory prevents the calculation of the total amount of biowaste generated), while this indicator measures the biowaste not following the correct route. Biowaste generally has the largest share in terms of quantity within MSW and the reduction of biowaste being landfilled (i.e. increasing the amount correctly managed, e.g. with anaerobic digestion) is of utmost importance in the EU strategy for waste management.

Figure 2-10. Data requirements to calculate the "Biowaste in mixed waste" indicator

This indicator can be calculated from the stream number 1, mixed MSW, in Figure 2-10.

Data needs and potential sources of data

This indicator requires, besides the mixed waste generation data, the result of the composition analysis.

Furthermore, the number of residents is required for the calculation and it would be preferable for it to refer to the same year as the waste statistics. As mentioned previously, in areas with a relevant tourist presence, the population equivalent can be calculated instead.

Advantages and disadvantages

As well as other indicators, the indicator on biowaste in mixed waste requires a composition analysis of the mixed waste. Furthermore, what is included in the biowaste and what is not needs to be stated in a note to the calculation, e.g. is cooked food collected together with garden waste? Since the amount of biowaste generated changes over the season, i.e. more biowaste is generated in summer than in winter, it is worthwhile having several composition analyses available and comparing the results of the indicator for different seasons.

Overall, since this is a quantitative indicator, the results might vary significantly across different municipalities. Amounts may depend on external factors that are not directly related to MSW management, such as climate and consumption patterns.

Interpretation of results

This indicator is very comprehensive and provides a good overview of how much biowaste is not sorted at source and captured by a separate collection system. Over time, it can provide insights into how waste generation and awareness for sorting change from year to year. In some cases, it could also provide an indication of the status of waste prevention and it can help to identify how the separate collection of biowaste evolves over time.

The result, however, depends on several aspects. For the interpretation of the indicator, it is important to reflect on the type of waste included in the fraction (i.e. which groups of organic waste can be disposed of) as well as on the number of households conducting home/community composting. The type of biowaste allowed (e.g. cooked and uncooked food vs. only uncooked food) in the biowaste collection system influences the amount of biowaste that will remain in mixed waste, e.g. if only uncooked waste is collected there will be more cooked waste in the mixed waste. If large amounts of biowaste are observed in mixed waste although a separate collection system is in place, local waste authorities can reflect on the effectiveness of this system and plan measures to increase the separate collection rate in order to avoid biowaste in mixed waste. Such measures may include awareness-raising, e.g. by sending flyers to households or starting a campaign with waste advisers.

In general, it is important that local waste authorities work with the aim of reducing the amount of biowaste in mixed waste as much as possible. However, this indicator is also influenced by the presence of other fractions (e.g. plastics, glass, metals) in mixed waste (i.e. the more other fractions are separated out of mixed waste, the higher the share of biowaste would be, for a constant amount of biowaste).

2.5. Additional waste-stream-specific indicators

The assessment of the municipal solid waste management performance based on the indicators described in Section 2.4 provides a general understanding of the performance of the system. Some municipalities might want to assess their performance beyond the waste streams and indicators covered there. The following additional indicators can be used by those municipalities that have a collection system established for glass bottles and/or textiles, to help them assess the performance of waste management in this regard too.

However, it can be noted that these indicators are not accompanied by the same level of information as the environmental performance indicators presented in the previous section, as they are used for additional measurement of the performance of collections not necessarily established in most municipalities in the EU. The following sections include an explanation of the additional indicators, a calculation method, and some conclusions on limitations, advantages, disadvantages, etc., if possible.

Collection scheme for glass bottles

Overview of indicator	
Phase of waste management	Waste collection
Name of indicator	Presence of a deposit refund scheme for glass bottles
Type of indicator	Qualitative indicator [y/n]
Explanation	The indicator on the presence of a deposit refund scheme (DRS) for glass bottles (y/n) is needed to complement the capture rate and the impurity rate for glass waste, because of the very significant influence of such a deposit refund scheme on the results obtained by the capture and impurity rate indicators.
Calculation method	Yes/No; no calculation needed Description of the deposit refund scheme
Conclusions on indicator	This indicator does not provide an overview of the actual performance of a waste management system for glass waste as it is a purely qualitative indicator whose assessment gives no information on the functioning of the DRS itself. However, the presence of a DRS for glass bottles can have a significant impact on the amounts of glass waste collected and also on the capture rate (higher since there is a strong incentive to collect glass bottles separately). When interpreting the results of the capture and impurity rate for glass, it is useful to take the presence of a DRS into account. Please note that it is possible that such DRS are also established for other streams, not only for glass, such as plastic bottles or cans. However, these systems are rarely implemented in the EU and are therefore not considered in this document.

Overview of indicator	
	Another dimension of glass deposit refund schemes is the possibility that they target glass bottles for refilling. In such cases, the waste management phase addressed by the DRS is waste prevention. However, such DRS for refillable glass bottles are rarely in place.
Benchmark of excellence	N/A

Amount of used and waste textiles collected separately

Overview of indicator	
Phase of waste management	Waste collection
Name of indicator	Amount of textiles collected separately
Type of indicator	Quantity-based indicator [kg/capita/year]
Explanation	The indicator reflects the annual amount of used and waste textiles collected separately through the collection scheme established by the local waste authority. This includes both used textiles sent for reuse and waste textiles sent to either preparation for reuse or recycling.
Calculation method	$Textiles\ collected\ separately = \frac{used\ and\ waste\ textile\ collected\ per\ year\ [kg]}{number\ of\ residents}$ Please also note that the figures used should be presented in kg and that separately collected textiles from bring points and civic amenity sites can be considered, if applicable. If available, data from private collections can be added.
Conclusions on indicator	This indicator has several limitations in terms of the interpretation of the results as it is a quantity-based indicator, not one providing reliable evidence on the performance of the collection system. Another relevant aspect is that most of the used and waste textile collections are performed by private companies, charities or associations and the local waste authorities only rarely have access to collection data; or there is no monitoring of the amounts collected.

Overview of indicator	
	This indicator can provide the local waste authority with an indication of the collection performance over the years, i.e. increase or decrease in the collected amounts.
Benchmark of excellence	N/A

Textiles in mixed waste

Overview of indicator	
Phase of waste management	Waste generation, collection and sorting
Name of indicator	Share of textiles in mixed waste
Type of indicator	Performance ratio [%]
Explanation	The share of textiles found in mixed waste can be used to monitor the correct source separation by households of waste textiles and the efficiency of the used and waste textlies collection system. This metric allows the assessment of the quantity of textiles that are not correctly source separated and are thus disposed of in the mixed waste.
Calculation method	$Textiles\ in\ mixed\ waste = kg\ of\ mixed\ waste\ collected * \%\ of\ textiles$ Where the % of textiles in mixed waste is calculated from the composition analysis of the mixed waste. As far as possible, all figures used should refer to the same year.
Conclusions on indicator	The indicator provides information about the performance of sorting at source and on the collection of waste and used textlies. The share of textiles in mixed waste is an important indicator to assess the functioning of sorting at source and may change over time, e.g. when awareness-raising measures are effective, more collection points are installed. This metric allows the assessment of the quantity of waste and used textiles that do not follow a correct route (to reuse or recycling) and are disposed of in the mixed waste. It is important that local waste authorities work with the aim of reducing waste textiles in mixed waste as much as possible. However, this indicator is also influenced by the presence of other fractions in

Overview of indicator	
	mixed waste (i.e. the more other fractions are separated out of mixed waste, the higher the share of waste textiles would be, for a constant amount of textiles). Therefore, this indicator can be used for regular monitoring and, possibly, improvement of waste textiles collection within the same local waste authority's territory but comparisons based on this indicator with other local administrations are difficult.
Benchmark of excellence	N/A

Capture rate for textiles

Overview of indicator	
Phase of waste management	Waste collection
Name of indicator	Capture rate for textiles
Type of indicator	Performance ratio [%]
Explanation	The capture rate is the share of the estimated generation of a specific waste fraction that is collected separately. It provides insights into the efficiency of a separate collection system. The precondition for the calculation of the capture rate for textiles is that a composition analysis of the mixed waste has been performed. In addition, all the amounts of waste textiles collected by each collection system (public and private) are needed in order to calculate the indicator.
Calculation method	$Capture\ rate\ textile = \frac{kg\ of\ textiles\ separately\ collected}{kg\ of\ total\ textile\ waste\ generation} * 100\ \%$ Where $Total\ textile\ waste\ generation$ $= kg\ separately\ collected\ textiles + kg\ of\ textiles\ in\ mixed\ waste$ With $kg\ of\ textiles\ in\ mixed\ waste = kg\ of\ total\ mixed\ waste *$ $\%\ of textiles\ in\ mixed\ waste$ Where the % of textiles in mixed waste is calculated from the composition analysis of the mixed waste.

Overview of indicator	
Conclusions on indicator	The capture rate for textiles is the best indicator for understanding the actual collection performance of this waste stream. It is useful if the amounts collected via all routes (private and public collections, bring points and civic amenity sites) are available to calculate this indicator. However, unlike for other MSW streams like paper, waste textiles are mainly collected by private collectors that are often not regulated by the municipalities, meaning that municipalities do not have a complete overview of the amounts collected. This can pose problems in identifying the actual amount of textiles collected separately and will decrease the reliability of the calculated indicator. It is recommended to use this indicator for the assessment of the performance of the separate collection of waste textiles only when the local waste authority has data on the amounts collected by private companies or when there is no/almost no private collection, in order to ensure the reliability of the indicator. Otherwise, it is advisable to use the indicator on the amount of textiles in mixed waste.
Benchmark of excellence	N/A

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3. Cross-cutting issues

3.1. Introduction

Looking at the current economic system (see Figure 1-10), thousands and thousands of products including packaging are produced and consumed, and these all end up as waste at a certain point. In order to reduce the environmental impacts of waste management and, especially, production, the objectives are to significantly increase the resource efficiency of the economic system by developing waste prevention, and to establish a circular economy to reuse, recycle and recover the waste materials. Following the overall scope of this report, the cross-cutting issues are those concerning municipal solid waste, construction and demolition waste, and healthcare waste. Specific best practices for these different waste streams are described for each of them separately in the following chapters (Chapters 4, 5 and 6).

3.2. Technique portfolio

The focus is on the development of a waste strategy. This strategy is based on a detailed analysis of the waste situation for a given municipality, city, county or region which should include knowledge of the quality and quantity of as many waste streams as possible. The waste strategy could also be called a waste management plan, which includes waste management targets in terms of rates for waste prevention, reuse, recycling and recovery, as well as the treatment and its efficiency of the different waste fractions, such as not to landfill any untreated waste. Of course, such a strategy or plan has to respect existing regulations but should also represent the pathway towards greater resource efficiency and a circular economy. The efficient collection of the different fractions is also part of it. In the following chapters, for the three waste groups mentioned, a number of techniques to consider when defining best environmental management practices are described in detail. Thus, when defining the waste strategy, the different techniques are only mentioned without describing them in more detail.

Sometimes, there are different options for certain waste streams and it may be that it is not obvious which of those is the most environmentally friendly or most sustainable. In this case, it is adequate to use life-cycle considerations in order to identify the best option or to justify the selected one (see Section 3.3.2).

The financial dimension of waste management is also considered through the application of economic instruments. Given the right conditions, the application of these by waste authorities at local level can produce a remarkable change in the amount of wastes generated (see Section 3.3.3).

3.3. Cross-cutting BEMPs

3.3.1. Integrated waste management strategies

Summary overview
It is BEMP to develop and implement an integrated waste management strategy that considers: - the current and future expected trends of waste streams; - the waste hierarchy[30], prioritising measures according to the hierarchy (firstly waste prevention, secondly preparation for reuse, etc.); - the availability and capacity of nearby waste sorting/treatment facilities; - the current environmental attitudes and perceptions of residents; - any other specific condition affecting waste management (e.g. the significant presence of tourists/commuters, specific economic activities, climate). The development of a waste management strategy requires knowledge of the quantity and quality of each major waste stream through an appropriate data monitoring approach and a sound evaluation of waste management options. This may require, in some cases, the use of a life-cycle assessment (LCA) to identify options associated with the best environmental performance (see BEMP 3.3.2), which may sometimes depart from the waste hierarchy.

Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**

Applicability
This BEMP is primarily targeted to waste authorities with control, or at least significant influence over, waste management strategy at the local or regional level – primarily local authorities. The waste authority may need to outsource aspects of strategic planning where particular specialist expertise, such as analytical data skills and knowledge of waste treatment processing, are required.

Specific environmental performance indicators
- Overall targets for the improvement of the waste management system (e.g. based on the indicators defined in this report) are in place (y/n).

[30] The waste hierarchy consists of the following steps: waste prevention, preparation for reuse, recycling, recovery and disposal

- Specific targets for waste prevention and reuse are in place (y/n).
Benchmark of excellence
- An integrated waste management strategy that includes long-term (i.e. 10–20 years) and short-term (i.e. 1–5 years) overall targets for the improvement of the performance of the waste management system is in place and regularly reviewed (at least every 3 years).

Description

Waste management deals with a considerable number of different waste streams, including MSW, but also various hazardous wastes, construction and demolition waste (Chapter 5) and healthcare waste (Chapter 6), and a multitude of processes.

For the development of an integrated local waste management strategy, the local authority and waste management company need to carry out a detailed assessment of the current situation of waste generation and collection in the territory, evaluate which options for the collection and treatment of waste are available, what is the current environmental level of education and perception of residents and identify the specificities (e.g. presence of tourists, prevalence of specific economic activities) which influence the local waste management system.

A key starting point for the development of a waste management strategy for MSW (the approach would also be the same for CDW and healthcare waste) is to monitor the current situation of the waste management system and to calculate, for the total municipal solid waste and for its different fractions available, the common environmental performance indicators presented in Section 2 and the spatial density of the waste generation (e.g. tonne/km^2) in the territory. By doing so, a detailed picture of the current waste generation, the capacity of the system to capture (any) specific recyclable streams and the quality of the streams collected separately can be evaluated.

Based on the assessment of the existing waste stream quantities and qualities, the waste strategy can define:

- the targets for waste prevention/reuse/recycling/recovery for the different waste streams;
- the most environmentally friendly disposal route for residual waste;
- the mix of techniques/instruments/approaches to achieve the targets.

During the development of the integrated waste management strategy, technical and economic instruments as well as psychological aspects of citizens' behaviour, such as environmental awareness, have to be taken into consideration. The integrated waste management strategy has to follow the waste hierarchy (Figure 3-1), prioritising prevention, minimisation and reuse as the most sustainable options for waste management, followed by recycling, with energy recovery and disposal as the least sustainable options. Based on this, a key decision when establishing a waste management strategy is the identification of the trade-offs between high recycling rates (normally leading to low-quality recycling) and lower recycling rates but with high-quality recycling. This choice is based on local conditions, namely current recycling levels, urban or rural environment, environmental consciousness of citizens, availability and capacity of nearby recycling plants and incinerators, market value of recyclables and incineration and landfill gate fees. In general, life-cycle thinking can support choices, selecting the most environmentally friendly and sustainable options for waste recycling and mixed waste treatment. In some cases, a detailed evaluation of alternatives through a life-cycle assessment (LCA) may be required to identify options with the best environmental profile (see BEMP in Section 3.3.2 on LCA of waste management options). LCA can lead to choices which may depart from the waste hierarchy, since local conditions can improve or worsen the environmental performance of the different stages of the waste hierarchy (e.g. long transport distances to civic amenity sites which leads to higher GHG emissions compared to a close-by recycling plant).

Figure 3-1. Waste hierarchy according to the Waste Framework Directive (2008/98/EC)

The development of the waste management strategy for municipal solid waste can rely on a number of instruments and approaches, presented in Section 4.3, such as the analysis of the estimated costs and revenues for the waste management, the economic tools suitable for charging residents, performance-based contracts and awareness-raising campaigns.

At local level, the possibilities to implement waste prevention measures are limited; however, in this document, two BEMPs for waste prevention are presented in Section 4.4, and these can support shaping the strategy for the waste prevention and minimisation steps.

In the same way, BEMP 4.5.1 on waste collection strategy provides an in-depth overview of the different options for the collection systems that can be adopted in the waste management strategy, presenting also their advantages and disadvantages. Once the strategy has defined the types and quantities of materials suitable for collection and recycling the system for their collection needs to be defined.

Finally, after the BEMPs on collection of waste, BEMP 3.3.4 provides guidance (for MSW but also for CDW and HCW) for the development and implementation of a waste management strategy, also for the waste management phases with low priority in the waste hierarchy pyramid (i.e. waste treatment and disposal facilities, recycling and recovery operations).

In the case of CDW and HCW, as for municipal solid waste, the development of the waste management strategy needs to be based on the waste hierarchy and several measures which can be adopted and go in that direction are presented in Chapters 5 and 6.

When establishing a waste management strategy, long-term planning is required, as the implementation of the strategy can only be achieved step by step, i.e. waste stream by waste stream. Therefore, prioritisation is needed and the starting point should target the most relevant waste streams, where the relevance takes into account quantity and hazard. Short-term and long-term targets are a useful tool to monitor progress and they can be calculated using the indicators employed for the systematic monitoring of the waste management system.

Achieved environmental benefits

The implementation of an integrated waste management strategy is normally associated with environmental benefits, specifically with the reduction of mixed waste and a significant increase in the percentage of waste reduction, reuse and recycling.

Appropriate environmental indicators

The most appropriate environmental performance indicators for the development and monitoring of a waste management strategy for municipal solid waste are the ones reported in Chapter 2.

Additionally, specific indicators to assess the level of implementation of this technique are as follows:

- Overall targets for the improvement of the waste management system (e.g. based on the indicators defined in this report) are in place (y/n).
- Specific targets for waste prevention and reuse are in place (y/n).

Indicators that can be used to monitor the waste management strategy for CDW and HCW instead can be the ones presented in the different BEMPs of Chapters 5 and 6.

Cross-media effects

There are no relevant environmental cross-media effects when developing a waste management strategy.

Operational data

Aschaffenburg (Germany)

The county of Aschaffenburg (Germany) is an excellent example of the development of an integrated waste management strategy and its systematic implementation and improvement. As far back as 20 years ago the local authority established an ambitious waste management strategy, based on collected and analysed waste data. The strategy has been regularly updated and improved based on the waste streams arising and the needs of the residents.

Table 3-1 shows the important milestones for the waste management strategy in the county of Aschaffenburg.

Table 3-1.Important milestones in the implementation of an integrated waste management strategy of the county of Aschaffenburg (Germany)

Measure as part of the strategy	Year
Introduction of an identification system with weighing both for residual and biowaste, and later also for bulky waste; close cooperation with the municipalities including financial support, installation and continuous development of recycling stations in the municipalities and one central recycling station of the county (Aschaffenburg, 2013, Aschaffenburg, 2014)	1996/1997
Introduction of paper/paper board collection in dedicated bins from all households (no weighing system) (Aschaffenburg, 2002)	2002
Analysis of the composition of residual and bulky waste in order to identify additional recycling options (Aschaffenburg, 2011)	2011
Systematic weighing of green cuttings	2012
Reassessment of the collection and disposal of green cuttings (Morlok, 2013)	2013
Waste sorting analysis of residual waste, biowaste, paper, light packaging, glass and metal packaging in order to identify additional optimisation potential (Hoeß and Ammon, 2014)	2014
Latest annual waste management report, for 2013 (Aschaffenburg, 2014)	2014

Capannori (Italy)

Capannori is a municipality of 46 700 inhabitants near Lucca, in Tuscany. Door-to-door collection was introduced in stages across the municipality between 2005 and 2010, starting with small villages, where any mistakes could be identified and corrected early on, then extended to cover the entire municipal area in 2010. By that time, 82 % of municipal waste was separated at source, leaving just 18 % of residual waste to go to landfill (Figure 3-2) (Zero Waste Europe, 2013a).

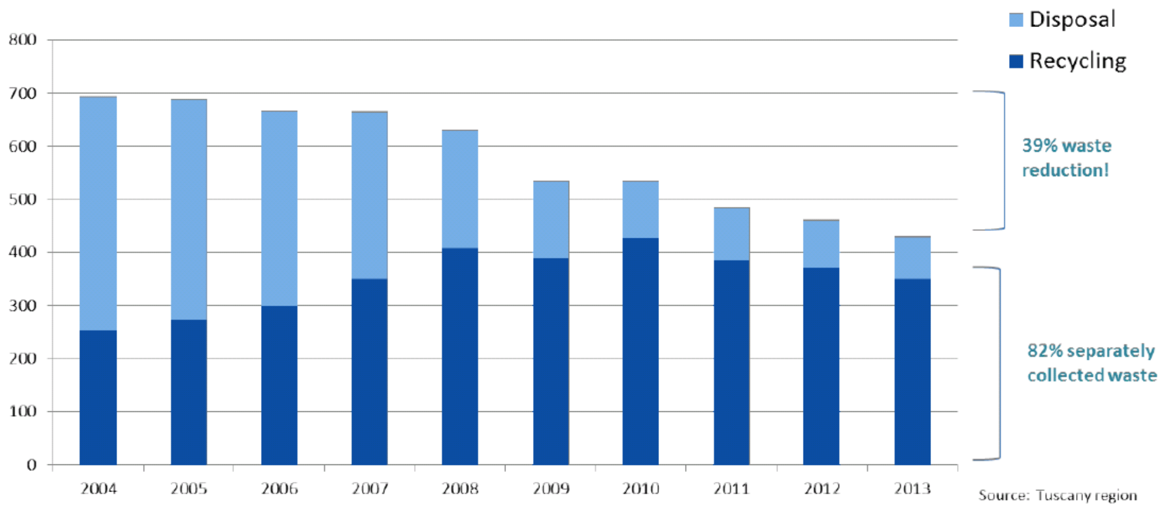

Figure 3-2. Evolution of separate collection and waste generation in Capannori (Italy) (kg/person/year) (Zero Waste Europe, 2013a)

The savings from no longer sending most waste to landfill, and earnings from the sales of materials to recycling plants make the scheme economically self-sufficient, even saving the city council over EUR 2 million in 2009. These savings are ploughed back into investments in waste reduction infrastructure, and reducing fixed waste tariffs for residents. It has also funded the recruitment of 50 employees in the local waste management company, boosting employment in the region (Zero Waste Europe, 2013a).

Capannori, in addition to increasing the recycling rates of waste collected, has also worked to improve reuse of used items to reduce the amount of waste generated (between 2004 and 2013 the overall volume of waste generated per person dropped by 39 %, Figure 3-2). The municipality opened its own Reuse Centre in 2011, where items such as clothes, footwear, toys, electrical appliances and furniture that are no longer needed but still in good condition can be repaired where necessary and sold to those in need, thereby diverting them from entering into the waste management system and serving also a social function. Additionally, as part of Capannori's Zero Waste Strategy, eleven areas for action were identified, such as the sale of products loose or on tap: the municipality provides tax incentives to small local businesses to stock products that could refill customers' own containers, such as liquid detergents. Other measures have been implemented to support local agriculture and the local market of products (reducing the need for transport and packaging), to promote the use of tap water rather than bottled water and to use washable nappies rather than disposable ones (Zero Waste Europe, 2013a).

Treviso (Italy)

Contarina is the publicly owned waste management company responsible for waste management in the province of Treviso, where about 550 000 inhabitants live. Separate collection of waste reached about 85 % and residual waste generated is only 53 kg per inhabitant and year. In contrast, the EU average level is about 40 % separate waste collection and 285 kg of residual waste per inhabitant and year.

The high level of separate waste collection could be achieved with the introduction of an intensive and adapted kerbside collection combined with a pay-as-you-throw system. Municipal solid waste is collected in five or six major waste streams: unrecyclable dry, organics (food scraps), garden waste, paper and cardboard, glass, plastic and tin (in some municipalities glass is collected alongside plastic and tins) (Zero Waste Europe, 2013b).

Figure 3-3. Types of bins for kerbside collection in Treviso (Zero Waste Europe, 2013b)

The collection of different waste streams takes place on different days of the week; the fraction collected the most often (twice a week) is food waste, and the one collected the least often is residual waste, which is also the least important one in terms of volume. Paper, green waste and other recyclables are collected between once and three times per week.

Kerbside collection is supplemented by the EcoCentri (civic amenity sites) which are equipped with large containers for other types of urban waste: from aggregates to bulky waste, from electrical and electronic appliances to hazardous waste (Zero Waste Europe, 2013b).

Applicability

This BEMP is primarily targeted to waste authorities with control, or at least significant influence over, waste management strategy at the local or regional level – primarily local authorities. The waste authority may need to outsource aspects of strategic planning where particular specialist expertise, such as analytical data skills and knowledge of waste treatment processing, are required. Once the strategy has been developed, the waste management company needs to fully engage its staff in order to ensure its effectiveness.

Economics

When developing a systematic waste management strategy for the first time, it may be appropriate to ask for external assistance from experts. At least larger municipalities and cities, and certainly counties and regions, usually have their own in-house experts.

There is no information available concerning the costs for the drafting of a waste management strategy for the first time and its continuous development. The initial costs may be recovered by revenues from recyclables or from optimising the different activities and operations.

Driving force for implementation

The drawing up and further development of waste management strategies is usually driven by the need to move towards a more sustainable society. Currently, a lot of attention at national and European level has been focused on circular economy, and a waste management strategy which promotes prevention, reuse and recycling is well aligned with this circular view of the economy and society.

Reference organisations

The county of Aschaffenburg (Germany) is an excellent example, including with respect to the annually published waste management report (Aschaffenburg, 2014). The Val di Non (Italy) is another good example of waste management strategy and reporting of data (Comunitá Val di Non, 2017). The counties of Rems-Murr (Germany) and Breisgau-Hochschwarzwald (Germany) and the cities of Besançon (France), Vienna (City of Vienna, 2012) and Munich (Schmidt, 2013) are good references too.

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Morlok, J. (2013). Options for actions with respect to managing green cuttings and biowaste (in German). Conference on bio energy on 11-12 June 2013, http://www.kommunales-informationssystem.de/, Last access February 2015.

Schmidt, H. (2013). Waste Prevention and Resource Conservation – The Munich Way. Presentation at the Vienna Waste Management Conference on 7-11 October 2013

Zero Waste Europe, 2013a. The story of Capannori - case study. Available at: http://www.zerowasteeurope.eu/zw-library/case-studies/, Last access June 2017.

Zero Waste Europe, 2013b. The story of Contarina - case study. Available at: http://www.zerowasteeurope.eu/zw-library/case-studies/, Last access June 2017.

3.3.2. Life-cycle assessment of waste management options

Summary overview
It is BEMP to embed life-cycle thinking and assessment into waste management strategy and operations, with steps 1 and 2 (below) being essential and steps 3 to 8 needing an ad-hoc life-cycle assessment (LCA) to be carried out and not always necessary: 1) Systematic application of life-cycle thinking throughout waste management strategy design and implementation (to complement the waste management hierarchy). 2) Review of relevant LCA literature to rank the environmental performance of alternative waste management options, where studied systems are directly comparable with available options. 3) Application of LCA to specific management and technology options for which no reliable published literature can be found; this requires procurement of LCA services, or in-house use of relevant LCA software. 4) Careful consideration of system boundaries to ensure an accurate comparison across options, including system expansion and/or LCA for avoided processes (e.g. grid electricity generation). 5) Compilation and documentation of life-cycle inventories in relation to reference flows, if possible using primary data recorded along the value chain, noting data quality and uncertainty ranges. 6) Selection of pertinent impact categories to capture the major environmental burdens. 7) Presentation of normalised results for relevant impact categories to evaluate complementarities or trade-offs, with clear indication of uncertainty errors and sensitivity analyses. 8) Validation of the LCA study by an independent third party (essential requirement under ISO 14044 for external dissemination of results, but good practice even when only used internally).

Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**

Applicability
A full life-cycle assessment is not always necessary. Basic prioritisation of the waste management options indicated in the waste management hierarchy may be sufficient to inform best practice in some cases. However, detailed comparison of options ranked similarly in the waste hierarchy, and of management changes that affect the overall

waste chain performance are often required. Waste management organisations of any size may apply life-cycle thinking and review LCA studies. Buying bespoke LCA services and/or paying for staff training in LCA may only be economically viable for larger organisations.
Specific environmental performance indicators
- Systematic application of life-cycle thinking, and, where necessary, undertaking of life-cycle assessments, throughout waste management strategy design and implementation (y/n).
Benchmark of excellence
- The waste management strategy is designed and implemented on the basis of systematic application of life-cycle thinking and, when needed, ad-hoc life-cycle assessment studies.

Description

Why undertake a life-cycle assessment?

Life-cycle assessment (LCA) was pioneered in the 1970s and 1980s to evaluate the environmental efficiency of packaging options (Hunt et al., 1974; Boustead, 1989), and has since developed further for wider application such as the comparison of different waste management options (White et al., 1995). LCA provides a comprehensive framework to evaluate the overall resource and environmental efficiency of different waste management strategies, practices and technologies (ISO, 2006a). Crucially, indirect and upstream effects, such as raw material extraction, transport and processing to replace resources removed from circulation in the economy, are accounted for in LCA, thus enabling comparison of recycling and extraction of virgin raw materials for example.

The waste hierarchy provides clear guidance on the prioritisation of management options. However, in order to compare the environmental efficiency of options within the same stratum of the waste hierarchy, or that transcend strata (e.g. anaerobic digestion that both *recycles* nutrients and *recovers energy* via biogas), LCA may be required. In particular, the move towards a circular economy, with circular flows of materials through multiple recycling loops and material to energy transformations (e.g. refuse-derived fuels, biogas and wood chips), necessitates an "expanded-boundary" LCA approach that considers for example the avoidance of fossil energy generation associated with use of biogas.

From a strategic policy perspective, "consequential LCA" may be the most appropriate framework to evaluate the *net environmental change* associated with prospective waste management strategies that are likely to involve multiple product outputs and multiple system substitutions and indirect (market) effects (Weidema, 2001, Ekval and Weidema, 2004).

Thus, life-cycle thinking and LCA are crucial elements of best practice in devising integrated waste management strategies (Section 3.3), and are integral components of strategic environmental assessments undertaken by local authorities to evaluate development plans in relation to national sustainability targets.

Best practice measures

The steps below represent important best practice measures to successfully embed life-cycle thinking and assessment into waste management strategy and operations. Steps 1 and 2 represent essential minimum requirements for best practice that may be undertaken universally by any waste management organisation (however small) to ensure that operations are fully informed by life-cycle thinking. Steps 3 to 8 involve the undertaking of an LCA study, and are only necessary where conclusions from published studies are not transferable to the options being compared by the waste management organisation.

1. Systematic application of life-cycle thinking throughout waste management strategy design and implementation, wherever necessary to augment the recommendations of the waste management hierarchy.
2. Review of relevant LCA literature to rank the environmental efficiency of alternative waste management options, where studied systems are directly comparable with available options.
3. Application of LCA to specific management and technology options for which no reliable published literature can be found, procurement of LCA services, or in-house use of relevant LCA software.
4. Careful consideration of system boundaries to ensure an accurate comparison across waste management options, including system expansion and/or application of consequential LCA to account for avoided processes (e.g. grid electricity generation) where appropriate.
5. Thorough compilation and transparent documentation of life-cycle inventories in relation to reference flows, using primary data recorded by organisations along the value chain where possible, and noting data quality and uncertainty ranges.
6. Selection of pertinent impact categories to capture the major environmental burdens.
7. Presentation of normalised results for relevant impact categories to evaluate complementarities or trade-offs, with clear indication of uncertainty errors and sensitivity analyses around variable parameters.
8. LCA studies should be validated by an independent third party (essential requirement according to ISO 14044:2006 'Environmental management - Life cycle assessment - Requirements and guidelines' for external dissemination of results, but good practice even when results are only used internally).

Achieved environmental benefits

Embedding life-cycle thinking and LCA into strategic planning and technology selection decisions can maximise environmental efficiency and reduce overall direct and indirect (life-cycle) environmental burdens. The realisation of environmental benefits referred to throughout this report, in Chapter 1 and subsequent BEMP techniques, is at least partially attributable to life-cycle (systems) thinking and assessment.

Appropriate environmental indicators

The most appropriate indicator for the assessment of the implementation of this BEMP is:

- Systematic application of life-cycle thinking, and, where necessary, undertaking of life-cycle assessments, throughout waste management strategy design and implementation (y/n).

Cross-media effects

Consideration of life-cycle performance across waste management strategies and technologies should help to minimise cross media effects.

The process of normalisation may be helpful to evaluate trade-offs across impact categories associated with cross-media effects.

Expansion of the LCA scope to undertake social LCA can identify any trade-offs between environmental, economic and social pillars of sustainability.

Operational data

A) Case study on the use of LCA for food waste treatment options

A case study is presented below, in which consequential LCA is applied to evaluate the net environmental change associated with the deployment of anaerobic digestion (AD) to treat different food waste streams, replacing three existing waste management options: (i) landfilling; (ii) in-vessel composting; (iii) animal feeding. More detail on this is provided in Styles et al. (2016).

<u>Scope and boundary definition</u>

ISO 14040 and ISO 14044 (ISO, 2006a, 2006b) describe the framework for LCA application, according to four main phases:

1. Goal, scope and boundary definition;
2. Inventory compilation;
3. Life-cycle impact assessment;
4. Interpretation and reporting.

Getting the first phase correct is critical and represents a challenge when considering waste management alternatives. In the first instance, the correct LCA approach must be identified. Extensive guidelines produced for product carbon footprinting (e.g. BSI, 2011; Commission Recommendation 2013) or organisation carbon footprinting (WRI, 2004, 2011a; Commission Recommendation 2013) provide a detailed methodological basis and guidance to perform LCA of products and organisations.

Typically, two main LCA approaches are used: an ‘attributional’ one (intended to provide a static representation of average conditions, excluding market-mediated effects) and a ‘consequential’ one, which strives to identify the consequences associated with the change applied to the product/service system (Weidema et al., 2003, 2009).

Figure 3-4 provides an example of the boundaries and main processes considered for a generic LCA of organic waste treatment by anaerobic digestion (A) or incineration

(B). Additionally to fulfilling the 'main service/function' (i.e. the treatment of the organic waste in input), the digestion system generates two valuable products: biogas and digestate (organic bio-fertiliser). Similarly, incineration provides energy and aggregate-type material (bottom ash). Owing to these, the system under assessment (both A and B) becomes multifunctional as co-products are generated along with the fulfilment of the main waste system service/function (i.e. the mere treatment of the organic waste input). To address multifunctionality, two options are available (see Commission Recommendation 2013): i) subdivision or system expansion, ii) allocation (mass, energy, or price). Allocation principles, sometimes used in attributional studies, should only be applied when system expansion (or subdivision) is not possible conforming with the best practices described by Commission Recommendation (2013).

Keeping the above in mind, the practitioner, depending upon the scope of the study and following recommendations from the guidelines mentioned earlier, may in this specific case apply: i) system expansion considering average market processes (this is also referred to as an 'attributional approach', which applies system expansion using average market data), ii) system expansion considering marginal market processes (also called a 'consequential approach'), or iii) allocation principles (i.e. no system expansion; this is also referred to as an 'attributional approach').

In the anaerobic digestion example below, expanding the system would mean including energy generation and fertiliser manufacture/application which is displaced by bio-electricity and bio-fertiliser (digestate) application, respectively. In the case of incineration, in addition to the displaced energy, the LCA practitioner should account for the natural aggregates extraction, transport and processing that are avoided when bottom ash is recycled. Then, results may be expressed as comparisons between scenarios (e.g. A versus B below) or as environmental burden *changes* expected from a particular change of strategy, or the introduction of a new system (e.g. going from A to B or vice versa) – as appropriate to inform waste management strategy from a wider public good perspective. It is important to bear in mind that the magnitude of these changes is highly dependent on the 'displaced processes' (e.g. type of fertilisers, electricity). Consequential LCA should be based on predicted marginal effects, rather than average effects: e.g. the question to be asked could be: "what type of electricity generation is replaced by new bio-electricity fed into the grid from biogas generation"?

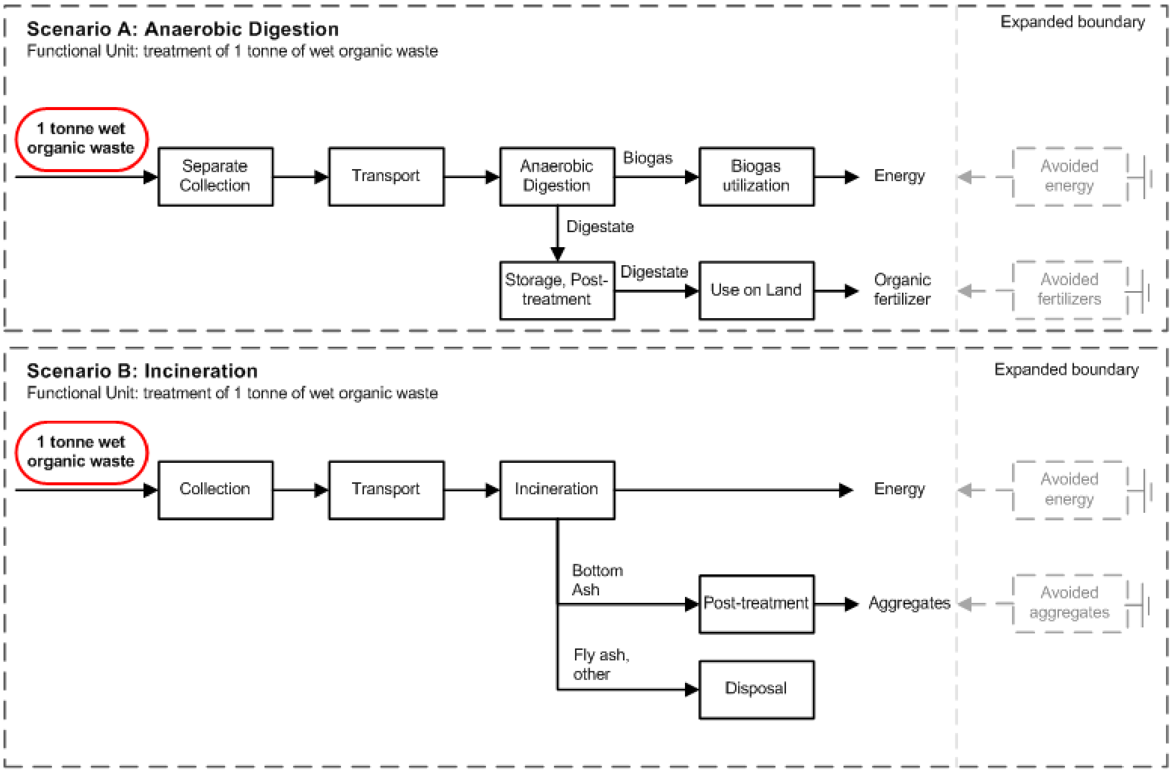

Figure 3-4. An example of the LCA system boundary for the comparison of two alternative management scenarios for wet organic waste: A) anaerobic digestion and B) incineration[31].

Finally, the environmental scope of LCA may be expanded to consider flows of money (life-cycle costing) and social capital (social life-cycle assessment). The United Nations Environment Programme provides guidelines on how to undertake social LCA (UNEP, 2009).

Once the LCA and system boundaries have been defined, the impact categories to be considered must be decided – see the section on *Life-cycle impact assessment indicators*, below.

In the AD case study referred to under "*Description*", boundaries were defined to include waste collection and transport, processing through the AD plant, digestate application including fertiliser replacement, biomethane upgrade and replacement of transport diesel, and also avoidance of pre-existing waste management options (landfilling, in-vessel composting and animal feeding – in the latter case avoided cultivation of wheat as an animal feed).

<u>Inventory compilation</u>

Inventory compilation is the second phase of LCA, in which data on activities and associated inputs, outputs and burdens are compiled for the system of study (e.g. AD

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[31] Note that system expansion is applied to handle multi-functionality (co-products, i.e. energy, organic fertiliser, and aggregates). Induced processes are represented with a continuous black line, while avoided processes are represented with grey dotted lines. In a consequential approach, avoided processes would be modelled with 'marginal market data', while in an attributional LCA 'average market data' would be used instead when system expansion is applied. In a hypothetical situation where allocation techniques are instead applied to handle co-products (in place of system expansion), the boundary would be as displayed here in light grey

system or in-vessel composting system). The International Reference Life-Cycle Data System (ILCD) provides a common basis for consistent, robust and quality-assured life-cycle data, methods and assessments (JRC, 2011), and hosts the European Platform on LCA (http://eplca.jrc.ec.europa.eu/) – an open-access life-cycle inventory database. Various commercial LCA databases also exist, such as Ecoinvent (http://www.ecoinvent.org/), that contain extensive data on common generic processes. Often, it is possible to simply multiply system-specific activity data (e.g. tonne-km of transport) with unit process data from LCA databases (e.g. environmental burdens, such as kg CO_2e, per tonne-km transport in a EURO V compliant 16-32 tonne truck) to generate burdens for particular processes, stages, and ultimately entire systems. In other cases, it may be necessary to use process-specific data to calculate burdens (e.g. measured or calculated methane leakage rates from fermentation, digestate storage and biomethane upgrade). For example, in the case of digestate and compost application to land, Bruun et al. (2006) propose long-term (100-year) soil organic carbon sequestration credit (a CO_2e "credit") equivalent to 13 % and 14 % of the organic C contained in digestates and composts, respectively. These values were used by Møller et al. (2009) to evaluate the life-cycle environmental performance of anaerobic digestion.

Owing to the number of actors involved in a typical product life cycle, or waste stream flow, it will often be necessary to obtain activity data from other organisations in order to complete an LCA. Care should be taken to evaluate the quality (accuracy and validity of the data) during data collation, so that appropriate uncertainty analyses and sensitivity analyses may be undertaken to facilitate interpretation. Data may be tagged as low, medium, or high uncertainty for example, or statistical distributions (e.g. 95 % confidence intervals) may be recorded.

Inventory data compiled for the AD case study example included:

- diesel consumption for transport of waste to the digester, calculated based on distance transported multiplied by burdens expressed per tonne-km in the Ecoinvent database;
- fugitive emissions of methane from the digester, from digestate storage and from biomethane upgrade, estimated from emission factors of 1 %, 1.5 % and 1.4 % of total biomethane yields, respectively;
- ammonia emissions from digestate storage, estimated from an ammonia-N emission factor of 10 % of ammonium-N in digestate;
- transport diesel fuel replaced calculated based on a biomethane yield of 440 m^3 per tonne of dry matter (food waste), a methane lower heating value of 34 MJ per m^3, 20 % of biomethane used on site to generate process heat and electricity, and a substitution efficiency of 1 MJ biomethane per 0.75 MJ diesel.

The above list is far from exhaustive, as it excludes, for example, diesel combustion, nutrient losses and fertiliser replacement incurred by digestate application.

<u>Life-cycle impact assessment (LCIA)</u>

Life-cycle impact assessment (LCIA) involves the characterisation of inputs and emissions according to their environmental damage potential, using factors derived from extensive fate and transport modelling (e.g. Huijbregts et al., 2001), thus synthesising inventories of inputs and outputs into a small number of environmental indicators representing key environmental burdens (Pennington et al., 2004).

LCIA involves the multiplication of inputs and outputs by relevant characterisation factors to represent contributions towards environmental burdens or impacts. LCIA is typically performed across three areas of protection: human health, natural environment, and natural resource use, and may include the following impact categories (JRC, 2011): climate change, ozone depletion, eutrophication, acidification, human toxicity (cancer- and non-cancer-related), respiratory inorganics, ionising radiation, ecotoxicity, photochemical ozone formation, land use, and resource depletion (materials, energy, water).

Table 3-2 summarises LCIA methods recommended for the International Reference Life-Cycle Data System (JRC, 2011).

Table 3-2. Midpoint life-cycle impact assessment methods proposed by JRC (2011) for the harmonisation of methods in the International Reference Life-Cycle Data System

Method	**Flow property**	**Reference unit**
Global warming potential, GWP100	Mass CO_2 equivalents	Units of mass (kg)
Ozone depletion potential, ODP	Mass CFC-11 equivalents	Units of mass (kg)
Cancer human health effects, CTUh	Comparative Toxic Unit for humans (CTUh)	Units of items (cases)
Non-cancer human health effects, CTUh	Comparative Toxic Unit for humans (CTUh)	Units of items (cases)
Respiratory inorganics, PM2.5 equivalents	Mass PM2.5 equivalents	Units of mass (kg)
Ionising radiation, ionising radiation potential	Mass U_{235} equivalents	Units of mass (kg)
Photochemical ozone formation potential, POCP	Mass C_2H_4 equivalents	Units of mass (kg)
Acidification, accumulated exceedance	Mole H+ equivalents	Units of mole
Eutrophication terrestrial, accumulated exceedance	Mole N equivalents	Units of mole
Eutrophication fresh water, P equivalents	Mass P equivalents	Units of mass (kg)
Eutrophication marine, N equivalents	Mass N equivalents	Units of mass (kg)
Ecotoxicity fresh water, CTUe	Comparative Toxic Unit for ecosystems (CTUe) * volume * time	Units of volume*time (m^3*a)
Land use, soil organic matter	Mass deficit of soil organic carbon	Units of mass (kg)
Resource depletion – water, fresh water scarcity	Water consumption equivalent	Units of volume (m^3)
Resource depletion – mineral, fossils and renewables, abiotic resource depletion	Mass Sb equivalents	Units of mass (kg)
Source: JRC (2011).		

Indicator results may be normalised (divided by "total" environmental loadings at a specified scale) to enable comparison of *relative* contributions across environmental impact categories. For example, Andersen et al. (2012) present LCIA indicator results normalised as milli-person equivalents (contributions to annual per capita loadings, divided by 1 000).

In Figure 3-5, burden data for a partial-expanded-boundary LCA of one tonne of organic waste treated by decentralised composting are presented after normalisation against average European citizen per capita loadings. Positive values indicate an adverse impact on the environment, whilst negative values indicate environmental savings compared with the alternative of separate waste collection (though the alternative waste management option is not accounted for in this particular partial LCA). Emissions of nitrous oxide and methane during composting give rise to a significant GWP burden, soil emissions of ammonia following application give rise to a significant AP effect, and replacement of fertilisers with organic nutrients following field application leads to significant EP, AP and FRDP savings (Figure 3-5).

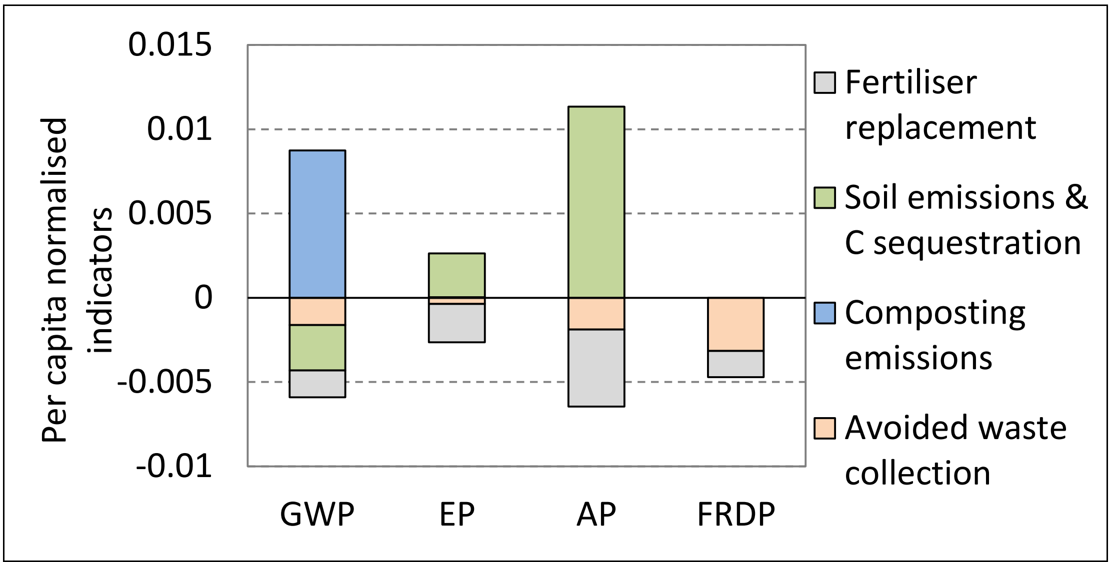


Figure 3-5. Results for global warming potential (GWP), eutrophication potential (EP), acidification potential (AP) and fossil resource depletion potential (FRDP) for decentralised composting of household organic waste (see Section 4.7.2)

A full consequential LCA would account for burdens and savings associated with alternative (replaced) waste management option(s), such as centralised composting, anaerobic digestion or MSW incineration. Results for the consequential LCA of the AD case study are displayed in the next section, expressed using the same four environmental indicators used in Figure 3-5.

Following on from the characterisation of input and output data to generate environmental indicators, ISO 14040 (ISO, 2006a) defines three optional steps:

- Normalisation: Indicator values (e.g. kg PO_4e) are converted into environmental loadings relative to a reference value – often "total" loading at national, EU or global scale, or for example per capita.
- Grouping: The impact categories are sorted and possibly ranked.
- Weighting: The different environmental impacts are weighted relative to each other so that they can then be summed to get a single number for the total environmental impact.

These procedures may facilitate an understanding of the relative importance of nominal indicator values across impact categories, but weighting is not recommended in ISO 14040 owing to the introduction of value judgements. In converting nominal indicator units into comparable burden fractions, normalisation facilitates the comparison of contributions to different environmental problems and relative trade-offs.

Interpretation and reporting

According to ISO 14044 (ISO, 2006b), the interpretation phase of an LCA study comprises the following elements:

- identification of significant issues based on the findings (life-cycle inventory (LCI) and life-cycle impact assessment (LCIA) phases);
- an evaluation that considers completeness, sensitivity and consistency;
- conclusions, limitations, and recommendations.

It is useful to structure results from the LCI and LCIA phases according to life-cycle stages and processes to underpin contribution analysis that in turn facilitates presentation, interpretation, validation and anomaly assessment (ISO, 2006b).

Mass or energy balance analysis of all input and output data may also be applied to check for anomalies, according to the law of conservation of mass and energy. The influence of uncertainty on final results can be tested using sensitivity analysis (e.g. Clavreul et al., 2013). Uncertainties for individual process interventions can be aggregated up to the system level based on error propagation methods.

Where results of comparative studies are intended for public disclosure they should be critically evaluated by an appropriate expert or panel of interested parties, and the results of the evaluation disclosed, according to ISO 14044 (ISO, 2006b). The critical review process shall ensure that:

- methods used to carry out LCA are consistent with the ISO standard;
- methods used to carry out LCA are scientifically and technically valid;
- data used are appropriate and reasonable in relation to the goal of the study;
- interpretations reflect the limitations identified and the goal of the study;
- the study report is transparent and consistent.

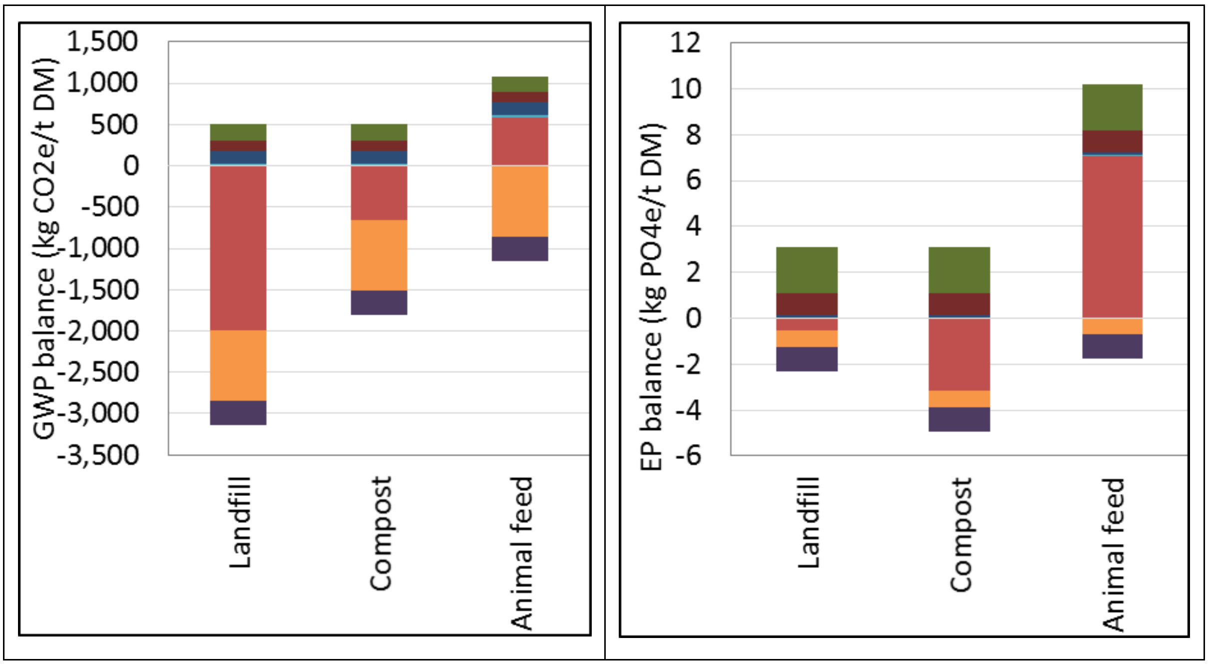
With respect to reporting LCA results, the goal, scope and boundaries applied should be clearly reported.

Table 3-3 and Figure 3-6 below summarise the environmental changes that arise, expressed as credits (negative values) and burdens (positive values) across avoided and incurred processes (Figure 3-6), and expressed as *net* environmental burden change (Table 3-3), in relation to one tonne of food waste dry matter – from the AD consequential LCA case study. Avoided waste management and avoided fossil energy (transport diesel) give rise to substantial environmental credits (negative values) in most cases, indicating that AD performs better than avoided waste management options – apart from in the case of animal feed. In this respect, it should be noted that

Styles et al. (2016) considered animal feeding as a particular food waste management option.

Where food factory waste can be used as animal feed for example, this avoids cultivation of wheat as an animal feed, and therefore generates significant environmental credits. These credits are no longer realised if waste is sent to AD rather than animal feed, and so become represented as a burden for AD (in Figure 3-6 see red "waste management" for the animal feed scenario).

These results are unique to the precise scenarios and underlying operational assumptions for typical UK conditions defined in Styles et al. (2016). Undertaking consequential LCA is associated with a high degree of specificity in relation to the *transitions* considered (from which baseline to which option), and a high degree of uncertainty. Results should include a robust sensitivity analysis on the scenario uncertainties (e.g. choice of displaced 'marginal' technologies/processes) and on the parameters' uncertainties (e.g. efficiencies, transport distances). On this basis, results should therefore be interpreted cautiously and always in relation to the specific scenarios and context considered.



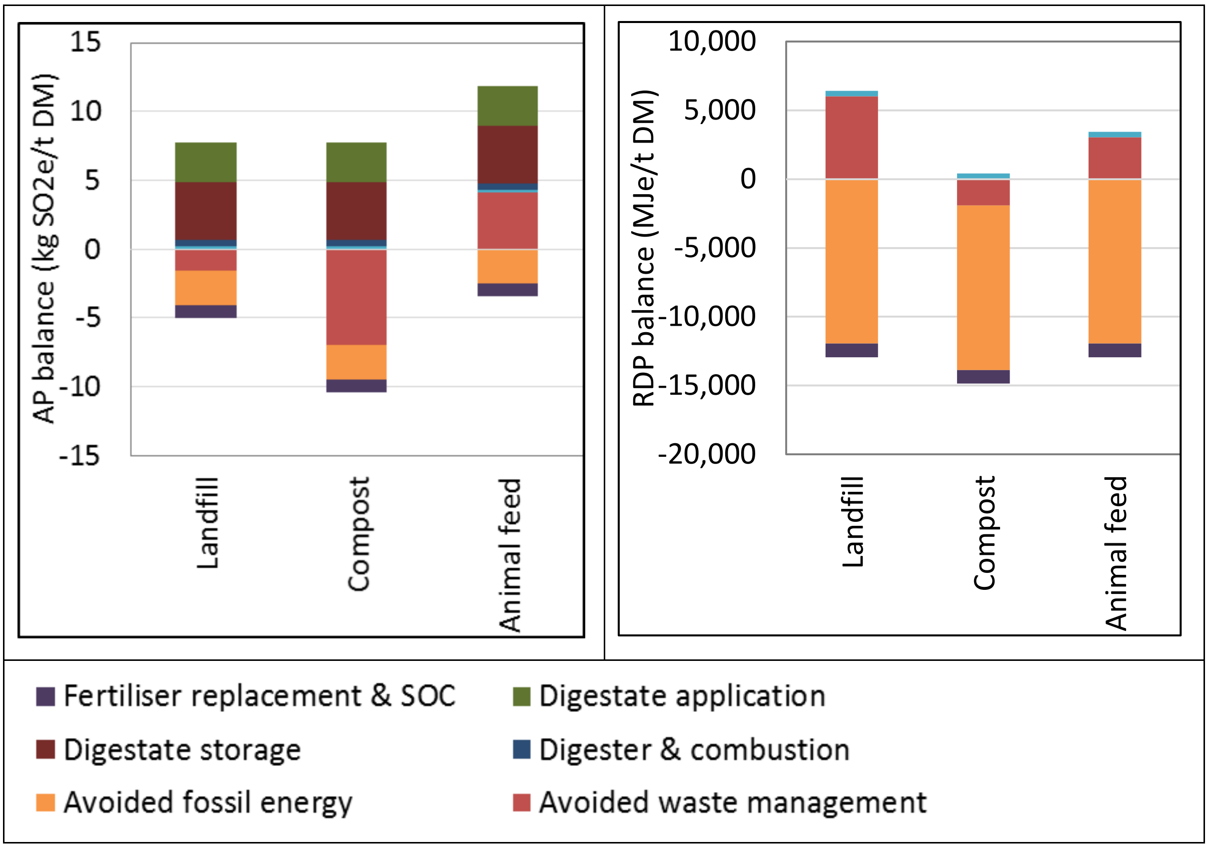


Figure 3-6. Net environmental burden changes, expressed per tonne of dry matter organic waste processed, when anaerobic digestion replaces landfilling, in-vessel composting or use of hygienic organic waste for animal feed

Table 3-3. Net environmental burden changes, expressed per tonne of dry matter organic waste processed, when anaerobic digestion replaces landfilling, in-vessel composting or use of hygienic organic waste for animal feed

	Landfill	Compost	Animal feed
Global warming (kg CO_2e)	-2 640	-1 306	-74
Eutrophication (kg PO_4e)	0.8	-1.8	8.4
Acidification (kg SO_2e)	2.7	-2.7	8.4
Fossil resource depletion (MJe)	-6 516	-14 449	-9 492

B) Case study on use of LCA for waste management options in Tuscany

SEI is the company in charge of waste management in 100 municipalities in the south of Tuscany (Italy). An LCA which calculated the carbon footprint of the waste management system in 2014 and for two different scenarios for 2021 was recently carried out. In 2014, the level of separate collection of municipal solid waste was 21 %, while for the first scenario for 2021 it was assumed to reach 45 % and for the second scenario for 2021 separate collection was assumed to be 48 %. In addition to the three different cases analysed (waste management system in 2014 and two scenarios for 2021), the carbon footprint differentiated between considering or not the benefits of energy generation from incineration of waste (avoiding the combustion of fossil fuel) and material recovery from waste reuse/recycling (avoiding raw material production). In Table 3-4, data on the carbon footprint of the SEI waste management system for 2014 (calculated for one tonne of waste) is reported. Note the translations from Italian to English for Table 3-4, Table 3-5 and Table 3-6: compostaggio domestico - domestic home composting; raccolta rifiuti - waste collection; trattamento – treatment; riciclo - recycling; riutilizzo - reuse; fine vita - end of life; senza benefici – without benefits (avoiding the combustion of fossil fuel and avoiding raw material production); con benefici – with benefits (avoiding the combustion of fossil fuel and avoiding raw material production) (Bolognani, 2016).

	Senza benefici (kg CO_2eq)	Con benefici (kg CO_2eq)
Compostaggio domestico	0,67	0,67
Raccolta rifiuti	60,82	60,82
Trattamento	37,83	37,83
Riciclo	1,48	-286,96
Riutilizzo	0,00	-6,29
Fine vita	190,65	84,04
TOTALE	291,45	-109,89

Table 3-4. Carbon footprint of SEI waste management system in 2014 (for 1 tonne of waste), with and without benefits (Bolognani, 2016)

Table 3-5 instead reports the carbon footprint calculation for the 2021 scenario of separate waste collection at 45 %.

	Senza benefici (kg CO_2eq)	Con benefici (kg CO_2eq)
Compostaggio domestico	2,40	2,40
Raccolta rifiuti	88,18	88,18
Trattamento	62,68	62,68
Riciclo	2,74	-659,24
Riutilizzo	0,00	-15,45
Fine vita	161,66	79,58
TOTALE	317,67	-441,86

Table 3-5. Carbon footprint of SEI waste management system in 2021 scenario (for 1 tonne of waste) with separate collection at 45 %, with and without benefits (Bolognani, 2016)

Finally, Table 3-6 reports the carbon footprint calculation for the 2021 scenario of separate waste collection at 48 %.

	Senza benefici (kg CO_2eq)	Con benefici (kg CO_2eq)
Compostaggio domestico	2,39	2,39
Raccolta rifiuti	92,94	92,94
Trattamento	61,69	61,69
Riciclo	2,95	-662,90
Riutilizzo	0,00	-33,89
Fine vita	144,29	70,70
TOTALE	304,26	-469,06

Table 3-6. Carbon footprint of SEI waste management system in 2021 scenario (for 1 tonne of waste) with separate collection at 48 %, with and without benefits (Bolognani, 2016)

From the three tables presented above, it is clear that the highest share of the carbon footprint in the cases without benefits (not considering the avoided combustion of fossil fuel and raw material production) is the end of life related to incineration and landfill. The situation changes substantially when separate collection of waste is increased and the benefits are considered in the carbon footprint calculations (Bolognani, 2016).

The LCA study allowed SEI to understand and assess the carbon savings achievable by improving the separate collection of waste and reducing the amount of waste sent for incineration and disposal in landfill. However, it was also clear from the study that the carbon footprint is not the only possible parameter to evaluate different waste management system options; other indicators (e.g. water use, waste water generation) can also be taken into account to more broadly evaluate the environmental performance.

C) Available software models and tools

One example of an LCA tool for evaluation of waste management technologies is "EASETECH" (Environmental Assessment System for Environmental TECHnologies), developed at the Technical University of Denmark. EASETECH enables users to perform LCA of systems handling heterogeneous material flows, accounting for resource use, recovery and emissions (e.g. Damgaard et al., 2011). Material flows are represented as a mix of material fractions with specified properties, partitioning and fates (e.g. rejects, slags, ashes and products), behind a toolbox interface that enables scenarios to be defined according to process and material flow combinations (DTU, 2015). The EASETECH and the module EASEWASTE are available for researchers, consultants, authorities and technology developers, after training in the use and interpretation of the model has been undertaken at a cost of approximately EUR 5 000 (DTU, 2015).

Various other LCA software tools are available, on a free-to-use or commercial basis, including the examples below:

- Open LCA: free LCA software available at http://www.openlca.org/;
- SimaPro: commercial LCA software available from PRé Consultants at http://www.pre-sustainability.com/simapro;
- GaBI: commercial LCA software available at http://www.gabi-software.com/.

Applicability

Life-cycle assessment is not always necessary. Basic prioritisation of waste management options indicated in the waste management hierarchy may be sufficient to inform best practice in some cases. However, detailed comparison of options ranked similarly in the waste hierarchy, and of management changes that affect whole-waste-chain performance, is often required.

Waste management organisations of any size may apply life-cycle thinking and review LCA studies. Buying bespoke LCA services and/or paying for staff training in LCA may only be economically viable for larger organisations.

Economics

LCA software and database access costs for commercial entities vary depending on the purpose of use and the number of individual (staff) users. Software licence fees are often bundled with database access fees and service contracts that provide support, software and database updates. For example, one provider offers commercial licences ranging from EUR 2 400 for a single-user "report maker" licence to EUR 22 000 for a multi-user developer licence (PRé Consultants, 2015).

Effective use of open-access LCA software such as Open LCA may require the purchase of a database access licence, and/or staff training: e.g. the Technical University of Denmark provides training courses in the use of EASETECH for EUR 5 000 per person.

Undertaking in-house LCA studies will also require significant staff time that should be accounted for in project costs. Alternatively, procurement of LCA services from a consultancy or academic institution is likely to cost tens of thousands of euros, but could avoid costs associated with licensing and staff time.

Efficiency benefits associated with systems thinking and optimisation informed by LCA could be orders of magnitude greater than these costs, but may be difficult to attribute directly.

Driving force for implementation

Waste management organisations may apply life-cycle thinking and assessment to:

- improve operational efficiency;
- reduce environmental impacts and potential liabilities;
- demonstrate the sustainability of their operations to stakeholders;
- comply with corporate social responsibility and stakeholder reporting obligations.

Reference organisations

Aschaffenburg local authorities demonstrate comprehensive and systematic life-cycle thinking in their waste management strategy, as described in the previous BEMP (Section 3.3.1).

The Technical University of Denmark (DTU) is a well-known organisation in LCA accounting for waste systems, and it provides software tools and training for waste managers.

An LCA study was undertaken to compare the current situation of MSW incineration in the Aalborg county of Denmark with an alternative scenario of anaerobic digestion of the separated organic fraction (Hill, 2010). The results of the LCA indicated that the current situation is the better option from an environmental perspective if the anaerobic digestion plant is managed in a "typical" manner, but that anaerobic digestion could be the better option if it is managed in accordance with best practice recommendations – highlighting the sensitivity of LCA results to operational parameters and assumptions.

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3.3.3. Economic instruments

Summary overview
It is BEMP to use economic instruments, to steer the behaviour of citizens and organisations generating waste towards more environmentally friendly results. Economic instruments can support: - reducing the amount of waste generated or reducing the proportion of hazardous waste; - encouraging preparation for reuse and recycling of waste; decreasing incineration and landfilling; - improving product design (e.g. encouraging the use of recyclable materials in products). The economic instruments related to waste management cover both incentives (positive economic signals, e.g. discounts, reward vouchers) and disincentives (negative economic signals, e.g. taxes, fees, penalties) and can take the form of: - taxes and tax modulation, e.g. waste disposal tax, landfill tax, incineration tax; - product levies (e.g. on plastic bags or construction aggregates); - waste pricing, such as unit-based pricing and pay-as-you-throw (PAYT) schemes; - deposit-refund schemes; - extended producer responsibility schemes; - others, e.g. tradable permits, recycling subsidies, VAT exemptions.

Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**

Applicability
The regulatory framework and its enforcement are the main barriers for the application of economic instruments at local level. In addition, the existence of environmental awareness, good management skills and innovation-driven behaviour at the local government level, with some good accounting practices, are prerequisites for the implementation of local economic instruments, which are complex to manage from the technical, managerial and social perspectives.

Specific environmental performance indicators
- Use of economic instruments at local level to stimulate good behaviour (y/n).

- Share of residents/businesses using a voluntary economic instrument (%).
Benchmarks of excellence
- Economic instruments set at local level in the form of taxes and tax modulation, product levies, waste pricing, extended producer responsibility schemes and deposit refund schemes are systematically implemented as a means to achieve the objectives set in the local waste management strategy. - For local authorities, a deposit refund scheme for glasses, cups, dishes and cutlery is in place for all festivals and large public events organised in the territory of the local authority.

Description

Aim

This BEMP gathers useful information and practical examples of economic instruments that can be applied by mainly local authorities and, possibly, by waste management organisations, in charge of the introduction of economic instruments, with the main focus on the local scope of its implementation. Although most of the measures described are oriented to municipal solid waste (MSW), there are several existing mechanisms oriented for industrial wastes, represented here mainly by construction and demolition waste (CDW). The term 'economic instruments' refers to regional or national policies or regulations. Herein, the term 'local economic instrument' is used to refer to an economic instrument applied at local level.

Introduction

As for environmental policies in general, waste management also includes a mix of complementary measures such as regulatory, economic, educational and informative instruments (OECD, 2007; van Beukering et al., 2009). Economic instruments are designed to persuade households and waste producers to strive towards diverting waste from landfills, recycle more waste and optimise the use of resources in order to prevent the generation of wastes, and, at the same time, contribute to financing waste management activities. From the economic point of view, these instruments are preferable to direct regulation due to their greater efficiency. While the polluter pays the abatement cost of the generated impact from waste generation and treatment, the existence of a tax, a levy, etc. is a clear incentive for the polluter to search for new abatement options (van Beukering et al., 2009).

Economic instruments belong to national or regional waste policies, usually responding to their particular objectives, and most of them fall outside the scope of this document. Also, the application of economic instruments is not a textbook solution but a tailor-made set of tools that may result in different performances in different regions or countries. Several approaches, however, fall under the decision-making process of waste authorities in charge of municipal waste, and, only to a certain extent, to private organisations in charge of other commercial and industrial wastes.

The application of economic instruments has been repeatedly recommended (EC, 2003, 2005, 2007, OECD, 2004, 2007). Some of the main applied instruments are detailed below:

- Taxes, e.g.
    - waste disposal tax;
    - landfill tax;
    - incineration tax;
    - product levies (e.g. on plastic bags or aggregates).
- Waste pricing, such as
    - unit-based pricing and pay-as-you-throw schemes;
    - differential and variable rates;
    - variable fee or charge systems.
- Deposit refund schemes.
- Extended producer responsibility systems.
- Others, such as:
    - tradable permits;
    - recycling subsidies;
    - VAT exemptions;
    - extension of depreciation periods;
    - positive incentives.

In general, economic instruments aim at:

- reducing the amount of waste generated;
- reducing the proportion of hazardous waste;
- improving product design;
- encouraging recovery, reuse and recycling of wastes;
- decreasing incineration and landfilling;
- minimising adverse environmental impacts related to solid waste collection, transport, treatment and disposal systems;
- encouraging the use of recyclables in products; and
- generating revenues to cover costs.

In any case, the economic instruments are implemented to link the cost of waste treatment charged to the waste generator (the citizen or the organisation) with the real amount of waste generated, i.e. by charging per unit of waste, charging for the consumption of avoidable products, and rewarding desirable practices.

Economic instruments applied to commercial and industrial wastes are essentially different from those applied to municipal solid waste. For example, unit-based pricing per type of treatment is a standard practice by waste service providers for CDW and HCW. However, MSW fees from public authorities are constant in many cases, independently of the amount generated by each citizen, due to the high dispersion of a large number of producers.

Local instrument for the management of MSW

Pay-as-you-throw (PAYT). In terms of municipal waste treatment, the economic instrument that works best is the pay-as-you-throw scheme. A specific BEMP on PAYT for MSW can be found in Section 4.3.3.

Recycling incentive schemes. Formally speaking, financial incentives include both rewards (to be described here as recycling incentives) and charges (defined here as pay-as-you-throw, and deposit refund schemes). But it is commonly accepted that recycling incentives schemes are essentially different from PAYT schemes. They consist of payments or rewards given to the users to encourage people to recycle more, typically with vouchers for individuals, vouchers for communities or payments to individuals (Holmes et al., 2014). In addition to direct incentives in the form of vouchers, an effective recycling incentive is also the reduction of waste fees for residents willing to separate more waste at source (e.g. accepting a new more advanced waste collection system) or when waste recycling targets at local level are achieved. Most of the examples that are applied in Europe are pilot schemes or partial coverage schemes implemented after the success of the pilot trial. Of these, some selected case studies are described in this document. It is important to note the following:

- Legal regulation at local level is a key factor for their implementation. While recycling incentive schemes are usually acceptable, PAYT has certain legal connotations that make its implementation difficult in particular regulatory environments. This is the case of the UK, where the debate is ongoing.
- Behavioural aspects need consideration. PAYT addresses the whole range of awareness levels, while reward schemes are generally oriented to recyclers. The study by Holmes et al. (2014) showed that "regardless of the reward type, personal or community, the majority of respondents claimed they already recycle as much as possible". However, a greater proportion of householders are likely to recycle more when rewarded individually.
- They tend to be self-funded. Some schemes are applied along with other measures to increase their efficiency. For instance, the 'Cash for Trash' scheme in the Netherlands applies increased charges to the final users, which is believed to have a significant impact on the results (OECD, 2015).

Given the right conditions (see applicability), recycling incentive schemes can be considered a best environmental management practice, due to their performance and costs. It is, however, difficult to benchmark such a system against PAYT, as their scope and applicability differ.

Local deposit refund schemes. A deposit refund scheme consists of a surcharge on the price of potentially polluting products. When pollution is avoided by returning the products or their residuals, a refund of the surcharge is granted (OECD, 2014). In the understanding of Ferrara (2008), deposit refund schemes are generally identified as the most effective option to improve the rate of recycling and they have been successfully applied to beverage containers, so their use is considered a best environmental management practice (Hogg et al., 2010; Schoenberger et al., 2013). However, their implementation goes beyond the municipal or county level, the usual geographical scope for the techniques described in this document. Municipalities, however, can run their own deposit refund schemes or impose the use of one. Some examples are shown below:

- A deposit is charged for portable batteries by the local government of Osthamar, Sweden (OECD, 2014), achieving a capture rate close to 100 %.
- Police regulation, e.g. City of Schwäbisch Gmünd (2005), Germany: mandatory deposit of at least EUR 2.00 for glasses used during the city festival.
- Waste management statutes, e.g. City of Nuremberg (2009), Germany: § 7 of the waste management statutes prescribes for all events in public institutions and on any parcel of land belonging to the city of Nuremberg, including public

transport areas, the use of reusable containers and reusable cutlery, supported by a deposit.

- Participation conditions/city market rules, e.g. City of Reinheim (2012), Germany: participation conditions/regulation for Christmas market: prohibition of single-use tableware, mandatory use of reuseable glogg cups, mandatory deposit of at least EUR 1.00, or City of Graz, Austria: charge of EUR 1.00 per beverage containers in football stadiums to limit littering.

Construction and demolition waste and healthcare waste

As this BEMP refers to cross-cutting issues, it is worth mentioning the different approaches to several economic instruments for different types of wastes. CDW management contracts include a fee per unit of collected volume, which varies for different fractions, the most expensive being for the mixed waste fraction (up to EUR 100 per tonne) compared to metals or clean concrete (from EUR 5 to EUR 25 per tonne). A very similar approach is observed in the management of HCW: the waste contractor usually charges the waste treatment cost per bin or container in which the waste is collected and stored. So, the healthcare organisation producing the waste may consider the implementation of best practices in its in-house waste management system to reduce costs.

For commercial and industrial waste, the business-to-business (B2B) approach is successfully applied. The existence of a B2B deposit refund scheme is sometimes a common practice for highly reusable packaging, like pallets, construction packaging, drums and others (Lundesjo, 2011; WRAP, 2008), and these practices have extensively reduced the amount of waste generated, e.g. at construction sites. Although waste managers are not involved in this particular approach, they are key in the management of the required reversed logistics, e.g. in the London Construction Consolidation Centre, partially run by the local government through Transport for London, and operating under a deposit refund scheme (WRAP, 2010).

Some municipalities have applied traceability requirements of CDW in their local licensing. All municipalities in Spain charge a deposit for the estimated amount of wastes reported in the site waste management plan, and it is an essential requirement for the operating licenses. The deposit is repaid to the contractor when "waste management certificates" are submitted to the authority. This deposit system managed by municipalities has the potential to become a BEMP, but its current performance is far from such consideration due to the following reasons:

- It is oriented towards avoiding illegal dumping. Direct landfilling of mixed waste is accepted as a correct management treatment, and is eligible for deposit return; this would not lead to best performance.
- Legally, municipalities do not need to issue permits for their own construction sites. The waste management deposit then becomes voluntary.
- The lack of enforcement affects the performance of the scheme. While large construction companies and contractors were already applying BEMP without the deposit, small producers are still failing to fulfil this practice.

Other successful economic instruments for CDW or HCW are applied at national or regional level. For instance, HCW extended product responsibility schemes, e.g. for waste medicines, or CDW product levies, e.g. adaptation of VAT for natural or recycled aggregates.

Achieved environmental benefits

Municipal solid waste

The performance of several case studies on the application of local economic instruments in municipalities is shown in Table 3-7.

Table 3-7. Examples of reward schemes and PAYT performance[32]

Municipality or county	Instrument	Results	Additional comments	Refe-rence
Bracknell Forest, UK	Recycling incentive scheme	Enhanced public perception and wide acceptability of recycling Increase of a total of 1 000 tonnes of recyclables in one year of implementation (around 91 kg per household per year)	Urban, all recyclables	BFC, 2012; BFC, 2015
Torelles de Llobregat, ES	Pay-as-you-throw, unit-based	Increase of separately collected materials from 33 % to 89 %, reduction of mixed waste by 38 %	Urban, all waste streams	OECD, 2006
Landkreis Schweinfurt, DE	Pay-as-you-throw, weight-based plus fixed fee	Total waste collected reduced by 28 %, and mixed waste reduced by 46 %	Urban, all waste streams	OECD, 2006
Ghent and Destelbergen, BE	Pay-as-you-throw, volume- and unit-based	Total waste arisings reduced, but not only attributable to PAYT	Urban, all waste streams	OECD, 2006
Valongo and Gondomar, PT	Recycling incentive scheme at drop-off sites (collection centres)	Paper and cardboard increased by 14 %, plastic 9 %, glass 75 %, batteries 24 % and used cooking oils 74 %.	Urban, waste streams at 2 collection centres	R4R, 2014a
Limerick, Clare, Kerry regions, IE	Pay-as-you-throw, weight system	Reduction of mixed waste from 79 % to 65 %, and increase in collection of recyclables from 21 % to 32 %	Urban and rural, all waste streams	R4R, 2014b
Aschaffenburg, DE	Pay-as-you-throw, weight system	Increased collection of recyclables up to 86 %, decrease of mixed waste disposal costs, reduction of residual costs down to around 50 kg per capita per year	Urban and rural, all waste streams	Section 4.3.3
Rotterdam, Barendrecht and Krimpen aan den Ijssel, NL	Recycling incentive system	Increased collection of 24 % (total waste), reduction of mixed waste of 37 %	Called 'Cash for Trash', rewards are cash paid directly back to citizens	OECD, 2015

[32] The most practical definition of "mixed waste" from the perspective of waste authorities in this BEMP is the remaining fraction of unsorted waste destined for disposal (e.g. incineration), either at the time of collection, or at the time of being sent to final treatment when the waste management company is involved in subsequent sorting (e.g. in sorting plants following co-mingled collection, or in mechanical and biological treatment plants).

Table 3-7. Examples of reward schemes and PAYT performance[32]

Municipality or county	Instrument	Results	Additional comments	Reference
Bradford, Aire Valley Recycling, UK	Recycling incentive scheme	Increase of 36.5 kg of recyclables collected per participant per year	Urban, all recyclables	Defra, 2013
Bath and North Somerset, UK	Recycling incentive scheme	Increase of 57 kg of recyclables per participant per year	Urban and rural, all recyclables	Defra, 2013
Birmingham, UK	Recycling incentive scheme	Increase of 5.2 kg of recyclables per participant per year	Urban, paper and cardboard	Defra, 2013
Gloucestershire, UK	Recycling incentive scheme	No increase or decrease of recyclables per participant per year	Urban and rural, all recyclables	Defra, 2013
Norfolk, UK	Reuse and recycling incentive scheme	Increase of 99 kg of reusables and recyclables per participant per year	Urban and rural, implemented through reuse shops	Defra, 2013
Student association in Bristol, UK	Recycling incentive scheme	Increase of 57 kg recyclables per participant per year	All recyclables	Defra, 2013
Preen Community in Bedfordshire, UK	Reuse incentive scheme	Increase of 67 kg recyclables and reusables per participant per year	Urban and rural, implemented through reuse shops	Defra, 2013
Westminster, UK	Recycling incentive scheme	No increase or decrease of recyclables per participant per year	Urban, all recyclables	Defra, 2013

Benefits in B2B deposit schemes for CDW

WRAP (2012) studied the environmental benefit of two different approaches for the reuse of three very common packaging items used for construction products: pallets, plastic folding boxes and bulk bags. Deposit refund schemes were used and waste collectors were involved in the application of reverse logistics (i.e. products to be reused are also transported by the waste manager). The results were compared to a hypothetical 100 % recycling scenario for the wood and plastic of the packaging materials, and CO_2 savings were calculated along with the theoretical minimum number of trips required to achieve those emission levels (Table 3-8). It can be seen that the performance of reverse logistics is significantly better.

Table 3-8. Greenhouse gas emissions savings and minimum number of trips of reusable packaging compared to single-use packaging (WRAP, 2012)

Packaging	Reverse-logistics		Separate collection and return	
	CO_2e savings	Minimum trips	CO_2e savings	Minimum trips
Trademarked pallets	81 %	2.3	38 %	3.4
Plastic folding boxes	50 %	10	15 %	15
Reusable bulk bags	85 %	1.2	75 %	1.2

Appropriate environmental indicators

The most important environmental performance indicators to monitor the implementation of this BEMP are:

- use of economic instruments at local level to stimulate good behaviour (y/n);
- share of residents/businesses using a voluntary economic instrument (%).

Cross-media effects

The risk of illegal dumping increases when applying economic instruments to MSW (van Beukering et al., 2009), but the associated costs of littering management seem to be much lower than the savings that economic instruments could bring. Waste authorities relatively isolated in the application of PAYT in their geographical area for example may have a *waste tourism* effect, i.e. disposing of waste to other neighbouring regions without similar charge systems.

Operational data

Implementation of an incentive- or unit-based pricing system at municipality level for municipal solid waste

Several steps can be defined in the implementation of a system that would allow the use of local economic instruments for waste separation in households:

- Produce a cost estimation that allows the waste authority to identify the priority areas of action and design how the new system would be integrated in the existing structure.
- Based on the results of the cost audit, set quantifiable objectives (Section 3.3.1), set cost benchmarks (Section 4.3.1), and establish a reliable waste accounting system (Section 4.3.2).
- Create a deposit fee response model that allows further optimisation.
- Enforce implementation by avoiding so-called waste crime.

The next subsections elaborate on each of the aforementioned steps, except for the second point, as it refers to other parts of the document.

Cost estimation

Although they are environmentally sound, local economic instruments are designed as a cost-saving measure. They directly affect the costs and budget management of waste authorities, so good bookkeeping practices are required. In order to establish a fee per kg of waste or the value of rewards, it is essential to identify the main revenues and costs of the system and disaggregate them per element of the service.

In addition, the accounting and auditing of public administration is regulated by each Member State under national regulations and legislation, but harmonisation through European standards is still poor (Brusca et al., 2015). Municipalities within a Member State are responsible for fulfilling these national requirements. However, cost allocation per municipal service or cost allocation per section of municipal service (as required for this BEMP) is far from being nationally homogeneous and usually requires cost audits performed by specialists and public or private consultants. The allocation method is then known to its practitioner, but may be confidential in cases where a private consultant audits municipalities (as the BEMP example case of Section 4.3.1 on cost benchmarking). Therefore, within this subsection, a short description of some general principles and guidelines will be given.

The general principles of cost estimation per municipal service are: direct, causal and allocation. The direct and causal methods are based on real outlays, i.e. the link between a service and yearly expenditure. So, in this way, the direct method would only include annual costs that are directly linked to the service (e.g. fuel spent by a truck), while the causal method is not linked to the service but to an activity, which may include more than one service.

Cost allocation, although less precise, is considered to be a better method of calculation, since it assigns a whole range of real costs to every service. The Network of Associations of Local Authorities of South East Europe (NALAS) recommended in 2009 the use of Full Cost Accounting (FCA) in order to estimate the real cost of public services in Europe (NALAS, 2009). This is a well-reported method used by the US Environmental Protection Agency (EPA) for waste services. Some of the principles used by FCA are detailed below:

- Cost is the monetary value of resources used or obligated for solid waste management, and outlays are the expenditure of cash to acquire those resources.
- Waste management is divided into the following management areas: collection, disposal (landfilling and waste-to-energy) and recovery (consumer products and packaging, and composting).
- Costs per area are:
    - upfront costs: public education and outreach, land acquisition, permitting, building construction and modification;
    - operating costs: normal costs (operation and maintenance, capital costs, debts), unexpected costs (usually as a percentage of normal costs);
    - back-end costs: site closure, decommissioning, post-closure care, retirement and benefits for employees;
    - remediation costs at closed sites: investigation, containment and clean-up of known releases, closure and post-closure care at inactive sites;
    - contingent costs: remediation costs (undiscovered and future releases), liability costs (property damage, accidents, etc.);
    - environmental costs: environmental degradation, use of waste of upstream resources, downstream impacts;
    - social costs: effect on property values, community image, aesthetic impacts, quality of life.
- Each municipality should define an appropriate set of each of these costs given their management practice and calculate indirect costs related to each category. The real cost of services contracted out should include what the consumers pay and not what the local government pays to the contractor. Volunteer costs also need to be included.
- Depreciation (of capital investment) and amortisation (of future outlays) should be included in the final cost estimation. Overhead costs in each of the category costs for the management, supervision, human resources, etc., of the service should be allocated a fair share from the local government expenses.
- The allocation method of shared costs between areas can be as follows:
    - Per budget (only for administration services): the allocation of a shared cost is calculated as the proportion of the total municipal

budget. This would allocate an administration cost to the whole waste management service with respect to other services.
  - Personnel share method: Similar to the budget method, but taking into account the number of people working in each service. This can be applied to waste management areas if the percentage of full-time equivalent for shared personnel is taken into account.
- Revenues are:
  - service revenues: as fee charges for the users of the system, both households and commercial businesses;
  - by-product revenues: from the sale of marketable products, as recyclables, compost, fuels or electricity;
  - tax revenues: income from taxes not directly linked to waste management;
  - transfer revenues, as subsidies or other funding received.

Regarding the above, a cost per tonne can be calculated per area of waste management (e.g. recycling, composting, disposal, collection). In order to calculate the potential reward or deposit fees, a behaviour response model would be required (see below) or, at least, a reasonable estimation of the performance of the system, taking into account that these schemes tend to be self-funded (as deposit refund systems and reward schemes) or tend to lower the costs of management (PAYT).

The text above is a full rationalisation of all the elements of a cost balance. Further simplification is always possible. A good example of such simplification is the award calculation made by Bracknell Forest Council, which is based on the expected savings from landfill fees (BF, 2012).

Behaviour response model

A key decision for any economic instrument is the fee to be charged per waste or the type and quantity of rewards in recycling incentive schemes. While all the systems should be designed under a self-funding principle, it is not easy to predict the increase in recycling that can be achieved, along with the amount of residual waste that will be reduced or the changes in costs derived from the impact of the system in transport and logistics. While the best starting point is to calculate the fee according to an expected frontrunner performance by following the principles stated in the 'cost estimation' section, a deeper study can be oriented to a behavioural model. As an example, the correlation between capture rate (or return rate) and deposit fee was modelled by Hogg et al. (2010) as:

$$\textit{Return rate} = 0.0529 \,\mathrm{Ln}(\textit{Deposit (EUR)}) + 0.725.$$

Kopytziok and Pinn (2011) performed a study on waste prevention and separation at markets and street festivals. In their experience a deposit has different effects, depending on the amount of the deposit. If a beer costs EUR 3.00 per cup and the deposit is EUR 0.10, the majority of cups are not returned, but are thrown into the rubbish. The provider makes a considerable profit from the non-reimbursed deposit and has no expenses related to sinks and returns logistics. Therefore, a low deposit for the caterer is lucrative as long as the cup costs are low. If, however, expensive hard plastic cups are used, a high deposit (EUR 1.00 to EUR 5.00) and returning of cups are attractive for the caterer. As the cups are returned, the loss is small. A profit can be achieved when the deposit is far above the cup price and the cup has a souvenir effect. Without requirements from the organiser or the local authority, the caterers generally tend to use simple cups and low deposits.

Case study on the implementation of a recycling incentive scheme

Bracknell Forest Council, in the south of England, manages the waste from a total population of 118 000 citizens through a contract with SITA UK. Given the low recycling rate, and of the increasing price of the landfill tax in the region (up to GBP 80 per tonne), the council decided to implement a pilot self-funded incentive scheme, for which they received funds from Defra (GBP 108 000). The implementation of the scheme followed these principles (BF, 2012):

- Objectives: The council decided to implement a system to save costs from the landfill tax. The system was implemented following advice from their waste contractor (note that in the UK waste cannot be charged through pay-as-you-throw schemes and a fixed fee is charged to citizens through the 'Council Tax'). It is considered that a potential saving of GBP 300 000 could be achieved only from avoidable landfill tax in three years. The key objectives were to increase the number of households participating in the kerbside recycling service from 75 % to 82 % in two years and to reduce the rate of recyclable materials in residual fractions from 13 % to at least 8 %.
- Scale of implementation: A first phase, as a pilot scheme, was successfully implemented and then extended to the whole town. Citizens can opt out and there is no mandate to be part of the reward system.
- Technology: Every citizen opting in is given an *e*+ card where points are accumulated. Blue bins are supplied at no cost for the final user. Points are given per pick-up of these bins, which are emptied if eligible by the personnel of the waste truck. No weight system is necessary and no fee reduction is offered in the management of the residual waste bin.
- Portfolio of rewards: No cashable value is given to the users of the system, but a maximum total value of GBP 26 in credits (points) per year. Rewards that can be redeemed with the points accumulated are seen as a marketing aspect of the scheme. Some of the rewards are as follows:
    - Council services rewards: The main rewards were offered as leisure rewards, e.g. as discounts or direct access to sports facilities, membership to local clubs, gyms, pools, etc.
    - Green rewards: These are designed to help the municipality to achieve further landfill reductions, while making them freely available if enough credit is accumulated on the *e*+ card. For instance, composters and water butts are offered.
    - Items: Although not used in the pilot scheme, some rewards include offers in local shops.

The implementation was considered successful by the council of Bracknell Forest (BF, 2015), as at least 11 000 households joined the scheme (a quarter of the total number of households). The amount of residual waste was reduced by 1 000 tonnes, representing a saving of GBP 90 000 (from 1 April 2013 till July 2014), achieving the objectives of the pilot trial; therefore, the system is now implemented at full scale. Feedback from the citizens was positive and many indirect benefits were achieved, such as the possibility of targeted awareness campaigns via the e-mail of system users, insights gained into waste management practices, and the construction of a new waste monitoring system. This also developed the required awareness for further waste reduction opportunities.

Enforcement

Enforcement consists of all the measures that can be organised by law, leading to discovery, deterral, rehabilitation and punishment. Enforcement is the last option that should be contemplated to raise the environmental awareness required for the performance of economic instruments (or any other best environmental management practice). These techniques are usually associated with a high risk of illegal disposal. Best practitioners should be recognised and rewarded by authorities; waste-regulationcompliant citizens should be engaged in the community to keep them fulfilling their obligations. Enabling and educating citizens should also be considered appropriate measures to reduce the extension of enforcement. In general, enforcement is outside the scope of this document, which covers best practices and frontrunner approaches at technical level (i.e. the document does not cover the remediation of bad or illegal practices). However, it is acknowledged that enforcement and, especially, a lack of it, plays a role in waste policies. Some examples can be found in the literature:

- SEPA and Zero Waste Scotland produced a set of guidelines for the enforcement of waste legislation for businesses and public contracts, with an extensive set of measures covering planning, designing, execution and assessment of public contracts (SEPA, 2015).
- Municipalities can establish for example a “Waste management enforcement policy”. For instance, Dudley in the UK established a policy to tackle problems associated with abandoned vehicles, untaxed motor vehicles, fly tipping, litter, dog fouling and accumulation of waste (Dudley, 2008). The policy remains open to new obligations or instruments derived from local legislation. Measures include visits, inspections, verbal and written advice on legal requirements and assistance with compliance, written warnings, penalty notices, prosecution, seizure and detention, etc. It also provides guidance to police officers for *informal* enforcement, where they need to be supportive of those willing to fix any non-compliant situation that they are not aware of.

Case study on deposit refund schemes: Cadaqués pilot test

As an example of the involvement of local authorities in the implementation of deposit schemes, the city of Cadaqués, Catalonia, implemented a pilot test to evaluate its effect on the municipal waste management system, from the environmental and economic point of view. The experiment was promoted by Retorna through the support of a number of agencies and waste managers in the region (Recuperadors de Catalunya, Internaco SA, Rhenus Logistics and Tomra SA). The exercise was supervised by the Catalonia waste agency. The effect on municipal waste management economics was relevant, reducing collection costs from 9.5 % to 6.5 %. However, a reduction in the income from recycling was detected, as well as a reduction in collection costs. Collected packagings were sold at prices 20 % to 40 % higher than usual due to the good quality of the waste streams. In addition, the cleanliness of public spaces in the city was evident (Retorna, 2013).

Other case studies in PAYT schemes

Box 3.1. Torelles de Llobregat (OECD, 2006)

This is documented as the first differential and variable-rate waste pricing system in Spain.

Implementation of the system

- Biowaste (food waste), collected three times per week (four in winter), no charge, 25-litre capacity bins supplied by the municipality.
- Paper and card collected once per week, no charge.
- Glass, no charge, bring scheme.
- Other packaging waste and residual waste, 40-litre bags (EUR 0.60 per bag) or 100-litre sacks (EUR 1.50 each), supplied by the municipality.
- Nappies, white sacks, no charge.
- Garden waste, EUR 0.40 per 50-litre sack, supplied by municipality, same collection as biowaste. Large branches excluded.
- Garden waste such as large branches, no charge, bring scheme.

Results

- Reduction of residual waste of 38 %.
- Increase of separately collected materials from 33 % to 89 %.
- Net private costs of EUR 11.58 per household (if avoiding landfill) or EUR -9 if avoiding other treatments, i.e. the system has a positive cost for the household if it is avoiding only low-cost landfilling.
- External benefits around EUR 11–20 per household (or EUR 8–10 if extra time spent by users is factored in), calculated from the avoidance of treatments. Increase in private transport and illegal disposal not included in the balance.

Box 3.2. Landkreis Schweinfurt (OECD, 2006)

Landkreis Schweinfurt (OECD, 2006)

Implementation of the system

- Fixed annual fee. This covers the costs of collection infrastructure, bulky waste collection, tyres, fridges and special waste. Around EUR 8 per month or EUR 16 per month per 240-litre bin.
- Emptying charge, calculated as EUR 0.20 per emptying.
- Weight-based fee. EUR 0.25 per kg for residual waste and EUR 0.15 for biowaste.

Results

- Total waste collected reduced by 28 %, and residual waste reduced by 46 %.
- Increase of separately collected materials from 64 % to 76 %.
- Net private costs of EUR -6 per household (i.e. cost reduction). The balance does not include a reduction or an increase in the deposit refund system.
- External benefits around EUR 8 per tonne (or EUR 14 per household).
- Increase in private transport and illegal disposal not included in the balance.

Box 3.3. Limerick, Clare and Kerry regions (R4R, 2014b)

Implementation of the system

- Customers are charged on residual wastes of average weights in the preceding six months, directly per kg of residual waste at collection, and/or per removal.

- A fixed fee, e.g. as an annual service charge, is also paid by the user.
- Recyclables, biowaste and glass are usually free of charge.
- The charge per kg is EUR 0.12–0.27.

Results

- Total waste per household was reduced in systems with charges per kg of waste.
- Recyclables collection was increased substantially in systems with charges per kg of waste.
- Illegal disposal of waste was detected; users opt out of the system due to high charges.
- Higher costs detected for smaller households.

B2B approaches

The implementation of deposit systems for several types of industrial packaging is usually performed in order to save costs and increase the efficiency of the logistics through reverse logistics, rather than improving the environmental performance, as private business would only apply such a measure if it is an opportunity for cost savings. The technical report on best environmental management practices in the building and construction sector (EC, 2012) identified pallets as one of the main reused packaging materials in the sector. Lundesjo (2011) reported on a pilot experience of Aggregates Industries, UK, on the implementation of reusable pallets. Although the motivation is essentially to reduce operational costs, the environmental savings are very relevant, compensating the production of new pallets after only two or three trips. At least 1 000 tonnes of wood are saved per year and 200 tonnes of CO_2e are avoided in one year.

The operational challenges of the implementation of a returnable system with industrial customers were the following:

- Two new types of pallets had to be purchased for the trial and redesigned in order to strengthen them with the objective of at least three trips before recycling or incinerating the waste pallet. The pallets were labelled as returnable and numbered in order to trace the results of the trial. After the first experience with local, small businesses, 40 % of pallets were returned.
- The experience was extended to large customers in order to achieve higher savings. B&Q (retailer) accepted to return the pallets from stores to the distribution centres by applying reverse logistics.
- The large-scale experience was applied to larger pallets that could not be stacked with other pallets and some resizing was required. This generated other problems, as the pallets were larger than the product size, therefore reducing the space efficiency during its transport.

Applicability

The regulatory framework and its enforcement are the main barriers for the application of some local economic instruments described in this section. Some countries, such as the UK or Greece, do not allow (or do not *facilitate*) the implementation of variable waste collection rates based on generated waste per household. For those countries, positive incentives are considered to be the best option.

In addition, the existence of environmental awareness, good management skills and innovation-driven behaviour at the local government level, with some good accounting practices, are prerequisites for the implementation of local economic instruments, which are complex to manage from the technical, managerial and social perspectives.

Economics

A study from the OECD for pay-as-you-throw, and a Defra study on recycling incentive schemes showed that, in general terms, the social benefit of local economic instruments in the monitored case studies is positive and justify their implementation. However, the studies point out that when the cost of treatment is low (e.g. cheap landfilling), the waste management system running costs are higher than for conventional waste management (see case studies described in Operational data).

Costs of implementation of pilot recycling incentive schemes in the UK

The study from Defra (2013) was performed on several case studies. Table 1-9 shows the costs of the different systems. Bracknell Forest, shown in Operational data, was one of the funded municipalities but not included in the first reported assessment by Defra. Conclusions from the study and the cost efficiency of the system are to be published by Defra. The costs shown in Table 1-9 do not include revenues from produced secondary materials; the balance has yet to be assessed and studied. The county of Norfolk and the Bristol students' association case studies refer to reuse shops that also produce recyclable materials.

Table 3-9. Disclosure of costs for Defra's pilot recycling scheme case studies in the UK (Defra, 2013)

	Cost breakdown													
Municipality	**Capital cost**	**Opportunity cost**	**Staff costs**	**Rewards**	**Communication**	**Monitoring and evaluation costs**	**In-kind contributions**	**Volunteers**	**Participants**	**Households**	**Total cost (GBP)**	**Cost per participant or household (GBP)**	**Potential cost per participant or household (GBP)**
Bradford, Aire Valley Recycling, UK	0%	0%	57%	8%	14%	12%	2%	5%	-	637	33 144.00	52.03	20.06
Bath and North Somerset, UK	15%	11%	25%	10%	5%	31%	3%	0%	-	3 866	104 116.00	26.93	20.49
Birmingham, UK	24%	0%	23%	6%	8%	38%	0%	0%	-	3 426	63 500.00	18.53	14.46
Gloucestershire, UK	2%	10%	17%	2%	11%	58%	0%	0%	-	7 008	60 343.00	8.61	5.96
Norfolk County, UK	0%	12%	5%	48%	33%	2%	0%	0%	258	-	27 371.00	106.09	
Student association in Bristol, UK	0%	7%	56%	6%	2%	28%	1%	0%	2 710	-	65 338.00	24.11	5.76
Preen Community in Bedfordshire, UK	0%	0%	21%	21%	55%	0%	3%	0%	7 505	-	61 240.00	8.16	5.83

N.B. Opportunity costs are those staff costs involved in the programme but not on a full-time basis. In-kind contributions include also stakeholders' contributions and volunteers unless disclosed in the volunteers column.

Final results and cost efficiency of the scheme yet to be published.

Driving force for implementation

Cost saving is a main driving force of economic instruments, along with the improvement of performance of waste management systems and the derived environmental benefits. The amount of waste is not reduced through these economic instruments, so waste prevention cannot be considered a driver of implementation, except for those B2B schemes and deposit refund systems applied in the industry. Recycling incentive schemes are also very popular among citizens and tend to give an environmental reputation to the local government.

Reference organisations

Supra-municipal organisations:

- Defra, on the study of the performance of recycling incentives schemes.
- LIPOR, on the application of recycling incentive schemes.
- ACR+, on the study of economic instruments.
- WRAP, on the application of B2B schemes.

Municipalities applying an economic instrument:

- Recycling incentive schemes:
    - Rewards: Bracknell Forest (UK), Valongo and Gondomar (PT).
    - 'Cash for Trash': Rotterdam, Barendrecht, Krimpen aan den Ijssel (NL).
    - Reduction of waste tax fee to residents source separating waste: villages in Mallorca.
- Deposit refund schemes at events:
    - Directly applied: Graz (AT).
    - Locally regulated: Schwäbisch Gmünd, Nuremberg, Reinheim (DE).

B2B approaches:

- BEMP: London Construction Consolidation Centre (UK).

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3.3.4. Link to other relevant reference documents for best practices

Summary overview							
It is BEMP to implement state-of-the-art techniques that maximise resource efficiency and minimise environmental impact in the areas of waste treatment (including material recycling, energy recovery and waste disposal). Useful reference documents (non-exhaustive list) on relevant state-of-the-art techniques that organisations can refer to are: - Reference Document on Best Available Techniques for Waste Treatment; - End-of-waste criteria; - Reference Document on Best Available Techniques for Waste Incineration; - EU Landfill Directive (99/31/EC).							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
This BEMP is targeted to local waste authorities and waste management companies planning and carrying out operations in the areas of waste treatment, material recycling, energy recovery and waste disposal.							
Specific environmental performance indicators							
- Relevant state-of-the-art techniques described in the reference documents listed in this BEMP are implemented (y/n).							

In their integrated waste management strategy (see Section 3.3.1 for guidance on its development), local waste authorities and waste management companies define a coherent set of actions to implement over the whole waste management cycle, including the final waste treatment steps (e.g. material recycling, energy recovery and waste disposal). These areas, which are not directly covered by BEMPs in this report (see Section 1.2.2 and Figure 1-18), also have a very large resource efficiency and environmental improvement potential. When planning and carrying out operations in these areas, it is BEMP for local waste authorities and waste management companies to implement state-of-the-art techniques that maximise resource efficiency and minimise environmental impact. To this aim, organisations can refer to the following (non-exhaustive) list of reference documents, which are useful sources of information about relevant techniques:

- Reference Document on Best Available Techniques for Waste Treatment[33];

[33] For more information on the content of the Best Available Techniques Reference Documents and a full explanation of terms and acronyms, refer to the European Integrated Pollution Prevention and Control Bureau website: http://eippcb.jrc.ec.europa.eu/

- End-of-waste criteria[34];
- Reference Document on Best Available Techniques for Waste Incineration;
- EU Landfill Directive (99/31/EC)[35].

The appropriate environmental performance indicator for this BEMP is:

- relevant state-of-the-art techniques described in the reference documents listed in this BEMP are implemented (Y/N).

[34] End-of-waste criteria were introduced by Article 6 of the Waste Framework Directive of December 2008 (2008/98/EC). More information is available at: http://ec.europa.eu/environment/waste/framework/end_of_waste.htm

[35] For more information on the content on the landfill directive and access to the full text, refer to the following website: http://ec.europa.eu/environment/waste/landfill_index.htm

4. Municipal solid waste (MSW)

4.1. Introduction

This chapter contains the best practice in relation to management of municipal solid waste (MSW). MSW is generated primarily by households, and also by commercial enterprises, and includes a wide range of fractions including organic materials, plastics, paper, glass and metals. In 2012, each EU citizen generated 492 kg MSW on average (Eurostat, 2014), of which only 40 % was recycled, with the rest being landfilled (37 %) or incinerated (23 %). EEA (2013) concludes that the majority of Member States will have to make unprecedented progress in increasing recycling rates in order to meet the Waste Framework Directive's target for 50 % of MSW to be recycled by 2020.

According to Eurostat (2014), 3 % of EU GHG emissions are directly attributable to waste management activities. However, MSW disposal represents the loss of products with high embodied GHG emissions and other environmental burdens associated with raw material extraction, processing, manufacture and transport. Consequently, disposal of the MSW fraction is associated with high indirect environmental burdens. As highlighted in Chapter 1 with respect to embodied GHG emissions, approximately 1.8 tonnes of CO_2e are embodied in the MSW generated by an average EU citizen over one year. At the EU-28 level, this represents over 890 Mt CO_2e/year of indirect GHG emissions, suggesting that waste management is actually associated with over 20 % of EU GHG emission. Food waste, textiles and nappies/sanitary products make the largest contributions to GHG emissions, followed by plastics.

4.2. Technique portfolio

This chapter will sequentially address a range of best practices to manage MSW, starting with the formulation of an overarching waste strategy in Section 4.3, and finishing by dealing with waste treatment in Section 4.7.

Section 4.3 provides waste authorities and waste management companies with an overview of best practice measures and indicators related to the development of waste management strategies that systematically and comprehensively deliver the best environmental outcomes.

Section 4.4 describes best practice techniques for waste authorities and waste management companies to drive waste prevention through local waste prevention programmes, schemes for product reuse and for the preparation for reuse of waste.

Section 4.5 covers waste collection and all its different aspects which are key for a well-implemented and -performing waste management system.

Section 0 presents the best practice for producer responsibility organisations (PROs) to enhance the performance of the extended producer responsibility (EPR) schemes.

Section 4.7 addresses waste treatment options that are not described in other best practice documentations, in particular IED BREFs.

Reference literature

EEA (2013). Managing municipal solid waste — a review of achievements in 32 European countries. EEA, Copenhagen.

Eurostat (2014). Statistics database. Accessed December 2014. Available at: http://ec.europa.eu/eurostat.

4.3. Strategy BEMPs

4.3.1. Cost benchmarking

Summary overview							
Choices related to waste management are greatly affected by economic factors; carrying out cost benchmarking by comparing the cost structure of a municipality with data of other municipalities is BEMP as it allows the identification of optimisation options which may open the door to more environmentally friendly practices. Cost benchmarking can be carried out internally, by an independent third party or in cooperation with other municipalities. Cost figures analysed typically include costs for waste management services and for the disposal of certain waste fractions as well as revenues gained from the sale of waste that is sent to preparation for reuse or recycling and other by-products. All relevant waste fractions generated within the territory considered and belonging to MSW need to be taken into account in the cost benchmarking. Comprehensive analyses include costs for waste collection, waste treatment (sorting, recovery, disposal, etc.) including the management of closed landfills, staff costs and all other waste-management-related costs.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
Cost benchmarking can be applied within an area (at local or national level) where waste management conditions are comparable and where there is a uniform legal framework. However, in some cases, strong deviations occur due to specific conditions. Cost benchmarking is particularly relevant for areas with poorly performing waste management systems, in order to support the shift to better performing waste management options.							
Specific environmental performance indicators							
- Regular participation in a detailed cost benchmarking study (y/n). - Total MSW management cost per resident[36] per year (EUR/capita/year).							

Description

[36] In areas where the presence of non-resident population (e.g. tourists, commuters) is relevant all over the year or during specific seasons, the number of residents can be adjusted and the number of population equivalent calculated, as presented in section 2.4. The same considerations on resident and non-resident population applies to all the relevant environmental performance indicators reported in the following sections of this report (i.e. 0, 4.4.1 and 4.4.2).

Waste management is greatly affected by economic factors; therefore, it is very helpful to carry out cost benchmarking in order to reflect the cost structure of a certain municipality (city, village or county) and to eventually identify optimisation options.

Cost benchmarking can be carried out by an independent third-party organisation, or internally by a local public administration of a considerable size, or in cooperation with other municipalities. Cost figures analysed can include costs for waste management services and for the disposal of certain waste fractions as well as revenues gained from the sale of waste that is sent to preparation for reuse or recycling and other by-products. All relevant waste fractions generated within the territory considered and belonging to MSW (paper/cardboard, glass, plastics, biowaste, green cuttings, scrap metal, non-ferrous metals, residual waste from households etc.) must be taken into account in the cost benchmarking study.

In more detail, in the evaluation of total costs, the following costs are usually considered:

- costs for collecting the different waste fractions (e.g. residual waste, biowaste, paper);
- costs for the treatment/disposal of residual waste (e.g. incineration) and recycling/energy recovery of waste fractions with distinction between municipality-owned plants and third-party plants;
- costs for operation, closure and management of closed landfills (leachate treatment, recultivation, etc.);
- costs for staff and administration related to waste management;
- miscellaneous costs.

In addition, the total costs can also include costs for services provided:

- by private waste management companies on behalf of the municipality;
- by the municipality itself;
- by municipalities providing services for another municipality.

In the evaluation of revenues from recycling/recovery activities, the following ones can be considered:

- selling electricity or/and heat from incineration of refuse-derived fuels, residual waste, biogas from anaerobic digestion of biowaste or landfill gas;
- selling biogas from anaerobic digestion;
- selling separately collected or separated paper/board;
- selling separately collected packaging;
- selling separately collected glass;
- selling separately collected or separated scrap metal;
- selling compost;
- fees charged to businesses for waste collection and disposal.

The difference between the total costs and the revenues is called “uncovered costs” and they are usually paid by the annual waste fee charged to the citizens of the municipality.

Once the cost benchmarking study is completed, analyses on the data could support the identification of improvement options in waste management processes (e.g. collection of the different fractions) or in the waste strategy (e.g. type of fractions collected) implemented at local level.

Cost benchmarking can also be used to compare the costs of waste prevention measures with the cost savings due to the decreased amount of waste to be managed.

Figure 4-1 shows an example for the evaluation of the main cost categories for 33 counties and 11 cities in Germany (ia GmbH, 2015).

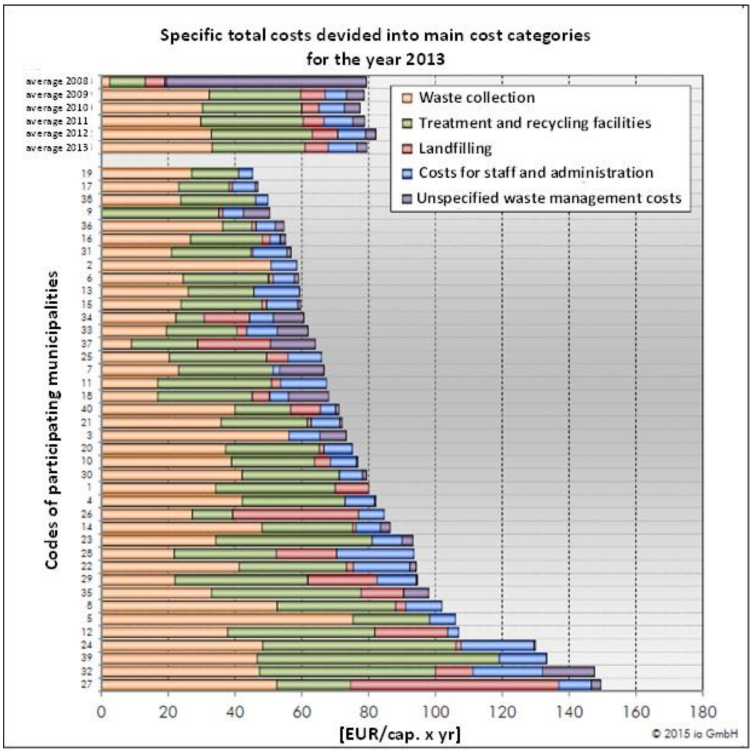


Figure 4-1. Specific waste management costs for the main cost categories for 2013 of 33 counties and 11 cities in Germany providing waste management services to 6.3 million citizens in total, based on ia GmbH (2015)

The corresponding annual waste quantities per capita are illustrated in Figure 4-2.

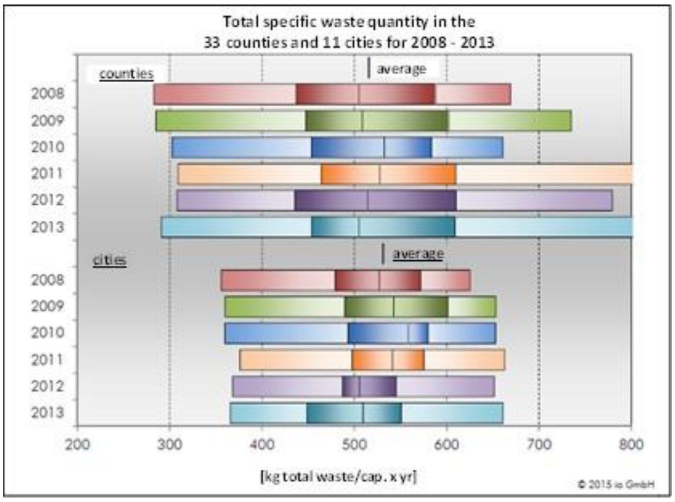
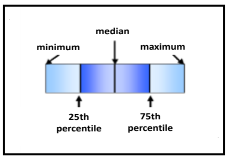


Figure 4-2. Total specific waste quantities of the participating 33 counties and 11 cities in Germany from 2008 to 2013, based on ia GmbH (2015)[37]

37 

The values are presented as median, minimum, maximum and 25th/75th percentiles as indicated in the figure above.

Achieved environmental benefits

Cost benchmarking is not directly associated with an improved environmental performance. However, it can contribute to an optimisation of services such as the collection of the different waste fractions. In this respect, it can encourage municipalities to increase the number of waste fractions that are collected separately as the figures demonstrate that advanced collection systems do not necessarily lead to significantly higher costs (Figure 4-3).

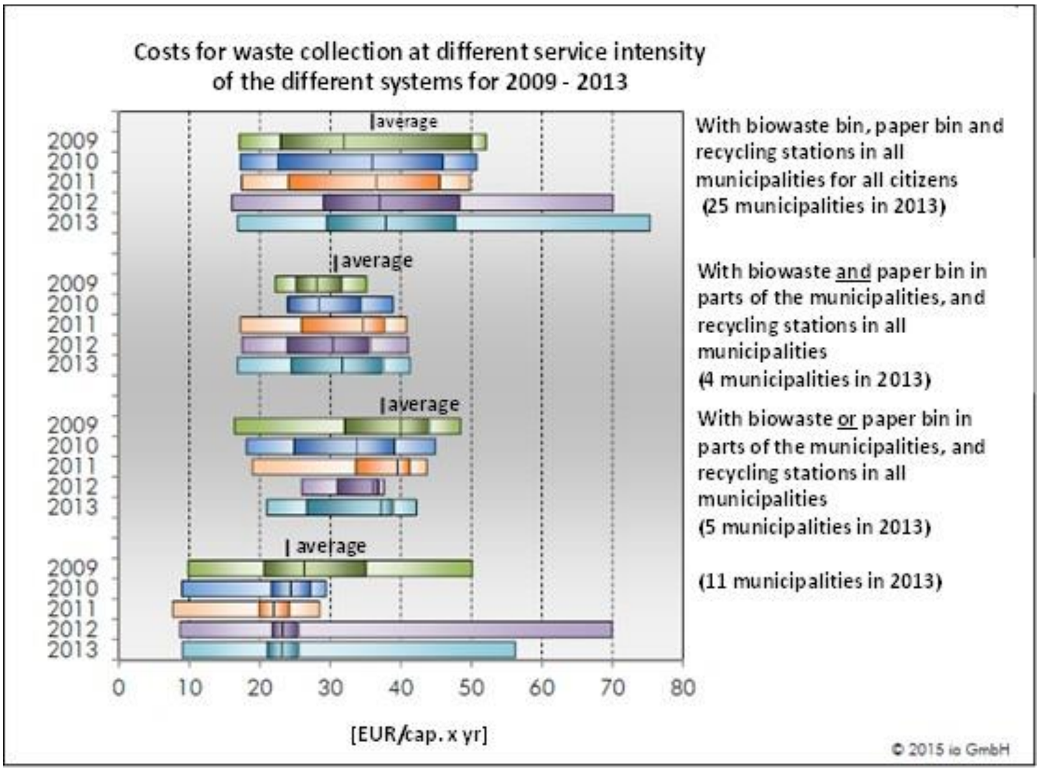
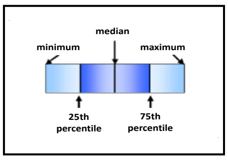


Figure 4-3. Costs for waste collection at different service intensities of the different systems for 2009–2013, based on ia GmbH (2015)[38]

Appropriate environmental indicators

The most appropriate indicators to monitor the implementation of this BEMP are as follows:

- Regular participation by the local authority in a detailed cost benchmarking study (y/n).
- Total MSW management cost per inhabitant per year (EUR/capita/year). For this indicator, it is important to keep in mind that for comparability over different

38 

The values are presented as median, minimum, maximum and 25th/75th percentiles as indicated in the figure above.

years, waste management costs need to be homogeneous, including therefore the same services and activities. Moreover, when the available data allow, the cost of MSW management can be disaggregated for the main different waste streams, in order to identify how costs are allocated to specific waste streams.

Cross-media effects

There are no cross-media effects as the technique is not associated with any significant energy or material consumption or emissions.

Operational data

Any municipality, city or region can participate in the cost benchmarking exercise by using and providing data to other organisations that participate in or carry out the benchmarking study. The more organisations that take part, the more reliable their assessment.

A specific case of a cost benchmarking exercise has been carried out by a network of municipalities and local authorities in Germany, called ForumZ, which promotes the inter-municipal cooperation in the field of waste management (www.forumz.de).

In order to collect data from the different municipalities included in the network, a questionnaire for data collection was developed by a working group comprising waste management experts from the different municipalities (counties and cities). Not only technical information is required to optimise waste management but also systematic and robust data on costs. The questionnaire was developed in a practice-oriented way in order to create helpful benchmarks.

As the cost benchmarking in ForumZ has been carried out six times so far (status: April 2015), increases and decreases in costs can be indicated as illustrated in Figure 4-4.

While developing the questionnaire, the working group decided that, based on the annual data collection and responses from the participating municipalities, the questionnaire may be (slightly) adapted year to year.

In the case of ForumZ, the data collection also comprises information on whether the services are carried out by private waste management companies on behalf of the municipality, by the municipality itself, or by municipalities providing services for another municipality. The collection of these data allowed ForumZ to also investigate whether the uncovered costs depend on the percentage of private services. Figure 4-5 shows that uncovered costs do not depend on the percentage of private services carrying out waste management.

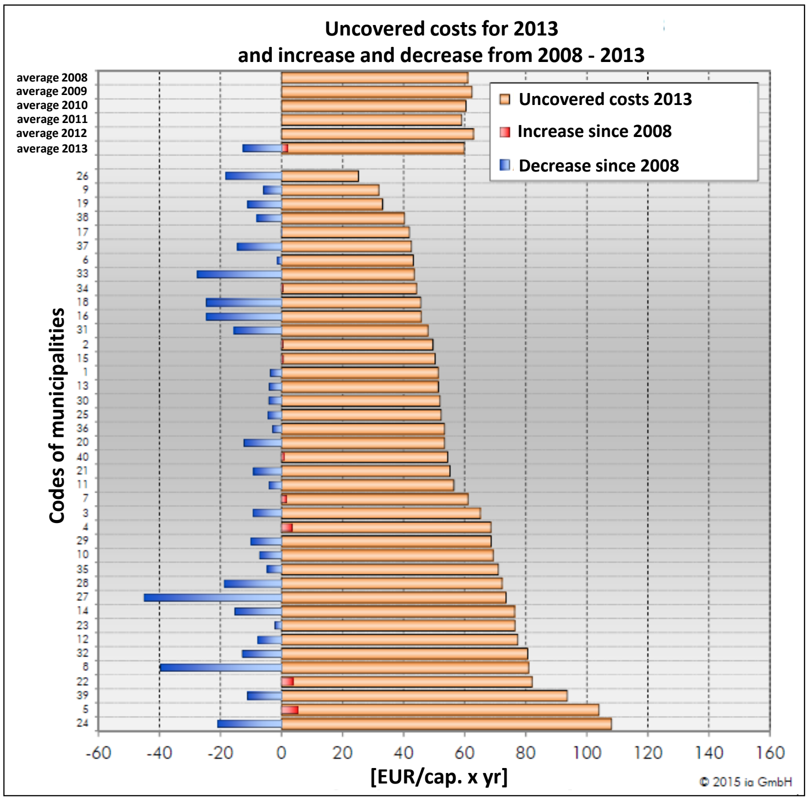


Figure 4-4. Increases and decreases in uncovered costs in 33 counties and 11 cities in Germany from 2008 to 2013, based on ia GmbH (2015)

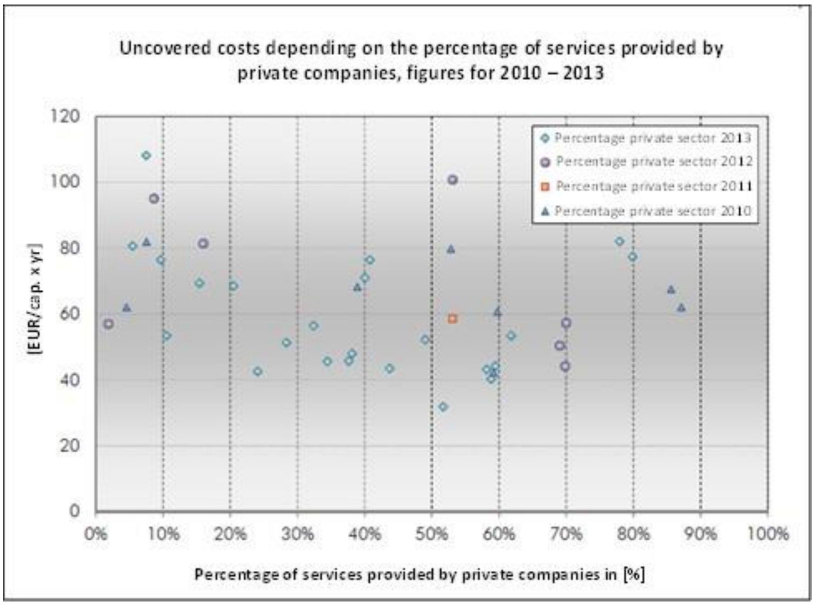


Figure 4-5. Uncovered costs and percentage of services provided by private companies in 33 counties and 11 cities in Germany in 2010–2013, based on ia GmbH (2015)

Applicability

Cost benchmarking can be applied in a county/region or at the national level, where waste management conditions are comparable and where there is a uniform legal framework. In order to carry out a cost benchmarking of waste management, the public administration or the waste management company need to have a full and detailed view/control of all operations and mass flows involved. Concerning comparability of cost figures, there may be individual cases where strong deviations occur due to specific conditions. For instance, for municipalities with a high number of tourists the cost figures in [EUR/capita per year] are significantly different; as a consequence, in this case, a cost indicator [EUR/t total waste] may be more appropriate. Cost benchmarking could be very useful when assessing existing poorly performing waste management systems in order to support the shift to more efficient ones.

A municipality or a county joining a cost benchmarking system should be able to produce cost estimations based on its accounts. For those, full cost accounting is preferred against yearly outlay balances, and an appropriate allocation procedure should be applied. A detailed description of cost estimation and allocation procedures is included in Section 3.3.3.

Economics

Municipalities taking part in the cost benchmarking exercise performed by the independent third-party organisation ForumZ (presented in the Operational data section) pay an annual fee to ForumZ which organises the collection and evaluation of cost data. This fee is in the range of EUR 1 000 and EUR 4 000 per year, depending on the size of the municipality.

According to Figure 4-1, waste management costs of different cities, counties or municipalities vary by up to a factor of 3. For individual services, the range can be bigger, e.g. up to a factor of 8 for waste collection. For instance, in 2013, the cost for waste collection with biowaste bins, paper bins and recycling stations in all municipalities for all citizens varied between EUR 17 and EUR 76 per capita per year. If the costs for the waste management of a region, e.g. a county, with 200 000 citizens at the upper end of the range can be reduced by only EUR 5 EUR per capita per year thanks to cost benchmarking and the improvement of the waste management system, the total cost savings in that region could reach EUR 1 million per year. This can be achieved by cost benchmarking for which the expenditure as a network member is EUR 0.02–3 per capita per year.

Driving force for implementation

The improvement of the waste management system and the consequential potential cost reduction for waste management is the main driving force for implementing cost benchmarking.

Reference organisations

ForumZ, a network including a number of municipalities and counties in Germany, is so far the only one which has been carrying out cost benchmarking for several years (2008–2013). The latest report for the figures of 2013 is dated March 2015.

In Germany, the Association of Municipal Waste Management and City Cleaning (VKS) as part of the Association of Municipal Enterprises (VKU) is also carrying out benchmarking studies both for technical and cost aspects, but not as regularly and specifically as ForumZ. However, so far a benchmark exercise has been carried out nine times (VKS, 2015); thus the development can be visualised and used for optimisation strategies. In the last rounds, about 70 counties, cities and municipalities took part. The data are processed and evaluated by third parties (Dornbusch, 2015).

The French Agency for the Environment and Energy Management (ADEME) has developed a cost matrix, which is available for local authorities and allows cost benchmarking

(ADEME, 2015). Also, the Paris Region Waste Observatory (ORDIF) is applying cost benchmark tools (ORDIF, 2015).

Reference literature

ADEME (2015) information on the concept of cost benchmarking is available on the ADEME website: http://www.ademe.fr/collectivites-secteur-public/integrer-lenvironnement-domaines-dintervention/dechets/maitriser-couts-ajuster-financement/dossier/connaitre-couts/outils-gestion-dechets-matrice-couts-methode-comptacoutr, last access June 2017.

Dornbusch, H.-J. (2015). Benchmarking und Erfahrungsaustausche für die Abfallwirtschaft – aus der Praxis für die Praxis (Benchmarking and exchange of experiences – from practice to practice. Presentation at the VKS/VKU-Landesgruppenfachtagung "Leinen los!" in Hamburg in October 2015, http://www.iswabeacon.obladen.de/images/presentations/Dornbusch.pdf, last access September 2017.

ia GmbH (2015). Abfallwirtschaftliche Gesamtkosten (total costs for waste management). Report on cost benchmarking for the waste management of 33 counties, 12 cities and one community in Germany for the year 2013 (in German – unpublished). ia GmbH is a small engineering company with about six employees which already started to systematically collect and evaluate data on waste management at municipality level in 1996 (see more information on ia GmbH on www.ia-gmbh.de).

Paris Region Waste Observatory (ORDIF) (2015). Connaître, analyser, et comparer ses coûts de gestion de déchets. March 2015.

VKS im VKU (Association of Municipal Waste Management and City Cleaning (VKS) as part of the Association of Municipal Enterprises (VKU)) (2015). Das Benchmarking-Projekt (The Benchmarking Project). http://www.vksimvku-benchmarking.de/das_projekt.php?thema=projekt, last access September 2017.

4.3.2. Advanced waste monitoring

Summary overview							
The development and implementation of an efficient and effective waste management strategy is based on detailed knowledge of statistical data for the waste streams collected and managed at local level. It is thus BEMP to: - regularly collect and process available data at single waste stream level, and for the different steps of the collection, reuse/preparation for reuse, sorting, recycling, recovery and disposal processes; - regularly carry out a composition analysis of the mixed waste; - when waste management operations are contracted out, include contract clauses for the systematic communication of comprehensive data. Waste monitoring data are useful both for internal analysis (such as evaluating the potential implementation of a new measure) and for sharing with the relevant public administration and citizens to drive improvement and awareness.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
Detailed waste monitoring is applicable to all local authorities and waste management companies managing municipal solid waste. For organisations starting the process, waste monitoring may focus first on the most relevant waste fractions and eventually be extended to all fractions step by step.							
Specific environmental performance indicators							
In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: - use of web-based tools for tracking and reporting waste data (y/n); - frequency of composition analysis of mixed waste (one composition analysis every # months or years).							
Benchmark of excellence							
- Composition analysis of mixed waste is carried out at least four times a year (during different seasons) every three years or after any substantial change of the waste management system.							

Description

An efficient and effective waste management strategy is based on detailed knowledge of statistical data for the waste streams collected at local level and treated. Data collection and management can be carried out in detail: initially defining which information should be collected and then keeping a good and updated database, which allows the extraction and processing of the required information in order to implement a number of analyses on the management of waste. A detailed example of advanced waste monitoring, which phases to analyse (i.e. waste generation, collection, sorting, recycling, recovery and disposal) and the most suitable environmental performance indicators (e.g. capture rate, impurity rate), which can be calculated from the data collected, is provided in Chapter 2.

A key aspect of improved waste monitoring is the ability to track information along the entire value chain of the collected waste, not only for the operations managed in-house (e.g. collection) but also on the fate of waste afterwards, when it may be managed by external companies and contractors (e.g. waste sorting and recycling). In such cases, it is important to include in the contractual agreement with the external organisation the provision to regularly communicate relevant data on waste management operations (e.g. sorting, recycling, energy recovery and disposal) as, for example, mentioned in different cases in Section 2.4.1.

Thanks to advanced monitoring of waste operations directly managed by the local authority/waste management company or outsourced to other organisations, local authorities are able to track waste streams throughout their presence in the waste management system and even further (e.g. when used as recycled or reused materials and items). Web-based tools can be adopted for tracking and reporting waste data and for ensuring the easy access of the local authority or residents to all data on waste management.

Waste management systems are complex and their monitoring is an activity which requires human resources and a full understanding of the system. As a simplified example, Figure 4-6 illustrates the waste streams derived from households and household-type commercial waste.

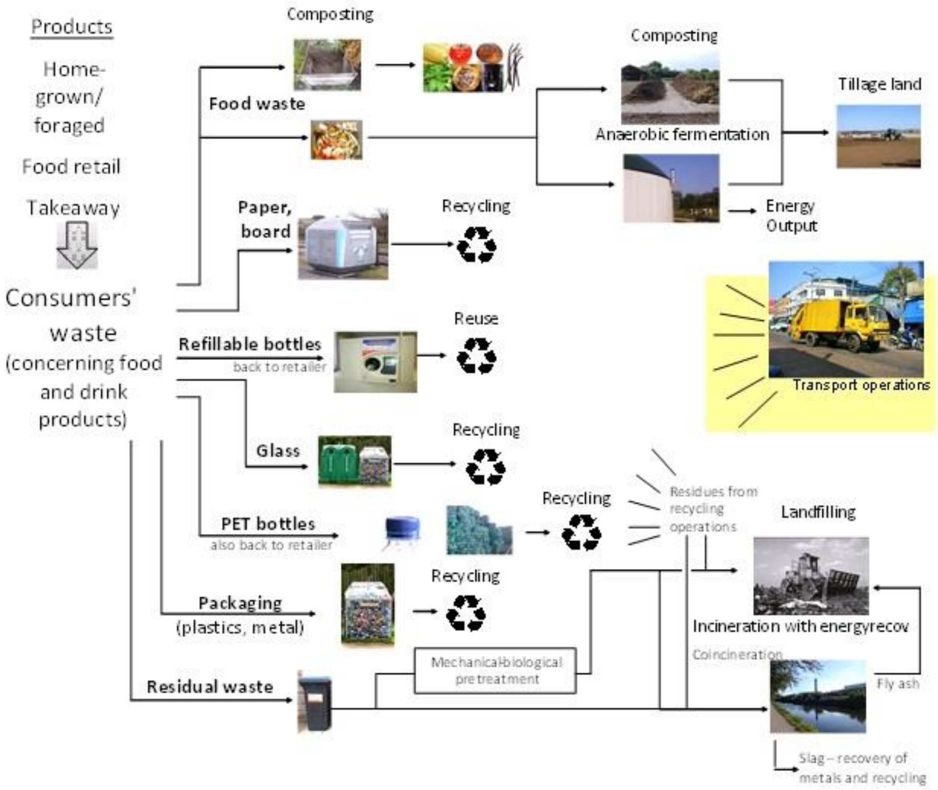


Figure 4-6. Important waste streams concerning municipal waste

Detailed waste monitoring requires regular analysis of the composition of mixed waste: as explained in detail in Section 2.4, this activity needs to be carried out in detail, (i) selecting a representative waste sample and (ii) in different periods of the year (i.e. to reflect seasonal changes). Knowledge of the composition of mixed waste then drives the improvements to the waste management strategy to further increase the capacity of the system to recycle and recover materials.

An important aspect of improved waste monitoring is the timely availability of data, which allows monitoring of the waste management system based on updated information. Data collected should be available for processing and analyses in a few weeks and the monitoring system should be continuously improved not only for the quality and amount of data collected but also for the time needed to obtain data to be processed.

Data collected and analysed can be used for internal purposes (e.g. evaluating the potential implementation of a new waste management measure, driving improvement of the waste management system) and for providing the required transparency to citizens. In fact, an annual waste management report can be published, providing an overview of the operation of the existing facilities and of the quantities of all collected, processed and recycled waste streams.

Additionally, in the coming years, advanced waste monitoring and web-based tools will be able to gather data on the waste streams collected at single household level and share them with citizens (called know-as-you-throw). The information could be used, apart from for defining the variable part of the PAYT (if present) tariff, to inform residents of their specific waste generation, increasing environmental awareness, promoting waste prevention and helping them in improving separate collection at source.

Achieved environmental benefits

Improved waste monitoring does not lead directly to any environmental benefit. However, detailed knowledge of the quantities and quality of the waste streams collected and treated can lead to a better waste management system with a consequent improved environmental performance (e.g. higher recycling rates). In fact, on the basis of exact quantities of the different waste streams, the efficiency of measures adopted in the waste management system can be determined and optimised, e.g. the management capacity of treatment plants can be improved, the collection of the different waste fractions can be optimised and a more accurate post calculation of fees can be achieved.

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- use of web-based tools for tracking and reporting waste data (y/n);
- frequency of composition analysis of mixed waste (one composition analysis every # months or years).

Cross-media effects

Due to the improved waste monitoring, there are no known significant environmental cross-media effects.

Operational data

Table 4-1 shows an excellent example of monitoring of the quantity of the different waste fractions collected in a German county (Aschaffenburg) from 1989 to 2013 (Aschaffenburg, 2013). A total of 17 recyclable streams and 4 unrecyclable, bulky, hazardous and commercial waste streams have been systematically recorded over the last 20 years.

Table 4-1. Example for the determination and documentation of the quantity of the different waste fractions of a county (county of Aschaffenburg in Germany) from 1989 to 2013, in kg/capita per year, (Aschaffenburg, 2014)

kg/cap x yr	Recycables																	Residual, bulky, hazardous and commercial waste				Total waste
	Waste glass	Waste paper	Scrap metal	waste tyres	Waste plastic	Textiles	Shoes	Green cuttings	Bio-waste	Waste wood	Windows/ flat glass	Alu-minium	Waste cable	Cork	Demo-lition waste	WEEE	Other recycables	Residual waste	Bulky waste	Hazardous waste	Commercial (household-type) waste	
1989																		191,4	25,7		444,2	
1995																		134	26,5	1,48	66,9	
1996	34	82,2	21,2		13,3	5,4	0,1	79,4	2,5	19,1	0,4	0,4	0,1	0	27,4							
1997	32,9	89,2	21,9		16,7	3,2	0,1	81,4	25	25,4	0,6	0,7	0,1	0,1	40,5			68,2	27,4	1,56	27,5	462,5
1998	33,5	97,7	21,9		16,4	3,2	0,1	62,5	24,2	30,5	1,1	0,9	0,1	0,1	42,4			44,6	35,5	1,4	20,9	437,0
1999	32,6	96,8	17,1		19,9	2	0,1	59,1	24,4	17,3	1,4	0,9	0,1	0,1	50,4			47,7	1,8	1,08	14,3	387,1
2000	32,1	100,8	19,7		21,4	2,3	0,1	74	24,2	20,2	2,6	0,1		0,1	44,7			48,8	2,7	0,56	10	404,4
2001	30,8	99,6	20,2		22,1	3,2	0,1	79,8	23,8	22,5	2,3	0,1		0,1	46,8			47,6	1,3	0,87	9,6	410,8
2002	29,2	98,7	20,4		23,3	3,1	0,1	81,2	23,5	23	2,5	0,1		0,1	54,1			47,1	0,8	0,58	8,7	416,5
2003	27	94,8	19,1		22	3,5	0,1	83,3	23,7	23,2	2,6	0,1		0,1	50,8			46,1	0,7	0,74	8	405,8
2004	24,8	84,1	15,4		22,1	3,8	0,1	85,3	25,7	22,9	2,6	0,1		0,1	51,1		0,2	47,9	0,7	0,83	6,9	394,6
2005	28,6	89,2	14,2		22,2	4,8	0,1	82,6	25,9	24,1	1,8	0,6	0,1	0,1	49,5		0,3	48,3	0,9	0,77	9,1	403,2
2006	29,4	92,6	13,5		22	6,6	0,1	83,9	26,7	23,1	5,9	0,6	0,2	0,1	49,5	5,5	0,2	49,3	1,1	0,86	11,2	422,4
2007	28,4	94,4	11,3		23,9	5,5	0,1	76,2	26,5	25,4	6,6	0,1	0,2	0,1	47,2	4,9	0,1	50,1	1,2	0,77	7,9	410,9
2008	26,3	94,3	12,2		25	2,9	0,1	72,1	27,1	26,2	7,6	0,1	0,1	0,1	47	5,7	0,1	50,4	1,5	0,83	8,3	407,9
2009	18,4	92,5	14,2		26,2	3,7	0,1	51,1	27,5	27,6	8,2	0,1	0,2	0	49,7	6,1	0,1	51,9	1,5	0,94	9	389,0
2010	27,3	91,5	12,8		26,5	5,6	0,1	90,2	28,1	28,5	8,5	0,1	0,2	0,1	50,2	5,7	0,1	51,7	1,7	1	9,7	439,6
2011	27,1	92,4	12,1		27,3	6,1	0,1	94,4	29,1	30,2	9,4	0,1	0,1	0	55,7	5,5	0,1	52,8	1,5	1	11	456,0
2012	27,2	91,6	11,2	0,2	24,2	5,4	0,1	97,5	29	29,8	9,8	0,1	0,1	0	52,5	5,6	0,2	52,3	1,6	0,99	9,1	448,5
2013	27,1	90,4	11,3	0,1	26	7	0,2	130,3	29,7	29,9	10,1	0,1	0,1	0,1	53,2	5,6	0,2	52,9	1,8	0,94	10,7	487,7

Such detailed waste monitoring reveals the drastic change in the waste management system of the county in the last 20 years. The quantity of residual waste decreased considerably and the quantities of recyclables sharply increased. The county introduced a weight-based pay-as-you-throw system for residual waste, biowaste and paper/cardboard. At the same time, the waste management infrastructure was significantly improved in order to drastically increase the recycling rates. Thus, today the percentage of recyclables is more than 85 % and the specific quantity of mixed household waste is about 50 kg/capita per year. These analyses and the successfulness of the implemented waste management system would not have been recognised and improved without such detailed waste monitoring.

With respect to evaluation of data, specific circumstances may have to be taken into account, such as the influence of tourism, and the collection of paper and cardboard by third-party organisations such as clubs of a municipality, etc.

In connection with the PAYT BEMP (see BEMP 4.3.3), it is easily possible to monitor which citizens have individual bins and which use common bins. Then it can be investigated where the collection and capture rates can be optimised most. The same is true for the collection frequency for citizens' waste as each collection is recorded and documented for all citizens. In this case, the data is available very quickly, practically just-in-time, and an evaluation and assessment is possible within a few weeks or months (Aschaffenburg, 2014).

Applicability

Detailed waste monitoring is applicable to all local authorities and waste management companies managing municipal solid waste. For organisations starting the process, waste monitoring may focus first on the most relevant waste fractions and then it can be extended to all fractions, step by step.

Economics

No detailed information about the costs for establishing and running improved waste monitoring is available. Economics is affected by the level of monitoring adopted, the frequency, the number of fractions monitored, the human resources involved, and the tools used for data analysis.

Driving force for implementation

The legal requirements at EU and national level concerning recycling rates and the rates for diversion of organic waste away from landfills as well as the need to determine the efficiency and effectiveness of waste management systems are the driving forces for improved waste monitoring.

Reference organisations

Many cities and counties throughout Europe, (for example Copenhagen, Hamburg, Barcelona, Bristol, Milano, Val di Non, Aschaffenburg, Schweinfurt and Lombardy) have detailed waste monitoring of waste fractions. In the specific case of Lombardy, the Regional Waste Monitoring Centre (O.R.So - Osservatorio Rifiuti Sovraregionale) of the Regional Agency for Environmental Protection of Lombardy (Agenzia Regionale per la Protezione dell’Ambiente della Lombardia) has set up a system to systematically collect data on single waste streams; this system is subject to continuous improvement. http://www2.arpalombardia.it/siti/arpalombardia/imprese/rifiuti/Pagine/ORSO.aspx

In the case of Val di Non (Italy), regular (four times per year) and comprehensive waste monitoring (quantification of separate amount of fractions collected and composition analysis of residual waste) is carried out in the local area (Comunitá Val di Non, 2017). Some results are publicly available at http://www.comunitavaldinon.tn.it/Aree-Tematiche/Gestione-rifiuti/Statistiche-raccolta-differenziata

Another relevant reference organisation is ORDIF (Île-de-France Regional Waste Management Observatory) which every year issues a dashboard summarising the main figures related to waste management in the Paris region (prevention, collection, treatment, costs, environmental impact, etc.) extracted from its various studies and surveys. The dashboard has proved to be a comprehensive and practical reference document for regional waste stakeholders. The dashboard is available here: http://www.ordif.com/public/document.srv?id=18805.

The Swedish waste management company Avfall Sverige, a public organisation managing the waste collection and treatment of the vast majority of household waste in Sweden, has also developed a detailed monitoring system, which can be easily consulted online, via a website (Avfall web). The online system allows the user to check the waste management data of specific municipalities and waste treatment facilities. A number of indicators (e.g. waste generation per capita, waste sent to energy recovery) can be freely consulted and results compared thanks to an easy interface (Svensson, 2015, personal comm.).

In terms of methodology, the Regions for Recycling project (R4R, 2014) can be considered a reference. In this EU-funded project, due to the difficulties of comparing data from different territories across Europe (see also http://www.regions4recycling.eu/upload/public/Reports/R4R_Data_comparison_main_findings.pdf), 13 EU partners together defined a common method to monitor, present and compare waste management data and recycling performances. The method is based on several elements (R4R, 2017):

- a common scope for municipal waste;
- a common indicator called 'DREC' (Destination RECycling) that only includes homogeneous waste fractions sent and accepted by the recycling sector (i.e. no significant contamination is included);
- a framework for detailing 'external factors' (i.e. factors having an impact on waste management performances over which the territory has little to no influence), acting as parameters allowing the identification of comparable territories;
- a framework for 'local instruments' detailing all the different policy instruments at the disposal of public authorities to organise and improve waste management/recycling;
- an online tool allowing any territory to input its own data according to the R4R method and to benchmark it against other territories.

In terms of advanced waste monitoring systems and web-based tools able to gather data on the waste streams collected at single household level and share information with residents (know-as-you-throw), there are currently some examples of municipalities experimenting with them within the framework of a PAYT system or without the adoption of PAYT. Relevant references are the municipality of Brive-La-Gaillarde (FR) http://incitation.sirtom-region-brive.net/ or webtools such as http://garbagesportello.harnekinfo.it/ElencoAziende.aspx where each user can access their real-time waste production, or the app http://www.riciclario.it/cosa-fa-riciclario/.

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Regions for Recycling (R4R) (2014). Regions For Recycling – R4R Methodology [on-line]. http://www.regions4recycling.eu/R4R_toolkit/R4R_methodology, last access September 2017.

Regions for Recycling (R4R) (2017). Regions For Recycling EU-funded project website: http://www.regions4recycling.eu/home, last access September 2017

Svensson (2015). Eva Svensson Myrin personal communication on Avfall web system on 19/10/2015.

4.3.3. Pay-as-you-throw

Summary overview							
The aim of pay-as-you-throw (PAYT) is to enact the polluter pays principle in a fair way by charging users of the waste management system according to the amount of waste they generate. It is BEMP to charge waste fees to users based on a fixed plus variable fee component, to reflect the cost structure of waste management and align incentives for users (i.e. lower fee when less waste is produced) and waste collectors (i.e. revenue stability from the fixed fee component). In practice, the system can be implemented in various forms, typically: - volume-based schemes (choice of container size); - sack-based schemes (number of waste sacks used), e.g. with prepaid specific sacks; - weight-based schemes (the weight of the waste collected in a given container); - frequency-based schemes (the frequency with which a container is left out for collection – this approach can be combined with volume- and weight-based schemes). The scheme can be focused on charging for residual waste only or also separated streams, still with the aim of fostering source separation and waste prevention. The four key elements enabling the implementation of a PAYT scheme are: - the identification of individual users; - the measurement of waste streams at the individual user level (e.g. from door-to-door collection, street containers or at civic amenity sites); - the definition of a unit pricing that effectively drives behavioural change; - the engagement of residents to ensure a correct understanding of the features of the scheme and their buy-in and commitment (this is important to avoid illegal dumping or the transfer of waste in other territories not served by a PAYT scheme).							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
While the approach is broadly applicable, existing infrastructure must be adapted (e.g. collection). Door-to-door collection is usually necessary to fully implement PAYT principles. Precautions must be taken to ensure that enforcement is ensured (e.g. no 'leakage'							

into the MSW of adjacent local authorities with no PAYT or into litter bins on the streets). This is more feasible when there is already an existing awareness of users regarding source-separation and broader environment and waste issues.

Depending on the implementation (e.g. in case of user identification of individual bins or bags), appropriate measures are needed to deal correctly with data privacy and confidentiality (e.g. secure data storage).

Specific environmental performance indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- A pay-as-you-throw system is in place (y/n);
- inclusion of waste conferred to civic amenity sites in the PAYT system (y/n);
- share of users with zero waste generation (%).

Benchmarks of excellence

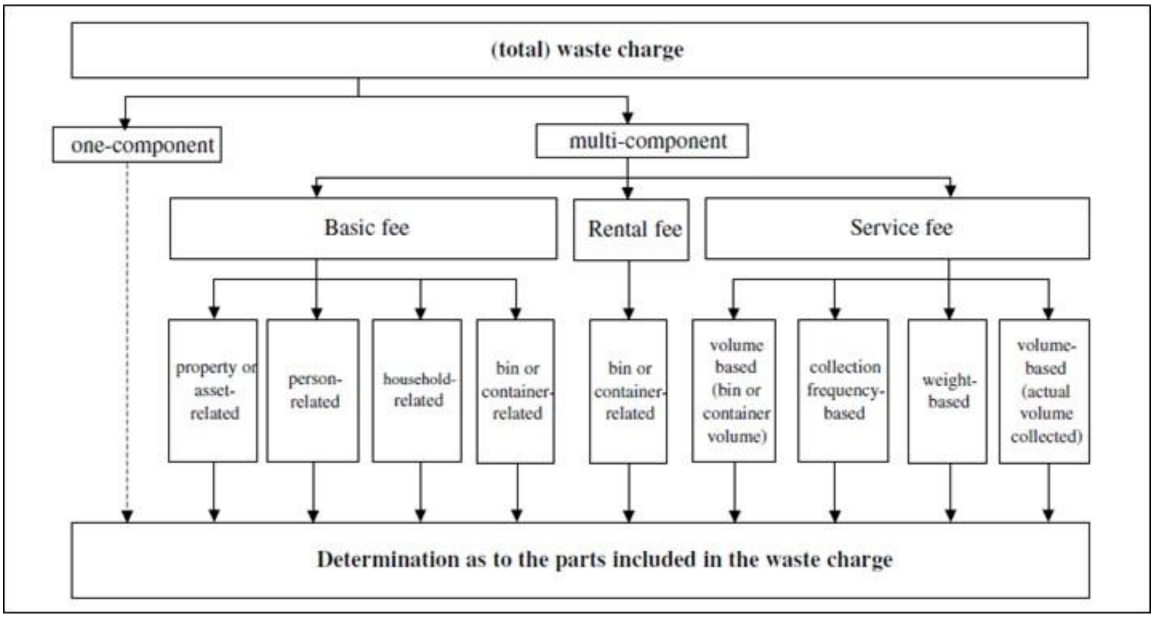
- A pay-as-you-throw system is in place, according to which at least 40 % of the cost is charged to the users depending on the quantity (kg or m^3) of mixed waste collected, the size of the waste collection bins and/or the number of collection rounds.
- The PAYT system also includes the waste conferred to civic amenity sites.

Description

The approach of "pay-as-you-throw" (PAYT) (also known as unit pricing (Dijkgraaf and Gradus, 2009), differential and variable rates (OECD, 2006; van Beukering et al., 2009) and variable fee or charge systems) is to apply the 'polluter pays' principle in a fair way by charging inhabitants according to the amount of waste they generate (Bilitewski et al., 2004).

The experience gained so far has revealed that the waste fee should not only comprise the single component "amount of waste generated" but should ideally consist of fixed and variable (service-based) fees (Bilitewski, 2008). On the one hand, this reflects the cost structure of waste disposal, which consists of fixed and variable costs (Bilitewski et al., 1995), and, on the other hand, the inclusion of a fixed (basic) fee helps to avoid illegal disposal practices, which can increase in the event that the fee is only charged for the variable amount of waste collected (Reichenbach, 2008; Puig-Ventosa, 2008). Waste fees applied to residents should have the right balance between variable and fixed fees. Local authorities aim at revenue stability, thus high fixed fees, but it is the variable fee (unit rate) that leads to behavioural change of residents, driving waste prevention and better waste separation at source. When establishing the waste fees, an economic balance of waste management should also be sought by covering as much as possible residual waste management costs with PAYT revenues.

Figure 4-7 shows the different possible components of a waste fee.



Source: Bilitewski (2008)

Figure 4-7. Different suitable components for the design of waste fees

In Figure 4-7, the service fee represents the service-related part of the fee. Consequently, the PAYT approach means that a substantial part of the overall fee is allocated to the amount of waste generated in order to stimulate waste prevention and recovery.

In this context, PAYT schemes can be implemented in different ways as illustrated in Figure 4-8.

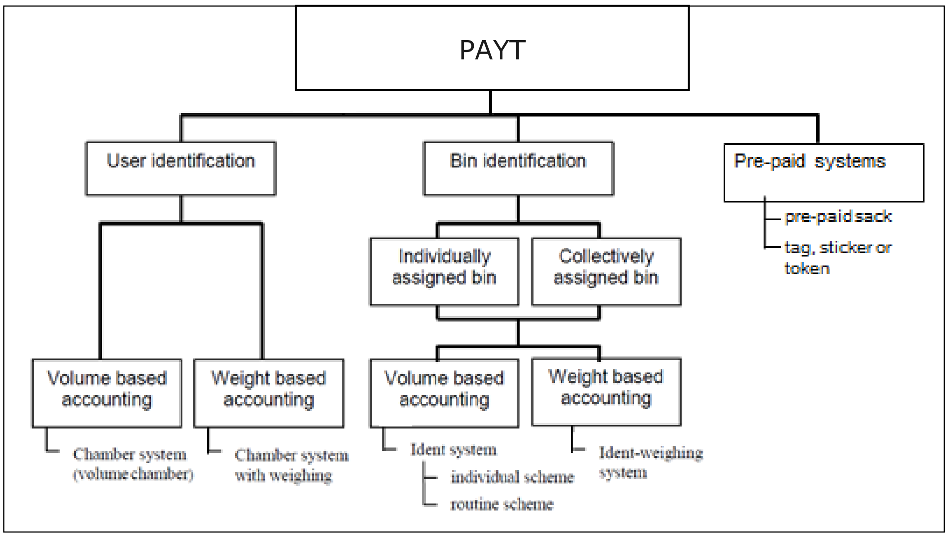


Figure 4-8. Overview of the different possibilities to implement the PAYT approach (based on Reichenbach, 2008)

The prepaid sack system is also considered to belong to the volume-based systems but here it is presented as an additional system as for solid household waste; the volume of a sack directly correlates with its weight and the fee has to be paid for each sack. Therefore, it is different from common volume-based schemes where citizens pay for the choice of container size. The most important PAYT schemes (Watkins et al., 2012) are:

- volume-based schemes (choice of container size);
- sack-based schemes (number of sacks left out for collection);
- weight-based schemes (the weight of the waste collected in a given container);
- frequency-based schemes (the frequency with which a container is left out for collection – this approach can be combined with volume- and weight-based schemes).

Best practice is that weight-based door-to-door collection is carried out not only for residual waste but also for organic waste and bulky waste. The successful implementation of an efficient PAYT system requires that the waste delivered to civic amenity sites is also covered by the PAYT system; therefore, a well-developed network of civic amenity sites (see Section 4.5.3) is key for a well-performing PAYT system in order to offer the citizens a comfortable way to dispose of materials that they no longer need. In addition, awareness-raising is also an important element for PAYT systems; if the citizens are aware, well-informed and supportive of the system, they will contribute to its success.

The experience shows that the best results can be achieved with weight-based schemes but that with prepaid sack schemes good performances are also achieved whereas volume-based systems impart the weakest incentive for waste prevention and recycling (OECD, 2006; Watkins et al., 2012). In contrast, the highest recycling rates and lowest residual waste quantities are achieved with weight-based systems accompanied with well-developed infrastructure and citizens with high awareness. Consequently, a case study is presented in more detail. For such a system, the technical elements of the PAYT scheme are based on the following four pillars:

- the identification of individual users;
- the measurement of waste streams at the individual user level (e.g. from door-to-door collection, street containers or at civic amenity sites);
- the definition of a unit pricing that effectively drives behavioural change;
- the engagement of residents to ensure a correct understanding of the features of the scheme and their buy-in and commitment (this is important to avoid illegal dumping or the transfer of waste in other territories not served by a PAYT scheme).

In other words, the waste producer has to be identified, the amount of waste delivered is recorded by weight, and there is a price per unit of waste which has to be paid in addition to the fixed fee.

Achieved environmental benefits

The amount of residual waste significantly decreases and the amount of recycled waste increases accordingly – if the infrastructure to collect and to process the recyclables is available and efficient and the citizens have adequate awareness and actively support the system. Recycling rates of 70 % and higher (Reichenbach, 2008), up to 86 % in case of weight-based systems (Aschaffenburg, 2013), are achieved. Figure 4-9 shows the development of the quantities per capita for the total waste, the waste disposed of and the recycled waste from 1991 to 2013 for the county of Aschaffenburg, Germany. The PAYT system with identification and weighing of the waste bins (for residual waste as well as for biowaste), collected door-to-door, was introduced in 1997 and the subsequent increase in recycled waste and the decrease in disposed of waste are obvious. In principle, this example is representative; as the weight-based system is applied, the recycling rates are particularly high.

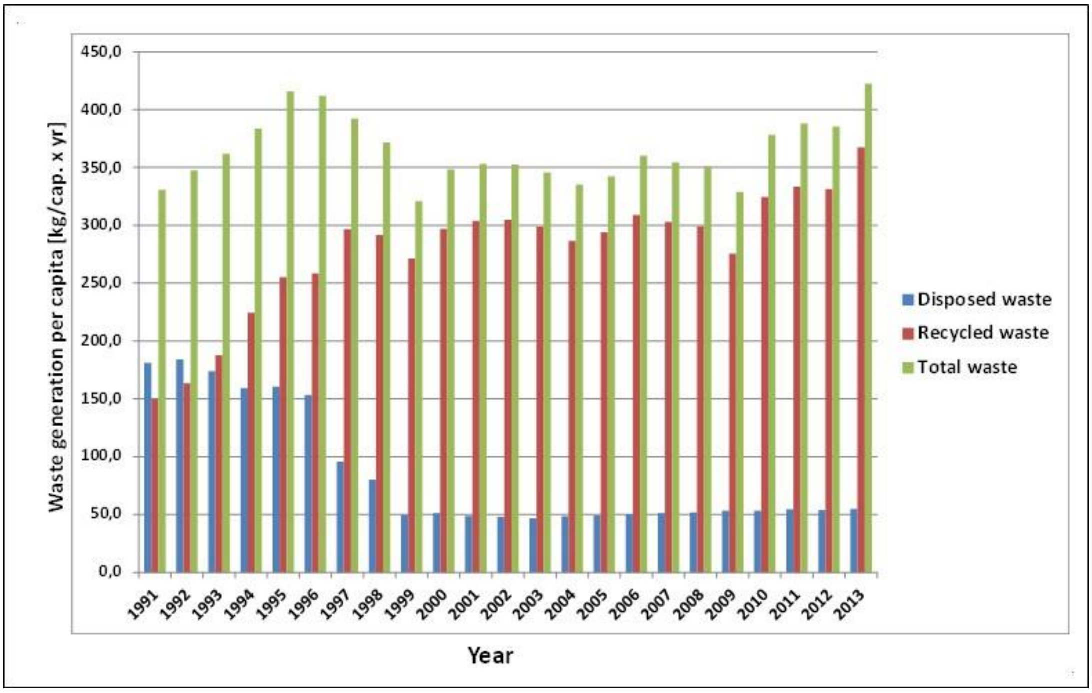


Figure 4-9. Development of the quantities of total waste, waste disposed of (i.e. mixed waste) and recycled waste from 1991 to 2013 in the county of Aschaffenburg (Germany) (County Aschaffenburg, 2013)

The reported recycling rates for weight-based systems vary significantly due to the different levels of waste collection infrastructure and public awareness. Another example with a very good performance is reported from Italy, where high recycling rates and low residual waste quantities were achieved. In the Treviso region, only 55 kg residual waste per capita were reported for 2015 (Contó, 2015; Contarina, 2015) and in the municipality of Trento in the year 2014 the residual waste quantity was 102 kg/capita per year (see Figure 4-10).

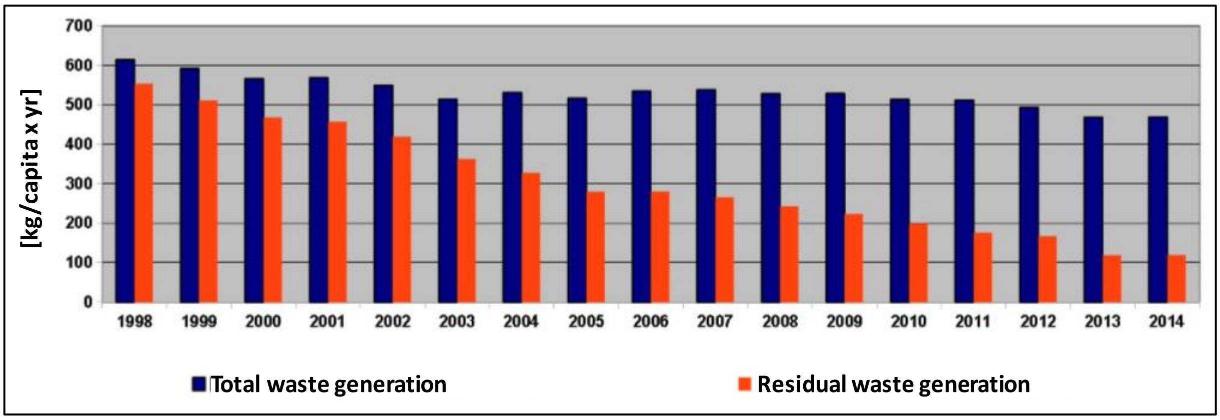


Figure 4-10. Development of the total and residual waste quantity in the municipality of Trento from 1998 to 2014 (Fedrizzi, 2015)

The same is true for Flanders, a region of Belgium, where first prepaid sacks were used and later weight-based systems. The recycling rate could be significantly reduced and the residual waste quantity reduced down to 149 kg/capita per year (Regions for Recycling, 2014a). The development is indicated in Figure 4-11.

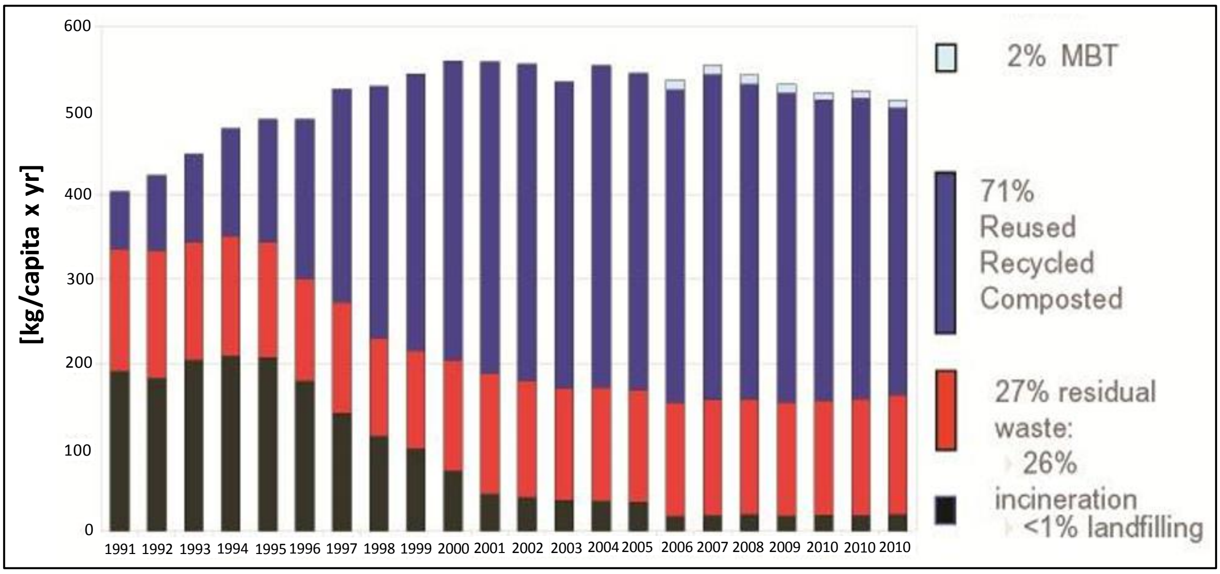


Figure 4-11. Development of recycled and residual waste as well as incinerated and landfilled waste in Flanders from 1991 to 2012 (Regions for Recycling, 2014a)

The prepaid sack systems also show a significant decrease in the quantity of residual waste but the achievable figures are lower compared to optimum weight-based systems.

- In Switzerland, on average 391 kg/capita per year are recycled which corresponds to 53.5 % of the total waste quantity (Switzerland, 2015).

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- A pay-as-you-throw system is in place (y/n);
- inclusion of waste conferred to civic amenity sites in the PAYT system (y/n);
- share of users with zero waste generation (%).

Cross-media effects

The implementation of PAYT increases the risk of waste leakages from the system (waste going to nearby municipalities without PAYT, illegal dumping, littering, etc.). A well-developed and easy-to-use infrastructure for the collection of waste reduces the risk of waste leakages together with adequate environmental awareness of residents. Local authorities, in addition, can monitor the leaked waste, for instance investigating residents/users with zero waste generation in the PAYT system. This method helps identify those residents disposing of their waste through alternative channels (which could include illegal dumping), so corrective actions can be implemented.

The implementation of PAYT systems may also lead to higher levels of impurities in waste fractions (e.g. recyclables) that can be collected for free or at a lower cost than mixed waste.

Operational data

The principal scheme of the weight-based system is illustrated in Figure 4-12.

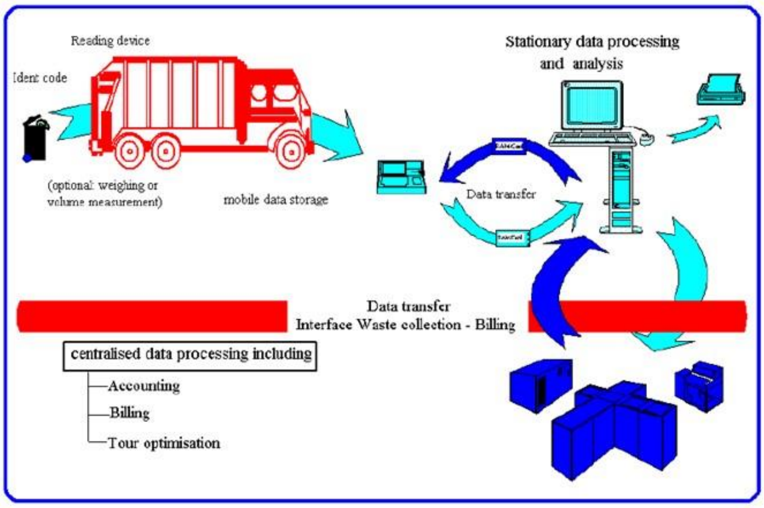


Figure 4-12. Process chart for electronic identification and data transfer in a bin identification scheme (Bilitewski et al., 2004)

Figure 4-12 also indicates the option of a volume-based system but this is not considered further as the reduction rates with this system are very low. In contrast, the weight-based system, accompanied by well-developed infrastructure and citizen awareness, can achieve the highest recycling rates and lowest residual waste quantity.

In the example of weight-based PAYT schemes, all the waste bins are equipped with a chip and a barcode that can be read by a transponder or barcode reader. An example for a barcode is given in Figure 4-13 and examples for chips in Figure 4-14.

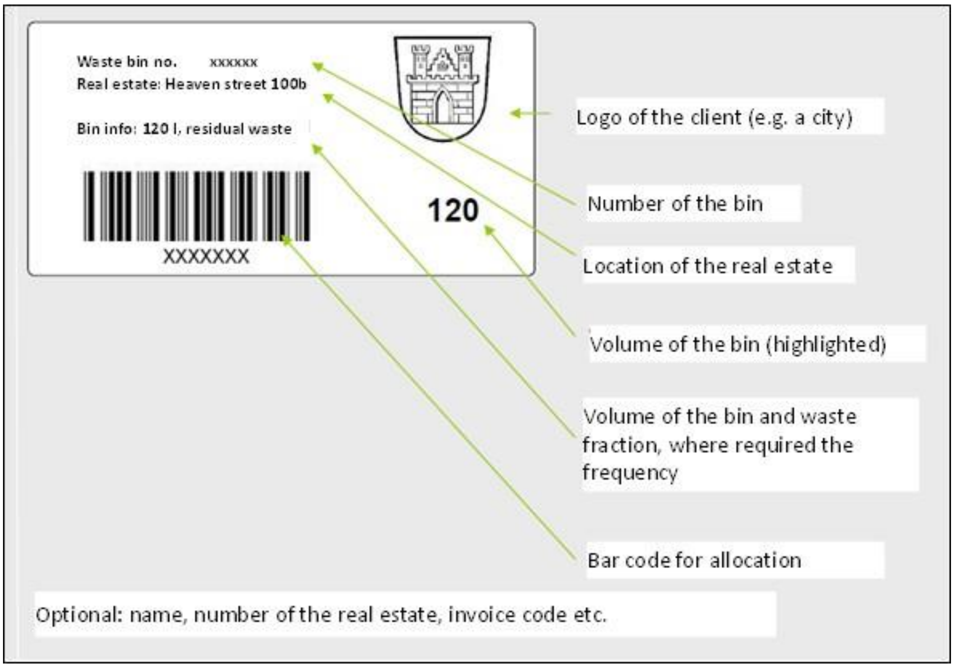
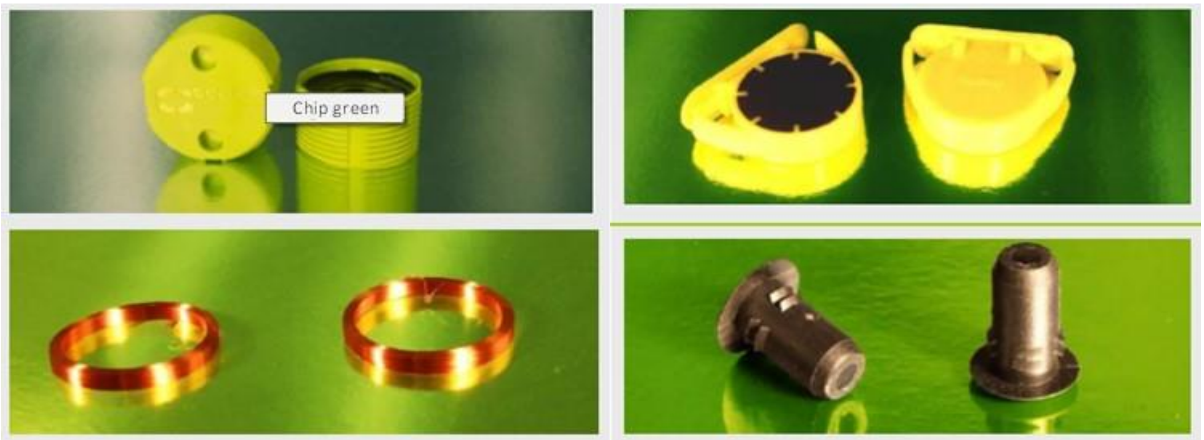


Figure 4-13. Example of the information automatically read by an identification system



Standard chip for new bins Chips for the retrofit of existing bins

Figure 4-14. Examples for chips for new bins (on the left) and for retrofitting existing bins (on the right)

Figure 4-15 shows a waste collection truck which is equipped with a waste identification system and a weighing system. The latter cannot be seen in the picture.

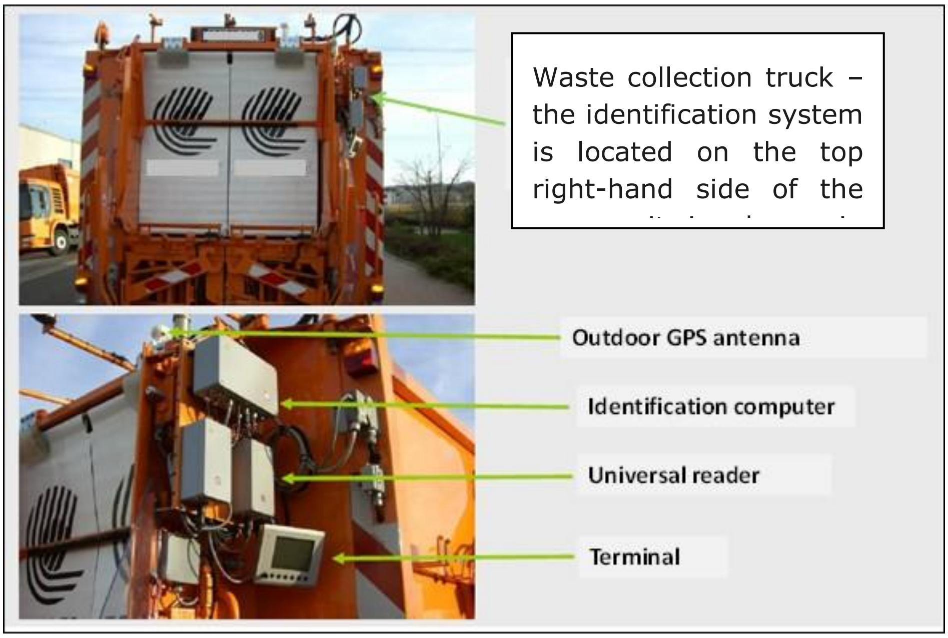


Figure 4-15. Example of waste collection truck equipped with a waste identification system

The weight-based system requires greater efforts to maintain and to calibrate the scales.

Where the infrastructure to separately collect and to process the different fractions, such as residual waste, glass, paper/board, plastics, organic waste, green cuttings, demolition waste, bulky waste, is well established and functioning, the difference in reduction of residual waste between the identification system and the weight-based system can be significant. Additionally, in such well established seprate waste collection systems, illegal dumping is usually negligible (County Aschaffenburg, 2013).

For densely populated areas and high-rise buildings, container systems are in use to which only assigned people have access. In Figure 4-16, two examples for such container systems are presented.

Source: (Bilitewski et al., 2004)

Figure 4-16. Large bins or containers to which only defined persons have access

The success of the system is directly associated with its environmental, economic and customer-friendly (level of service) performance. This is especially true for the infrastructure to collect and to process recyclables.

Applicability

From a technical point of view, the PAYT system can be implemented in any municipality. The weight-based system requires more technical equipment and staff but can achieve very high performance levels; it requires a detailed inventory of all households and individual bins and containers. Confidentiality aspects can be managed and have not limited the application of the techniques so far; for instance, the privacy fears in the UK could be managed (e.g. Holmes et al., 2014).

At the time of introducing the system, there is a significant workload peak for the municipality, city or county concerned as well as for the service provider (collector of the bins and containers).

Furthermore, as already stressed, a well-established infrastructure for the collection of the different waste fractions is required in order for the citizens to dispose of certain waste fractions in an easy and comfortable way.

The environmental awareness of citizens is also a factor that has to be considered, especially with respect to illegal dumping of waste to save money. If the environmental awareness is low, information campaigns are required. Specifically, with respect to possible illegal dumping, adequate enforcement must be in place (see Section 3.3.3, Operational data, for more information). Meanwhile, as mentioned above, where environmental awareness is already well developed, the introduction of PAYT does not lead to relevant problems with illegal dumping.

Economics

In the county of Aschaffenburg, after implementing the whole current waste management system, the 2013 waste fee was lower compared to the initial situation 16 years before. The PAYT system in Aschaffenburg included: the PAYT scheme with weight-based waste collection of residual waste and biowaste as well as separate collection of paper from all households, the operation of recycling facilities and composting/incineration of green cuttings in all bigger municipalities, the PAYT approach for collection, processing and disposal of bulky waste since 1999, disposal of the residual waste in an incineration plant according to BAT standards, anaerobic digestion of biowaste, subsidies for composting at the household level, for the use of reusable nappies, and for families with incontinent persons).

The calculation of the fee (just before and just after introducing the weight-based system) is publicly available (County of Aschaffenburg, 1995; County of Aschaffenburg, 1997). Despite the manifold additional activities (separate collection of the different fractions, erection of the first facilities to recycle or to recover waste streams), the fee significantly decreased after the change. So, the fear that the weight-based system is more expensive (e.g. Slavik and Pavel, 2013) is not reflected in the case of Aschaffenburg. However, the extent of the cost can vary from case to case. After the change, the disposal cost decreased by 46 %, especially because the residual waste was incinerated and the incineration costs were high at that time

(EUR 232/t in 1997) and decreased to EUR 52.80 in 2014. In 1999 and 2000, the fee had to be increased by 20 % to cover all the costs; the fee estimation had been based on a part of the county but the costs in other parts were higher. But, from 2002 to 2013, the fee significantly decreased, by about 23 % (see Table 4-2), although the county further invested in anaerobic digestion of the biowaste, in collection centres, in weighing the green cuttings, etc.

The same has been observed in Italy. The region of Treviso also has an advanced waste management system (high recycling rates and low quantities of residual waste) and also has low waste fees; the average waste fee is about 27 % lower than the average waste fee in Italy (Contó, 2015; Contarina, 2015). Currently, in Treviso, 60 % of the waste fee for a household is calculated based on the number of people living in the same place and 40 % varies according to the amount of mixed waste collected from the household (Contarina, 2016). Discounts are applied if home composting is implemented, while an increase in the tariff is applied if the household also delivers green cuttings to the waste management system (Contarina, 2016).

Table 4-2. Development of the waste fees in the county of Aschaffenburg from 1997 (the year the PAYT system for residual waste was implemented) to 2012 for an average four-person household.

Year	Annual basic fee for a 120-l bin	Fee for the weight of the waste	Fee to collect the waste (emptying the bins)	Total annual fee (without a bin for organic waste	Total annual fee (with a bin for organic waste)
	[EUR]	[EUR]	[EUR]	[EUR]	[EUR]
1994-95				171.8 / 245.4[1]	
1996-97				158.0 / 225.50[1]	
After the introduction of the weight-based system in mid-June 2007					
1997	50.31	44.54	21.47	116.33	148.67
1998	50.31	47.92	18.41	116.64	148.97
1999	55.22	53.87	20.25	129.34	165.52
2000	62.58	59.93	21.47	143.99	184.91
2001	62.58	59.30	21.47	143.36	182.05
2002	63.00	46.22	21.60	130.82	162.90
2003	63.00	45.80	21.60	130.40	162.70
2004	63.00	48.50	21.60	133.10	168.33
2005	60.00	40.04	19.60	119.64	147.76
2006	60.00	40.13	19.60	119.73	148.20
2007	60.00	40.66	19.60	120.26	149.49
2008	54.00	37.28	19.60	110.88	138.72
2009	54.00	37.76	19.60	110.36	139.50
2010	54.00	37.20	19.60	110.80	138.65
2011	54.00	38.32	19.60	111.92	140.94
2012	54.00	37.68	19.60	111.28	140.14
2013	54.00	37.60	19.60	111.20	140.38

[1]Lower figure for a 35-litre bin, higher figure for a 50-litre bin.

Note to the table: Columns 2-5 provide the figures for the case where the household has no bin for organic waste and column 6 gives the total fee where the household also has a bin for organic waste (County of Aschaffenburg, 2013)

The fee after the introduction of the weight-based system represents an average value as all the bills are individual due to the variable fee for the weight.

The fee in the county of Aschaffenburg consists of the basic fee, the collection fee (to empty the bins) and the weight fee. In 1997 and in 2012, the percentages were as follows (County of Aschaffenburg, 2013):

	1997	**2012**
Basic fee	32 %	47.0 %
Collection fee	17 %	18.5 %
Weight fee	51 %	34.5 %

The percentage for the weight part decreased but is still high enough to motivate waste prevention/recycling. However, the effect on prevention is low. Figure 3.18 shows an example of the annual bill of the county of Aschaffenburg indicating the basic fee, the service charge to collect the waste (collection fee) with a certain frequency and the weight fee, separately for the biowaste, for which the basic fee is zero, and the residual waste.

Landkreis Aschaffenburg
- Müllgebührenstelle -

Öffnungszeiten Müllgebührenstelle
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1

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SPK Aschaffenburg-Alzenau **BLZ** 79550000 **Kto.Nr.** 60954
IBAN DE04 7955 0000 0000 0609 54 **BIC** BYLADEM1ASA

Mandatsreferenznummer
PK33458/1KD13442-21-A-0
(bei Überweisung unbedingt angeben!)
Bescheidnummer 2900929 vom 09.01.2015

Note on the waste disposal fee

1. **Determination for the estate**
BEISPIELSTR. 21 A, WALDASCHAFF

Final bill for 2014 **173,01 EUR**

2. **Fee calculation**
For the time period 01.01.2014 – 31.12.2014

								Fee	Sum
Biowaste		60 L, bin no. 101625, 01.01.2014 – 31.12.2014							
a)	Basic fee esidual waste	12 months		x	0.00	EUR	=	0.00 EUR	
b)	Collection fee	collect. frequ. per yr	25	x	0.45	EUR	=	11.25 EUR	
c)	Weight fee	weight	343.0 kg	x	0.18	EUR	=	61.74 EUR	72.99 EUR
Residual waste		120 L, bin no. 604576, 01.01.2014 – 31.12.2014 014							
a)	Basic fee esidual waste	12 months		x	4.05	EUR	=	48.60 EUR	
b)	Collection fee	collect. frequ. per yr	12	x	2.50	EUR	=	30.00 EUR	
c)	Weight fee	weight	119.0 kg	x	0.18	EUR	=	21.42 EUR	100.02 EUR

Final billing 173.01 EUR
Already paid amount 153.72 EUR
Remaining amount to be paid **19.29 EUR**

Please check your bin number! Residual waste: 604576 Bio waste: 101625 Paper: 732590

3. **Remaining amount 2014**
The remaining amount mentioned under no 2 for the year 2014 is payable on:
16.03.2015: 19.29 EUR

Figure 3.18: County of Aschaffenburg example of the annual bill for the waste fee of a four-person household having separate bins for residual waste (120 l), biowaste (60 l) and paper/cardboard

In a country with a hot climate, the collection frequency for biowaste will be higher, which may be associated with higher collection costs, but the collection frequency for residual waste can be as low as indicated.

Driving force for implementation

In many cases, waste managers in municipalities were motivated to implement the PAYT approach where landfill capacity was exhausted, where fees were high and/or public environmental awareness called for a change. Furthermore, in some Member

States, the landfill of untreated municipal waste was already banned before the EU-wide restrictions came into force[39].

Reference organisations

Many municipalities in Germany apply the weight-based system (e.g. counties of Aschaffenburg, Schweinfurt, Garmisch-Partenkirchen, Landsberg am Lech) as well as municipalities in the Netherlands (Rijkswaterstaat, 2014), France (city of Besançon) and Ireland (Regions for Recycling, 2014b). It is also practised in the US (Skumatz, 2002, 2008; Hall et al., 2009).

The prepaid sack system is widespread in Switzerland (Bilitewski et al., 2004, Switzerland, 2015) and is applied in Belgium, the Netherlands, Denmark and in a few cases in Italy and Spain (Catalunya, 2010).

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4.3.4. Performance-based waste management contracting

Summary overview

It is BEMP for local authorities that contract out the delivery of certain MSW management services to private suppliers to include performance-based contract clauses. Performance-based contracting can ensure that both environmental and financial objectives are met.

Three main characteristics are inherent to a performance-based contract:

- definition of a series of objectives and indicators to measure contractor performance;
- collection of data on the performance indicators to assess the implementation of the service;
- good or bad performance impacting the contractor (higher revenue or penalties).

It is important for local authorities to base the performance clauses on a full set of indicators (for example taking inspiration from the indicators presented in Chapter 2) and appropriate monitoring. Special care needs to be taken in defining a baseline and bearing in mind the influence of the variation in external conditions (economic, social, regulations, etc.) on the benchmark mechanism.

Waste management area

Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability

The existence of an effective waste management performance monitoring system is a prerequisite to performance-based waste monitoring system (building on internal management practices to expand to contract management).

When switching to a performance-based contract for the first time, it is also important to establish a dialogue with the prospective contractors and all stakeholders involved, in order to learn what is technically achievable and economically feasible.

Specific environmental performance indicators

- Share of the contract value depending on the achievement of the environmental objectives or of the defined environmental performance levels (%).
- Customer satisfaction (% of residents satisfied with household waste collection and specifically with the collection of the separately collected fractions).

Description

Municipality-contracted services are usually in efficient when, once a private service provider is in place, the cost efficiency and cost savings of the system come at the expense of its performance, i.e. costs are reduced due to a lower quality of the service. To avoid that, the municipality can put in place a binding contract that articulates robust performance standards. If the contractual mechanisms needed to encourage the right results are inadequate or are even missing, the contract will result in a failure (Chamberland, 2011). Performance-based contracting (or *resource management*) is a common technique used in other areas of public and private contracting. A performance-based management contract is an agreement for the management of waste that, through the action of a contractually agreed payment mechanism related to defined performance indicators and targets, incentivises the movement of waste management further up the waste hierarchy, and enhances the prospects for improved resource efficiency and the flourishing of a circular economy (EUNOMIA, 2014). The waste authority establishes a contract with an entity where the payment obligation for each year, including the year of implementation, is either (a) set as a percentage of the municipal solid waste cost savings attributable under the contract, or (b) guaranteed by the entity to be less than those solid waste cost savings (WSL, 2007).

In contrast to energy contracting, waste management performance-based contracts are not so common. The main example is the case of Bristol, which implemented a green public procurement system based on a performance-based contract. Although in all waste management contracts there are clauses and schedules on performance and its monitoring, no incentive or penalty system has been detected to constitute a best practice. Also, the Regions for Recycling (R4R) programme did not include any example of performance-based best practice in their analysis of economic instruments at local scale (R4R, 2014). The International Institute for Sustainable Development (IISD, 2014) argues that performance-based contracts do not necessarily ensure any degree of environmentally or socially beneficial performance if these are not correctly targeted, while the public sector shifts to an evaluation or measuring only role. Also, Eunomia (2014) performed a theoretical study of the plausible impact of performance-based contracts and some conclusions were derived:

- the municipality needs to develop a full set of indicators, for example taking inspiration from the ones presented in Chapter 2, and develop monitoring practices;
- a baseline has to be defined, and the influence of the variation in external conditions (economic, social, regulations, etc.) has to be properly taken into account in the benchmark mechanism.

The study does not include any example of its application, but the analysis of plausible scenarios in a theoretical context. In light of these conclusions, it is concluded that the application of best environmental management practice (e.g. waste monitoring, PAYT) enables the use of performance-based contracts. For systems with an outstanding performance and a solid strategy, performance-based contracts would be a tool for optimisation. Unfortunately, no example has been derived in this regard.

The key is to create a *win-win* situation for both the customer and the contractor, since both participate due to the achieved cost savings. Three main characteristics are inherent to a performance-based contract:

- definition of a series of objectives and indicators to measure contractor performance;
- collection of data on the performance indicators to assess the implementation of the service by the contractor;
- good or bad performance leading to consequences for the contractor (higher revenue or penalties).

A public organisation, in a performance-based setting, identifies the problem to be solved and the supplier must convince the public organisation with a solution. Then, the public organisation is required to develop or use clear standards to measure the performance of the service and penalise non-compliance (Chamberland, 2011). Conventional contracts, even including performance-based clauses, do not include win-win situations or the measures to achieve the performance are not left to the decision of the contractor. The contractual economic arrangements for the waste management service should be based on three pillars (U.S. EPA, 2004): (i) cost-effective opportunities to reduce waste, (ii) financial incentives to contractors to pursue the recycling and reduction of waste, and (iii) financial incentives are generated from cost savings. In most of the examined literature, performance-based contracting in the waste management sector focuses on waste collection, but the applicability can cover the whole spectrum of techniques (prevention, reuse, treatment, etc.).

Performance-based contracting can be applied to several contract arrangements in public-private utilities. In 2011, the OECD reported the following contractual formats for municipal services:

- Service contract: the private organisation carries out technical and/or administrative tasks (e.g. repairs, meters).
- Management contract: the private organisation takes over operation and management, although the user or client remains legally responsible for the public entity.
- Lease contract: the private company under a management contract also assumes the legal responsibility for operating the service in exchange for payments for the use of the fixed assets.
- Build-Operate-Transfer contract: the private organisation designs, builds and finances a new project that it also has to operate and maintain for the concession period.
- Concession contract: similar to the lease, but the contractor is in charge of financing the expansion or the rehabilitation of the service.
- Joint venture contract: the municipality and the private cooperator co-own the service (in these cases, the municipality usually has a golden share).
- Full divestiture: the asset is entirely sold to the private sector, with the private organisation bearing the risks. Public sector and independent regulatory agencies are in charge of supervision of the performance.

Table 4-3 shows how these contractual arrangements distribute responsibilities in the different stages of a performance-based contract.

Table 4-3. Allocation of responsibilities in a performance-based contract

	Responsibility for

Type of contract with the private organisation	Setting performance indicators and benchmarks	Asset ownership	Capital investment	Operation	User fee collection	Oversight of performance and fees
Fully public	Public	Public	Public	Public	Public	Public
Service	Public	Public	Public	Private	Public	Public
Management	Public	Public	Public	Private	Private	Public
Lease	Public	Public	Public / Private	Private	Private	Public
Concession	Public	Public	Private	Private	Private	Public
Fully private	Public	Private	Private	Private	Private	Public

Source: Adapted from OECD (2011).

Achieved environmental benefits

Performance-based contracting eases the implementation of best environmental management practices, and, therefore, may result in a better environmental performance by the following:

- Establishing a funding mechanism for a better performance, e.g. through incentives to the contractor or penalties due to low performance, without extra burdens to the public authority.
- Establishing an appropriate link between the waste hierarchy and the waste management contract. Part of the contractor revenues would be directly linked to the environmental performance. This is as opposed to conventional contracts, paid per volume collected or treated, so the reduction of waste volume generated is in contrast with the economic performance of the service, while recycling is sometimes not even considered in terms of the contractor performance.

Appropriate environmental indicators

The most appropriate indicators to monitor the implementation of this BEMP are:

- Share of the contract value depending on the achievement of the environmental objectives or of the defined environmental performance levels (%);
- customer satisfaction (% of residents satisfied with household waste collection and specifically with the collection of the separately collected fractions).

Cross-media effects

Performance-based contracts are designed to remove cross-media effects from conventional contracting. The environmentally beneficial performance of performance-based contracts is not always ensured and their benefit compared to conventional contracts can be disputed: for instance, if the contracting authority has not developed the metrics for the system or established a baseline (IISD, 2014). In that case, technical specifications in conventional contracts may produce better performance results.

Operational data

The United States Environmental Protection Agency uses the term *Resource Management* for performance-based contracting for waste management, under their WasteWise program (U.S. EPA, 2013). The original idea comes from General Motors' contracting practices, intended to achieve a better resource efficiency through cost reduction and conservation of manufacturing resources. EPA, through the WasteWise programme, shows that resource management contracting is applicable to businesses, institutions and municipalities.

In terms of waste management, clear differences are established between performance-based and conventional services (Table 4-4).

Table 4-4. Differences in management of waste management services

Features	Traditional hauling & disposal contracts	Performance-based contracts
Contractor compensation	Unit price based on waste volume or number of pick-ups.	Capped fee for waste hauling/disposal service. Performance bonuses (or liquidated damages) based on value of resource efficiency savings.
Incentive structure	Contractor has a profit incentive to maximise waste service and volume.	Contractor seeks profitable resource-efficient innovation.
Waste generator-contractor relationship	Minimal generator-contractor interface.	Waste generator and contractor work together to derive value from resource efficiency.
Scope of service	Container rental and maintenance, hauling, and disposal or processing. Contractor responsibilities begin at the dumpster and end at processing site.	Services addressed in hauling and disposal contracts plus services that influence waste generation (i.e. product/process design, material purchase, internal storage, material use, material handling, reporting).

Source: U.S. EPA (2013)

What the EPA detected through the analysis of several case studies is that traditional waste contracts typically pay a unit price based on the weight of the waste collected, the number of pick-ups and the container rental fees, while recycling is not considered a driver for any contractor. In terms of performance-based contracts, the contractors' profitability depends directly on, for example, recycling rates, diversion from landfill, and other indicators. Performance-based contracts can therefore establish a fixed price for the waste management service and introduce bonuses for good performance and penalties for deviations. The bonuses would come from the avoided disposal costs and marketed recovered materials. As a result, the contractor shares the incentive of the customer (the municipality) and creates a win-win situation: the good environmental performance of the contractor in charge of collection is directly linked to the profits.

Conventional waste contracting also results in little communication between the contractor and the municipality except for problem resolution or special requests. Under a performance-based contract, strong links are required and improved communication is usually achieved, resulting in refined and better strategies over time (Tellus Institute, 2002).

Bristol, in the UK, started in 2009 a new contract service for its waste management service. A dialogue with pre-qualified companies was established in order to define the approach of the new contract, in order to achieve the maximum recycling rates and a reduction in emissions (Bristol City Council, 2013). For the first time, the call for tenders included desired outcomes instead of conformance-based technical specifications. These were:

- reduce the 'carbon footprint' associated with the service in line with the agreed 2020 target for Bristol;
- increase waste reduction, reuse, recycling and composting, towards an aim of zero waste;
- deliver significant reductions of untreated waste sent to landfill;
- maximise the efficient recovery of resources, i.e. recyclates and energy from residual waste;
- tackle and reduce the incidents of environmental crime (e.g. by storing and collecting evidence from 'fly tipping');
- enhance community understanding of sustainable waste management.

The performance clause of the contract was set by establishing a CO_2e reduction target to be met by 2020. As the duration of the contract is 2011–2017, a pro rata basis of 25 % was defined in the call for tender, using as a baseline the emissions data from the previous contractor in the 2009–2010 period. No shared benefit is defined, but a penalty is defined for each 1 % above the target to a maximum of 0.375 % of the annual contract value. Money raised this way is used for environmental improvements that the contractor failed to make.

As a result, all bidders included a carbon emissions management plan committing to a new collection regime and offering solutions oriented to reducing the number of journeys necessary, e.g. by using multi-compartment trucks, using telematics and monitoring driver behaviour. The winner offered a 32 % CO_2e savings by 2017. During the first year of the contract, the recyclable materials collection rate increased from 38 % in 2010 to 50 % in 2011–2012. However, the penalty clause for not achieving the carbon reduction could not be implemented in the contract due to the high risk of supplier failure, which would imply a price increase for the final user (Bristol City Council, 2013).

Applicability

The existence of a well-standardised waste management performance monitoring system is a prerequisite before starting the procedure of a performance-based waste monitoring system. For instance, Bristol could implement a performance-based approach based on the existing CO_2e monitoring system and indicators system, derived from the EMAS-registered environmental management system (Bristol City Council, 2014). Another prerequisite, especially when changing to a performance-based contract, is to establish a dialogue with the prospective contractors and all stakeholders involved, in order to learn what is technically achievable and economically feasible. The city of Bristol may have failed at involving all required stakeholders, as, finally, the penalty clauses could not be implemented in the contract due to budgetary restrictions, i.e. the city council would never be able to absorb the higher price of the service that would then be charged to the citizens' waste fees.

Finally, another key aspect for applying this BEMP is the need to ensure, among the different parties involved in the contract, the traceability and transparency of data to which the performance-based contract is linked as well as the need for independent (i.e. third-party or joint) monitoring of results and the performance achieved.

Economics

Compensation options

According to U.S. EPA (2004), there are basically two compensation options for the contractor. However, the specifics of contracts may change depending on the negotiation phase; there will then be as many compensation options as contracts signed under performance-based clauses.

- Option 1. Pass-on of service costs with shared savings and performance bonus. Costs are established from the basic financial proposal in the bid, then cost savings are shared between the waste authority and the contractor. Examples of savings opportunities are diversion of materials towards recycling, more efficient handling and hauling through right-sizing, behavioural changes, etc. (all to be implemented by the contractor). The split of savings depends on the contract, the main example is 50/50 %. Other approaches could be for example 30/70 % for the contractor if the overall savings are over 5 %. Below 5 %, all savings go to the public authority. Then the performance bonus/penalties can be given through the increase/reduction of the savings share.
- Option 2. Fixed cost with guaranteed cost reductions. A fixed amount for the basic service is given to the waste management company, which is calculated on the previous year's total costs, and with a guaranteed cost reduction. For instance, if the cost was EUR 100 000 per month during the last year, the contractor may offer a 5 % cost reduction based on its own confidence of achieving that result. So, the public authority would pay EUR 95 000. All further savings would benefit the contractor. This is the option preferred in many US municipalities, as it is the one with less uncertainty for year-to-year accounting.

Examples of implementation

The case in Bristol, UK, showed that the time taken to prepare the tender and the dialogue and negotiation was twice that of a conventional contract, although its evaluation is not more complex. This factor adds an extra administrative burden and a resource-intensive tender process. In the case of Bristol, it also added a restricted budget, so no incentive or penalty clauses were finally introduced in the contract.

In Europe, not many references to the implementation of waste management performance-based contracts could be found. However, these examples have been successfully implemented in other areas of public procurement, such as energy efficiency of buildings, information technology, road construction, transport fleet and railways (IISD, 2014).

Driving force for implementation

In general terms, this technique is meant to align the waste management hierarchy with economic drivers. For instance, in conventional contracts an increase in the total amount of waste can be assumed as positive from the contractor's perspective. However, performance-based contracts would link waste prevention actions or programmes executed by the contractor to the actual revenues. Therefore, the main driver is the enhancement of the environmental performance of the waste system and the improvement of its management which will eventually reduce costs.

Reference organisations

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4.3.5. Awareness-raising

Summary overview
Best practice in awareness-raising is to effectively encourage waste prevention, reuse and recycling behaviour within the waste collection catchment area. Ultimately, this should translate into improved performance across key waste generation and separation indicators. Best practice awareness-raising campaigns need to: - ensure continuity, consistency, complementarity and clarity of all communications with well-defined aims and objectives; - create clear messages appropriate to, and directed at, well-defined target audiences; - ensure efficient delivery through the integration of activities and clear lines of responsibility. Examples of two major barriers to recycling that may be overcome by awareness raising are: - lack of knowledge: not knowing which waste materials to put in which container, or not understanding the local recycling scheme (e.g. collection days, etc.). - attitudes and perceptions: not accepting there is a need to recycle, being insufficiently motivated to avoid and sort waste. Awareness campaigns for citizens may be delivered directly by the waste management organisation, by professional agencies on their behalf, or by partner organisations (including stakeholders in other sectors). A whole range of communication channels can be used, which can include advertising, public relations, direct marketing, community engagement, online engagement, social media and product labelling.

Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**

Applicability
Awareness-raising can be implemented at some level in any context.

Specific environmental performance indicators
In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: - budget spent on awareness-raising per resident per year (EUR/capita/year);

- share of total MSW management budget spent on awareness-raising (%); - share of population in the waste management catchment area having received awareness-raising messages over a given time period (e.g. % of population per month).
Benchmark of excellence
- Awareness campaigns are systematically implemented for different types of target groups (e.g. pupils, general public, users of civic amenity sites) and the annual budget devoted to awareness-raising activities is at least EUR 5 per resident.

Description

Background

Effective communication between waste management organisations and citizens is integral to the efficient operation of waste management services. For instance, WRAP (2015a) cites research that found unwanted or broken waste electronic or electrical equipment (WEEE) items are commonly stored at home because citizens are often unsure of how to dispose of them. Citizens need to know what services are available to them, and the schedule and requirements of that service, in order for those services to be efficiently used. Citizens are also more likely to undertake waste sorting and recycling activities if they know what happens to waste that is sent for recycling, and the associated environmental benefits (Zero Waste Scotland, 2012). Thus, a key component of this BEMP is influencing large-scale behaviour change among citizens not yet fully engaged in good waste management practice.

Zero Waste Scotland (2012) identified two major barriers to recycling that may be overcome by awareness-raising:

- lack of knowledge: not knowing which materials to put in which container, or not understanding the local recycling scheme (e.g. collection days);
- attitudes and perceptions: not accepting that there is a need to recycle, or being insufficiently motivated to sort waste and recycle.

A particularly effective way to improve attitudes towards waste reuse and recycling is to embed waste management education into the school curriculum, teaching children about the causes and consequences of waste disposal and the importance of waste prevention and recycling through fun activities (e.g. R4R, 2014a). Local authorities and/or waste management organisations can facilitate this by undertaking outreach activities, sending representatives to local schools or inviting schoolchildren to facility tours or open days, etc.

Awareness campaigns for citizens may be delivered directly by the waste management organisation, by professional agencies on their behalf, or by partner organisations (e.g. R4R, 2014b). Paying for professional assistance, especially during the development of communication strategies, can significantly improve the effectiveness and “payback” of communication campaigns. The establishment of networks across key stakeholders can help to achieve a critical mass, reach a wider audience, and reinforce messages through repetition and validation.

Producers may also contribute to awareness-raising, directly in relation to responsible storage, use and disposal of their own products, and collaboratively with waste management organisations, including via "producer responsibility organisations" (PROs). PROs (see also Section 4.6.1) are collective entities set up by producers or through legislation with responsibility for meeting the recovery and recycling obligations of the individual producers.

Best practice measures

Best practice in awareness-raising is to effectively encourage waste prevention, reuse and recycling behaviour across citizens within the respective municipality or waste collection catchment. Ultimately, this should translate into improved performance across key waste generation and separation indicators. Particular emphasis is placed on reaching *all* stakeholders, including non-native speakers via multilingual or pictorial communication and via school activities. Additionally, awareness-raising activities/campaigns/meetings could integrate the aspect of collecting feedback and possibly complaints from residents on the waste management system in place. Waste advisers (see Section 4.3.6) could be useful to this aim, since they could directly answer the comment or report the issue to the local authority/waste management company. Such inputs, when useful, could then be considered for the revision of the waste management strategy, waste collection system, etc.

The following critical elements of effective awareness-raising should be embedded in all awareness-raising campaigns (Zero Waste Scotland, 2012):

- ensure continuity, consistency, complementarity and clarity of all communications with well-defined aims and objectives;
- create clear messages appropriate to, and directed at, well-defined target audiences;
- ensure efficient delivery through the integration of activities and clear lines of responsibility.

Best practice involves the use of a wide range of communication methods deployed through appropriate communication channels tailored to the target audience and to the message to be delivered, as indicated below in Table 4-5.

Table 4-5. Communication channels appropriate to various methods of awareness-raising

Methods	Communication channels
Advertising	Radio, printed press, TV, outdoor billboards, mobile, online, cinema spots.
Public relations	Media relations via radio, press, TV and online.
Direct marketing	Door-to-door canvassing, leaflet/information distribution, exhibitions and events.
Community engagement	Outreach to schools, support for local community groups, collaboration with third-sector organisations (see examples of best practice for product reuse schemes in Section 4.4.2). Also roadshows, seminars and door-to-door campaigns.

Table 4-5. Communication channels appropriate to various methods of awareness-raising

Methods	Communication channels
Online engagement	Local authority, waste management organisation, public agency or third-sector websites. Online calculators, interactive activities and videos, and apps, e.g. providing information on nearest collection points.
Social media	Social media is an effective way for citizens to access real-time or location-specific information, and provides a convenient and flexible form of communication. Social media channels include YouTube, Facebook, Twitter. See some examples below: https://www.youtube.com/watch?v=PZEA63TPYT0 (DE, video) https://www.youtube.com/watch?v=jo-nPS3VWvw (GB, video) https://www.youtube.com/watch?v=q3deji0AGys (GB, video) https://twitter.com/ACRplus (EU, Twitter) https://twitter.com/2EWWR (EU, Twitter) https://twitter.com/LetsCleanUpEU (EU, Twitter)
Product labelling	Producers may engage with other stakeholders, especially waste management organisations, to communicate with consumers via all of the above pathways within extended producer responsibility schemes. In addition, producers may clarify use-by dates, storage instructions and recycling options on packaging to minimise consumer waste.
Internal communication	Waste management organisations may inform their staff of the latest initiatives and plans via: staff magazines, intranet, information folders, activity reports, events, competitions (slogans, etc.), suggestions for improvements. ZeroWastePro have produced a training manual for staff of waste management companies http://www.zerowastepro.eu/publications/.

Source: Zero Waste Scotland (2012), Vienna City Council (2013), R4R (2014a), (EC 2014), own elaboration.

Awareness-raising campaigns, thanks to the use of a wide range of communication methods, go through four different progressive steps of residents' engagement, i.e. the number of people reached by the awareness-raising campaign (e.g. who received a leaflet), the number of people that read it, the number of people that understood it and the number of people who took action. The ultimate aim of any awareness-raising campaign is when people addressed decide to take action (in relation to the message of the campaign).

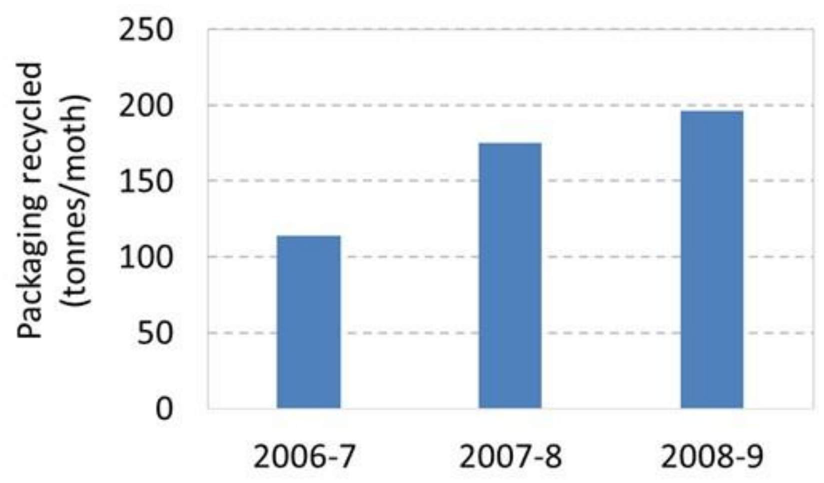
Examples of how some awareness-raising actions have been implemented using different channels are provided under Operational data and Reference organisations below.

Achieved environmental benefits

Effective awareness-raising should achieve significant environmental benefits through reductions in resource extraction and final waste disposal, as outlined in Chapter 1 of this report. However, it is often difficult to attribute changes in the rate of reuse or recycling to specific communication campaigns.

The Ecological Recycling Society in Attiki, Greece, ran a door-to-door information campaign to promote recycling of packaging, biowaste, batteries and WEEE between

2007 and 2009 within the municipality of Elefsina (R4R, 2014b). Data recorded for the total weight of packaging recycled in the locality showed a 72 % increase in the second year of the campaign, compared with the beginning of the campaign (Figure 4-17).



Source: Derived from data in R4R (2014b)

Figure 4-17. Total packaging recycling in the Elefsina municipality of Attiki, Greece, before (2006–2007) and during a door-to-door information campaign

Vienna City Council (2013) reported significant reductions in litter within the city for the time period from 2008 to 2012, following the principles of a provocative and humorous anti-littering advertising campaign: illegal dumping of white goods and shopping trolleys fell 68 % and 38 %, respectively, and cigarette butt littering dropped by 31 %. They also reported that 1 100 tonnes of dog poo is collected every year in disposable bags provided from street dispensers.

Appropriate environmental indicators

The success of any awareness-raising action should be assessed by monitoring the number of residents addressed who have decided to take action (in relation to the message of the campaign). However, this is practically speaking very difficult and some alternative indicators, in addition to the common environmental performance indicators presented in Chapter 2, have been identified for this BEMP:

- budget spent on awareness-raising per resident per year (EUR/capita/year);
- share of total MSW management budget spent on awareness-raising (%);
- share of population in the waste management catchment area having received awareness-raising messages over a given time period (e.g. % of population per month).

Cross-media effects

Information campaigns may involve transport and the production (and ultimately disposal) of paper-based advertising materials, or energy and material consumption, e.g. energy use for online media (Greenpeace, 2014). The magnitude of resultant environmental burdens will vary considerably depending on the type of campaign but should be significantly exceeded by the benefits associated with even small increases in waste prevention or recycling rates.

Operational data

Steps to implementation

Zero Waste Scotland has produced a guide for effective communication on waste management. Below is a synthesis of key information from that guide (Zero Waste Scotland, 2012) distilled down into a sequence of five steps (Table 4-6).

Table 4-6. Five steps for delivering effective communication on waste management to citizens

(1) Evaluate current situation	• Classify local demographics – based on government statistics and information from local agencies/companies • Evaluate current waste (recycling) performance – based on monitoring data • Define collection systems and strategy in the focus area – consultation with relevant waste management operational staff • Evaluate current levels of awareness – research based on monitoring of facility usage rates, survey questionnaires, etc. • Identify key barriers to recycling in the focus area
(2) Define objectives	• Identify key waste management performance deficiencies identified from information gathered in (1) • Consult relevant waste management staff to target priority performance aspects and metrics for improvement • Identify key demographic group(s) or area(s) to drive improvement • Establish specific, measurable objectives linked with performance monitoring
(3) Develop communication strategy	• Link with national campaigns where possible to improve recognition • Develop a strong visual (brand) identity, including icons, using focus groups • Relate appropriate messages and mediums of communication to relevant objectives and target groups • Devise lists of actions for each message and target group, based on available resources and specified timeframe
(4) Deliver communications	• Deploy a range of appropriate actions as defined in (3) • Plan and organise specific events, carefully considering locations and timings to suit target audience • Brand all actions and information material using visual identity icons defined in (3) • Ensure strong overlap across events to maximise recognition and reinforce effectiveness
(5) Measure impact	• Evaluate the influence of particular campaigns on key performance indicators at the relevant geographic scale (if possible) • Seek feedback from target audience on campaign efficacy, during, immediately after, and some time after, the campaign is run • Document which actions or messages worked well, and which did not work so well

Target audience

Defining the target audience is a key step of any communication campaign. Campaigns may be more general, e.g. to advertise a new service, or highly targeted, e.g. to promote recycling within localities, such as an apartment block with a low recycling rate. Some target audiences may be difficult to reach or engage with owing to socio-economic circumstances and lifestyles, requiring additional effort such as door-to-door direct marketing.

Zero Waste Scotland (2012) provides the following guidance to select the most appropriate medium of communication for various target audiences:

- TV is good for targeting people across an entire region with the same message;
- radio, depending on its coverage, is better to target people in smaller areas, say a single local authority area (although broadcast areas will probably overlap with other local authorities);
- local weekly newspapers may target people in particular areas of a local authority;
- door-to-door canvassing is effective if used in a targeted way in relatively small areas;
- signage at recycling sites will only target people visiting that site.

In addition, social media is an effective channel through which to reach younger generations and office-based professionals who spend a lot of time "connected" to desktops and mobile devices.

General marketing/information campaigns

Coordinated and consistent use of positive slogans and sound bites can be an effective way of raising awareness and conveying simple messages to citizens. For example, WRAP in the UK has a "Love food, hate waste" campaign, which provides an overarching theme for many communication initiatives. Using an appropriate "tone of voice" is very important – lighthearted and encouraging messages work best (Zero Waste Scotland, 2012).

Messages must be designed to engage, inform, educate and motivate target audiences. According to Zero Waste Scotland (2012), an effective message should:

- be personal;
- be simple, clear and consistent;
- address barriers for the target audience;
- focus on a single action or an issue and how to overcome it.

Partners in the ZeroWastePro project have developed templates for waste management information campaigns, freely available to download from the following website: http://www.zerowastepro.eu/tools/.

Public engagement activities

WRAP has produced guidance for local authorities and waste management organisations on how to run public engagement activities promoting the prevention of food waste under the "Love food, hate waste - save more" campaign (WRAP, 2015b). The guidelines describe activities that address the topics of meal planning, best-before/use-by dates, food storage, portion sizes and using leftovers, and emphasises how reducing food waste can save money. A screenshot of the guide is shown in

Figure 4-18, and highlights how most of the best practice in relation to food waste prevention is concordant with good household management. Each activity has an appealing title, such as: "It pays to plan", "Too good to waste", "Your freezer is your friend".

Love Food Hate Waste - Save More

Activities	Ease	Save Us More For use with small groups, community groups, families and those living communally.	Save Me More For use on a one-to-one basis e.g. friends, family members and volunteers.	Saving Even More	PLANNING	DATES	STORAGE	PORTIONS	LEFTOVERS
1. **Most Wasted** A guessing game, that's good to start with, about the most wasted foods in the UK - and in our own homes ...	o	✓	✓	✓					
2. **It Pays to Plan** Self-scoring tick list and discussion about top tips for better planning.	oo	✓	✓	✓	✓✓	✓	✓		✓
3. **What's for dinner?** Meal planning exercise using magazine recipes to inspire, and a simple menu plan and shopping list to save time and money.	oo	✓	✓	✓	✓✓				✓
4. **Shopping Savvy** Top tips for before and whilst you shop and an activity about getting the best deal ... or is it?	oo	✓	✓	✓	✓✓	✓	✓		
5. **Keep the Date** Card activity that makes sense of date labels. Includes a store cupboard basics list and date labels fact sheet.	o	✓				✓✓	✓		
6. **Dates Round-up** Spin the bottle with date labels! Decide whether food is safe or good to eat ... or not.	oo	✓	✓	✓	✓	✓✓	✓		
7. **Too Good to Waste: Storage** Simple, verbal storage tip-sharing game requiring nothing more than a scrap of paper and a pen. A good ice-breaker.	o	✓					✓✓		
8. **Savvy Storage** Quiz and storage do's and don'ts with a reminder sheet to take home.	o	✓	✓	✓	✓		✓✓		
9. **Your Freezer is Your Friend** Can we freeze it? Yes we can! Mostly. Quiz about what can and cannot be frozen and good ideas for how to do it.	o	✓	✓	✓	✓		✓✓		✓

Source: WRAP (2015b).

Figure 4-18. Screenshot of a guide produced by WRAP providing an overview of various activities, highlighting suitability for different audiences and topics addressed

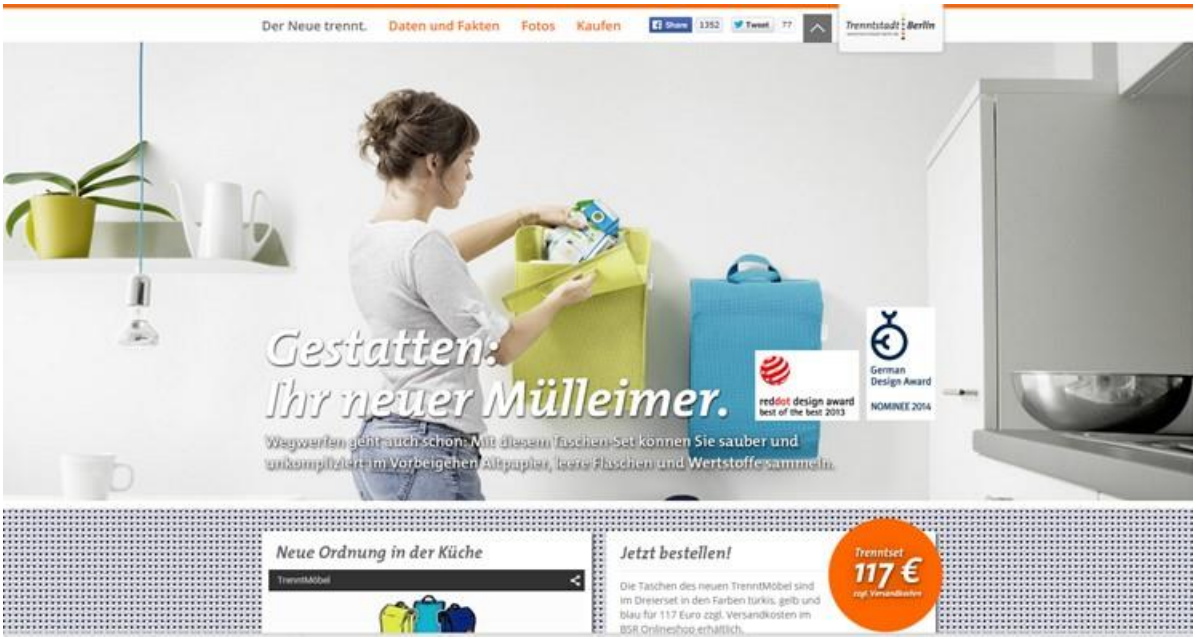
The ZeroWastePro project has produced similar guidance, in the form of a recommended educational programme template that can be implemented by schools or other public organisations:

http://www.zerowastepro.eu/images/educatioal_kit_24_06.pdf

BSR, the waste management utility in Berlin, started the *Trenntstadt* campaign in 2010, aimed at encouraging Berlin citizens to improve on already high (80 %) packaging recycling rates through a trendy campaign. BSR (2013) summarise the following attributes of their effective approach:

- avoid preaching;
- present waste sorting – the prerequisite for effective recycling – as a contribution to environmental protection;
- commend Berliners for their efforts and motivate them to continue waste sorting;
- use examples from Berlin to highlight issues of environmental protection and resource conservation;
- in addition to “classic” advertising, use new media, promotions and special campaigns.

The *Trenntstadt*[40] campaign makes extensive use of social media sites, and includes the marketing of attractive recycling storage bags as “fashion accessories” (Figure 4-19).



Source: http://www.trenntmoebel.de/

Figure 4-19. Screenshot of the online shop marketing fashionable recycling storage bags and bins as part of BSR’s *Trenntstadt* campaign for Berlin residents

[40] Trenntstadt is a pun: Trend = trend, “Trendstadt” = trendy city, trennt = separate

SYBERT in France has employed a range of media to deploy important waste management messages via humour. Among numerous videos is this example advertising the utility of "gourmet bags" (or "doggy bags"): https://www.youtube.com/watch?v=OBBdOvXCS_s

To the left there is a poster advertising a new campaign to take "selfies" with gourmet bags and post them on social media. This campaign by SYBERT and partner restaurants is intended to target younger generations with this important message to reduce food waste generation.

LIPOR in Portugal has created a public space (Horta da Formiga) dedicated to awareness-raising for good farming and environmental practices. The good practices promoted aim to increase the awareness of the population about biowaste generation, preventing organic waste production through home and community composting. The area measures 1 hectare and includes a number of different facilities, such as a composting area (with more than 15 demonstration composting bins) and a training centre. The awareness-raising activities are free for residents, schools, etc. to visit. Trainings are organised throughout the year, providing a set of different short theoretical and practical courses about composting, organic farming, sustainable gardening and sustainable cooking. All courses about environmental awareness are regularly organised and free to attend (Lopes, 2015).

Apps and online engagement

The "Don't bin it, bring it" campaign run by *Recycle Now* (2015) aims to raise awareness about where to dispose of small items of household WEEE. The campaign includes a webpage where citizens can type in their postcode to locate their nearest WEEE collection point (Figure 4-20).

Decision support tools can be used to highlight the environmental performance of alternative waste management options. Typically, these tools are more useful for businesses and waste management organisations than for the general public, but making them freely available to the public offers an avenue of information exchange for motivated citizens and for businesses. Three examples of such tools are:

- the Scottish Carbon Metric Calculator (Zero Waste Scotland, 2015);
- Benefits of reuse tool (WRAP, 2014);
- CO2ZW Calculator (ZeroWastePro, 2015).

Social media is becoming increasingly important as a form of communication, and as a cost-effective advertising medium. Examples of waste management communication campaigns via Youtube videos and Twitter feeds are given in Table 4-5.

Source: Recycle Now (2015).

Figure 4-20. Screenshot of "Don't bin it, bring it" website with a function to locate the nearest WEEE collection point

Vienna City Council provides an online map of recycling locations and collection points: http://www.wien.gv.at/stadtplan/.

Education for children

The city of Tallinn operates a *Waste Wolf (Prügihunt)* waste awareness campaign, which involves events, competitions, information seminars, public surveys and excursions to waste management facilities (R4R, 2014a). An important component of this campaign is the *Sustainable Consumption and Waste Information Trailer* which is a mobile learning class for children that is set up alongside *Waste Wolf* events. Pedagogical materials, including educational play cards and exercise books, are produced and updated every year by the Tallinn Environment Department. *Waste Information Trailer* presentations are delivered in spring and autumn, either in the trailer or in workshops, and are designed for children in kindergarten and elementary school (1st to 2nd grade). In addition, *Waste Wolf* visits nursery schools and schools to teach children about how to sort waste, consume and behave in an environmentally responsible manner through the use of games and interviews. Outreach activities are supported by a *Waste Wolf* mascot, online videos and a Facebook page. In 2013, 320 presentations were delivered and 6 691 children participated in the campaign (see photos below).

Source: R4R (2014a).

The city of Vienna also provides a range of children's activities and materials for application in school, kindergarten, holiday camps, sports and waste management facility settings (Vienna City Council, 2013).

In October 2013, the LIPOR Generation+ Project (PLG+) began in Portugal, with the aim of creating an educational programme for application in associations, educational institutions, social institutions or other organisations and entities interested in promoting better waste management (Lopes, 2015). The PLG+ programme promotes good environmental practices to citizens, facilitating the acquisition of skills and enabling greater civic intervention in order to promote the growth and consolidation of sustainable processes. Activities are based on four essential stages:

1. **Intervention Diagnosis**: aims to identify the set of needs of institutions, appoint points of improvement and build a plan for sustainable responses.
2. **Intervention Strategy**: development and implementation of methods and practices contained in the *Intervention Plans*, promoting significant changes in the community's environmental performance, ensuring effective results that facilitate the final certificate. This phase of the project is divided into two distinct strategic plans – *Initial Intervention Plan* and *Advanced Intervention Plan*, according to the initial evaluation of the institutions.
3. **Certification**: the conclusion of LIPOR and institutions' work, done by evaluating the results obtained and the consequent recognition of the effectiveness of these results, through the award of certification.
4. **Certification Management**: monitoring certified institutions and promoting a best practice maintenance plan, which ensures the continuity of good environmental behaviour in the institutions, allowing the certification renewal.

Features of the project considered innovative by LIPOR include the diversity of the target audience, the required development of the activity in a global network strategy, and the absence of a deadline for completion of the project – which is exclusively associated with the fulfilment of objectives, not compromising the normal activity development of these institutions.

The PLG+ currently involves 141 institutions and the intervention will reach over 40 000 citizens directly, consolidating LIPOR's regional strategy. So far, LIPOR has undertaken 137 environmental audits, covering 1 215 activities and 23 489 people. Waste separation is one of the most common actions across institutions, promoted by 97 % of participating institutions (Lopes, 2015).

Producer responsibility

Labelling is an important method of communication between producers and consumers that can be used to help reduce food waste and encourage appropriate recycling options. WRAP (2011) undertook a detailed study on the influence of labels on consumer behaviour in relation to food waste. They found that consumers could be confused about how best to store certain products (e.g. unaware that some fruit and vegetables are best stored refrigerated and/or in their packaging), and by "best before" and "display until" labels which could be confused with the more critical, food-safety-related "use by" dates. Unambiguous and prominent labelling by producers can reduce some of this confusion and therefore contribute to the avoidance of food waste (WRAP, 2011).

EC (2014) suggests that there is considerable scope for coordinated approaches for communication and awareness-raising across specific product streams, citing an obvious lack of harmonisation between WEEE and battery and accumulator PROs.

PROs are often established under extended producer responsibility policies, which may involve regulation in some Member States. Producers may be obliged to finance and coordinate communication and awareness-raising efforts, e.g. to reduce litter and improve source segregation by consumers (EC, 2014).

Applicability

All waste management organisations can employ communication to raise awareness about their services at some level.

Economics

Citizens

It is estimated that households in the UK throw away EUR 635 worth of food every year on average (WRAP, 2015b). Possible financial savings provide a strong motivation for waste prevention across all types of product category, and represent a useful focal point for information campaigns to encourage waste prevention actions, and leverage-related recycling actions.

Waste management organisations

Awareness-raising is an integral operational cost for all waste management organisations. Indeed, for private service providers it may be largely accounted for within the advertising budget.

Typical costs for a standard communication campaign are between GBP 1 and GBP 2 per household, where the average household size in Scotland is about 2.2 residents (Zero Waste Scotland, 2012). Therefore, communication costs for awareness-raising campaigns can be estimated as about EUR 1 per resident.

School activities and events may be paid out of national, regional or local government education budgets.

Producer responsibility organisations (PROs)

Most EPR schemes at least partly cover administrative, reporting and communication costs relative to the operation of collective schemes. According to EC (2014), this includes public information and awareness-raising (in addition to a PRO's own communication initiatives), to ensure participation of consumers within the scheme (i.e. through separate collection), and surveillance of the EPR system. The degree of “full cost coverage” by the producers in EPR schemes varies, depending on the distribution of responsibilities between stakeholders (EC, 2014). In Portugal, regulation requires that 5 % of PRO budgets must be dedicated to communication and awareness-raising activities (EC, 2014).

LIPOR’s PLG+ programme incurs relatively small direct costs for communication (EUR 3 000), but incurs significant personnel costs, with five technicians promoting and supporting the project (Lopes, 2015).

Driving force for implementation

The main driving force for this technique, as with most others referred to in this document, is to reduce waste generation and increase waste recycling, driven by regulations and/or financial considerations.

Economic factors are particularly important for this technique: improving the uptake of existing waste management services almost always improves economic performance.

Reference organisations

- BSR, Berlin, Germany, is a reference organisation for implementation of the *Trenntstadt* campaign that aims to engage younger and trend-conscious citizens in recycling efforts.
- Càmara Municipal de Lisboa, Portugal, is a reference organisation for efforts in educating schoolchildren in waste prevention and recycling through school campaigns (R4R, 2014c; Càmara Municipal de Lisboa, 2015).
- The Ecological Recycling Society in Attiki, Greece, ran a successful recycling campaign to reduce of packaging, biowaste, batteries and WEEE.
- SYBERT, France, has an extensive campaign educating citizens on waste management using various media, including theatre and videos.
- Tallinn City Council, Estonia, promotes waste awareness among children and adults with interactive outreach activities, including a touring trailer.
- Vienna City Council, Austria, uses a wide range of communication channels to raise awareness, ranging from humorous anti-litter campaigns to online apps displaying the nearest waste collection points.
- WRAP, UK, supports local authorities in the development of a wide range of communication activities, from online apps to workshops, and has developed a number of effective advertising campaigns including “Love food, hate waste”.
- Zero Waste Scotland, UK, similarly supports local authorities in engagement activities, and has directly developed a number of online tools to inform and engage citizens.
- Scania, a region in the south of Sweden, has a programme named 'cut the crap 2020' which implements measures for waste minimisation by changing attitudes of residents and providing methodological support to individuals.

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4.3.6. Establishment of a network of waste advisers

Summary overview
It is BEMP to set up a network of waste advisers (also called "waste (prevention) officers", "recycling officers", "waste (prevention) consultants") at local level in order to raise the awareness of the general public (residents and small businesses delivering their waste to the local MSW management system). The use of waste advisers is especially relevant to address specific issues by targeting a specific territory or audience with a poor separate collection rate or high contamination in separately collected fractions in order to deliver an adapted answer, as waste advisers can interact face to face. Waste advisers typically have a prior qualification in the environmental field as well as knowledge of the practices of waste minimisation, reuse and recycling, and can be volunteers, part-time or full-time staff. Waste advisers can perform a range of activities, such as: - make residents and small businesses aware of the environmental issues related to waste generation and management; - inform residents and small businesses about the waste collection rules and how the different fractions are treated and recycled; - provide residents and small businesses with guidance to identify possibilities to reduce or better manage (e.g. better source separation) their waste; - work with residents and small businesses on specific waste streams that are considered more problematic (food waste, textiles, nappies, etc.); - carry out engagement actions targeted to specific audiences (e.g. children/teenagers, pensioners, businesses, foreign-language speakers); - gain a better understanding of what happens on the ground (drivers, reasons, shortfalls).

Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**

Applicability
This BEMP can be implemented at any level. However, their scope of action is more focused on the local level since they address operational issues (waste prevention and recycling guidelines).

Specific environmental performance indicators
In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: - Share of population in the waste management catchment area advised by waste advisers over a given time period (e.g. % of population per month);

- Number of waste advisers per 100 000 residents.
Benchmark of excellence
- A network of waste advisers is in place with at least one waste adviser per 20 000 residents.

Description

Effective environmentally friendly management of municipal solid waste, as well as other types of waste, e.g. commercial waste, relies to a large extent on the individual choices and behaviour of citizens. Convenient waste collection infrastructure and efficient collection services are important facilitators in this process. Furthermore, in order to improve waste management performances, local authorities and waste management companies can further play a crucial role in:

- making citizens and entrepreneurs/small businesses **aware of the environmental issues** related to waste generation and management;
- **informing** them about the waste collection and treatment rules;
- **providing them with guidance** to identify possibilities to reduce or better manage their waste.

Different types of communication instruments are used by local authorities and waste companies: sorting guidelines, information letters, websites, etc. However, one of the most effective methods of communication is interactive face-to-face communication where citizens, entrepreneurs and small businesses, delivering their waste to the local waste management system, can directly engage in discussions about waste issues.

This BEMP deals with the setting up of a network of "waste advisers" (also called "waste (prevention) officers", "recycling officers", "waste (prevention) consultants") at local level in order to **raise the awareness of the general public** (citizens and entrepreneurs/small businesses delivering their waste to the local waste management system) in the field of waste management. The idea is to **target issues at source** through awareness-raising activities, instead of applying technical end-of-pipe solutions. Waste advisers can support the adoption of a correct waste hierarchy by residents (by focusing on prevention and reuse). Also, through direct interaction with them, they can achieve **more engagement and long-lasting behaviour changes**. This is therefore a continuing process that requires strategic and long-term planning.

The involvement of waste advisers is an effective way to disseminate waste management and prevention actions. They can be especially relevant to address very specific issues by targeting a specific territory or audience with a poor sorting performance or high contamination in order to deliver an adapted response.

Waste advisers can be **employees** of the local waste authority or waste management company. They can also be **volunteers** who receive some public or private funding. Employing full-time waste advisers usually implies creating a dedicated team/unit within the organisation, with an appropriate management structure and procedures to ensure good coordination with other relevant departments. Interaction with other actors and institutions outside the organisation also needs to be considered.

Waste advisers typically have a prior qualification in the environmental field as well as some degree of knowledge of the practices of waste minimisation, reuse and recycling. Depending on the exact tasks and setup of the network, project management may also be an important element of the required profile. The ability to communicate effectively and to present information in a clear, concise and straightforward manner are essential skills for waste advisers. Taking into account previous experiences and specific roles in the position, additional targeted trainings can be provided on a case-by-case basis (public speaking, improving presentation skills, use of specific analytical tools, etc.).

The activities performed by the waste advisers can be more general (raising overall environmental awareness) or targeted towards the following:

- Specific **waste management actions** such as prevention, reuse or recycling/source separation.
- Specific **waste streams** that are considered more problematic (food waste, textiles, plastic bags, nappies, etc.).
- The target can also be a specific **audience**. For instance, children and teenagers are seen as an important audience whose awareness of waste management issues needs to be particularly stimulated (because they are the future generation of citizens and can also have a big influence on others, for instance by "educating" their parents). Concrete communication actions could include open classes at schools and kindergartens, arranging educational visits to waste management facilities (composting/recycling sites), screening of educational movies, providing teaching resources to be integrated in the curriculum, etc.
  Further target audiences can be businesses or public entities, for which waste advisers can provide practical advice or develop tailored waste management plans and resource-efficient strategies. Average households can also be a target audience and best reached through personal visits in their homes to help individual citizens understand how to correctly separate waste in their own home environment.

In comparison to using conventional communication activities, there are a number of elements that make the involvement of waste advisers potentially more effective, especially in the long term, such as a consistent message, the possibility to develop expertise in different topics, feedback and capacity-building among the team and transfer of the accumulated knowledge externally (i.e. waste advisers acting as enablers). Elements that can be considered best practices for having an effective network of waste advisers include the following:

- **Holistic approach**: Even if some campaigns have a specifically targeted focus, all materials and waste streams should be taken into account within a broader environmental strategy. Awareness-raising actions should be prioritised in line with the waste management hierarchy. Focus should be on prevention and reuse.
- **Cross-cutting issues**: The activities of waste advisers should not only tackle waste but should also make connections to other environmental issues (including energy, biodiversity, climate, etc.) in an effort to achieve a real and lasting change of mindsets. The target audience's interests should also be taken into account (for example promoting reduction of food waste to save money, promoting reuse to stimulate local employment, etc.).
- **Consistency** of the message delivered by waste advisers in the territory should be sought, making sure that it is in line with the national/regional policy framework and existing technical and logistical solutions.
- **Coordination** with other organisations with the same aim in order to find possible synergies and enhance the effect of the communication.

- **Capitalising on the knowledge** waste advisers gain through their direct contact and work with the citizens in order to boost the general communication strategy and to identify specific possibilities for improvement.

Achieved environmental benefits

An effective and active network of waste advisers can foster good source separation of the different waste fractions by the general public as well as waste prevention. This leads to various environmental benefits, such as higher recycling rates, improved quality of the recyclable fractions collected, reduced quantities of residual waste and optimised management of associated treatment costs.

Although directly linking communication activities with changes in behaviour and related environmental benefits is challenging, as behaviour can be affected by many different external factors, there is evidence of the positive effects of introducing a system of waste advisers. For instance, there are reports of improved separate collection after the introduction of a system of waste advisers, or of a relation between the number of waste advisers per inhabitants and the waste management performance of an area (Schleich pers. comm., 2016). This is developed in the next section.

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- share of population in the waste management catchment area having contact with waste advisers over a given time period (e.g. % of population per month);
- number of waste advisers per 100 000 inhabitants.

Cross-media effects

The cross-media effects of waste advisers are considered marginal when compared to the environmental benefits resulting from their work.

These cross-media effects could include greenhouse gas emissions due to their travels, and printing of materials and creation of other material supports needed for educational purposes.

Operational data

This subsection will present information on actual successful implementation of waste adviser networks in various local and regional authorities across the EU.

Austria

In Austria, municipal waste advisers are seen as one of the biggest success stories in public waste management. Over a period of three decades since they were first created they have been contributing to raising separate collection rates (in some regions raising them from around zero to over 70 %), saving costs and generating new follow-up jobs.

Municipal waste advisers were first established in the country in 1986 as permanent full-time employees of regionally or locally based public waste authorities. They can be employed in public entities at different levels:

- municipalities/local authorities;

- towns with more than 3 000 inhabitants;
- cities;
- associations of towns/districts;
- provincial authorities;
- associations under public contract;
- waste management entities at a municipal level.

Since the beginning, the underlying idea of employing waste advisers was to use human resources prior to legal restrictions and industrial investments to minimise environmental problems and reduce public expenses (“prevention” instead of “end-of-pipe-treatment”). The concept is: educating the population to prevent and separate waste instead of paying for expensive technical solutions to deal with the waste once it has already been (incorrectly) disposed of. As of 2016, 410 municipal waste advisers are the backbone of public waste management communication and public relations (PR) work. This means an average of one adviser for 20 000 inhabitants.

Municipal waste advisers can also cover other environment-related areas such as sustainability and consumption, but their main focus is on awareness-raising, public education of the population and PR in the field of municipal waste management. Their communication work is focused on waste prevention, reuse, separate waste collection and sustainable consumption and lifestyles in general within the local/regional context. Their target groups are children from schools and kindergartens, private households and small and medium-sized enterprises in their region. Interaction can be either direct (personal) or via dedicated service hotlines or electronic newsletters. Additionally, they consult their regional waste management organisations in planning and implementing collection schemes, and communication projects and campaigns. They further cooperate with private waste management companies and provincial and federal authorities for the development of (innovative) waste management strategies and concepts.

Waste advisers in Austria typically receive a dedicated training. During the years between 1986 and 1995, it was a six-month training programme. Partly due to shrinking public funding and saturation of the job market, this initial permanent training programme has progressively been substituted by shorter training courses and learning "on the job".

Nuremberg - Germany

A network of waste advisers is present across the whole of Germany. According to national legislation (Law on Circular Economy and Waste), every municipality is obliged to have waste advisers. They are employees of the municipality or of the public waste management company and inform companies, private households and institutions, associations, schools and the like on issues of waste prevention, recovery and treatment (Abfallberatung, personal communication, February 2017).

In Nuremberg, fr example, the public waste management company employs a number of permanent waste advisers. They are in charge of communication with the households – their work includes giving presentations and being in direct contact with the citizens in order to explain prevention measures and to inform them about correct waste separation. They can also be present in schools (on invitation); for this the waste advisers have developed targeted information materials and several different programmes depending on the age of the children, level of knowledge and time available. Moreover, the Public Waste Management Authority of the City is in partnership with a City Environmental Museum for Children which includes a section on resources and waste. The Authority subsidises the entrance to the museum so that children from schools and kindergartens can enter with a reduced fee. Waste advisers are present on site to lead different activities. Finally, waste advisers also provide tailored consulting services to commercial and industrial entities on demand.

In addition to permanent staff, trained advisers have been supporting the work of permanent waste advisers on a voluntary basis since 1991. The voluntary waste advisers operate an "Infomobil" (special waste information vehicle), give advice at different events and take part in different campaigns. Since there is an important migrant population, it is important to ensure that foreign language speakers are equally well informed about good waste management practices. Foreign language competency is therefore an important aspect when considering the choice of volunteers.

North London - UK

The North London Waste Authority (NLWA) comprises seven North London districts (boroughs) covering a population of nearly 1.9 million residents. The NLWA is responsible for helping the seven North London boroughs dispose of the 827 000 tonnes of waste they collect every year. In 2012 the NLWA established a team of waste advisers with the objective to deliver the waste prevention message through direct contact with residents (partially or fully replacing an equivalent team in each of the boroughs) (NLWA, 2016). The team consists of five people, works according to an elaborated plan and is funded through levies coming from boroughs for disposal costs.

The team is guided by a plan of activity for two years with an associated budget and works in close partnership with all NLWA boroughs. The elaboration of the plan starts in June of the previous year with a number of consultation meetings with the seven borough councils. At this stage the discussions are more strategic; the aim is to define the global priorities for the period to come. Several considerations are taken into account: EU and national policies, legislation, available research evidence, etc. The starting point is the waste hierarchy which means that the main aim is always to prevent and reduce waste at source. In consultation with different stakeholders, a decision is taken on where to focus the actions in the upcoming period. This decision prioritises waste streams by taking into account several aspects, i.e. whether:

- they are generated in large quantities;
- they have high tonnage diversion potential;
- they provide financial savings to residents;
- they are difficult to reintroduce into production cycles;
- regulatory and legislative instruments already exist;
- they have seen a significant increase in generation in the recent years; and
- they are emblematic waste streams that provide the opportunity to promote further waste prevention and recycling messages.

Once the priority streams are decided, four types of measures and instruments are identified – technical, economic, educational/behaviour and organisational (involving various stakeholders and institutions) – and a number of concrete actions are derived from these measures and instruments. Once the key programme of activity is developed according to this procedure, another consultation phase starts with all relevant bodies in order to identify concrete steps for implementation on the ground.

This phase also allows identification of similar ongoing actions in order to combine and optimise resources. This is also a time to define which actions are going to be performed in-house and which will be subcontracted.

Priorities emerged based on reduction of waste streams, with each person in the team having a specific focus:

- two people are in direct contact with the **citizens** (recycling and engagement waste prevention officers);
- two people deal mostly with **subcontractors and other stakeholders** and organisations performing waste prevention activities (waste prevention officers);
- one person **coordinates and manages** the work, and is responsible for the preparation, execution and evaluation of the work programme.

The programme itself is constantly reviewed and monitored, and there is an effort to evaluate the outcomes and to measure what can be achieved and improved. There is also a continuous communication and exchange with relevant actors: every two to three months regular updates are provided to key stakeholders, bilateral meetings with members of each borough are held twice a year, etc. The waste prevention officers themselves hold quarterly meetings in order to coordinate, exchange information and share best practices for waste prevention, recycling and education. At the end of each implementation period a final report and a brochure are produced to provide a summary of the year's activities. These documents are widely distributed to all involved and interested stakeholders.

Training is delivered during the first six months in the position. It involves information on how the areas operate, many site visits (reuse and recycling centres, material recycling facilities, etc.), as well as individual meetings with representatives of each of the seven boroughs.

The idea for the future is that waste prevention officers not only deliver trainings, knowledge and information to other people/organisations, but that they also act as "enablers" – enabling others to deliver and the local community to be able to set up and run projects itself. The focus will therefore not be on getting more staff in-house, but to enhance what others are already doing to complement and increase the overall impact.

Brussels-Capital Region - Belgium

In view of the observation that businesses do not necessarily have the time or knowledge required to optimise the prevention and management of their waste, the 2010 Waste Management Plan of the Brussels-Capital Region included a proposal for putting waste adviser services at the disposal of businesses. This is how the Brussels Waste Network programme was created as a joint initiative of the environmental administration of the region (Bruxelles-Environnement) and the Brussels Enterprises Commerce and Industry in Brussels (BECI). The aim is to organise and coordinate a network of waste advisers that corresponds to the needs and challenges faced by businesses in the region. The implementation and operations of the network are based on the assumption that waste and related problems differ across the sector of activity but that businesses within the same sector face similar problems. This is why waste advisers have been created within sectoral/business federations and in addition a dedicated waste adviser at BECI plays a coordinating role for all running projects and all advisers who are part of this network.

The main objectives of the Brussels Waste Network are to:

- develop a platform for exchange of information between the private and the public sector;
- manage a network of "waste resources" advisers from different sectors of activities and encourage the sharing of best practices among them;
- offer information and advice in terms of prevention and management of waste/resources (via service helpdesk, webpage, newsletter);
- disseminate practical tools to prevent and manage waste/resources.

The Brussels Waste Network organises regular calls for projects which lead to the creation of different tools, allowing companies to improve their waste management. Altogether these projects have achieved the following results since the creation of the programme:

- 140 businesses in Brussels have taken effective actions to improve their internal waste management;
- 55 persons within the companies participating as project developers have been trained or involved;
- 15 tools for improved waste management have been created;
- 41 plans of action have been implemented in businesses.

These measures have had a beneficial impact on different aspects of internal waste management, for instance enhancing prevention/reduction of waste or improving separation at source (for better reuse/recycling).

Applicability

The introduction of waste advisers in a territory is a BEMP that can be implemented at any level. However, their scope of action is more focused at the local level since they address operational issues (waste prevention and recycling guidelines). No specific instrument is required to set up a network of waste advisers; the main issue is rather how to fund their operations.

National/regional subsidies or financing through PROs can contribute to the development of such networks. The latter can be on a voluntary or regulatory basis. For instance, specific legislation in some countries (e.g. Austria, France) requires that EPR schemes for household packaging contribute to finance the activities of waste advisers at local level since their tasks involve communication on packaging waste management, which falls under the responsibility of the respective scheme.

According to the experience of organisations that have successfully deployed waste advisers networks (e.g. North London and province of Styria, Austria), it is better to implement it on a larger scale – at least a region, province or big city (in both cases mentioned above a territory of more than 1 million inhabitants). This is seen as beneficial to ensure the optimisation of resources, the economic feasibility of the development and implementation of a qualification/training programme as well as the continuity of step-by-step implementation of waste advisers in all regions and municipalities. For instance, in North London in 2008 each of the seven boroughs had a recycling officer, an education officer and a waste prevention officer. Today, the resources are reduced because the councils have had to make significant budget cuts. As a result, nearly all positions have been removed and a team of only five waste prevention officers acts at the level of the NLWA instead.

On the other hand, it has to be ensured that the territory covered by the action of waste advisers shares the same objectives and sets similar priorities.

It is also important to have good collaboration between the different stakeholders involved and a good flow of information between them in order to achieve synergies and avoid inconsistencies or duplicating work.

Economics

Economic data mainly includes staff costs and the production of communication material and the organisation of events. The practice of involving waste advisers also involves indirect financial benefits as a consequence of the positive effects they achieve (improvement of separate collection, better recycling, etc.) and the related decrease of costs for waste treatment and/or increased revenues from sale of materials, but no actual data on these could be obtained.

Austria

In Austria, the initial funding in the first years was provided by the Federal Labour Agency (AMS), serving to finance concept and qualification measures. In the starting period until 2000, the AMS continued to provide funding for staff costs for consultants during training and employment in municipalities amounting to either 50 % of total staff costs for one year or 30 % for two years. After the AMS funding expired in 2000, the municipalities which employed waste advisers were responsible for the financing and there were limited staff cost contributions from packaging waste collection scheme(s) (amounting to around 20–30 % of total staff costs). In Styria, one of the provinces, there was a limited provincial subsidy until 2008 (amounting to 10 %) (Styria, 2014).

Currently, the financing of the staff costs comes from the overall municipal waste management budget which in Austria consists of residual waste fees from households and small enterprises (larger enterprises are fully self-responsible for their waste and are usually not covered by municipal waste management). The mandatory federal guidelines for municipal waste fee calculation also include the costs for waste advisers. Since 1993 the packaging collection scheme(s) has partly contributed to the staff costs, and in return the municipalities provide the service of also covering the communication work for prevention and collection of packaging waste which is legally the obligation of the scheme(s).

North London

The team of waste officers is financed through the levy for disposal of waste, which is itself the responsibility of the NLWA. The interests are therefore aligned to reduce the generation of waste, have better separation at source and consequently reduce expenses for treatment.

Driving force for implementation

Organising a network of waste advisers is a way to improve the communication strategy of the waste authority. Waste advisers act as a friendly interface between the waste producers and the waste management systems and are generally put in place to address very specific issues, e.g. the improvement of sorting performances, the reduction of contamination in the sorted fractions, or the implementation of participative actions such as home composting. It is an interesting way to address more complex or unpopular issues or to target areas of the population that are not complying with the requirements of the waste strategy.

It can also be used to help with the implementation and the coordination of other technical or financial instruments, e.g. a new collection scheme or a PAYT system.

Moreover, developing a network of waste advisers allows the creation of jobs in the environmental sector.

Austria

The concept of "municipal environment and waste advisers" was invented as an innovative solution to a number of severe waste problems Austria was facing in the 1980s, which were causing broad political discontent. The waste advisers' network was implemented within a decade, transforming the public discontent into highly motivated action and contributions of the majority of citizens to separate waste collection. Subsequently this led to political acceptance of building new waste treatment facilities and even landfills with the highest technical standards of that time. In the early years, the Federal Labour Agency (AMS) provided an important financial contribution within a broad national initiative for the creation of new and innovative jobs. This happened against the background of rapidly rising unemployment rates (also within well-qualified groups) in the 1980s and early 1990s. The funding was a long-term political commitment of the AMS within a long-term general national funding programme for the creation of new jobs ("Aktion 8000", the "experimental labour market policy"), which facilitated funding for municipalities intending to employ waste advisers and send them on the training programme.

Between 1990 and 1993 some provincial waste laws (Styria, Salzburg, Tirol, Upper Austria) integrated obligations for municipalities or regional municipal associations to provide waste management advice for their populations. Meanwhile all provincial waste management plans as well as the federal waste management plan and integrated prevention programme contain further detailed provisions on waste management advice.

North London - UK

The position of "waste officer" was created in 2007 and each of the seven NWLA boroughs had dedicated staff working with waste prevention. The NLWA itself had only one such employee who was working with and relying on the staff of the seven boroughs. Around 2010, because of the recession and related restructuring, many positions of officers dedicated to waste prevention were removed. At the same time, the NLWA received funding of GBP 200 000 from WRAP to run a food waste prevention campaign. It was very well received and had positive outcomes. Given the positive results achieved through optimised resources (although local budgets were increasingly tightening after the financial crisis), it was decided to build a in-house team and to continue its operations with a slightly modified role (as compared to the previous situation).

Brussels Capital Region - Belgium

Waste management in a company is often synonymous with high costs. However, an adapted strategy within the company generally allows a decrease in the quantity of generated waste and related costs. As a consequence, this also leads to the company having a reduced environmental impact, to financial savings, to compliance with existing relevant legislation and not least to a “greener” image in the eyes of the consumers/general public. However, business managers often do not have the technical knowledge necessary to develop adequate waste management policy for their companies and therefore access to qualified advisory services can be very beneficial.

Reference organisations

Waste advisers with a focus on household waste and awareness-raising of citizens:

Austria – Styria: the province of Styria has developed a network of waste advisers since the early 1980s (Styria, 2014). It is regarded as one of the key instruments for the development of the Styrian Waste Strategy.

- The Office of the Federal State Government of Styria (Division Waste Management and Sustainability). More information available: http://www.awv.steiermark.at/cms/ziel/27514100/DE/
- Austrian Association of Waste Prevention (ARGE): the association developed the first training concept for municipal waste consultants. More information available: www.arge.at/
- Austrian Association of Waste Consultants (VABÖ): the association representing municipal advisers on environment and waste in Austria. All municipal waste consultants in Austria are members of VABÖ with the main aim to foster the exchange of experiences and ideas. More information available: http://www.vaboe.at/

France

- Trivalis: the public authority that manages municipal waste in the department of Vendée. It manages a team of waste advisers with a specific focus on school campaigns: http://trivalis.fr/pedagogie-scolaire/
- Eco-Emballages and Ecofolio: French EPR schemes for packaging and graphic paper which provide financial support for waste consultants.

Germany

- Abfallberatung: communication and information exchange platform for waste advisers in Germany: http://www.abfallberatung.de//kommunen/kommunen.aspx
- Nuremberg: waste advisers on a volunteering basis: https://www.nuernberg.de/internet/abfallwirtschaft/abfallberatung.html

Italy

- Hera SpA (an Italian waste management company), in cooperation with some municipalities in the Emilia Romagna Region (e.g. Modena, Bologna, Ferrara), has trained and involved volunteers as waste advisers: http://www.comune.modena.it/salastampa/archivio-comunicati-stampa/2016/8/raccolta-rifiuti-il-bilancio-dell2019attivita-di-gel-e-gev/#null

UK

- NLWA: North London Waste Authority: The NLWA has an extensive programme of awareness-raising activities on most aspects of waste prevention, with a particular focus on food waste, bulky waste and textiles. A team of waste prevention officers and advisers has been set up to deliver the waste prevention message through direct contact with residents: http://www.nlwa.gov.uk/about/authority-services

- WRAP: Waste and Resources Action Programme, "Organisation helping businesses and individuals reduce waste, develop sustainable products and use resources in an efficient way"; it provides information and guidance to help local authorities deliver waste and recycling services, notably trainings for recycling officers: http://www.wrap.org.uk/

Waste advisers acting as consultants for businesses:

Belgium – Brussels-Capital Region:

- Brussels-Environment: the environmental administration of Brussels-Capital Region has concluded a partnership with the Brussels Enterprises Commerce and Industry in Brussels (BECI) to support businesses in prevention and management of their waste by creating a special programme (Brussels Waste Network): http://www.environnement.brussels/thematiques/batiment/la-gestion-de-mon-batiment/pour-vous-aider/brussels-waste-network
- Brussels Waste Network: initiative of the Minister of Environment, Brussels-Environment and the BECI in Brussels. The aim of this programme is to inform, to develop and to encourage a network of "waste advisers" who are employed at different sectoral federations and companies: http://www.brusselswastenetwork.eu/ and http://document.environnement.brussels/opac_css/elecfile/IF_BrusselsWasteNetwork_FR

Ireland:

- The Southern Waste Region: The region currently employs a Regional Industrial Waste Minimisation Officer (RIWMO) who works specifically with the business sector across the region in order to raise environmental awareness among employees and assist companies in their waste reduction programme. Environmental Awareness Officers (EAOs), based in each of the local authorities within the region, also work with the business sector in pursuit of best environmental practice, and work closely with the RIWMO. The RIWMO has set up a number of Networks and issues a newsletter two to three times per year. The region also employs a Waste Prevention Officer who has responsibility for implementing the EPA-funded Local Authority Prevention Network (LAPN) programme which works on the delivery of specific prevention initiatives. The EAOs based in each of the local authorities within the region also work with the programme. The region also funds a number of prevention and reuse programmes.

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4.3.7. Home and community composting

Summary overview
In cases when home and community composting is the most appropriate waste management option for biowaste based on the waste management strategy adopted and/or on an LCA study on waste management options (see Sections 3.3.1 and 3.3.2), it is BEMP to: - Systematically deploy and promote home and community composting, keeping track of the number of residents involved, registering where composting equipment is installed and operated. - Organise initial awareness-raising campaigns through graphic material, public meetings, waste advisers, etc. (see Sections 0 and 4.3.6) informing and training residents about home and community composting, its benefits, its correct operation (in order to limit methane emissions and pollution to soil, and ensure that the output is good quality compost), which biowaste is suitable, etc. - Regularly update and train residents on the correct operation of home and community composting. - Regularly monitor home and community composting sites. A number of representative sites can be inspected every year to check the correct operation of composting and ensure its environmental benefits.

Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**

Applicability
In cases when home and community composting is the most appropriate waste management option for biowaste, there are no major restrictions to implementing this BEMP. However, the success of home and community composting as an environmental management strategy is highly dependent on the management of the waste separation and composting process by citizens who must be first engaged to motivate them to separate organic waste, and then trained to correctly manage the composting process. Additional effort is required to organise home and community composting in urban areas.

Specific environmental performance indicators
In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: - share of population doing home composting or to which community composting is available (% of total population in the waste management catchment area); - share of population implementing home/community composting correctly, on

the basis of an annual visit and analysis of the compost produced (% of the population doing home composting or to which community composting is available); - system in place for regular follow-up with residents doing home composting (y/n); - Share of home composters visited annually (% of the households doing home composting).
Benchmark of excellence
- All residents have access to either separate collection of biowaste or home and community composting of biowaste.

Description

Home and community composting refers to the composting (i.e. the managed aerobic decomposition) of domestic organic waste from kitchens and gardens by householders or in small community composting facilities. Home and community composting avoids the economic costs and environmental burdens associated with organic waste collection. Home and community composting can be adopted when other biowaste management options (anaerobic digestion and centralised composting) are less appropriate based on the waste management strategy adopted and/or an LCA study on waste management options (see also Sections 3.3.1 and 3.3.2 on the establishment of a waste management strategy and life-cycle assessment of waste management options).

A major advantage of home and community composting in regions with low organic waste recycling rates is that it can generate "buy-in" from citizens who are otherwise less likely to separate organic waste, thus significantly decreasing residual waste volumes and increasing overall recycling rates (SYBERT, personal communication 2015). Such an effect could be particularly important among lower socio-economic classes in inner city areas (WYG Environment, 2011). Another important benefit of home and community composting is the replacement of peat used in hobby gardening (Andersen et al., 2012).

When implementing home and community composting, it is BEMP to:

- Systematically deploy and promote home and community composting, keeping track of the number of residents involved, registering where composting equipment is installed and operated.
- Organise initial awareness-raising campaigns through graphic material, public meetings, waste advisers, etc. (see Sections 0 and 4.3.6) informing and training residents about home and community composting, its benefits, its correct operation (in order to limit methane emissions and pollution to soil, and ensure that the output is good quality compost), which biowaste is suitable, etc.
- Regularly update and train residents on the correct operation of home and community composting.

- Regularly monitor home and community composting sites. A number of representative sites can be inspected every year to check the correct operation of composting and ensure its environmental benefits.

Source: E3 Environmental Consultants Ltd

Figure 4-21. Example of a community composting point in Besançon, France

Achieved environmental benefits

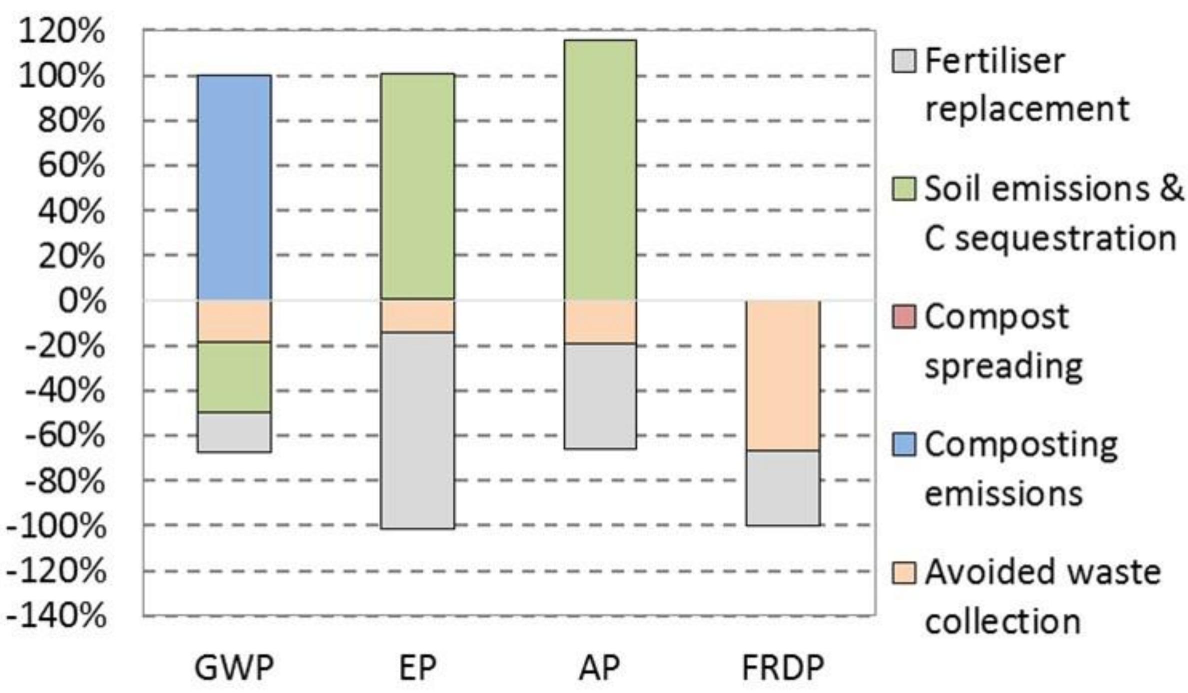
Life-cycle assessment of composting

Table 4-7 and Figure 4-22 summarise life-cycle environmental burdens and credits for home composting of organic household waste (OHW), comprising food waste and green waste, based on data from various sources. Some aspects are uncertain and highly dependent on specific management practices. Although EC (2010) reported significant methane and ammonia emissions for in-vessel composting, Andersen et al. (2012) report negligible ammonia emissions and variable methane emissions of between 0.4 kg and 4.2 kg per tonne of wet OHW. These emissions are highly dependent on process management and can be minimised under best practice. The proportion of organic N added to soils in compost that replaces fertiliser manufacture and application is highly dependent on the type of land to which the compost is applied, the precision of any nutrient management planning applied to calculate fertiliser application rates, and the period of time considered. In the short term (two years), only 11 % of organic N is likely to be available to plants and could potentially replace fertiliser-N (Nicholson et al., 2013). But, over the longer term, organic N mineralisation could result in considerably greater fertiliser-N replacement. For the LCA calculation here, it was assumed that 20 % of organic N could replace fertiliser-N in the long term (Andersen et al., 2012). Unlike centralised composting, home composting does not require diesel or electricity input (unless an automatic composter is used).

Table 4-7. Environmental burdens and credits calculated for home composting using life-cycle assessment

Environmental burdens	Environmental credits
• Methane emissions during composting	• Avoided fertiliser manufacture and

of 2.3 kg CH_4-C per tonne of wet waste, median of 0.4 kg to 4.2 kg CH_4-C reported in Andersen et al. (2012). • Nitrous oxide emissions of 0.075 kg N_2O per tonne of wet waste (Saer et al., 2013), which corresponds closely with an N_2O-N emission factor of 0.6 % total N cited in IPCC (2006). • Ammonia volatilisation during spreading equivalent to 3.6 % of compost N (Nicholson et al., 2013). • Soil N_2O emissions of 1 % of applied N (Tier 1, IPCC, 2006). • Nitrate leaching based on Nicholson et al. (2013) for food/green compost.	application emissions based on long-term fertiliser replacement values of 20 % for applied N (Andersen et al., 2012), and 50 % and 80 % for applied P and K, respectively (Nicholson et al., 2013). • A long-term (100-year) soil organic carbon sequestration credit equivalent to 14 % of C in the compost (Bruun et al., 2006; Møller et al., 2009). • Avoided food waste collection (7.2 litres of diesel per tonne).



N.B. This represents the best case where available nutrients in the compost replace fertiliser application.

Figure 4-22. Environmental credits (negative values) and burdens (positive values) for home composting of organic household waste across four environmental impact categories (global warming potential (GWP), eutrophication potential (EP), acidification potential (AP), fossil resource depletion potential (FRDP)).

Based on Table 4-7, the following net burdens were calculated for composting one tonne of OHW (wet weight basis):

- global warming potential: 32 kg CO_2e;
- eutrophication potential: 0.0 kg PO_4e;
- acidification potential: 0.18 kg SO_2e;
- fossil resource depletion potential: -359 MJe.

Thus, home composting leads to relatively minor net burdens across three of the four impact categories considered, and a significant fossil resource depletion of -359 MJ equivalent per wet tonne of OHW composted if the avoidance of waste collection is considered. However, there is considerable uncertainty over CH_4 and N_2O emission factors. If the highest CH_4-C and N_2O-N emission factors reported in Andersen et al. (2012) are applied, then the GWP of home composting increases over tenfold to 331 kg CO_2e per wet tonne of OHW.

However, Andersen et al. (2012) reported a modest additional GWP credit for OHW compost on the assumption that approximately 20 % of the home compost produced in Denmark replaces peat used in hobby gardening.

Comparison with alternative waste treatment options

Andersen et al. (2012) found that home composting performed comparatively well against landfilling and incineration in terms of nutrient enrichment, acidification and ecotoxicity in water, but less well in terms of GWP owing to energy recovery from the other two options in the Danish context. However, under a scenario of some landfill methane leakage, perhaps more typical of European landfills overall, composting performed considerably better than landfilling in terms of GWP.

Biogas electricity generation can avoid 1 227 MJe of fossil energy per tonne of food waste. Anaerobic digestion thus performs considerably better than composting in terms of global warming potential and fossil resource depletion, but less well in terms of eutrophication and acidification owing to ammonia emissions from digestate. Styles et al. (2015) calculated the following life-cycle net environmental burdens for anaerobic digestion of one wet tonne of food waste:

- global warming potential: -95 kg CO_2e;
- eutrophication potential: 0.5 kg PO_4e;
- acidification equivalent: 0.59 kg SO_2e;
- fossil resource depletion potential: -1,340 MJe.

Soil quality improvement

Compost returns almost three times more carbon to the soil than digestate, per tonne of food waste treated, leading to greater soil quality improvement, which will lead to indirect environmental benefits in terms of soil biodiversity and functioning, including crop yields, not accounted for in the above LCA.

The greatest degree of soil improvement and associated environmental benefits arise when compost is applied to soils with a low organic matter content, especially heavily cultivated soils on arable farms. Although compost produced by home and community composting is more likely to be used locally, in household or public gardens, this may result in more compost being available elsewhere for agricultural use via market displacement. Communal home and community composting schemes could also provide compost (free or at a price) to local horticulture enterprises.

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- share of population doing home composting or to which community composting is available (% of the population in waste management catchment area);
- share of population implementing home/community composting correctly, on the basis of an annual visit and analysis of the compost produced (% of the population doing home composting or to which community composting is available);
- system in place for regular follow-up with residents doing home composting (y/n);
- Share of home composters visited annually (% of the households doing home composting).

Cross-media effects

Home and community composting has the objective of producing compost of a good quality which concerns three main aspects:

- organic amendment properties;
- fertilising effect;
- innocuousness of its application on land.

However, if home and community composting is not correctly implemented it may generate compost of a low quality, generating a negative environmental burden (IRSTEA, 2012).

Operational data

Home composting

Waste management organisations can promote home composting by providing free or low-cost equipment, such as small kitchen bins and composting bins, alongside information that may be disseminated by posted leaflets and web pages. An example is provided by Leicester County Council, referred to under "Reference organisations". They support a home composting club and disseminate a short 10-page illustrated guide produced by WRAP to promote home composting (Figure 4-23).


6

MAKING GOOD COMPOST

The key to good compost lies in getting the mix right. You need to keep your greens and browns properly balanced. If your compost is too wet and gives off an odour, add more browns. If it's too dry and is not rotting, add some greens. Air is essential to the composting process and by mixing material up, as you fill your bin, it will create air pockets and help keep your compost healthy.

PUT THESE IN

Like any recipe, your compost relies on the right ingredients to make it work. Good things you can compost include vegetable peelings, fruit waste, teabags, plant prunings and grass cuttings. These are considered "greens." Greens are quick to rot and they provide important nitrogen and moisture. Other things you can compost include cardboard egg boxes, scrunched up paper and small twigs. These are considered "browns" and are slower to rot. They provide fibre and carbon and also allow important air pockets to form in the mixture. Crushed eggshells can be included to add useful minerals.

'Greens'

- Tea bags
- Grass cuttings
- Vegetable peelings, salad leaves and fruit scraps
- Old flowers and nettles
- Coffee grounds and filter paper
- Spent bedding plants
- Rhubarb leaves
- Young annual weeds (e.g. chickweed)

7

'Browns'

- Crushed egg shells
- Egg and cereal boxes
- Corrugated cardboard and paper (scrunched up)
- Toilet and kitchen roll tubes
- Garden prunings
- Twigs and hedge clippings
- Straw and hay
- Bedding from vegetarian pets
- Ashes from wood, paper and lumpwood charcoal
- Sawdust and wood chippings
- Wool
- Woody clippings
- Cotton threads and string (made from natural fibre)
- Feathers
- Vacuum bag contents
- Old natural fibre clothes (cut into small pieces)
- Tissues, paper towels and napkins
- Shredded confidential documents
- Corn cobs and stalks

KEEP THESE OUT

Certain things should never be placed in your bin. Do not put in...

- Cooked vegetables
- Meat
- Dairy products
- Diseased plants
- Dog poo or cat litter, or baby's nappies.

Putting these in your bin can encourage unwanted pests and can also create odour. Also avoid composting perennial weeds (such as dandelions and thistles) or weeds with seed heads. Remember that plastics, glass and metals are not suitable for composting and should be recycled separately.

Figure 4-23. Screenshot of one page from the WRAP guide to composting at home

Communal home and community composting

WRAP (2008) found that households with larger gardens are more likely to compost waste than households with smaller gardens. In urban areas where a large proportion of the population live in apartment blocks, there are obvious constraints to home composting. However, these can be overcome by implementation of community or district composting schemes, which may achieve various social and educational benefits besides diverting waste from the residual waste stream.

SYBERT is a waste management company located in Besançon in France. They are undertaking various initiatives to overcome the challenges of community and urban composting, and have established over 230 community compost points throughout Besançon, including the examples below.

 Source: *E^3 Environmental Consultants Ltd*	As of 2015, 11 composting sheds were installed in very dense areas, with 10 of them in operation. 5 380 households have access to them, representing about 10 450 people. Among these, 24 % participate in their operation. There are 3 sheds in the city centre, 3 in the Chaprais district, 2 in Planois and 2 in Palente centred around dense social collective housing. These sheds are open two to three times per week at convenient times (including Wednesdays and Saturdays) for local residents to bring food (excluding meat, fish and dairy to avoid rat infestations) and green waste. Volunteers from the local community manage the stations during opening times to ensure the correct waste is fed to the closed-shed composters, and also to turn the compost. Wood chips are added to ensure structure and aerobic conditions, and waste is composted over six months, and compost used for local community areas and by residents.
As of June 2015, 251 collective composting facilities within the apartment buildings were in service. These facilities are managed by two volunteers each and are open all the time, with a 40 % use rate. It is a challenge to find volunteers, who need to be trained for a few days on compost management, and guided for the first year. In total, 8 901 households (about 22 000 inhabitants) have access to composting facilities within the apartment buildings. Since 2012, 740 tonnes of organic waste have been diverted from residual waste incineration through these facilities.	 Source: *E^3 Environmental Consultants Ltd*

 Source: *E[3] Environmental Consultants Ltd*	One automatic rotating drum composter is in service at a large apartment block, serving over 2 000 households. This is opened three times per week to receive waste, including meat, fish and dairy products, along with wood pellets for structure/aeration. Leachate enters the sewer. Compost is generated over four weeks, leaving the composter only after it has achieved a temperature of 50 °C, followed by three to four weeks maturation in outdoor boxes.

Source: SYBERT (2015).

In total, nearly 30 % of households living in collective housing have access to one of these three types of local composting, representing more than 33 000 inhabitants. In 2014, 330 tonnes of organic waste were diverted from incineration through home and community composting.

Source: WORMS (2015).

Figure 4-24 presents a map of district composting locations in Brussels, from a screenshot on the WORMS (Waste Organic Recycling and Management Solutions) website. WORMS is an organisation that promotes composting of household organic waste in Belgium.

Source: WORMS (2015).

Figure 4-24. Screenshot of district composting locations in Brussels.

Training

The example of SYBERT (above) included significant efforts in training of local volunteers to manage home and community composting facilities, and awareness-raising among citizens about how to separate and manage their organic waste. In the case study of organic waste management in Flanders promoted by Vlaco detailed under "Reference organisations", it can be seen that Vlaco train volunteer 'Master Composters' to inform local citizens on management of home composting systems. Over 2 700 of these Master Composters are currently active, representing 1 per 2 000 inhabitants. Consequently, 40 % of home composters are managed according to best practice, and 91 % produce compost of an acceptable quality. Appropriate training is essential to avoid some of the negative environmental outcomes that can arise from poorly managed composting (see "Cross-media effects").

Applicability

In cases when home and community composting is the most appropriate waste management option for biowaste, there are no major restrictions to implementing this BEMP. However, the success of home and community composting as an environmental management strategy is highly dependent on the management of the waste separation and composting process by citizens who must be first engaged to motivate them to separate organic waste, and then trained to correctly manage the composting process. Additional effort is required to organise home and community composting in urban areas.

Economics

Costs

Eunomia (2007) estimated the costs for the waste management company/authority for instigating household recycling (Table 4-8). The net cost of bins will depend on their specification, and whether, and at what level, householders are charged for them. Arcadis (2010) estimate that bin costs should not exceed EUR 25 per household, leading to a total annualised cost of just over EUR 2.50 per household to support home and community composting, assuming a bin lifespan of 10 years.

Table 4-8. Costs of instigating household composting

Cost item	Cost per household
Marketing, literature and support	EUR 6.76
Net bin cost (after sales revenue)	EUR 3.38
Delivery and storage	EUR 14.86
Annualised cost	**EUR 2.50**
Source: Eunomia (2007).	

The main cost to the householders is their time.

Benefits to the waste management organisation

Home and community composting avoids a number of costs for waste management organisations, most notably:

- waste collection costs;
- waste management or disposal (landfill) costs.

According to cost benchmarking data presented in the BEMP on cost benchmarking (Section 4.3.1) provided by ia GmbH (2015), average waste collection and treatment costs amount to approximately EUR 80 per capita per year. It is difficult to estimate the proportion of these costs attributable to organic waste collection and treatment, but a crude estimation based on the 30 % relative mass of organic waste in MSW (Eurostat, 2014) would suggest that avoided organic waste handling costs could amount to approximately EUR 25 per capita per year.

However, in addition to avoiding costs associated with organic waste collection and treatment, the waste management organisation may also forego income from the sale of centrally produced compost, in the region of EUR 18/t (Aschaffenburg Local Authority, 2015).

Benefits to the compost user

Compost produced in home and community units can be used by householders in private gardens, housing associations or local authorities in public gardens. The fertiliser replacement value of compost based on food waste is displayed in Table 4-9.

Compost may be used as a substitute for peat or purchased compost products, leading to avoided purchase costs considerably greater than the fertiliser replacement value. These avoided costs are highly dependent on the type of product substituted.

Table 4-9. Fertiliser replacement value of compost derived from food waste, expressed per wet tonne of food waste (26 % dry matter)

Nutrient	Fertiliser nutrients replaced (kg per tonne of food waste)	Avoided fertiliser costs (EUR per tonne of food waste)
N	1.4	1.70
P_2O_5	0.6	0.65
K_2O	2.7	2.19
Total		**4.54**

Driving force for implementation

Legislation and financial incentives to divert organic waste from landfill, established in EU Member States in response to Directives 1999/31/EC and 2008/98/EC, are major driving forces for the composting and anaerobic digestion of organic wastes. In countries that offer feed-in tariffs for renewable electricity, or other financial incentives for biogas production, economic factors may drive implementation of incineration with energy recovery and/or anaerobic digestion. Otherwise, economic factors may favour home and community composting as the cheapest option to divert organic waste from landfill.

Another important factor driving home and community composting is the fact that it counts as “waste prevention” under statistical accounting rules, because it avoids the collection and classification of “waste”. Thus home and community composting may count towards waste prevention targets established by local authorities and/or waste management companies, even though it does not achieve genuine waste prevention (and may in fact lead to higher environmental burdens than management options, such as anaerobic digestion, for collected waste: see BEMP on integrated waste management).

Reference organisations

Box 4.1. Example of support for home composting provided by Leicester County Council, UK

Leicester County Council established and supports the “Rot-a-Lot Compost Club”, a free-to-join home composting club that assists Leicestershire residents with home composting. Residents joining the club receive a member’s pack to help them get the most from their compost bins, including a kitchen caddy with biodegradable liners and a book about composting. Club members are kept up to date with club news and composting events through regular newsletters. Leicester County Council also distributes the WRAP guide to home composting:

http://www.leics.gov.uk/composting_at_home.pdf.

Source: Leicester County Council (2015).

Box 4.2. Example of home and community composting implemented by SYBERT in Besançon, France

SYBERT is a waste management company in Besançon, France, that is pursuing a strategy of home and community composting. Owing to the absence of high feed-in tariff subsidies for bio-electricity and the high cost of collection, and possibly reflecting small local agricultural areas for digestate disposal, SYBERT did not pursue anaerobic digestion. It provided food collection boxes to all households to encourage composting. Single households were quick to take up composting, with 80 % now composting their organic waste. However, SYBERT had to invest significant resources in establishing over 230 community composting schemes throughout the city to cater for households in apartment blocks (described under "Operational data" above).

Nantes and Rennes are the only other examples of home and community composting that SYBERT know of in France.

Source: SYBERT (2015).

Box 4.3. Example of home composting and organic waste management promoted by Vlaco npo in Flanders

In Flanders, Vlaco npo supports and implements sustainable biowaste management, especially through home composting. Vlaco is a membership organisation with representation of both the Flemish government (OVAM and inter-municipal waste associations) and the private sector (private waste treatment companies). The 'Biocycling at home' unit of Vlaco focuses on raising environmental awareness concerning organic waste management via a twofold awareness approach.

An initial 'Home Composting' scheme evolved to the 'Closed Loop Gardening' scheme and finally, since 2012, the 'BioCycle at Home' scheme that includes communication about food losses and how to prevent them. The Vlaco unit 'Biocycling at Home' has trained several thousand volunteers called 'Master Composters' or 'Biocycle Volunteers' to assist the municipality in promoting recycling of food waste, lawn clippings and prunings via home composting and compost use, and chicken keeping. About 40 teachers are available to regularly train these volunteers and to update them. In total, 4 000 of those volunteers have been trained in the last 20 years. For the moment, 2 700 of these Master Composters / Biocycle Volunteers are still active (which is about 1 per 2 000 inhabitants). Volunteers are claimed to have better credibility compared with 'officials', as they have a rapport with local citizens.

Vlaco also approaches the public directly by: organising courses (about the prevention and processing of organic waste); (co-)organising campaigns and events (Closed Loop Weekend, Closed Loop Festival, Floralies 2016, etc.); distributing leaflets, brochures, posters (and booklets for those who want to know more about a specific theme); communicating by several types of (social, internet or paper) media, and through inter-municipal waste associations and local environmental services; using other educational materials (demonstration tools about processing organic residues, compost boxes and bins, wormeries, insect hotels, mulch mowers, wood chippers, school games, compost information box, etc.).

Results are tracked through screening of the behaviour of citizens every five years. In 1991, 5 % of the people in Flanders were composting at home. By 2012, this percentage had increased to 52 %. Vlaco estimate that 106 000 to 120 000 tonnes of organic waste is processed at home by composting, equating to between 16 kg and 19 kg per inhabitant per year. Their research indicates that 40 % of home composters are managing the process exactly according to best practice, and the vast majority of the home-produced compost is of an acceptable quality. 91 % of respondents that are compost at home do not experience problems with the composting itself or with the quality of the home compost. Almost all the compost produced is used at home.

Source: Vlaco npo (2015).

Box 4.4. Example of community composting in the province of Gipuzkoa, Spain

In November 2011, the first pilot project of community composting in the Basque Country (Spain) was launched in Usurbil (Gipuzkoa) and achieved a quick uptake by residents. By the end of 2013, 1 500 families in Gipuzkoa (600 000 inhabitants) were managing their food waste with community composting and there was a waiting list of 1 131 families. Today there are about 4 000 families in Gipuzkoa involved in community composting (up from zero in 2011) and many more doing home composting, since in the small villages the organic waste is no longer collected because it is all managed at source.

Source: Simon J. M. (2015).

Box 4.5. Example of Horta da Formiga training and awareness-raising for organic waste management in Portugal

Horta da Formiga is an educational farm managed by LIPOR in Portugal to educate citizens and institutions on the prevention and good management of organic waste, and also on good farming practices that can use composted waste. Horta da Formiga covers 1 hectare and includes demonstrations of composting bins and an organic kitchen garden.

The awareness-raising activities are free visits for groups of citizens, schools or other institutions, and a training service is provided comprising short theoretical and practical courses about composting, organic farming, sustainable gardening and sustainable cooking targeted at any citizen that intends to replicate the practices at home. The three-hour composting course is free.

More than 16 100 trainees have participated in the Horta da Formiga training plan since 2002, and more than 15 100 people have undertaken the home composting course. The farm has received over 26 500 visitors since 2002.

Source: Lopes A. (2015).

Communal home and community composting or district composting is realised in several cities or counties in Belgium (WORMS, 2015), Switzerland and Spain (Öko-Institut, 2012).

The county (Gemeinde) of Muttenz (Switzerland) offers assistance with information leaflets and a model contract concerning the maintenance of the district composting facility. Examples of leaflets are given in the links below (German language only):

- Infoblatt_Quartierkompost_Seemaettli.pdf;
- Infoblatt_Pflichtenheft_Quartierkompost.pdf;
- Infoblatt_Leitfaden_Quartierkompost.pdf).

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4.4. BEMPs on waste prevention and reuse

4.4.1. Local waste prevention programmes

Summary overview

It is BEMP to put in place waste prevention measures that target both households and public and private organisations. Some examples are adoption of local plastic bag charges, support for the setup of repair shops, introduction of product/material exchange areas in the territory as well as cooperation with social economy organisations, NGOs and restaurants to encourage the development of agreements for the reduction of food waste, thanks to donations. Waste prevention measures can be identified by:

- assessing current waste generation patterns in the territory;
- prioritising the most relevant waste streams in terms of prevention potential, such as food waste and biowaste, paper/cardboard, plastic (packaging), glass and textiles;
- Elaborating a local waste prevention strategy involving the relevant stakeholders (e.g. residents, local businesses, social economy organisations, NGOs);
- Monitoring the results of the waste prevention measures adopted and, in light of the results, reviewing the waste prevention strategy.

Waste management area

Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability

Waste prevention measures need to be carefully selected based on local circumstances and well implemented (e.g. some may need support by financial incentives) but there are suitable measures for any context.

Although some key waste prevention instruments can only be pursued at the international or national level (e.g. product policy, value-added taxation), there is also scope for action at the regional and local levels.

Specific environmental performance indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- establishment of a local waste prevention plan, including long-term and short-term targets and provisions for regular monitoring (y/n);
- budget dedicated to waste prevention programmes per resident per year (EUR/capita/year);

- share of total MSW management budget devoted to waste prevention (%); - number of stakeholders involved in prevention programmes.
Benchmark of excellence
- Waste prevention has strategic relevance in the waste management strategy, which includes a local waste prevention programme underpinning long-term (i.e. 10–20 years) and short-term (i.e. 1–5 years) waste prevention targets and including provisions for regular monitoring.

Description

The term `waste prevention' is defined in the Waste Framework Directive (WFD, 2008), and, being at the top of the waste hierarchy (Figure 3-1), prevention measures that lead to a reduction in the amount of waste are of utmost priority. In this respect, various instruments such as strong product policies are discussed in order to reduce the throughput of the economic system, i.e. reduction of raw material inputs and reduction of waste outputs (dematerialisation) (Kranert, 2009; Grooterhorst, 2010a, 2010b; van Ewijk and Stegemann, 2016; Gharfalkar et al., 2015; Defra, 2010). Such instruments can only be established and implemented at the global and/or European level (for some instruments also at national level) with policy approaches like ecodesign of products, extended producer responsibility, change of tax systems, etc. (EC Waste reduction, 2010; European Commission, 2012). In this document, the focus is on waste prevention measures that can be implemented at the regional and local levels.

Following the definition of waste prevention, the measures include those to avoid waste at source and those to reuse products and materials or prepare for reuse waste. For the identification of these measures, the following sources have been considered:

- the waste prevention programmes of the Member States, which have to be established according to Article 29 of the Waste Framework Directive (EIONET, 2015);
- guidance documents (e.g. ACR+, 2010; EC Guidance, 2012; EEB, 2012; INTERREG IVC, 2013; ADEME, 2015, Pre-waste, 2015); and
- waste prevention plans of regions, cities or counties.

In many cases, in these documents the focus is on general strategies and recommendations and only a few concrete measures are mentioned. The proposed approach for the development of a waste prevention programme is shown in Figure 4-25.

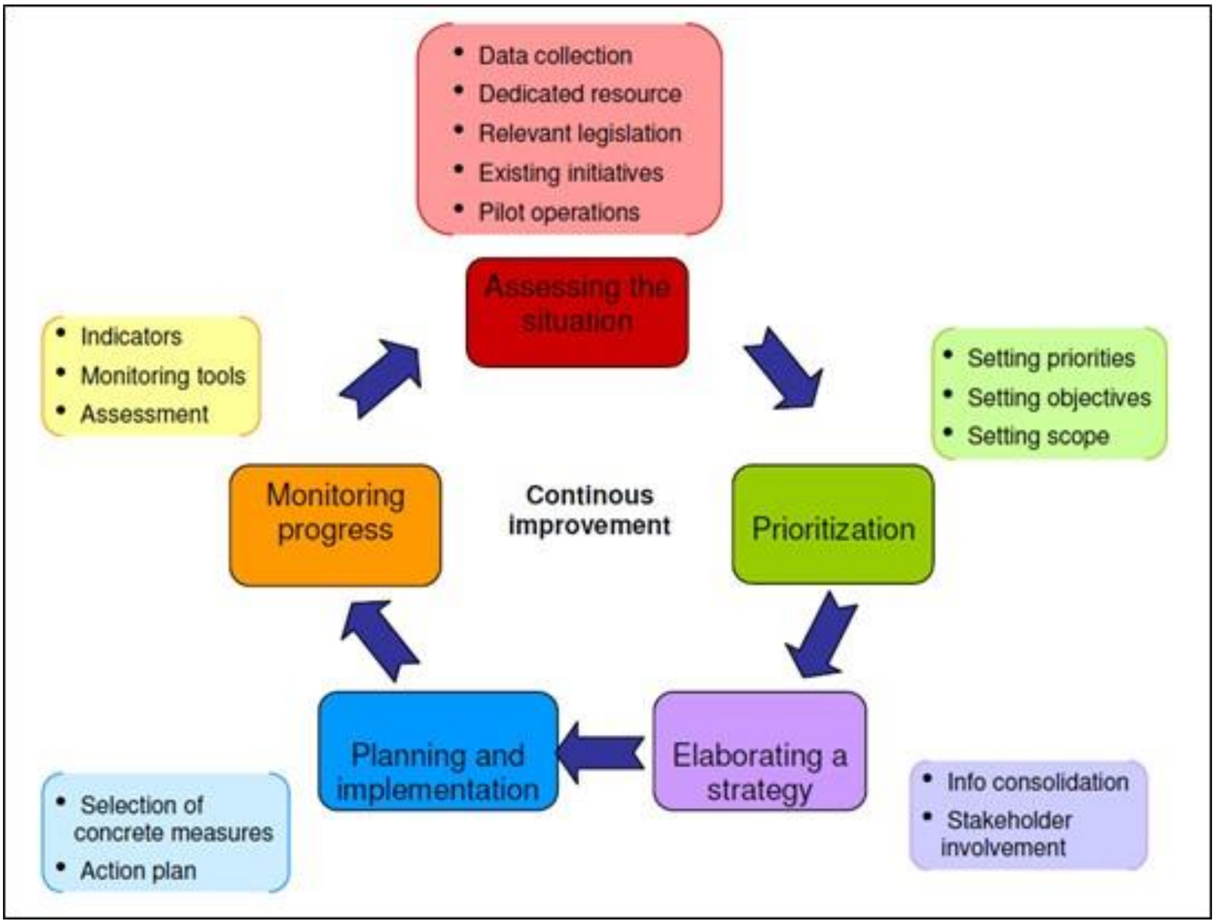


Figure 4-25.Developing a waste prevention programme (EEB, 2012)

When starting to identify waste prevention measures at the regional and local level, it may be appropriate to focus on the most relevant waste streams, such as food and biowaste, paper/cardboard, plastic (packaging), glass, and textiles (see for instance Welsh Government, 2013; Barcelona City Council, 2013; Phillips et al., 2010). In recent years, the prevention of food waste specifically has been discussed (Sharp et al., 2010a; Cox et al., 2010; European Commission, 2010, 2011a, 2011b). In Table 4-10, specific prevention measures are presented. They are grouped into measures for individuals and families and for municipalities, cities and counties or private organisations.

Table 4-10. Examples of waste prevention measures

Measure	Short description	Reference
For individuals and families (consumers)		
Little packaging	To buy things that are produced with as little packaging as possible	Kuriso/Bortelo, 2011
Bags	To use own bags when going shopping, rather than disposable ones provided by the shop	Kuriso/Bortelo, 201
Reusable packaging	To look for packaging that can be easily reused	Kuriso/Bortelo, 2011
Reusable product	To buy products that can be reused rather than disposable items	Kuriso/Bortelo, 2011

Table 4-10. Examples of waste prevention measures

Measure	Short description	Reference
Repair	To try to repair things before buying new items	Kuriso/Bortelo, 2011; City of Graz, 2015
Paper use reduction	To reuse paper for writing notes, to avoid printing or print double-sided, to ask for digital billing and invoicing services; in addition, to discourage unwanted mail, especially advertising, for instance by a "no junk mail" sticker on the mailbox	Kuriso/Bortelo, 2011
Container reuse	To reuse containers	Kuriso/Bortelo, 2011
Reusable dishcloth	To use dishcloths rather than paper kitchen towels	Kuriso/Bortelo, 2011
Refillable products	To try to buy refillable products (e.g. printing cartridges, hand soap, powdered cocoa drinks)	Kuriso/Bortelo, 2011
Donation	To donate old items to other possible users	Kuriso/Bortelo, 2011; Sharp et al., 2010a; Cox/Giorgi et al., 2010
Returnable bottles	To buy returnable bottles instead of single-use bottles	Kuriso/Bortelo, 2011
Own cup	To bring own cup, e.g. to school or office	Kuriso/Bortelo, 2011
Needless packaging avoidance	To refuse needless packaging	Kuriso/Bortelo, 2011
Needless product avoidance	To try not to buy needless products	Kuriso/Bortelo, 2011
Reuse shop/centre	To bring reusable products to shops for reselling	Kuriso/Bortelo, 2011; City of Graz, 2015
Bottled water avoidance	To try not to buy bottled drinking water	Kuriso/Bortelo, 2011; Province of Florence, 2014
Reduction of food waste	To try to buy only the quantity of food that one can consume, correctly store purchased food, cook adequate portions and use leftovers	European Commission, 2010, 2011a, 2011b and 2015; Sharp et al., 2010a; Cox/Giorgi et al., 2010
Reusable nappies	To use reusable nappies (supported by the county or city)	Morlok et al., 2017
Mobile dishwasher for festivals	To use a mobile dishwasher (provided by the county or city) for festivals to avoid single-use dishes and cutlery	e.g. Vienna, Rems-Murr County

Table 4-10. Examples of waste prevention measures

Measure	Short description	Reference
For municipalities, cities and counties or private organisations		
Mobile dishwasher for festivals	To provide dishes and cutlery along with mobile dishwashers for public festivals for free	e.g. Vienna, Rems-Murr County, City of Graz, 2015
Reduction of canteen waste	To provide reusable dishes, cutleries, napkins and tablecloths as well as tap water and draught beverages in canteens	
Reusable nappies	To financially support the use of reusable nappies	e.g. Enfield Council, County of Aschaffenburg, Besançon
Lunchboxes	To provide schoolchildren with reusable lunchboxes	e.g. Rems-Murr County, Barcelona
Repair shops	To support the setup of repair shops	e.g. Vienna, Wales, BMUB, 2013; City of Graz, 2015
Reduction of office paper waste	To promote/adopt reduction of paper consumption in offices (e.g. avoid printing of documents readable on screen, default double-sided printing and copying, use of electronic archives, reuse of envelopes)	City of Graz, 2015
Reduction of food waste	To support activities for the reduction of food waste produced in canteens and restaurants (e.g. staff training, promote customer behaviour change). To promote/support the collection of still edible but no longer sellable food from supermarkets for delivery to social canteens or similar. In addition, to continuously raise awareness so that citizens shall try to buy only the quantity of food they can consume	e.g. LIPOR, Portugal (LIPOR, 2015), Vienna, Wales; BMUB (2013), last minute market, Bologna (last minute market, 2017)
Pay-as-you-throw (PAYT) system	To introduce pay-as-you-throw systems	See the BEMP on PAYT

Source: Own elaboration from different sources

Many of the measures mentioned in Table 4-10 are for consumers. The change in consumption patterns requires targeted awareness campaigns taking into account psychological mechanisms and the multifaceted nature of waste prevention (Bortoleto et al., 2012; Bortoleto, 2015). Continuous awareness-raising of consumers is required to make them conscious of the waste issue and to keep them motivated (Cecere et al., 2014; Cole et al., 2014). However, economic incentives are much stronger driving forces as the example of charging for plastic bags, e.g. in Ireland, Spain or Japan or anywhere else, demonstrates.

Concerning product reuse (which is a specific aspect within the area of waste prevention measures) such as furniture, electrical and electronic equipment, clothes and home textiles, books, bicycles, etc., there is a specific BEMP (Section 4.4.2) dealing with the topic.

Achieved environmental benefits

Although waste prevention has high priority, the prevention potential appears to be relatively small in relation to the total municipal waste; only 1–3 % has been reported (Salhofer et al., 2008). For some individual waste streams, the percentage can reach the order of some 10 % (Salhofer et al., 2008). This is confirmed by Figure 4-26, which shows the development of the total municipal waste amount in Germany, consisting of the fractions: light packaging/plastic, glass, paper/cardboard, biowaste and residual waste. Despite the fact that waste prevention was always a top priority in Germany, the total waste quantity slightly increased. The increase would probably have been even higher without prevention measures but their impact does not seem to be significant. Thereby, quantitative measurement of waste prevention is notoriously difficult as the basic problem is measuring something that is not there (Sharp et al., 2010b; Zorpas and Lasaridi, 2013).

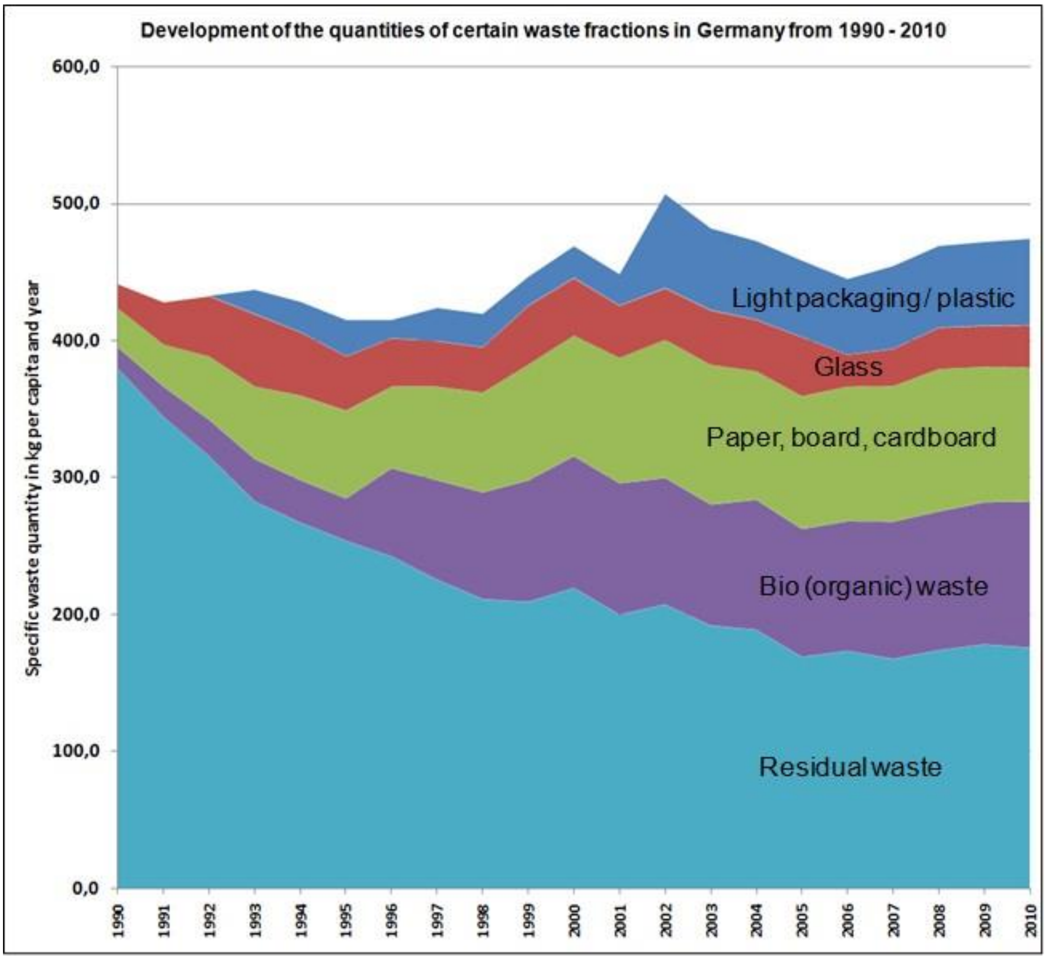
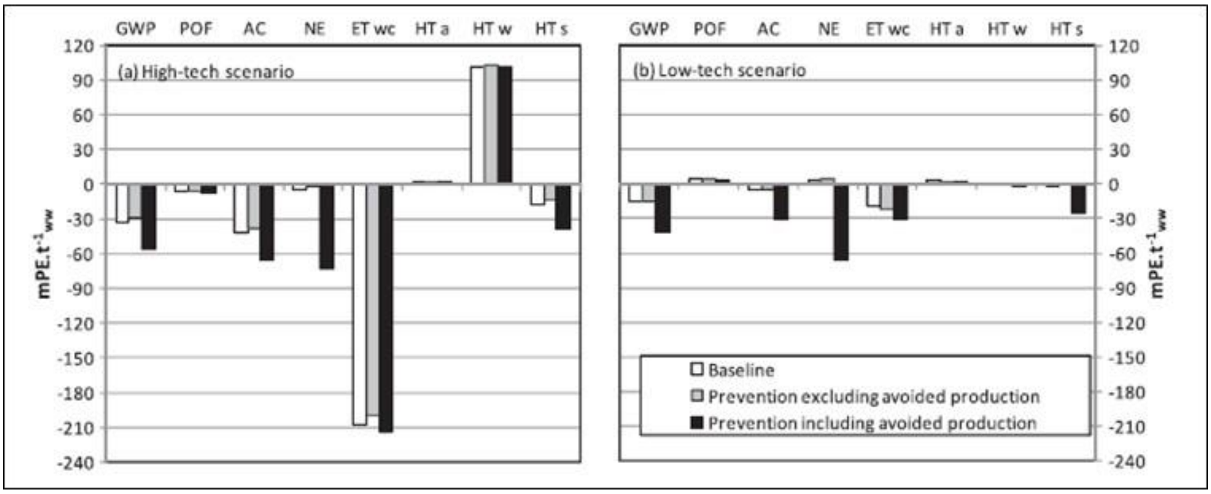


Figure 4-26. Development of the quantities of certain waste fractions in Germany from 1990 to 2010

The expectation that prevention means reduction of input mass streams and thus reduction of the environmental impact can be confirmed. Note: **The top of the vertical bars indicates 0 % waste prevention (baseline), the bottom of the vertical bar indicates 100 % waste prevention of the waste streams**

considered (unsolicited mail, vegetable and meat waste, plastic and glass beverage) (Gentil et al., 2011)

Figure 4-27 shows the related environmental impact assessment of integrated waste prevention on two waste management systems. Here, the comparison of the two systems is not important but the illustration that prevention is associated with a significantly lower environmental impact is. However, as indicated, the reduction rates of total municipal waste are low and so the environmental benefit is limited.



Legend:

Impact category	Acronym	Reference	Unit
Acidification	AC	74	kg SO_2-eq/person/year
Ecotoxicity water chronic	ET wc	$3.52 * 10^5$	m^3 water/person/year
Global warming (100 years)	GWP	$8.7 * 10^3$	kg CO_2-eq/person/year
Human toxicity air	HT a	$6.9 * 10^{10}$	m^3 air/person/year
Human toxicity soil	HT s	127	m^3 soil/person/year
Human toxicity water	HT w	$5 * 10^4$	m^3 water/person/year
Nutrient enrichment	NE	119	kg NO_3-eq/person/year
Photochemical Ozone formation	POF	25	kg C_2H_2-eq/person/year

Note: The top of the vertical bars indicates 0 % waste prevention (baseline), the bottom of the vertical bar indicates 100 % waste prevention of the waste streams considered (unsolicited mail, vegetable and meat waste, plastic and glass beverage) (Gentil et al., 2011)

Figure 4-27.Comparison of integrated waste prevention in two waste management systems.

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- establishment of a local waste prevention plan, including long-term and short-term targets and provisions for regular monitoring (y/n);
- budget dedicated to waste prevention programmes per resident per year (EUR/capita/year);
- share of total MSW management budget devoted to waste prevention (%);
- number of stakeholders involved in prevention programmes.

Cross-media effects

With respect to waste prevention, no significant cross-media effects are known.

Operational data

The development of waste prevention programmes/projects may take into account the aspects and steps indicated in Figure 4-28.

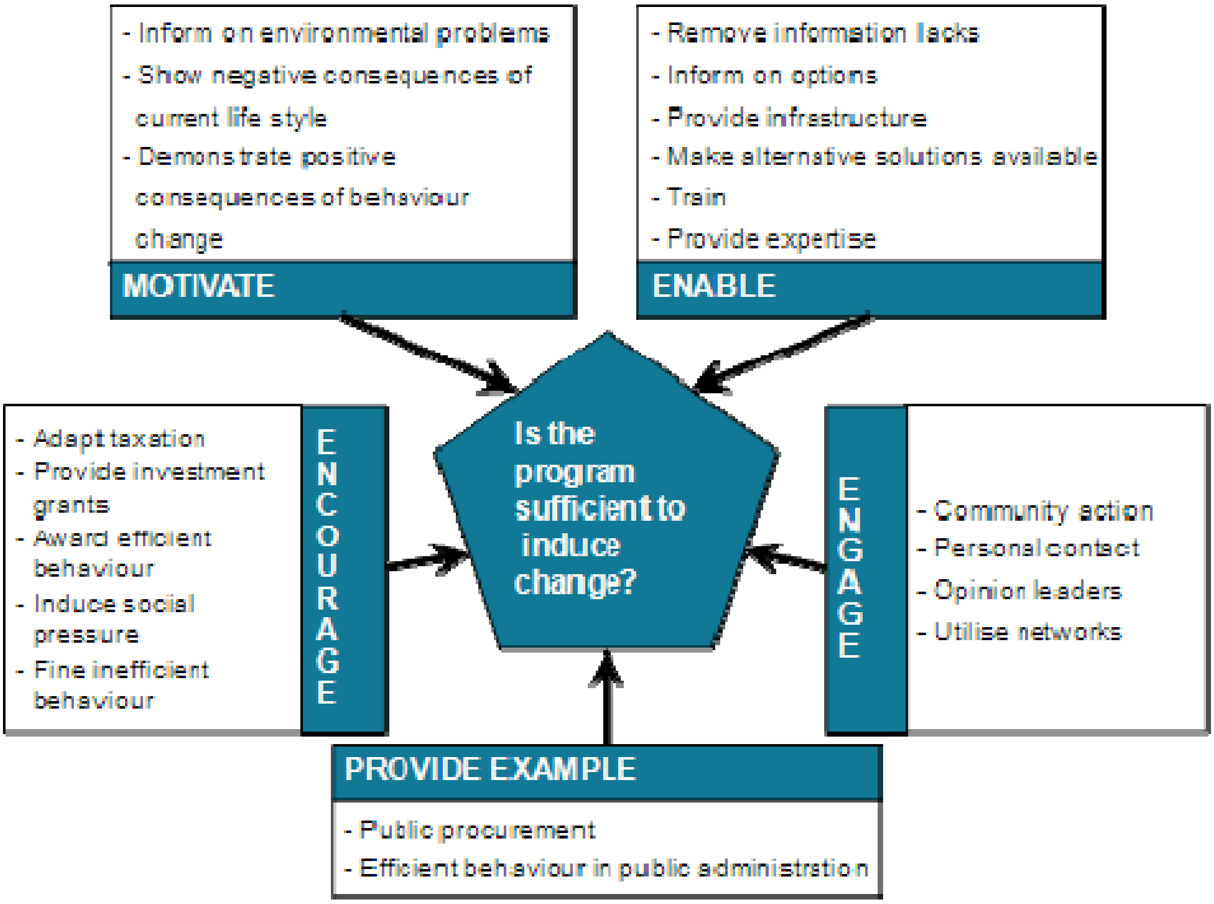


Figure 4-28. Aspects and steps to consider when developing a waste prevention programme (European Commission, 2011b)

It is important to develop a waste prevention programme/project specific to certain waste streams such as biowaste, food waste, packaging, paper/cardboard, etc. The efficiency of waste prevention can be measured best for such waste streams. It can be expected that the highest reduction rates can be achieved for food waste as the potential is high and citizens may develop adequate awareness. The required campaigns should take psychological aspects into account and should provide concrete best and good practice examples. In addition, waste prevention measures should be combined with financial incentives. In bigger cities and in counties, qualified staff should be available to carry out information campaigns, to regularly inform the citizens and to respond to their questions.

Applicability

Waste prevention measures need to be carefully selected based on local circumstances and well implemented (e.g. some may need support by financial incentives) but there are suitable measures for any context.

Although some key waste prevention instruments can only be pursued at the international or national level (e.g. product policy, value-added taxation), there is also scope for action at the regional and local levels.

Economics

There is little information on economic aspects. The investment in awareness campaigns and monitoring of the quantities of the main waste streams will not have a significant impact on waste fees.

Driving force for implementation

Waste prevention is top of the waste hierarchy of the Waste Framework Directive. According to Article 29 of this Directive, the Member States have to establish waste prevention programmes. This legal background is the main driving force.

Reference Organisations

The cities of Barcelona, Vienna, Copenhagen and Besançon and the counties/regions of, Aschaffenburg, Schweinfurt, Flanders and Île-de-France are references with regard to waste prevention (programmes).

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4.4.2. Schemes fostering the reuse of products and the preparation for reuse of waste

Summary overview

It is BEMP to encourage diversion of reusable products away from waste streams and into reuse streams, through the active establishment or facilitation of second-hand and municipal exchange markets (via repair workshops where necessary) or charity collections. Additionally, waste management organisations can send certain waste streams to preparation for reuse by establishing or facilitating the creation of reuse/repair centres.

The BEMP covers four key measures:

- collect products suitable for reuse before these are considered waste, repair them if needed, and distribute or sell them to residents and organisations, including charities;
- collect waste items suitable for reuse, have them prepared for reuse, and distribute or sell them to residents and organisations, including charities;
- establish effective information exchanges to advertise the demand for, and market the availability of, reusable used products;
- monitor the output (regardless of whether their input is classified as waste or product) of repair and reuse centres which have been accredited based on Annex IV to the Waste Framework Directive.

Waste management area

Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability

This BEMP applies to all waste management organisations that handle any type of reusable products and waste, in particular garments, furniture and electrical and electronic equipment.

Specific environmental performance indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- Number of reuse centres/community repair points per 100 000 residents;
- Number or quantity (i.e. weight or volume) of end-of-life products collected for reuse and waste items sent for preparation for reuse;
- Annual number of customers of the reuse centres/community repair points;
- Availability of products/materials exchange areas aimed at fostering reuse in civic amenity sites (y/n).

Benchmark of excellence
- In civic amenity sites, product/material exchange areas aimed at fostering reuse are available.

Description

Background

Product reuse comes at the top of the waste hierarchy as a waste prevention measure that avoids the environmental burdens associated with product manufacture and disposal or recycling. This BEMP addresses the implementation of preparation for reuse and reuse schemes for end-of-life products, in particular products which tend to be replaced when still fully functioning owing to consumer trends and short innovation cycles, e.g. garments, furniture and electrical appliances. When such products are replaced, it is often convenient for previous owners to dispose of them into waste disposal or recycling streams. Castellani et al. (2015) applied life-cycle assessment to evaluate the environmental benefits of product reuse in second-hand shops, considering the new product replacement factor associated with reuse of different types of product. They found that the greatest environmental savings arise from reuse of apparel products, due to the volume of items sold, followed by reuse of furniture products, owing to the high environmental burdens from production of new items.

Best practice measures

It is BEMP to encourage diversion of reusable products away from waste streams and into reuse streams, through the active establishment or facilitation of second-hand and municipal exchange markets (via repair workshops where necessary) or charity collections. Additionally, waste management organisations can send certain waste streams to preparation for reuse by establishing or facilitating the creation of reuse/repair centres.

There are four key measures covered by this BEMP:

- collect products suitable for reuse before these are considered waste, repair them if needed, and distribute or sell them to residents and organisations, including charities;
- collect waste items suitable for reuse, have them prepared for reuse, and distribute or sell them to residents and organisations, including charities;
- establish effective information exchanges to advertise the demand for, and market the availability of, reusable used products;
- monitor the output (regardless of whether their input is classified as waste or product) of repair and reuse centres which have been accredited based on Annex IV to the Waste Framework Directive.

One way to foster reuse is establishing products/materials exchange areas in civic amenity sites where residents can deliver products which they wish to discard but still fully or partially functioning or usable (see also BEMP 4.5.3). Forming partnerships with social economy organisations and other stakeholders can be an important element of best practice.

In relation to electronic items, this BEMP covers reuse schemes that complement and go beyond the provisions for Waste of Electrical and Electronic Equipment (WEEE) established by Directive 2012/19/EU (known as the WEEE Directive). In particular, the WEEE Directive requires Member States to: (i) promote product design measures that facilitate reuse, upgrading and recycling of EEE, (ii) arrange return systems for WEEE that are free of charge to final holders, including consumers and distributors who are obliged to accept WEEE free of charge from consumers, (iii) comply with reuse and recycling targets established for national mass streams of WEEE.

Achieved environmental benefits

WRAP's Benefits of reuse tool (WRAP, 2014a) indicates the life-cycle benefits for reuse of different waste categories within the UK context (Table 4-11).

Table 4-11. Environmental benefits achieved per tonne of product category reused compared with prevailing counterfactuals in the UK

Category	Avoided global warming potential (kg CO_2e)	Avoided abiotic resource depletion (kg Sbe)	Avoided fossil resource depletion (MJe)
Clothing	7 510	0.039	57 100
Home furniture	30	0.004	5 000
Home electricals	3 290	0.030	67 100
Source: WRAP (2014a).			

The Surrey reuse network described below under "Operational data" achieved the following benefits within one year of establishment:

- a 22 % increase in diversion of furniture and white goods to reuse, to 600 tonnes per year;
- a 100 % increase in overall recycling rate.

Castellani et al. (2015) report on the following life-cycle environmental savings arising from product substitution through sales of reusable items in an Italian second-hand shop:

- 160 t CO_2e/year;
- 7 000 000 MJe/year;
- 170 kg PM2.5e/year.

WRAP (2015) estimates that, during 2012, the emission of 1.5 Mt CO_2e was avoided in the UK through product reuse. This translates into a CO_2e saving from reuse of 23 kg per capita per year.

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- Number of reuse centres/community repair points per 100 000 residents;
- Number or quantity (i.e. weight or volume) of end-of-life products collected for reuse and waste items sent for preparation for reuse;
- Annual number of customers of the reuse centres/community repair points;
- Availability of products/materials exchange areas aimed at fostering reuse in civic amenity sites (y/n).

Cross-media effects

Reuse of most products is not associated with any significant cross-media effects. Transport distances for collection of reusable items are unlikely to be greater than life-cycle transport distances associated with production and disposal or recycling of new products.

However, for some types of electrical equipment, from an energy and carbon perspective, it may be better to replace old, inefficient items with newer, more efficient items – recycling rather than reusing components from the old equipment. In addition, it is important to avoid risks associated with malfunctioning electrical equipment (e.g. microwaves).

Operational data

Guidance documents

Waste management organisations can play an important role by describing and disseminating best practice in the establishment and implementation of reuse schemes among the various stakeholders typically involved in successful implementation of such schemes, especially the third sector. WRAP has produced a number of guides on product reuse, available at the following link: http://www.wrap.org.uk/content/how-guides-0. They describe how to:

- make reuse a strategic priority;
- establish a reuse baseline for your area;
- set up and run a reuse forum;
- produce a reuse action plan;
- write a communications plan to boost reuse;
- provide for reuse in household waste collection centres (Source: **E3 Environmental Consultants Ltd**
- Figure 4-29);
- provide a reuse-focused bulky waste collection service.

Source: E^3 Environmental Consultants Ltd

Figure 4-29. Clearly identified bins accepting clothing for reuse at a community waste collection centre in Aschaffenburg, Germany

Establishing collaborative reuse networks

Successful reuse schemes involve multiple stakeholders, including third-sector organisations that sell reusable items to raise money for charitable causes or that distribute reusable items to people in need, businesses, local authorities and government agencies. Coordination among stakeholders can reduce the costs of collecting and distributing reusable items. Consequently, the establishment of local reuse networks comprising relevant stakeholders is an important aspect of best practice. Local authorities are particularly well positioned to coordinate, or at least catalyse, the development of these networks at an appropriate local-to-regional scale.

WRAP (2014b) describes the role of a local authority in catalysing the establishment of a successful reuse network in Surrey, England. Surrey County Council (SCC) was seeking ways to deliver ambitious targets to:

- reduce household waste by 30 000 tonnes;
- send zero household waste to landfill;
- achieve recycling rates of up to 70 %.

Furniture and white goods were identified as bulky waste streams that could be considerably reduced through reuse. SCC worked with numerous independent local furniture reuse organisations, and realised that they could become more efficient if they pooled their resources. SCC therefore embarked on a project to increase furniture reuse across the county by:

- enabling furniture reuse organisations to work as a county-wide network, delivering coordinated, high-quality services;
- building capacity of furniture reuse organisations to handle greater volumes of furniture and white goods;
- raising public awareness of the potential for reuse, and improving access to it.

Key steps and actions in the development of the reuse network are summarised below, based on information described in WRAP (2014b).

Table 4-12: Key steps and actions in the development of the reuse network

Step	Actions
Engaging furniture reuse organisations	SCC offered grants to build capacity and quarterly furniture reuse credits, as well as funding a county-wide communications campaign, providing marketing support, and part-funding an interim manager. In return, each furniture reuse organisation had to commit in writing to be part of a "Surrey Reuse Network" (SRN).
Agreeing a structure	SCC proposed to establish the SRN as a legal entity in the form of a constituted membership network, with its own board and constitution. A Memorandum of Understanding was agreed, and plans put in place for the SRN to become a registered charity and a company limited by guarantee.
Building capacity	Each member of SRN retained autonomy, and was encouraged to grow with tailored advice provided by WRAP-funded independent consultants. This ensured capacity growth across SRN members, individually and collectively.
Establishing a business plan	An interim manager was part-funded by WRAP to develop a three-year strategic plan for the SRN drawing on the skills and strengths of different

Step	Actions
	members. One deliverable was the establishment of a shared 0800 phone number for people to request collections, alongside development of a dedicated website to raise awareness of reuse in general and the SRN in particular.
Building relationships	One intention of the SRN was to leverage the combined capacity of the network to bid for collection of bulky waste from households, and for resale of reusable items from household waste collection centres. The SRN interim manager established relationships with contracting authorities and SCC departments, enabling the SRN to become integrated in the delivery of services across the county. The SRN also won a contract to supply goods to Surrey's Local Assistance Scheme that provides furniture and white goods to people in need.

Source: Based on information described in WRAP (2014b).

Promoting reuse markets

Managing schemes that directly engage with citizens to promote reuse markets is also an important component of best practice. Training in basic repair work and advertising repair services are two simple measures that could increase reuse rates. Area Metropolitana de Barcelona (AMB) provides an example of collaboration among different administrations and organisations, and manages a repair centre in Barcelona where technicians teach citizens how to repair products. The centre also functions as an exchange facility, where people can use and share tools. More information can be found at: www.millorquenou.cat

AMB and local municipalities around Barcelona also promote second-hand markets, and allow people to take materials from municipal waste centres for reuse. There may be restrictions on what can be taken away from municipal waste centres owing to health and safety concerns around potentially faulty electronic equipment, and hygiene, etc., and authorisation is required in some centres before objects are removed. In Barcelona, reuse is restricted primarily to books (Passalacqua, 2015).

The following video shows an example of a second-hand market in Sant Cugat, El Prat de Llobregat: https://www.youtube.com/watch?v=P1TEvhR-FxY. Meanwhile, the photos below show examples of trendy upcycling and reuse shops in the Basque region of northern Spain.

Source: Koopera (2015).

Koopera is a group of cooperatives and social enterprises. The Basque Government supported the creation of the Koopera reuse plant that takes, sorts and prepares goods for the stores. Koopera also creates social jobs for people at risk of exclusion, providing training in technological skills to all employees. Koopera has developed specific collection containers to facilitate separation and reuse discarded miscellaneous waste streams such as books, clothes and small electronic appliances. Purpose-built vehicles pick up goods from the containers, bringing them to the classification facility where manual sorting combined with voice identification systems separates textiles, small electronic devices, used toys and others.

Applicability

This BEMP applies to all waste management organisations that handle any type of reusable products and waste, in particular garments, furniture and electrical and electronic equipment.

Economics

Waste management organisation economics

Local authorities or waste management organisations may work in partnership with each other, and with third-sector reuse organisations, to efficiently design and implement reuse schemes. Such reuse networks can realise significant economies of scale, and achieve “critical mass” with respect to effective advertising and awareness campaigns, thus increasing both supply and demand for reusable items.

Budgetary constraints may decrease opportunities for local authorities to organise and advertise reuse schemes, and to commission agreements with third-sector reuse organisations (Ricardo-AEA, 2015).

Reuse schemes avoid recycling or disposal costs, and may even generate income if reusable items are sold on.

Societal cost-benefit analysis

In 2012, the third sector in the UK benefited by an estimated GBP 430 million through reuse, and reuse organisations created 11 000 full-time equivalent jobs (WRAP, 2015).

WRAP (2015) estimate that, by keeping goods in circulation for longer and by offering more affordable products, UK households benefitted by an estimated GBP 6 billion from product reuse in 2012. The Surrey Reuse Network described above provides goods to approximately 5 000 low-income household families each year (WRAP, 2014).

Reuse of materials can generate turnover of up to EUR 1 500 per tonne, over 10 times more than the turnover generated by recycled materials (TWG, 2015).

Driving force for implementation

Waste prevention through reuse of products as well as preparation for reuse of waste can significantly reduce waste handling and disposal costs for waste management organisations and facilitate compliance with applicable legislation and targets.

Another driving force is consumer demand for used products that are often considerably cheaper, and offer comparable functionality, compared with new products.

Reference organisations

In addition to the examples detailed below, WRAP has compiled a number of video and downloadable PDF case studies of local-authority-led reuse schemes in the UK, available at the following link: http://www.wrap.org.uk/content/how-case-studies-and-videos-0

CERREC – "Central Europe Repair & Reuse Centres and Networks" – is an EU-funded programme implemented through the CENTRAL EUROPE Programme and co-financed by the ERDF that started in April 2011 and lasted for 3.5 years. During this time the consortium of nine partners from seven different Central European countries carried out evaluation, quality management and dissemination activities in the field of reuse and repair of end-of-life items. The Municipal Waste Management Association Mid-Tyrol (ATM) in Austria was the lead partner on the project. Information can be found on http://cerrec.eu/, and a list of best practice examples at http://cerrec.eu/downloads/best-practises/

RREUSE is a network of social enterprises active in reuse, repair and recycling throughout Europe. Members of the network are listed, with links, at the following web address: http://www.rreuse.org/about-us/members/. Members include Repanet in Austria, Envie in France, EKON in Poland, Ateliere Fără Frontiere in Romania, AERESS in Spain and Reuseful in the UK.

Box 4.6. Establishment of Leicestershire and Rutland Reuse Network

WRAP contracted Ricardo-AEA to assist in the development of a reuse plan for Leicestershire County Council, Leicester City Council, Rutland County Council and local third-sector reuse organisations (TSROs). The objective was to support the development of a financially sustainable reuse sector in the region.

Stakeholders involved in the project included local authorities, TSROs, housing associations, waste management companies and businesses. Opportunities that could be realised via collaboration within a reuse network were identified.

A reuse mapping exercise quantified current levels of reuse for items within the bulky waste stream, and estimated the potential for increasing reuse across major material streams.

A four-year action plan for the delivery of the reuse network was devised, based around eight service options to improve rates of reuse and recycling of bulky waste. The stakeholders have adopted the four-year action plan and are exploring options for partnership working, including:

- members of Leicestershire and Rutland Reuse Network (LRRN) have signed a Memorandum of Understanding to work together;
- LRRN is working towards the incorporation of the Network;
- RRN is working with Leicestershire County Council to supply furniture items for the implementation of Leicestershire Welfare Provision (social fund);
- LRRN, with the support of the Producer Compliance Scheme in Leicestershire, is developing a WEEE repair workshop.

Source: Ricardo-AEA (2015).

Box 4.7. Example of the London Reuse Network

Waste reuse is prioritised within London’s Municipal and Business Waste Strategy plans, which identify the third sector as an important growth area and the London Reuse Network as a lead delivery partner to drive reuse targets.

The London Reuse Network comprises various reuse projects, including charities, that work together to collect, repair and sell unwanted furniture, appliances and household items, giving them new homes across London. In addition, the network arranges and provides employment, skills development, training and volunteer opportunities. It is organised around London Reuse Ltd, a central operating company.

London Reuse Network members work with a number of London waste authorities, and this collaboration will be strengthened by a new London Waste Authority Support Programme to be implemented by the London Waste and Recycling Board and WRAP. The London Waste and Recycling Board has a commercial approach to supporting the third sector, encouraging robust business practices.

Cllr Bassam Mahfouz, a London Waste and Recycling Board member, commented: “In order to accelerate the move towards a circular economy in London, reuse, repair and remanufacturing will have ever greater roles to play in our lives”.

Source: Waste Management World (2014).

Box 4.8. Waste prevention and reuse employing disadvantaged persons in Graz, Austria

“Waste Prevention, Responsible Use of Resources and Sustainable Development” is a non-profit company managed by Berthold Schleich that employs 140 disadvantaged persons to wash dishes, cutlery, drinking glasses, and plastic drinking cups from catering companies, festivals, etc. (waste prevention), and also to repair equipment such as mobile phones, table lamps, standard lamps, computers and other electronic and electrical equipment for sale in a reuse shop. The photo on the left below shows the repair desk and on the right mobiles repaired for reuse. The company is 30 %

funded by the region of Styria.

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Source: Schoenberger (2015).

Box 4.9.LIPOR Reuse Lab: repairing electrical and electronic equipment

'Reuse Lab' is a laboratory established by LIPOR (waste management company in the area of Porto in Portugal), where electrical and electronic equipment can be repaired to bring them back to their original functionality or regular maintenance can be carried out in order to extend the lifespan.

The laboratory is a space where it is possible to experiment and learn skills to recover the functionality of equipment, learn about maintenance and understand major malfunctions.

All activities of the Reuse Lab are supervised by experienced technicians and residents and local organisations are involved in the repair and maintenance of electrical and electronic equipment.

Source: LIPOR (2015).

Box 4.10.Flanders reuse shops

In the 1990 in Flanders (northern Belgium), reuse shops were established with social and environmental purposes. The products delivered to the reuse centres are considered "goods without value" by the parties that discard them and, consequently, are delivered to the reuse centres for free. Non-reusable goods are not accepted. The goods collected receive a monetary value following their sorting process and preparation for sale at the reuse centres. The number of reuse shops increased from 81 in the year 2000 to 124 in 2014, and at the same time the number of customers increased, from 1.56 million to 5 million over the same period. The trend to deliver goods to reuse shops has constantly increased since the 1990s and the turnover of reuse shops in the year 2014 reached EUR 45.4 million.

Source: OVAM (2015).

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4.5. BEMPs for waste collection

4.5.1. Waste collection strategy

Summary overview

It is BEMP to design and implement a waste collection strategy that considers:

- the main features of the waste management strategy (e.g. number of separately collected waste fractions);
- the targets set in the waste management strategy (e.g. share of separately collected waste out of the total waste collected, impurity rates of the separately collected fractions, revenues from recyclables);
- the characteristics of the collection area (e.g. population density and main housing types);
- the current environmental attitudes and perceptions of residents;
- any other specific condition affecting waste collection (e.g. the relevant presence of tourists/commuters, specific economic activities, climate).

The main goal of a waste collection strategy is to collect, in a timely and economical manner, as much correctly source separated waste as possible, in order to ease the subsequent waste sorting/treatment with the aim to maximise recycling. In many cases, these objectives can be pursued by setting up the following:

- frequent door-to-door separate collection of food waste (e.g. weekly or more often depending on the season and climate);
- less frequent collection of mixed waste (e.g. every two weeks);
- door-to-door collection of recyclables (e.g. paper, cardboard, cans, plastics, glass), individually source separated where public acceptability allows, otherwise co-mingled and sorted at a material recovery facility; glass, followed by paper and cardboard, is more often more effectively collected separately;
- a convenient network of civic amenity sites (see section 4.5.3) that accept all waste fractions not collected door-to-door or in street containers from households, including hazardous waste and biowaste.

Waste management area

Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability

The prevailing socio-economic status and recycling consciousness within the area from which waste is collected needs to be considered in the definition of the waste collection strategy. More costly strategies, such as door-to-door collection, may prove more cost-effective once fully running, but require initial investment.

Specific environmental performance indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this

BEMP are: - Participation rate, i.e. the share of the population using the waste collection system (%). - Share of the local area covered with a specific waste collection system (%). - Customer satisfaction (% of residents satisfied with household waste collection and specifically with the collection of the separately collected fractions). - Collection of bulky waste on demand (y/n).
Benchmark of excellence
- Door-to-door waste collection of at least four waste fractions[41] is implemented in the whole territory administered.

Description

Background

Collection of MSW can be undertaken via door-to-door (or kerbside) collection rounds from households and businesses or at municipal waste collection centres. Collection rounds are typically provided for the most voluminous MSW fractions, with municipal waste collection centres accepting a wider range of waste streams, including electronic and hazardous waste streams. Return schemes and electronic waste are addressed in other BEMPs; here the primary focus is on the following MSW fractions: biowastes[42], glass, paper and card, plastics, metals and mixed waste (where "mixed waste" refers to waste not separated at source and sent for incineration/final disposal).

A key measure of environmental efficiency for any waste collection strategy is the proportion of total waste collected that is *selectively* collected. ACR+ (2014) defined "selective collection" as the separation of waste materials at source with the intention of recycling them, and has benchmarked performance across European cities (Figure 4-30). The quantities of waste fractions selectively collected are also influenced by the quantities generated, and do not necessarily represent the highest *proportions* of waste being selectively collected.

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[41] In areas where different waste fractions are collected co-mingled (e.g. metal and plastic waste packaging) the co-mingled fraction is considered as one fraction.

[42] Biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants, excluding forestry or agricultural residues, manure, sewage sludge, or other biodegradable waste such as natural textiles, paper or processed wood (EC, 2015).

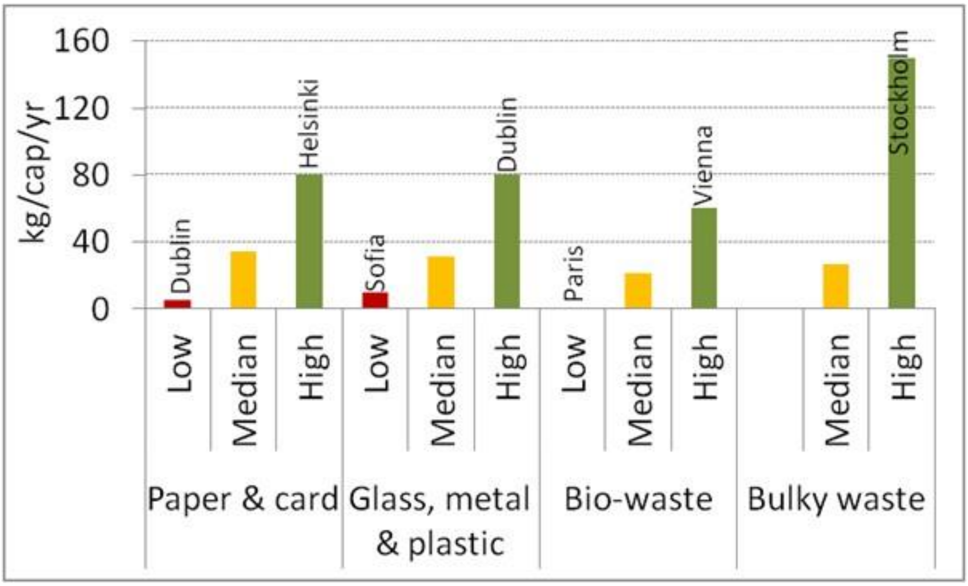


Figure 4-30. Range of quantities of different waste fractions selectively collected across European cities, according to ACR+ (2014)

Benchmarking, such as that undertaken by ACR+ (2014), can be a powerful driver to improve effectiveness and cost-efficiency (see also BEMP 4.3.2 on waste monitoring).

Types of selective waste collection

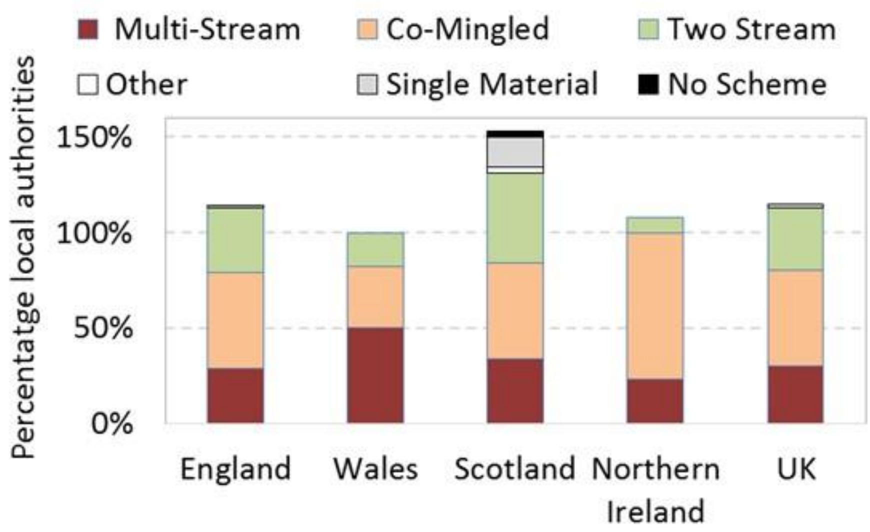
Various models of waste collection exist to deliver separated fractions for recycling, including separate door-to-door collection rounds for individual fractions, co-mingled recyclable material collection rounds with and without kerbside sorting, and community collection centres where citizens deposit waste fractions as required. Strategies for collection of dry recyclables (e.g. paper, card, cans, plastic bottles, mixed plastic, glass, aerosols, batteries, foil and textiles) are particularly varied (Table 4-13).

Table 4-13. Main types of waste collection strategies for dry recyclables

Collection type	Explanation
Door-to-door 	Within door-to-door collection systems the bins/sacks can be collected from the doorstep of the inhabitants, but also by kerbside collections. Kerbside collections are provided for flats where residents set out containers for collection from the street. This system can range from one bin for mixed waste collection to up to six separate bins/sacks (including the bin for residual waste) for other targeted waste streams. Most commonly this covers the waste streams paper, plastic, metal, glass, and biowaste. However, it is also possible that several materials are collected together in one bin/sack which is then called co-mingled collection; this is most common for metal and plastic together in one bin. This system can be supplemented by occasional collections by the municipality or other actors such as private operators, e.g.

	for bulky or hazardous wastes. The collection frequency varies in general, but it is mainly every two weeks for most fractions. Biowaste collection tends to be more frequent, presumably due to the nature of this fraction, while many cities apply more frequent collection during the summer. For some materials (e.g. glass), collection in some cases happens upon demand from the households.
Bring points 	Another system is bring point collection which is applied for the collection of recyclable materials and mixed waste, e.g. commonly for the collection of glass (mostly separate for white and coloured glass). But paper/cardboard, plastics, and metals are also collected at bring points. In addition, another form of bring point collection is the centralised/community composting of biowaste. Residents jointly share and manage a central composting facility. The advantage of this system is mainly that the collection points across the city are reduced substantially compared to door-to-door systems. Bring systems can also be complementary to door-to-door collection and they may target specific materials that are not covered by door-to-door collection.
Civic amenity sites 	**Civic amenity sites** or recycling centres are typically enclosed and sometimes staffed collection sites that are used as additional collection systems, usually accepting the same streams as collected in the door-to-door and bring point collection but also additional streams such as hazardous waste, garden wastes, and WEEE. Often civic amenity sites are operated by the municipalities themselves. Citizens can bring their waste there, which may or may not be free of charge.
Deposit and refund schemes 	**Deposit and refund systems** are typically applied for beverage bottles (cans) made of glass or plastic (metal) and are in most cases systems established at national level, e.g. by an EPR scheme.

Figure 4-31 shows the frequency of different types of waste collection across the UK.



Source: WRAP (2015).

NB: Values over 100 % owing to multiple collection frequencies across zones with local authorities. Includes textiles in some local authorities but not in others.

Figure 4-31. Percentage of local authorities operating each dry recycling scheme in 2013/14

The most appropriate collection strategies will depend on the characteristics of the collection zone (e.g. densely populated urban areas versus sparsely populated rural areas) and public acceptability of various strategies. Municipal collection points can be very cost-efficient and cost-effective in areas where citizens are sufficiently motivated to ensure widespread and effective separation (Table 4-14). Similarly, multi-stream collection systems such as *Optibag* and the *Quattro System* (see Operational data) have achieved very high separation efficiencies in Sweden, leading to 90 % recyclability (Björk, 2015; LAPV, 2012), but again require high levels of citizen engagement.

Waste collection strategy design

WRAP (2009) refers to the following four primary criteria that waste management authorities should consider when deciding on the type of waste collection system to implement or outsource for a particular waste fraction: (i) quality of material, (ii) cost-efficiency, (iii) cost-effectiveness, (iv) public acceptability. In terms of environmental performance, the separation efficiency and the quality of the separated material are the key criteria.

"Quality" is defined as "consistently delivering materials to the market that are effectively separated to meet reprocessor requirements, in the required volumes with security of supply, and at a price that sustains the market" (WRAP, 2009).

"Cost-efficiency" refers to the objective of minimising waste collection costs per household served, but may conflict with "cost-effectiveness", which ultimately represents the cost per tonne of final waste disposal avoided. From a societal perspective, "cost-effectiveness" represents a maximisation of resource efficiency and minimisation of environmental externalities associated with waste management per euro spent on waste management. From a narrower waste management authority perspective, "cost-effectiveness" can be defined as the economic balance of recyclable waste stream income minus collection costs and landfill charges. Thus, some low-cost collection strategies, such as alternate-week kerbside collection of co-mingled recyclable fractions may lead to a poor overall economic performance owing to reduced revenue for low-quality material streams. Table 4-14 under Operational data highlights some of the trade-offs in relation to glass collection.

"Public acceptability" is one of the prerequisites for establishing an effective system for separate collection of recyclables and waste materials. Varying public acceptability and engagement with recycling across Europe is a major reason why different waste collection strategies may be considered "best" across different Member States, and regions within them.

Key factors influencing separation efficiency

A best fit regression model developed in the UK explains 42 % of the variation in kerbside recyclable collection performance (kg/resident/year) across 434 local authorities using variables relating to socio-economic and regional characteristics and kerbside operational factors (WRAP, 2010). The frequency of residual waste collection was found to be an important driving force for the recycling rate. Fortnightly refuse collections were associated with higher dry recycling yields compared with weekly refuse collections, presumably because less frequent residual waste collection means a lower effective weekly capacity for residual waste, and increases citizens' consciousness of the need to reduce residual waste. Meanwhile, the number of recyclable fractions collected, the recyclable fraction containment volume and the frequency of collection were all positively associated with the recycling rate. These results highlight the importance of an integrated waste collection strategy that simultaneously:

- ensures adequate frequency (e.g. weekly) and containment volume for recyclable fractions, including separate collection of biowaste;
- minimises the residual waste collection frequency (climate-dependent, best achieved when the organic fraction is separated out);
- accepts a wide range of dry recyclable fractions.

ACR+ (2014), in its EU Capital Cities Study, notes that European cities with the highest rates of separate waste collection, such as Helsinki, have comprehensive door-to-door collection schemes alongside civic amenity centres which are free at the point of use. Meanwhile, analysis by WYG Environment (2011) showed that the best dry recycling performances in the UK were associated with:

- 100 % co-mingled dry recyclates collected fortnightly in wheeled bins; plus
- refuse collections being made fortnightly from wheeled bins; and
- at least the five main materials being collected for recycling: i.e. paper, card, cans, glass and plastic bottles.

Co-mingled collections were found to yield 30–40 kg more separated recyclable waste streams per household per year compared with kerbside sort collections, across the societal spectrum (WYG Environment, 2011). Although co-mingled collections have been found to be more expensive than kerbside sorting collections in the past, cost comparisons have often ignored the following factors for co-mingled collections: (i) the potential for fortnightly (rather than weekly) collections, (ii) higher recycling yields, (iii) reducing material recovery facility costs (WYG Environment, 2011).

Best practice

Ultimately, performance varies considerably depending on implementation, and there is significant potential to optimise all waste collection strategies in accordance with integrated waste management strategies (BEMP 3.3.1). Each local authority must decide on the most appropriate strategy for their area and residents, and under local conditions.

It is BEMP to design and implement a waste collection strategy that considers:

- the main features of the waste management strategy (e.g. number of separately collected waste fractions);
- the targets set in the waste management strategy (e.g. share of separately collected waste out of the total waste collected, impurity rates of the separately collected fractions, revenues from recyclables);
- the characteristics of the collection area (e.g. population density and main housing types);
- the current environmental attitudes and perceptions of residents;
- any other specific condition affecting waste collection (e.g. the relevant presence of tourists/commuters, specific economic activities, climate).

The main goal of a waste collection strategy is to collect, in a timely and economical manner, as much correctly source separated waste as possible, in order to ease the subsequent waste sorting/treatment with the aim to maximise recycling. In many cases, these objectives can be pursued by setting up the following:

- frequent door-to-door separate collection of food waste (e.g. weekly or more often depending on the season and climate);
- less frequent collection of mixed waste (e.g. every two weeks);
- door-to-door collection of recyclables (e.g. paper, cardboard, cans, plastics, glass), individually source separated where public acceptability allows, otherwise co-mingled and sorted at a material recovery facility; glass, followed by paper and cardboard, is more often more effectively collected separately;
- a convenient network of civic amenity sites (see section 4.5.3) that accept all waste fractions not collected door-to-door or in street containers from households, including hazardous waste and biowaste.

Achieved environmental benefits

Each kg of material diverted from landfill or incineration to recycling leads to significant resource and environmental savings, as outlined in Chapter 1 (e.g. Table 1-21). For example, sending biowaste for anaerobic digestion leads to avoided fossil fuel combustion and fertiliser production, and avoids significant GHG emission associated with the landfilling of biowaste. Recycling metal and plastic wastes avoids resource extraction and energy-intensive primary processing.

Implementation of an effective waste collection strategy can rapidly increase recycling rates. In Treviso, Italy, Contarina increased the MSW recycling rate from 55 % in 2013 to 85 % in 2014, simultaneously reducing residual waste to 53 kg per capita per year (Contarina, 2014).

Appropriate environmental indicators

Important parameters for monitoring the operation of the waste collection are waste collection frequency (per fraction), average distance between user and collection point (per fraction) and number of collection points (per fraction). However, these parameters do not allow the assessment of the environmental performance of waste collection. Instead, in addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- Participation rate, i.e. the share of the population using the waste collection system; data are usually available, based on estimations, surveys, how often the bin for recyclables is left out for collection, etc.;
- Share of the local area covered with a specific waste collection system (%).
- (Customer satisfaction (% of residents satisfied with household waste collection and specifically with the collection of the separately collected fractions).
- Collection of bulky waste on demand (y/n)[43].

Cross media effects

There may be a trade-off for waste collection strategies between maximising material recovery and minimising fuel consumption and emissions associated with collection. For PET plastic, for example, Bing et al. (2014) conclude that post-separation of co-mingled dry-recyclable collections is associated with higher costs and a higher environmental impact for the collection and transport stage owing to the limited number of separation centres compared with cross-docking sites for source separation. However, they note that post-separation is associated with a higher separation rate and lower installation costs for waste management organisations and householders, which is likely to result in a better life-cycle environmental performance.

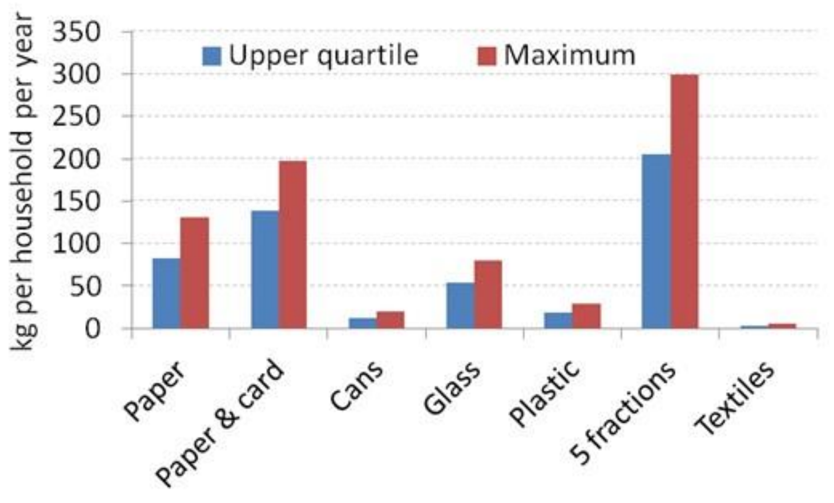
Operational data

Benchmarking

In the UK, WRAP has developed the "Local Authority Waste and Recycling Information Portal" which provides access to data on local authority recycling and waste schemes and performance benchmarks for kerbside dry recycling and residual collections: http://laportal.wrap.org.uk/UserHomepage.aspx.

WRAP (2010) identified best practice across UK local authorities for important dry recyclable fractions (Figure 4-32).

[43] Please note that this indicator referrers to a measure (availability of collection on demand of bulky waste) which is coupled with the presence of civic amenity sites collecting bulky waste. It is not meant that collection on demand of bulky waste substitutes the presence of the collection of bulky waste in civic amenity sites.



Source: WRAP (2010).

Figure 4-32. Top quartile and maximum achieved kerbside collection rates, expressed in kg per household per year, for waste management authorities throughout the UK in 2008/2009

The figures above correspond well with data on maximum selective collection rates across European cities provided by ACR+ (2014) and presented in Figure 4-30.

Multi-stream collection (source separation)

In terms of overall environmental efficiency, source separation of waste streams by householders is the preferred option in areas where there is a high level of public acceptance and engagement of residents, because it minimises contamination. Two examples of kerbside multi-stream collection of separated waste fractions are the Optibag system and the Quattro Select system (Björk, 2015).

The Optibag system comprises six colour-coded bags conveniently sized to fit within household kitchen or utility room cupboards, in order to separately collect the following waste fractions.

• Organic (green bag) • Plastic packaging (orange bag) • Metal packaging (grey bag) • Paper packaging (yellow bag) • Newspaper (blue bag) • Combustible (white bag)	 *Source:* Björk (2015).

The colour-coded bags can then be collected in a single refuse collection truck for transport to an integrated optical sorting plant where separated waste streams are checked and optically sorted for export to recycling facilities, or combustion/digestion on site. A total of 22 Optibag plants are currently in operation across Europe (Optibag, 2015). The modular approach maximises logistical efficiency and minimises collection costs but relies on a high level of householder motivation and engagement. More information is available at: http://www.optibag.com/reference-projects.

The Quattro Select system is based on householder separation of waste into eight separate fractions, stored in small multi-compartment containers that fit within two separate wheelie bins (Figure 4-33) collected in two collection rounds using vehicles with four separate compartments (Figure 4-34). The Quattro System has been in use in some areas of Sweden since 2004 (e.g. Lund), and has met with a high public acceptance, resulting in 90 % of all waste being recycled (LAPV, 2012). The tall wheelie bin format for the separated fractions improves health and safety for both householders and employees of the waste management companies by minimising the need to pick up heavy collection containers.

Source: Björk (2015).

Figure 4-33. Quattro Select bins

Figure 4-34: Multi-compartment collection vehicle (Svensson, 2016)

The bins in the Swedish Quattro system can also be equipped with extra small containers, for example to dispose of Lightbulbs and small batteries. The multi-compartment bin containing the residual/combustible waste and food waste is normally emptied once every two weeks. Bins that contain only packaging and newspaper can be emptied less often, usually once a month. The collection is conducted with special collection vehicles with four compartments. All four compartments separately compress the waste. Since the vehicles that have four compacting compartments are more complex than traditional collection vehicles, the need for maintenance is increased compared to standard single-waste-fraction collection vehicles (Svensson, 2016).

Civic amenity sites

Civic amenity sites or collection centres or bring banks are used for hazardous waste fractions such as used batteries, paints and other chemical products, electronic appliances, etc., and large waste objects that are not routinely collected. Moreover, civic amenity sites can also be used for a wide range of waste fractions that may otherwise be collected from households directly, with cost-saving and material quality advantages compared to household collection services owing to source separation (Table 4-14). However, an important criterion missing from Table 4-14 is public acceptance and motivation. In the example of the county of Aschaffenburg in Germany, described below, citizens are highly motivated and frequently use collection centres to dispose of waste not collected from households. Mixed dry recyclable fractions are collected in yellow sacks or bins from households in urban areas, but may be collected at central collection points in smaller villages.

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Figure 4-35. Metal collection bins in a collection centre in the county of Aschaffenburg, Germany

A wide range of fractions are collected in the 29 village collection centres located across the county of Aschaffenburg and operated by local citizens for limited opening hours, paid for by the county, including:

- eight fractions of non-ferrous metals;
- ferrous metal;
- batteries;
- glass;
- paper and card;
- plastics;
- non-impregnated wood;
- impregnated wood;
- three fractions of green cuttings (grass and leaves; wood, leaves and needles; trees without leaves);
- cooking oils;

- residual and biowaste (charged: EUR 0.18 per kg).

In addition to the 29 village collection centres, there are 131 smaller waste collection centres in Aschaffenburg and a few large centres where hazardous wastes, such as paints and solvents, can be delivered. Hazardous wastes are also collected twice a year from households using a mobile hazardous waste collection vehicle.

Important factors to maximise efficient use of bring centres are as follows:

- Accessibility – centres should be as widely distributed as possible so that most of the population has one in close proximity, and located conveniently close to major roads or frequently used amenities (e.g. out-of-town retail centres) so that citizens can drop in without taking long detours (and consuming extra fuel).
- Opening hours – well-publicised and extensive opening hours, including out-of-office hours and weekends, maximise social acceptance and use of collection centres. Waste management authorities may need to find a trade-off between the duration of opening times and the number (accessibility) of centres, especially in rural areas.
- Clear indications – clear indications are essential to improve ease-of-use and minimise contamination / maximise the quality of separated materials.

Mobile collection centres

The LIFE EMaRES project demonstrated application of the *Dynamic Ecopoint* concept in Italy (Umbria region); a mobile collection centre for low-volume hazardous waste items that circulates around convenient collection points within a region (e.g. shopping centres, markets, parks) according to a fixed timetable. Target waste streams are WEEE, used cooking oil and used batteries, which typically amount to just 3–5 kg/year/inhabitant, but improper disposal of which can have serious environmental consequences in terms of water pollution, toxicity and resource (rare-earth metal) depletion.

Source: EMaRES (no date).

Figure 4-36. The Dynamic Ecopoint "Ricimobile", a 7.5-tonne vehicle for the collection of small WEEE, used cooking oil and batteries

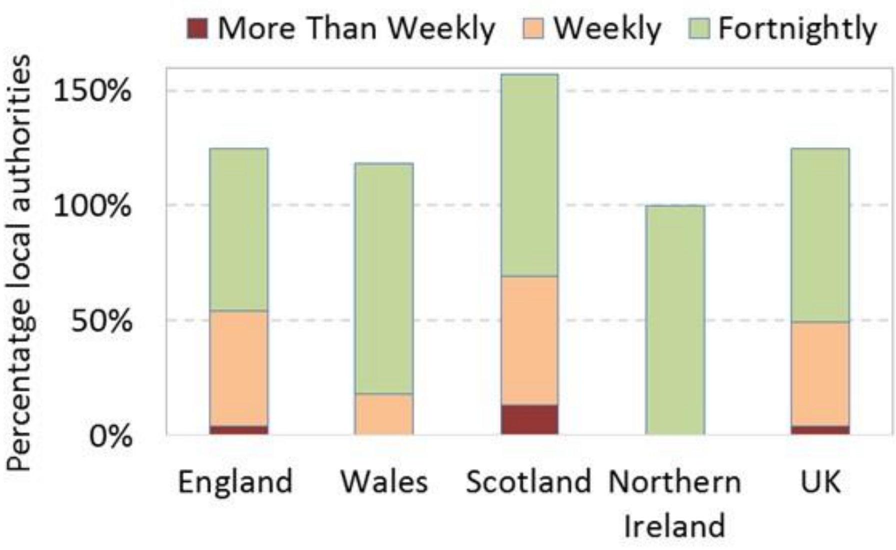
Preliminary activity as of September 2015, since the Dynamic Ecopoint started operating in May 2015, indicate an annual collection rate of about 2 000 kg/year. This represents 2 % of the static Ecopoint collection rate in the region for WEEE, but continues to increase as citizens become familiar with the service and schedule (EMaRES, no date). See also the Île-de-France mobile civic amenity service example under Reference organisations below.

Optimising the frequency of residual waste collection

Reducing the frequency of collection for residual waste bins provides a strong driver to recycle waste, whilst also reducing the cost of residual waste collection. Across the UK, there has been a move towards fortnightly collection of residual waste bins (Figure 4-37). Important points for reduced frequency of residual waste collection include:

- clearly publicised scheduling of collections;
- provision of durable closed bins (to avoid odour and pest problems);
- provision of wheeled bins[44] to "squeeze" waste (WYG Environment, 2011);
- separate collection of biowaste, especially in warmer climates.

[44] Provision of wheeled bins was mainly forced by the implementation of the European Directive 90/269/EEC - manual handling of loads, for preventing occupational disorders, particularly of back pain and injury, of the collecting staff.



Source: WRAP (2015).

NB: Values over 100 % owing to multiple collection frequencies across zones with local authorities.

Figure 4-37. Percentage of local authorities across the UK collecting residual waste by frequency, 2013/14

Clear instructions for households

It is crucial that whatever collection strategy is in place is clearly conveyed to citizens so that they know what to put in which bins/sacks, and when to leave them out for collection (see also BEMP 0 on awareness-raising of residents). Figure 4-38 displays information leaflets produced by Worcester County Council in the UK regarding mixed dry recyclable bin and sack collections. The authority also provides a website with full information about kerbside collection times (based on a postcode search) and alternative options[45]. Household calendars of collection dates are useful to remind citizens when to put out bins for collection.

[45] http://www.worcester.gov.uk/recycling


Please put these items LOOSE into your GREEN BIN
Worcester City Council
Tel: 01905 722233
customerservicecentre@worcester.gov.uk
Corrugated Cardboard
Plastics
Paper & Thin Card
YES
Tins & Cans
Cartons & TetraPak
Glass Jars & Bottles
DO NOT put these items in the green bin
Worcester City Council
Tel: 01905 722233
customerservicecentre@worcester.gov.uk
Garden Waste
Textiles & Shoes
Plastic Bags
NO
Contaminants such as these will result in non collection of your green bin.
Tin Foil & Foil Containers
Nappies
Food Waste

There are lots of other items that can be re-used or recycled in Worcester City.

Recycling centres have been placed around the city to allow you to recycle glass, paper, cans, textiles, shoes, books, CD's and Videos. Below is a list of the main recycling centres in Worcester City, for a full list please contact us on 01905 722233.

- Co-op, Ombersley Road
- Homebase, Hylton Road
- Sainsbury's, Swanpool Walk
- Tesco, Millwood Drive
- Tesco, St Peter's Drive
- Viking Afloat, Lowesmoor
- Household Waste Site, Bilford Road
  - electrical items, batteries, white goods, garden waste, used oil
- Household Waste Site, Hallow Road
  - electrical items, batteries, white goods, garden waste, used oil
- Pitchcroft Car Park, Severn terrace
- Worcester Golf Range, Weir Lane
- Countryside Centre, Wildwood Drive

Choose to reuse

Buying and donating goods for reuse helps:

- Prevent valuable resources form going to landfill
- Saves energy and raw materials
- Raise funds for charities
- Make good quality items available at affordable prices

Item	LifeTech 01905 756067	St Richard's Hospice 01905 745495	Worcestershire Lifestyles 01905 731352	Spokes 01562 861154
Computers	Yes			
Furniture inc beds		Yes	Yes	
Bicycles				Yes
Electrical Items		Yes	Yes	
White goods		Yes	Yes	

Source: Worcester.gov.uk (2015). http://www.worcester.gov.uk/recycling Last access April 2015.

Figure 4-38. Information leaflets provided by Worcester County Council (UK)

Table 4-14 summarises the advantages and disadvantages of six alternative approaches for collection in relation to glass fractions. It is important to note that there is a 10 % rejection rate for fractions collected in co-mingled streams (WRAP, 2010).

Table 4-14. Overview of the performance of six alternative approaches for glass collection

Criteria	Dedicated collection rounds (colour sorted)	Kerbside sorting (colour sorted)	Kerbside sorted dry recycling (clear and colour glass streams)	Mixed glass collections	Fully co-mingled recyclables	Household waste collection centres
Ease of collection	3/5. Collections are easy to operate but are slowed by the colour sorting process. Collected glass can be bulked at a transfer station prior to transfer or delivered straight to reprocessors.	3/5. Sorting material at the kerbside reduces the speed of collections compared to bin collections. However, innovative vehicle designs now exist to make the sorting process as easy as possible.	3/5. Similar to fully colour sorted. So not expected to make collections significantly easier, nor lead to significant reductions in required resources.	4/5. Kerbside sorting schemes are well developed for the collection of glass. For co-mingled schemes, the glass can be added to an extra compartment on a modified refuse collection vehicle.	5/5. Refuse collection vehicles can be used on alternate weeks for dry recycling (provided they are cleaned), and large round sizes can be achieved. Material is either taken to a transfer station for onward transport or delivered straight to a material recovery facility (MRF).	4/5. Collections are familiar to most authorities, and aided by more modern design of banks for easy collection. However, there is a need for servicing schedules that ensure banks are emptied at appropriate intervals.
Quality of recyclate	5/5. Colour-sorted cullet will be relatively free from contamination and can be used to create the full range of glass products. Probably the best quality cullet of all collection options (including bring sites).	5/5. The colour-sorted cullet will be relatively free from contamination and can be used to create the full range of glass products.	4/5. The level of variation in the coloured glass stream may prohibit closed-loop recycling. Technology at glass recyclers may allow for colour separation, in which case both streams of glass can be fully recycled.	3/5. A mixed recyclate will always be less acceptable to the container glass industry – but keeping the material separate from other dry recyclables is the key to maintaining an appropriate quality for creating new container products.	1/5. Of all the schemes described in this guidance, collecting glass co-mingled with other recyclables produces the lowest quality cullet. The majority of glass collected through this type of scheme can only be used for low-value applications, such as aggregate.	5/5. The quality of recyclate from bring banks is high, with only occasional contamination from incorrectly sorted glass.

Criteria	Dedicated collection rounds (colour sorted)	Kerbside sorting (colour sorted)	Kerbside sorted dry recycling (clear and colour glass streams)	Mixed glass collections	Fully co-mingled recyclables	Household waste collection centres
Environ-mental perfor-mance	5/5. Colour separating the glass reduces the energy requirement of both reprocessors and the glass industry.	5/5. Colour-separated cullet offsets the need for virgin raw materials in the glass industry, reducing energy requirements. Furthermore, the impact of the collection vehicles is greatly reduced, as is energy consumption at the MRF or transfer station.	4/5. The mixed colours in the coloured stream may prohibit recycling, depending on the technology available at the glass recyclers. The extra effort of separation at the kerbside would result in less energy being needed by the glass recycler for separating colour streams (resulting in higher revenue for the material).	3/5. Mixed glass collections are of more benefit to the environment when the glass can be colour sorted for closed-loop recycling. This step may require more energy than the alternative of sorting the material at the kerbside, depending on the type of scheme used.	2/5. The environmental performance of co-mingled collections is lower than those where glass is collected separately, as the benefits of closed-loop recycling have not been realised.	4/5. The environmental performance of bring banks is boosted by vehicles travelling less than for kerbside collections, and the ability to fully recycle the collected glass. However, depending on the location of the banks, residents' travel distances may outweigh any benefits. Location of the banks is therefore an important factor in their operation.

Criteria	Dedicated collection rounds (colour sorted)	Kerbside sorting (colour sorted)	Kerbside sorted dry recycling (clear and colour glass streams)	Mixed glass collections	Fully co-mingled recyclables	Household waste collection centres
Cost of collection	High. Relatively high operational cost partially offset by the revenues received for sale of materials.	Medium. Lower collection costs than a dedicated fully colour sorted glass collection, and, when whole system costs are considered, comparable if not lower cost than co-mingled collections. Revenues from the sale of materials can be used to offset the costs of collection whilst co-mingled schemes involve the payment of MRF gate fees.	Medium. Similar to the kerbside sorting option. The coloured glass stream will, however, generate lower revenue per tonne than a three-stream glass collection. WRAP studies show marginal differences in cost between collections that separate glass into three streams and those that separate into two streams in a kerbside sorting service.	Medium. A lower revenue per tonne will be received for the glass compared to colour sorted options. Cost impacts for a kerbside sorting service are likely to be negligible. Investment in new vehicles may be required if a two-stream co-mingled collection is introduced.	Low. Co-mingled collections can be less costly to operate but the collection cost is offset by a higher gate fee at the MRF and the lower revenue received for sale of the materials.	Low. The cost of operating banks is low compared with kerbside collection services. When run in parallel with kerbside glass collection, some banks may not be cost-effective, depending on the contractual arrangements in place.

Source: WRAP (2012)

Material recovery facility

Co-mingled collection of dry recyclable fractions is a popular strategy because it involves less effort from citizens than source separation, and is therefore considered to yield higher recycling rates in regions where there is less history of recycling. Co-mingled collections must be sent to a material recovery facility (MRF) for sorting and onward shipment to production facilities for final recycling into products. Modern material recovery facilities use a combination of sorting technologies, including rotating drum size sorters and opto-electronic (e.g. infra-red plus air pulse) sorters, alongside manual sorting. Note that two specific BEMPs on sorting of co-mingled light packaging or collected mixed plastics are presented in Section 4.7.

Applicability

The optimum approach to maximise the recycling rate whilst minimising costs will vary considerably depending on local circumstances, including human behaviour which is partly related to socio-economic situation. WRAP (2010) found that the prevailing socio-economic status within local authority areas was an important factor in determining the recycling rate, with lower recycling rates associated with a lower socio-economic status, perhaps reflecting a low prioritisation for waste management in poorer households.

Whilst bring centres can be an effective and cost-efficient strategy for waste collection in countries and regions where recycling is well established in the public psyche, in other areas, including poorer regions, waste collection at bring centres should be restricted to those waste types that really cannot be collected from households, such as bulky objects and hazardous wastes. More costly strategies, such as door-to-door collections (see Italian example in Box 4.12), may be required to achieve acceptable levels of recycling across the major dry recyclable fractions in such areas.

Less frequent (e.g. fortnightly) residual waste collection may not be practical in warmer climates owing to odour and hygiene issues if it contains biowaste. The separate collection of biowaste is crucial as then other waste fractions can be collected more efficiently (ACR+, 2014). In hot countries, the collection frequency must be higher. In Milan the biowaste collection is twice a week; in Germany it is usually once a week in summer and twice a month in winter.

Driving force for implementation

Targets established in the Landfill Directive and the Waste Framework Directive, alongside associated landfill charges and commodity prices (recyclate value), drive collection of separated recyclable fractions. Bans on biowaste and combustible waste being sent to landfill in Sweden helped to drive implementation of the highly effective Optibag and Quattro System collection systems (Björk, 2015). However, high levels of citizen awareness and engagement with waste recycling also played an important role in the efficacy of these systems.

Personnel costs drive optimisation of waste collection strategies in terms of the economic efficiency of collection (e.g. automation, side loaders for one-man-operation). In some cases, recyclate revenues are a driving force too.

Fuel costs drive optimisation of waste collection strategies in terms of the energy efficiency (minimisation of GHG emissions and reduction of air pollution) of collection.

Economics

Costs for the staff, for the collection fleet and bins, for treatment and for landfill are major determinants of the economics of different waste collection strategies. For example, it is essential for strategy and logistics optimisation to invest in "multi-modal" collection vehicles that are able to empty different kinds and sizes of collection bins (see the example of Vienna waste authority in BEMP 4.5.2). In some cases recyclate revenues are an additional determinant. For example, the price of cullet determines whether colour sorting of glass is economically attractive to waste management authorities (WRAP, 2012).

Bing et al. (2014) compared the GHG emission intensity of different collection strategies for plastics in the Netherlands. Results were highly region- (context-) specific, and in some scenarios separate collection of polyethylene terephthalate (PET) bottles was found to be both cost- and carbon-efficient. Bing et al. (2014) reported that post-collection separation scenarios were found to have the highest costs and environmental impacts owing to the limited number of separation centres compared with abundant cross-docking sites for source separation. However, post-collection separation achieves a higher separation rate and lower installation costs for municipalities and householders.

WYG Environment (2011) suggests that local authorities rarely undertake comprehensive comparisons of costs across waste collection strategies. It is essential that representative (optimised) collection frequencies and economic data on recyclate revenues, material recovery facility costs and landfill costs are accounted for in integrated cost-benefit analyses. Proximity to a material recovery facility can significantly influence the relative costs of co-mingled versus separated collection, and WYG Environment (2011) suggests that co-mingled collection can be a cost-effective collection strategy.

Quattro Select collection vehicles cost GBP 300 000 (EUR 420,000) each, over double the price of conventional single-compartment collection trucks. However, each Quattro Select vehicle has a capacity of 10 tonnes, can replace at least two conventional trucks, and requires less manpower (one person per truck). In Lund, eight Quattro Select vehicles and one truck cover up to 2 400 houses, equivalent to 4 800 bins, with each operator emptying up to 180 bins in one shift (LAPV, 2012). The need for just two separate vehicle collections per household can facilitate logistics optimisation further, whilst high separation efficiencies greatly improve the overall economic efficiencies of waste management companies by minimising residual waste disposal costs.

Reference organisations

Box 4.11. Gwynedd Council waste collection strategy, involving separate biowaste collection and kerbside sorting

The UK has only recently begun to recycle food waste in composting and anaerobic digestion plants; food waste recycling increased from 1 % in 2006 to 12 % in 2012 (Defra, 2014). Gwynedd Council collects food waste separately once a week from the kerb in 22-litre brown containers (left). The following fractions of food waste are collected in small kitchen containers and biodegradable bags provided by the Council (left): any food waste, cooked or raw, including fruit and vegetable peelings, cheese, bread, beans, meat, eggs, plate scraps, food passed its best before date, tea bags, fish, etc., but excluding liquids such as milk or oil. Food waste is sent for anaerobic digestion.

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Gwynedd Council collects the following dry mixed recyclable fractions in blue boxes once a week, on the same day as food waste collection, using a kerbside sorting service: paper (newspaper, magazines, office paper, junk mail, shredded paper), food and drink cans, glass bottles and jars, foil, aerosols, plastic bottles, plastic pots, tubs and trays, yoghurt and butter pots, plastic containers for fruit and vegetables and meat trays, food and drink cartons, fruit juice or soup cartons, cardboard.

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Green garden waste and residual waste are collected in separate brown and green 240-litre wheelie bins (right) on alternate weeks, coinciding with food waste and mixed recyclable waste collection days.

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Source: Gwynedd Council (2015).

Box 4.12. Example of twice-weekly biowaste collection in Milan

The municipality of Milan covered by Amsa comprises 1.281 million citizens, and first introduced door-to-door collection of household biowaste in November 2012 for one quarter of the city of Milan. The scheme was expanded to the entire city over four stages, and was fully implemented by June 2014. Compostable bags and 120-litre brown bins are used for collection from houses (smaller 35-litre brown bins are available on request). Small 10-litre aerated kitchen baskets, designed with an airy structure to minimise odours and anaerobic decomposition, are used in apartments. Biowaste is collected twice a week.

The waste management organisation coordinated activities with the City of Milan. Census data from the area were used to prepare the service setup. A software model was used to determine logistics requirements, based on factors such as bin weights, vehicle loads, route distances, crew productivity, etc. The model was validated using data from trial runs.

Following implementation of the plan across three quarters of the city, the recycling rate for food waste rose from 35 % in 2011 to 48 % in 2014, equating to 90 kg per capita per year. Composition analysis at the start of the service showed that just 3.8 % of the food waste fraction comprised non-compostable (contaminant) material. This increased to 5.1 % eight months into the campaign, but dropped back down to 3.7 % after the quality awareness campaign.

Source: R4R (2014c).

Box 4.13. Example of waste collection strategy operated by the county of Aschaffenburg in Germany

The county of Aschaffenburg in Germany collects residual waste in padlocked wheelie bins that contain identifier microchips and are weighed on the back of refuse collection trucks (see pay-as-you-throw BEMP), with rubble collected separately. Paper, plastic and metal cans are collected weekly from the kerbside in yellow sacks in urban areas, and in waste collection centres in villages (80 % of metal is collected in waste collection centres). Glass, garden waste and various other fractions such as batteries are collected in local waste collection centres (see description under Operational data). In small villages, local citizens are employed by the County to operate recycling stations.

Source: County of Aschaffenburg (personal communication).

Box 4.14. Mobile civic amenity sites in Île-de-France

This innovative solution addresses waste collection at source in an area where the implementation of traditional civic amenity sites is extremely challenging (because of urbanisation, high population density and limited access of citizens to personal vehicles for the transport of bulky waste). Collection containers are temporarily left in public areas such as town squares and marketplaces, and opening hours communicated to citizens by local authorities. The service is provided free of charge to citizens living within the municipality, and accepts construction and demolition wastes, mixed bulky wastes, garden waste, WEEE and textiles, among other fractions. The system is regarded positively by citizens and attracts increasing numbers of users.

Source: R4R (2014a).

Box 4.15. Initiating door-to-door collection in Lisbon

This example from Lisbon provides an example for municipalities with less developed waste collection strategies on how to rapidly upgrade the service offered, including the introduction of separate biowaste collection.

Selective kerbside collection of paper/ cardboard and packages was introduced gradually to replace bring banks and to complement kerbside collection of residual waste. Separate collection of biowaste was also implemented for small commercial premises such as restaurants, canteens and markets. The collection frequency was also adapted progressively, beginning with alternate collection of residual and recyclable waste fractions. Contact was made with waste producers during collection rounds to disseminate information material and to answer any questions on the new service. A communication campaign was used to generate public awareness of the new system, and local stakeholders were consulted and involved during implementation. The quantity of selectively collected recyclable material has increased significantly under the new system, from 6 % to over 20 % of the total MSW generated.

Source: R4R (2014b).

Box 4.16. Contarina SPA integrated waste management collection strategy

Contarina is a publically owned waste management compnay serving a region of 1 300 km^2 and a population of 554 000 inhabitants across 50 municipalities in the Veneto region (Italy), with 260 000 users across a range of urban and rural settlements. Contarina employs separate waste collection strategies for less densely populated areas and densely populated and often logically complex (historic) urban centres:

Standard service for less densely populated areas (below)

Service for densely populated urban areas, including small bags for users with limited space (below)

Contarina implements a PAYT approach (BEMP 4.3.3). Users are charged a 60 % fixed fee based on household numbers, plus a 40 % variable fee based on home composting (-30 %) and number of bin collections. Waste collection costs are less than half the Italian average, at EUR 104 per user. Contarina has successfully increased the recycling rate for MSW in Treviso from 55 % in 2013 to 85 % in 2014, simultaneously reducing residual waste to 53 kg per capita per year.

Source: ZeroWasteEurope (2015).

The European-funded project IMPACTPAPEREC provides guidance on how to improve separate collection for paper and cardboard recycling, presenting a number of case studies and best practices. More information is available at: http://impactpaperec.eu/en/home/.

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4.5.2. Inter-municipal cooperation (IMC) among small municipalities

Summary overview							
It is BEMP for small and medium municipalities to adopt inter-municipal cooperation that allows the implementation of measures that would be too costly for them to implement alone and can result in the improved environmental performance of the waste management system. Municipalities can join together to operate or contract out some waste management services, with the aim of delivering economies of scale and building critical mass. Inter-municipal cooperation makes it possible for the municipalities involved to: - share administrative overheads, - reduce unit costs and improve service quality through economies of scale, - attract investment funds reserved for projects of a specified minimum size (e.g. EU structural funds and other investment mechanisms) and - enhance economic performance through coordinated planning while allowing better environmental protection.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
There are no specific barriers for the application of inter-municipal cooperation in waste management. However, benefits from the economy of scale are only evident for small and medium municipalities.							
Specific environmental performance indicators							
In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicator to assess the successful implementation of this BEMP is: - implementation of inter-municipal cooperation with other municipalities (y/n).							

Description

Inter-municipal cooperation (IMC) is defined as the collaboration of several municipalities with the aim of providing a joint public service (Halmer and Hauenschild, 2014). This is not a new instrument, but just an approach taken by municipalities for decades to improve the economic performance of municipal services. It has been proven that IMC takes advantage of proven economies of scale in waste management for small municipalities, as illustrated by Bel and Fageda (2010) when studying the waste management costs of 65 municipalities from the Spanish region of Galicia. The advantages of IMC lie in the reduction of avoidable duplication of work and the creation of synergies. IMC improves resource efficiency and leads to improved services and less costs associated with public services conventionally with a high cost intensity, such as waste management.

The empirical evidence shows that, for small municipalities, the collaboration with other municipalities reduces the total cost of management. For larger populations, the effect of economies of scale is negligible or even opposite to that observed for small municipalities (Bel and Mur, 2009). The same authors found an interesting and somewhat unexpected effect of inter-municipal cooperation in small municipalities: under certain conditions, a high collection frequency does not increase the waste management cost. This is directly opposite to any other empirical observation but the authors identified this effect as coming from the same concept of economy of scale, as for example the same truck serves several municipalities. On the management side, inter-municipal cooperation is not necessarily a money-saving process, but, according to the Council of Europe (COE et al., 2010), the good practice application makes it possible for involved municipalities to:

- share administrative overheads,
- reduce unit costs and improve service quality through economies of scale,
- attract investment funds reserved for projects of a specified minimum size (e.g. EU structural funds and other investment mechanisms) and
- enhance economic performance through coordinated planning while allowing better environmental protection.

The crucial point for this BEMP is: What is the definition of 'a best practice in inter-municipal cooperation' for waste management and what is the real impact of such a measure? First, it should be clear that inter-municipal cooperation is an economic instrument implemented with the aim of saving costs, sharing risks and reducing cost intensity; technically, it does not improve the service (e.g. many cooperation agreements are based on the existence of a shared landfill). Certain requirements have to be met for best practice cooperation (COE et al., 2010):

- the building of central waste disposal or treatment plants;
- the development of joint policies for solid waste management; and
- the establishment of recycling to achieve better environmental protection.

Municipalities collaborating in the management of waste are relatively well established in Europe. A survey among the town halls of France’s large cities revealed that 63 % of them transferred waste management to a consortium of towns (Djemaci, 2009). So, inter-municipal cooperation is not a best environmental management practice that leads directly to a better environmental performance, but it is an approach that allows the implementation of best practices only achievable by organisations of certain size or that would be too costly for small municipalities to implement alone. The United Nations Development Programme emphasises that only the local scale is small enough to handle day-to-day communication with citizens and large enough to support the specialisation of functions; this can be achieved by sufficiently large municipalities or through the development of inter-municipal cooperation agreements (LDG, 2006).

According to the Council of Europe et al. (2010), there are at least 15 basic elements of a well-performing inter-municipal cooperation scheme (see Table 4-15).

Table 4-15. Basic structure of inter-municipal cooperation (IMC) (CoE et al. 2010)

PHASE	STEPS

I. INITIATING IMC (explore possibilities for cooperation with partners, examine risks/advantages of IMC, launch formal negotiations)	1. Identify needs and opportunities
	2. Identify potential partners and possible areas of cooperation
	3. Analyse the legal and economic environment
	4. Decide on entering into IMC and set up the negotiating platform
	5. Build awareness and support
II. ESTABLISHING IMC (build foundations of IMC and reach agreement with partners on IMC structures and operation)	6. Identify IMC scope
	7. Choose the legal form
	8. Determine the financial arrangements
	9. Define the institutional arrangements
	10. Finalise Agreement/Statute
III. IMPLEMENTING AND EVALUATING IMC (mechanisms to ensure effective IMC operation)	11. Establish management and representative structures
	12. Develop cooperation mechanisms
	13. Ensure continuous monitoring and self-assessment
	14. Ensure continuous and effective communication
	15. Conduct regular evaluation

Source: COE et al.(2010)

Achieved environmental benefits

The environmental benefits of inter-municipal cooperation in waste management services correspond to the benefits of the best practice that the arrangement between municipalities makes it possible to apply. The borderline of the applicability of a best practice to small municipalities is never clear, but some examples of the performance of cooperation are shown in Table 4-16.

Table 4-16. Application of BEMPs by inter-municipal cooperation examples and their environmental benefit

County	Member State	Applied BEMP	Environmental benefit	Comments	Reference
Grand Besançon	France	PAYT system	Immediate reduction of the residual waste by 1 % the year following the implementation of a volume-based PAYT scheme. In 2012, after weight-based PAYT implementation, the residual waste was reported to have been reduced by 10 %.	The IMC allowed the application of a different approach between the main town (Besançon) and the surrounding small towns.	Djemaci, 2009 Sybert, 2015
Harju	Estonia	Waste sorting of biological and paper waste	Enhanced collection efficiency of recyclable materials. Increased collection by 2.5 times compared to the current	This is an estimation of performance after a proposed route for IMC	Põldnurk, 2015

			situation.	implementation.	

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicator to assess the successful implementation of this BEMP is:

- implementation of inter-municipal cooperation with other municipalities (y/n).

Cross-media effects

No environmental cross-media effect is foreseen. However, the implementation of such a scheme requires a strong regulatory framework for its governance (see Bolgherini, 2011, for further details), to avoid the overlapping of responsibilities or a distortion of the primary objectives of the scheme (e.g. the IMC can improve efficiency and reduce management costs, but the fee or taxes paid may even increase given the introduction of new, less pollutant, waste treatments).

Operational data

IMC as a way of improving the performance of municipal services is basically a very old measure. However, it has only recently been identified as an effective measure for small-scale municipalities, as other systems (e.g. private outsourcing) have been given priority in terms of increasing the efficiency of the system. The current economic situation, however, has imposed very strict deficit objectives and austerity in public services, and a re-municipalisation effect is occurring to save costs and ending contracts with private companies. IMC has received far less attention (Bel and Warner, 2015). Small municipalities are more sensitive and have less experience when facing financial, organisational, dimensional and expertise problems, as well as having more problems to fulfil challenging objectives in the delivery of public services. So, IMC has been viewed recently as the most promising solution for them (Bolgherini, 2011).

Higher tier local government structures are usually responsible for the implementation of cooperation agreements, e.g. *comarcas* or *mancomunidades* in Spain, *communautés de communes* in France, *unioni di comuni* in Italy. However, one of the common elements of cooperation through these supra-municipal cooperation arrangements is its voluntary character. In some Member States (e.g. in Germany), waste management and disposal have to be organised at county level. Thus, IMC has a legally binding character.

Several European case studies[46] for IMC for waste are described in the boxes below. Not much detail is given on the specific administrative arrangement, as the regulatory framework is dependent on national and regional legislation, but the focus is instead

[46] Only three examples are shown in this current version of the text. More examples will be included as a result of the research exercise.

on the specific outcomes and benefits obtained in terms of waste management performance.

Box 4.17. Besançon (France) case study

The city of Besançon implemented an incentive-based financing scheme via the bin tax in 1999, called REOM (Redevance d'Enlèvement des Ordures Ménagères). Thanks to the participation of the city in the Greater Besançon waste authority, CAGB, the scheme was transferred to the ring of 59 municipalities. This bin tax is one of the multiple versions of the PAYT (pay-as-you-throw) system, charging per volume generated by household. In order to have a, somehow, fair scheme for the service rendered, the municipalities of the ring introduced a fixed part and a variable part according to the number of people in the household and the frequency of the service provided. The system ensured that an increase in waste volume would suppose an increase in the waste fee, increasing more with higher frequencies than with higher bin volumes. The measure had an effect after the first year of implementation, decreasing the residual waste by 1 % and increasing the recyclable fraction by the same amount, while the city saved EUR 5.25 per capita per year. The authorities also noticed a change in the citizens' habits regarding waste (Djemaci, 2009).

A new system was implemented after the LIFE project "Waste on a diet", with a higher impact in the municipality of Besançon, achieving an immediate reduction of 10 % of the residual waste fraction in the test phase and 7 % in the actual implementation (SYBERT, 2015; Pre-waste, 2012).

Box 4.18. Harju (Estonia) case study

A study on the optimisation of the waste services in the region of Harju in Estonia was published in 2015. It shows the probable impact of the implementation of centralised separate biowaste and paper collection in rural areas. The study identified the administrative, economic and logistical benefits of the adoption of inter-municipal cooperation. In rural areas, the main source of costs is transportation (i.e. the fuel consumed and the collection time per tonne of waste is higher). The administrative burden is identified as one of the main barriers for improvement. For instance, in the area analysed, there are 23 officials or more in charge of waste management in the 23 municipalities. However, the multiplicity of tasks of these officials, with a very low specific dedication to waste, could easily be solved with only four officials in charge of a supra-municipal waste structure. In total, 70 % of the municipalities in Estonia have less than 4 000 inhabitants and would benefit from such schemes (Põldnurk, 2015).

Box 4.19. City of Friedberg (Germany) (AWB, 2015)

In 2005, 209 tonnes of bulky wastes were collected in Friedberg by street collection. In 2010, there were only 125 tonnes. In the same period, the bulky waste delivered to the recycling centre in Friedberg increased from 121 tonnes to 604 tonnes. Waste fees could be reduced by 2011.

Applicability

There are no specific barriers for the application of inter-municipal cooperation in waste management. However, benefits from the economy of scale are only evident for small and medium municipalities.

Economics

In rural areas, there is an increased probability of administrative and logistical inefficiencies affecting the waste management service. High waste transportation costs, multiplicity of tasks, different pricing and lower control over the collection service are only some of the symptoms of such a problem (Põldnurk, 2015).

Three main factors affect the performance of inter-municipal cooperation: size of population, volume of service and dispersion of population (Bel and Warner, 2015). The effect of these variables can be translated into:

- **economies of scale**: they exist when the cost per tonne of managed waste decreases as the total volume increases (e.g. for the same truck, the higher the volume transported, the lower the cost per tonne of waste);
- **economies of density**: they exist when the fixed cost per tonne is spread across a large number of users (e.g. the water distribution network);
- **economies of scope**: they exist when the cost per unit of a certain service is reduced when other services operated by the same management structure increase.

Economy of scope affects the administrative burden of the service. It has been proven that the economy of density does not affect waste management costs, while economies of scale only affect the small municipalities when arranging inter-municipal cooperation agreements for the waste management service.

The influence of IMC alone on the economic performance of a waste management service is not easy to determine, as its implementation usually includes new treatments or sorting systems. Bel and Mur (2009) performed a statistical analysis and determined the "pure" influence of the existence of IMC in small municipalities: 16 % cost reduction in municipalities under 5 000 inhabitants, while the difference was not statistically significant for municipalities above that size. Djemaci (2009) attributed a cost reduction of EUR 5.25 per capita per year to the application of IMC in the area of Grand Besançon, although the fee system had to be changed to a PAYT system. In the Estonian region of Harju, the establishment of IMC would save around EUR 28 per inhabitant per year (including a raise in the residual waste fee) in an optimistic scenario and EUR 10 per inhabitant per year in a more realistic projection (Põldnurk, 2015). In Germany, the cities of Dreieich and Neu-Isenburg reduced their garbage fees in January 2015 by 10 % as a result of inter-municipal cooperation. This was possible because the expenditures on material resources decreased due to IMC. For example, the 120-litre residual waste bin is priced at EUR 20.20 instead of EUR 22.60 per month with fortnightly emptying. This means a saving of EUR 28.80 per year. In a four-person household, this is a saving of EUR 7.20 per capita per year (Werwitzke, 2013).

Driving force for implementation

The existence of vast experience in municipal cooperation in Europe has shown the feasibility and efficiency of cooperation schemes. However, the legal and regulatory framework needs to be well defined, which is usually done at regional level. The higher efficiency, the removal or reduction of tasks' multiplicity and the inherent cost savings of IMC implementation in small municipalities are also important drivers. In addition, new challenging recycling and material recovery goals from the waste management would demand techniques and technologies that require higher capital investment and would be unaffordable for a single, small municipality.

Reference organisations

Grand Besançon is considered to be a good example of the application of BEMPs. The IMC in place allowed the extension of BEMPs to small towns and villages in the area. For more details, see http://sybert.fr/presentation.html.

In addition, the establishment of new IMC schemes has been and will continue to be key in the achievement of new waste policy targets and it is the focus of new initiatives and research around Europe. A reference organisation on the development of IMCs is the Council of Europe and the United Nations Development Programme.

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4.5.3. Civic amenity sites

Summary overview
As a key complement to an effective door-to-door (kerbside) collection of the most common waste fractions, it is BEMP to run civic amenity sites (also called container parks, collection centres, clean points, ecopoints, recovery sites, waste parks, etc.) where citizens and small businesses can drop off as many waste fractions as possible for separate collection. Elements of best practice for civic amenity sites include the following: - presence of at least a civic amenity site in the local authority or regular periodical presence of a mobile site. - separate collection of as many fractions as possible and the possibility to drop off any household waste. - training of the staff of the civic amenity sites to maximise recycling, recovery and appropriate safe disposal. - watertight paved area and collection of run-off water for appropriate treatment. - proximity of the sites to citizens (e.g. accessible without a car by a large share of the population), also thanks to mobile/temporary collection sites. - long opening hours to enhance convenience for citizens. These may change across seasons (especially for green cuttings).

Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**

Applicability
The concept of collection centres is broadly applicable. The ultimate recyclability of the waste streams collected also depends on the availability of downstream markets.

Specific environmental performance indicators
In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: - number of civic amenity sites per 100 000 residents; - number of different fractions collected at the civic amenity sites; - availability of product/material exchange areas aimed at fostering reuse in civic amenity sites (y/n); - easy accessibility of civic amenity sites, e.g. without a car (y/n).

Benchmarks of excellence
- For municipalities with at least 1 000 residents, there is at least one civic

amenity site in their territory.

- At the civic amenity sites, at least 20 different waste fractions are collected.
- In civic amenity sites, product/material exchange areas aimed at fostering reuse are available.

Description

Efficient recycling and recovery (with recycling and recovery rates of at least 80 %) requires an adequate infrastructure to perform door-to-door (kerbside) collection of the paper/cardboard, biowaste, packaging and possibly glass fractions. In addition, at its best, every bigger municipality (> 1 000 inhabitants) has at least one easily reachable civic amenity site (also called 'container park' or 'collection centre') where citizens can drop off as many waste fractions as possible which can be recycled or recovered at reasonable costs. Civic amenity sites can also be complemented by mobile/temporary collection facilities at planned locations and times of the week, in order to increase the usability of such facilities by residents.

The county, city or region identifies the numbers and locations of civic amenity sites and provides a standard layout for them. The latter can be applied by municipalities. In addition, staff are trained to operate the centres in such a way that all fractions are well separated and deposited in the correct container, drum, box, etc. Concerning the location, it is important that it is easy for citizens to access, well connected to the road network and does not disturb the neighbourhood. The area must be watertight and paved in order to avoid soil pollution and the run-off water shall be adequately treated or discharged to a public sewer.

The opening hours should allow sufficient opportunities for citizens to drop off different waste fractions; an example is shown in Figure 4-39. In spring, summer and autumn, the opening hours are longer compared to winter when less material is delivered, especially green cuttings.

Figure 4-39. Opening hours of a collection centre of a German village with about 8 300 inhabitants (the opening hours are adapted to daylight and season, specifically there are extended opening hours in November to increase the reception of green cuttings)

The different fractions which can be collected in a civic amenity site are described under Operational data.

Achieved environmental benefits

The recycling of the manifold mentioned waste fractions corresponds with savings in raw materials and energy. The separate collection and environmentally friendly disposal of hazardous substances reduces the contamination of waste streams and the environment. The separate collection of the different fractions usually enables higher recycling rates and thus lower losses of raw materials.

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are:

- number of civic amenity sites per 100 000 residents;
- number of different fractions collected at the civic amenity sites;
- availability of product/material exchange areas aimed at fostering reuse in civic amenity sites (y/n);
- easy accessibility of civic amenity sites, e.g. without a car (y/n).

Cross-media effects

The transport of the different waste fractions to the collection centre by the citizens is a relevant cross-media effect.

Operational data

The following fractions can be dropped off at a civic amenity site:

- Green cuttings (with little structure, branches with leaves or needles, woody material without leaves or needles – see photos below). The green cuttings with little structure can be shredded and classified on demand. The green cuttings with branches and leaves or needles are shredded and classified whereas the fine fraction is composted and the coarse fraction is used for energy recovery.
- Rubble (small amounts, i.e. 0.25 m^3 per delivery, thus deliveries of rubble from commercial activities are avoided). It is important that citizens have the opportunity to drop off rubble in order to avoid illegal disposal in the countryside. Gypsum and gypsum board as well as Heraklith (wood wool insulation) panels and asbestos products can be dropped at the collection centre but have to be disposed of at a cost of about EUR 170 per tonne.

Green cuttings with low structure (lawn, grass, leaves, windfall, balcony plants)

Green cuttings with branches – with leaves and needles

Woody green cuttings – without leaves and needles

Rubble (max. 0.25 m³ per delivery)
- no gipsum or gipsum board
- no Heraklith panels
- no asbestos products

- Scrap metal and different non-ferrous metals (e.g. copper, aluminium, brass) as well as stainless steel, lead or lead-containing materials are also collected – see photos below).

- Paper, board and cardboard is collected separately at household level (door-to-door/kerbside). Nevertheless, a collection centre is also equipped with a suitable container. The same is true for glass; it is collected via containers distributed over the residential area where citizens can drop container glass in three colours (white, green and brown). The following photos show examples of paper/cardboard and glass containers at a collection centre.

- Metal tins are also collected separately at collection centres, as are polystyrene packaging (see photo below). Polystyrene is collected separately to enable high-quality recycling.

- In order to support take-back obligations, waste electrical and electronic equipment (WEEE)[47] is collected in the fractions ‘communication devices’, ‘small electrical and electronic devices’ and ‘screens’ (see photos below). This is also true for refrigerators, car batteries and small batteries (see photos below).

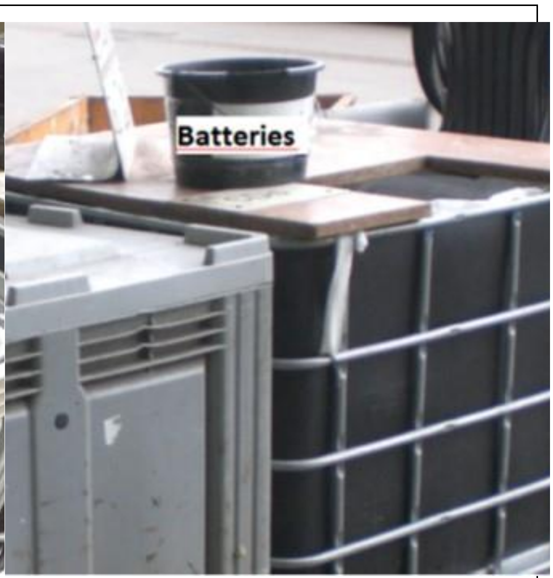
[47] Directive 2002/96/EC and Directive 2912/19/EC


Electronic waste –
communication devices
- computer, printer,
video recorder, DVD/CD
player
- radio, record player
- telephone, dax device
- loudspeaker, remote
control device


Electronic waste – small
devices
- electronic and electric
household devices
- electric tools
- electric toys


Electronic waste
Screens
- TV set
- screens
- flat screens


Refrigerators



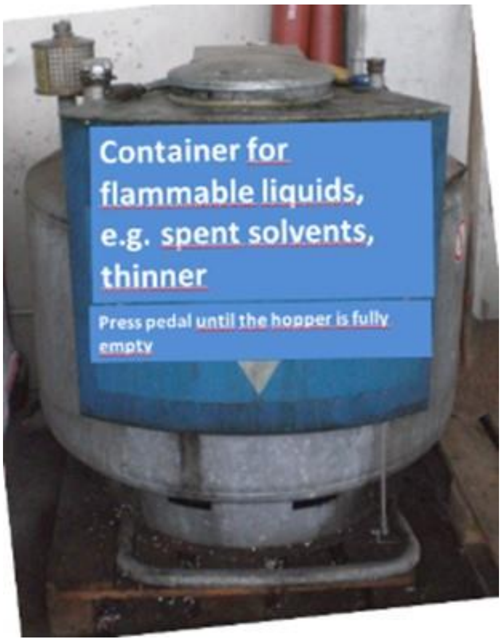
- Bulbs and fluorescent tubes are additional fractions that are collected separately and delivered to recycling according to legal regulations; no revenues are gained for them.

- Waste wood is collected in two fractions: untreated waste wood, i.e. wood which is not impregnated or soaked, and treated waste wood, i.e. wood which is impregnated (furniture with wood preservatives, such as window frames, exterior doors, wood from palisades and other outdoor applications – see photos below).

- More and more, small items are also recycled, such as polyurethane (PU) foam cans, CDs and DVDs, natural cork, toner cartridges but also waste vegetable fat, electric cable (although only about 100 g per capita per year, it is financially attractive) and items mainly made of lead (see photo below). Taken-back shoes, textiles and handbags can be recycled and the revenues can be donated to social projects (see photo below).

- It is very important to collect waste containing relevant amounts of hazardous compounds separately, such as acids, alkaline, solvents, wood preservatives, pesticides, paints, lacquers, oil-containing waste (oil filters, oil sludges, mineral-oil-containing fats, etc.), waste oil, disinfecting agents, waste containing metallic mercury (certain thermometers and electric switches), mercury-oxide-containing batteries, laboratory chemicals containing cyanide, cadmium or arsenic, etc. (see photos below). These wastes are collected in certain collection centres and by mobile collection trucks (see photo below). The time and location of their stops in all municipalities, city neighbourhoods, etc. are adequately communicated to citizens. Mercury-containing waste is kept strictly separate and is stored in special containers.

- Waste solvents and waste oil can be dropped at the central collection centre (see photos below).



- Devices containing lithium batteries have to be collected and disposed of separately; special provisions for transportation via road have to be followed (see photo below showing a special container).
- Solar panels are also collected separately (see photo below).

Figure 4-39 provides an example of the opening hours of a collection centre of a village with 8 300 inhabitants. The opening hours should depend on the population density and frequency of deliveries respectively. In areas with a low population density, it may be sufficient to open the collection centre for a few hours a week, preferably on Saturday, whereas in cities with high delivery frequencies the opening time may be more than 20 hours, in some cases even more than 40 hours. Figure 4-40 indicates the distribution of opening hours of almost 100 collection centres in Germany.

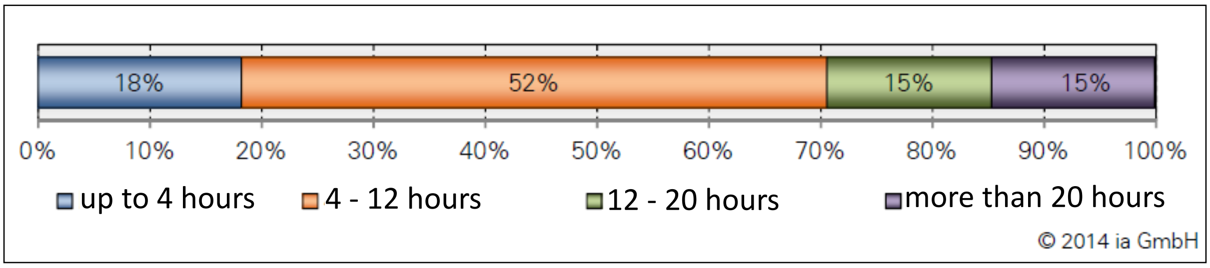


Figure 4-40. Distribution of opening hours of collection centres in Germany (ia GmbH / UMSICHT, 2015b, p 15)

Ideally, the average catchment area of collection centres in city areas is 34 km^2, in rural areas 43 km^2 and in individual municipalities 16 km^2. Thus, as an average, the distance of the inhabitants from a collection centre is only 3.3 km (city), 4.0 km (rural area), and 2.4 km (individual municipality). The maximum distance and the number of citizens are important parameters.

To improve the user-friendliness, with respect to bigger items (scrap metal, cardboards, green cuttings, etc.), it is advantageous to opt for so-called two-level solutions where the level of the delivering persons and the level of the container bottom are different (see Figure 4-41).

Figure 4-41. Two-level solutions for the delivery of materials (ia GmbH / UMSICHT, 2015b, p 16)

Putting a roof over the collection centre makes deliveries more comfortable (see an example in Figure 4-42) but is much more expensive compared to open-air facilities (see Economics).

Figure 4-42. Example of a roofed collection centre (ia GmbH / UMSICHT, 2015b, p 17)

It is important and required that skilled personnel of the municipality, county or city supervises the deliveries of the citizens in order to avoid cross-contamination of the different fractions. They are also trained with respect to safety aspects for themselves and citizens dropping off certain waste fractions.

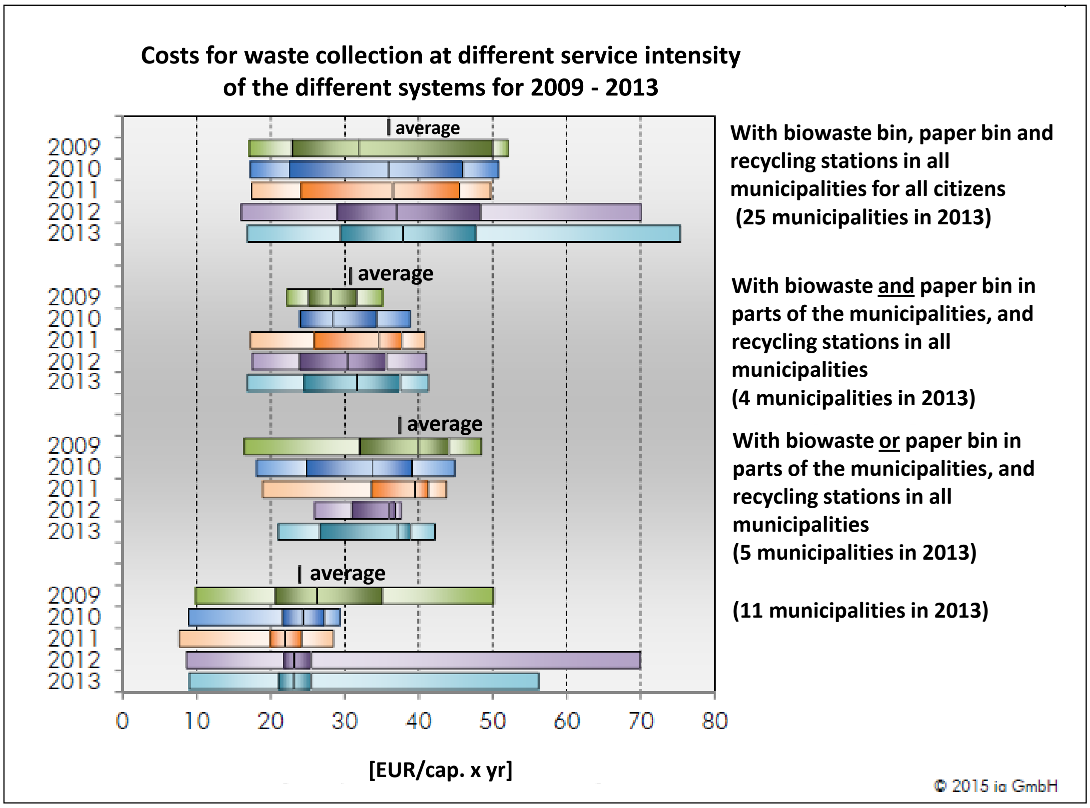
Applicability

In principal, the concept of collection centres is applicable to all municipalities, cities or counties. The introduction of collection centres in cities can be limited due to space constraints. The recyclability also depends on available markets, for instance waste vegetable fat can only be recycled if biodiesel is produced.

The application of this technique is strongly supported by other instruments such as the pay-as-you-throw system and cost benchmarking.

Economics

The costs for an efficient waste collection system and the operation of collection centres in all municipalities of a county vary considerably.According to Figure 4-43, in 2013, the range for counties or cities collecting biowaste, paper/cardboard and residual waste in specific bins as well as operating collection centres in all municipalities (upper part of the figure) is between EUR 17 and EUR 76 per capita per year. This indicates that an efficient system can be operated at a reasonable cost and that there can be significant room for cost optimisation. The cost figures already include the revenues gained from some of the recycled fractions.



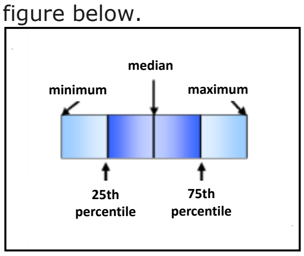
Note: see BEMP on cost benchmarking, see explanations in the footnote[48]

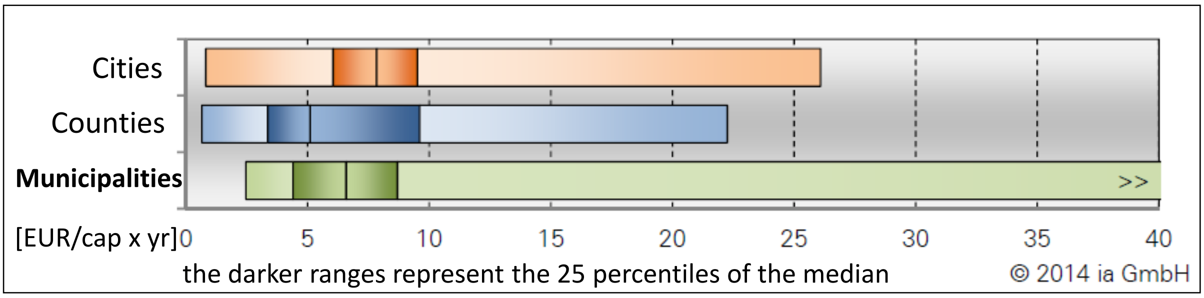
Figure 4-43. Costs for waste collection at different service intensities of the different systems for 2009–2013, based on ia GmbH (2015a)

The counties or cities to which the citizens pay their waste fee often cover the operating costs of the collection centres that are operated by municipalities (villages, small cities or city neighbourhoods).

Considering the collection centres only, the cost range is also large (Figure 4-44). In most of the counties, cities and municipalities (about 100 in total), the costs are between less than EUR 4 and EUR 10 per capita per year. For the cities evaluated, the average cost figure is EUR 7.8 per capita per year, for counties EUR 5.1 per capita per year and for individual municipalities EUR 6.6 per capita per year.

[48] The values are presented as median, minimum, maximum and 25th/75th percentiles as indicated in the figure below.





Note: see footnote to the previous figure

Figure 4-44. Costs for the operation of collection centres (ia GmbH / UMSICHT, 2015b, p 32)

The composition of the cost for collection centres is illustrated in Figure 4-45. Almost two thirds of the costs are those for personnel. The other shares of costs are much lower. Against this background, due to long depreciation times, it can be concluded that investment costs, e.g. for roofing or two-level solutions (see Figure 4-41), etc. will not significantly influence the total costs.

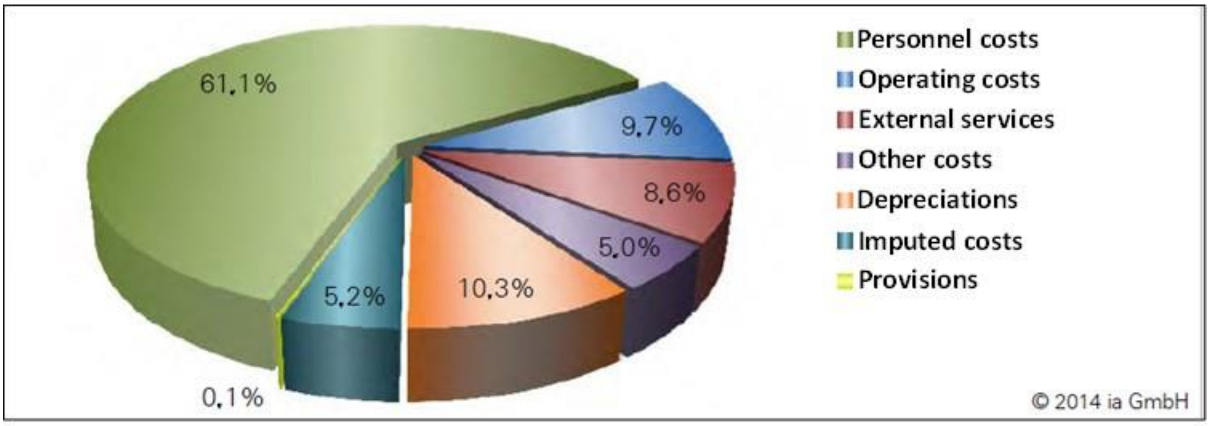


Figure 4-45. Composition of the costs for operating collection centres (ia GmbH / UMSICHT, 2015b, p 32)

It was already mentioned in relation to Figure 4-42 that investment costs for collection centres depend on their standard. They can be grouped into the categories simple, medium, high and very high. The definition of these categories is as follows:

- Category I: investment costs up to EUR 50 000 – simple enclosure, no operating building, no two-level solution;
- Category II: investment costs between EUR 50 000 and EUR 150 000 – container or roofing as "operating building", flat asphalted area;
- Category III: investment costs between EUR 150 000 and EUR 500 000 – solid, closed operating building, enclosed area, partly levelled area with ramps;
- Category IV: investment costs over EUR 500 000 – solid, closed operating building, storehouse, possibly reception of hazardous waste, levelled area with ramps.

Considering about 100 collection centres in Germany, about half of them fall into Category II, and about one fifth each into Categories I and III and only a few into Category IV (see Figure 4-46).

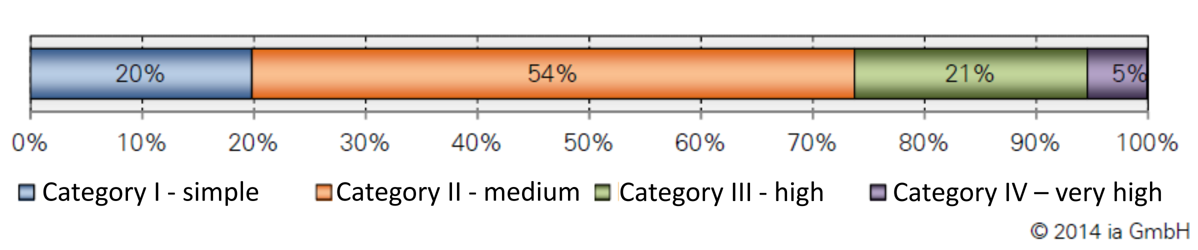


Figure 4-46. Different categories of collection centres (ia GmbH / UMSICHT, 2015b, p. 26)

Driving force for implementation

The rising awareness of the circular economy concept is a major driving force for establishing and operating collection centres. This awareness has often been driven by the limited availability of landfills, and, in some Member States, by the legal ban on landfilling of untreated waste. For instance in Germany, Austria and the Netherlands, the awareness was already starting to significantly increase more than 30 years ago.

Reference organisations

Germany: Counties of Aschaffenburg, Rems-Murr, Schweinfurt, Enzkreis. Cities of Munich, Hamburg, Berlin, Neumünster.

Odense (Denmark) established a network of eight civic amenity sites, i.e. approximately 24 000 inhabitants per site. The average distance to the nearest site is around 2 km. All sites have approximately 30 containers for different waste types (R4R, 2014a).

The Île-de-France region implements a network of mobile civic amenity sites. The service consists of the temporary installation of collection facilities in a public space (from 1.00 pm till 6 pm in summer time and till 5.00 pm in winter time). The location of the mobile civic amenity sites is fixed and the frequency of opening is decided with the local authorities and ranges from once to seven times a month (R4R, 2014b).

Trasimeno Servizi Ambientali (TSA) introduced, thanks to the LIFE EMaRES project, a mobile collection point ('Ricimobile') which, at planned times and locations in the territory, collects WEEE, batteries and spent cooking oil (Di Maria, 2015).

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4.5.4. Logistics optimisation for waste collection

Summary overview							
It is BEMP to optimise the logistics of waste collection by: - installing where appropriate an alternative collection system to road transport, such as a pneumatic system in urban areas; - using Computerised Vehicle Routing and Scheduling (CVRS) technology to optimise collection rounds; - exploring collaboration opportunities with neighbouring waste management organisations; - benchmarking fuel/energy consumption and/or CO_2 emissions; - incorporating one or more environmental metrics, such as cumulative energy demand (CED) and/or CO_2 emissions, into network design and route optimisation algorithms; - installing telematics equipment in collection vehicles for real-time route optimisation based on GPS and training drivers in eco-driving techniques.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
All organisations involved in waste collection can implement some degree of logistics optimisation (e.g. planning the location of waste bins). However, the actions are limited in some cases by existing organisational structures (e.g. on-going contracts for outsourced waste collection services). In terms of collection strategy optimisation, logistics optimisation is secondary to optimising recycling. Pneumatic waste collection systems are more suitable for densely populated areas and are easier to install in new developments than in existing urban areas.							
Specific environmental performance indicators							
In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: - fuel consumption per tonne of waste collected[49] (litres/t);							

[49] depending on the waste collection system in place (e.g. vehicles and/or pneumatic collection, type of vehicles) and the data available, more useful alternatives to this indicator can be: Primary energy

- GHG emissions per tonne of waste and km travelled (kg CO_2e/tkm).

Description

Overview

When developing a new waste collection strategy (Section 4.5.1), logistics optimisation is an important aspect to consider, since it can contribute to improving the economics and the environmental performance of the waste management system. For instance, as presented in Section 4.5.1, colour-coded bags for different waste fractions can be collected in a single refuse collection truck for transport to an optical sorting plant where separated waste streams are checked and sorted for further treatment in recycling facilities. The choice of this collection system, when defining the waste collection strategy, contributes to logistics optimisation, reducing the number of collection routes, lowering fuel consumption, traffic congestion and noise.

In general, and not only during the development of a new waste collection strategy, there is often scope for significant logistics optimisation in order to reduce the related fuel consumption, noise, traffic and costs.

Logistics optimisation ranges from the design of waste collection infrastructure and networks, including the installation of vacuum collection systems and the use of colour-coded bags, to real-time route optimisation based on GPS or geographical information system (GIS) software. The opportunities to implement the design of advanced waste collection infrastructure and networks may be limited depending on the existing organisational structures of waste collection providers – for example, outsourced collection providers may not have any opportunity to influence network design. However, all organisations involved in waste collection can implement some degree of logistics optimisation (e.g. location plan of waste bins).

Table 4-17 summarises the key measures to optimise logistics operations for waste collection, and the rationale underpinning them.

Table 4-17. Key measures proposed as BEMP and the underpinning rationale

Measure	Underpinning rationale
Install an alternative collection system, such as a pneumatic system in urban areas.	Pneumatic systems avoid the need for collection vehicles to enter built-up areas where traffic congestion, noise and air pollution effects are most problematic. They can therefore lead to significant improvement in urban environmental quality.
Utilise Computerised Vehicle Routing and Scheduling (CVRS) technology to optimise rounds.	Optimisation requires detailed modelling using specialist software, and may be undertaken in-house or outsourced. In any case, the EU rules for driving time and rest periods following (EC) 561/2006 have to be taken into account.
Explore collaboration opportuni-	Collaboration offers considerable scope for improvement

consumption per tonne of waste collected, cumulative energy demand per tonne of waste collected, GHG emissions per tonne of waste collected.

Measure	Underpinning rationale
ties with neighbouring waste management organisations.	through efficiency savings, such as route optimisation and depot rationalisation (AMEC, no date).
Benchmark fuel/energy consumption and/or CO_2 emissions.	Benchmarking fuel consumption and emissions per tonne of material collected and delivered facilitates continuous improvement in environmental efficiency, and also provides data necessary for LCA of material recycling chains, informing design of the circular economy.
Incorporate one or more environmental metrics, such as cumulative energy demand and/or CO_2 emissions, into network design and route optimisation algorithms.	The environmental impact of waste collection is dominated by fuel consumption and related combustion emissions, and is indirectly represented via fuel costs in economic optimisation of reverse logistics. Explicitly incorporating one or more environmental metrics, such as cumulative energy demand and/or CO_2 emissions, into optimisation algorithms can maximise the environmental benefits achieved through logistics optimisation.
Install telematics equipment in collection vehicles, and train drivers in eco-driving techniques.	Driving style (especially during stop-start collection) and routing depending on traffic conditions can have a significant influence on fuel consumption.

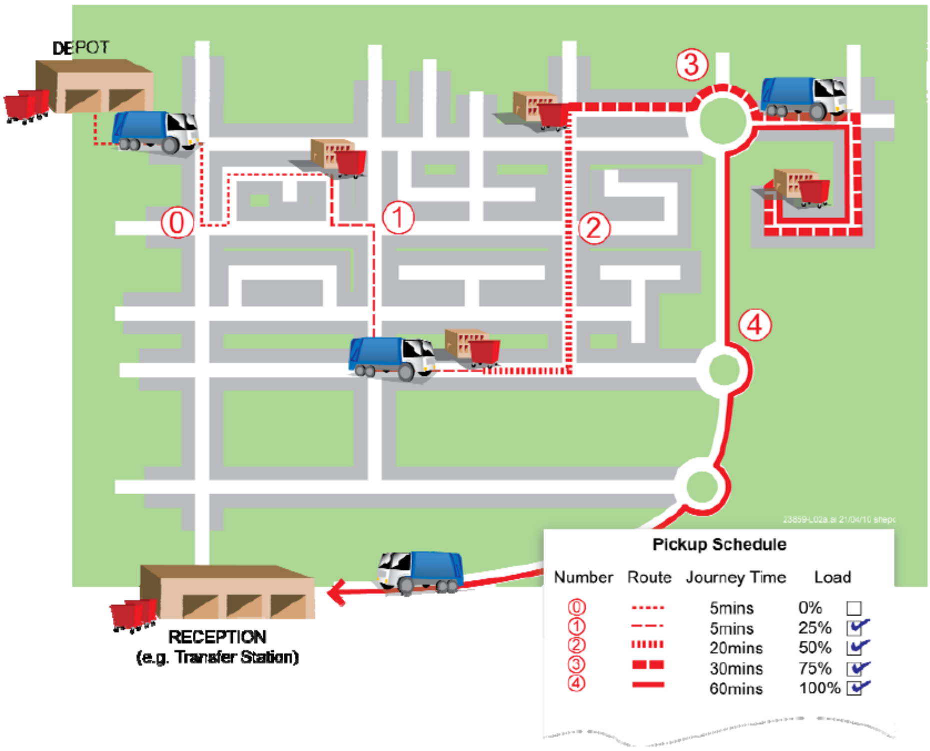
Route optimisation

Logistics operations for waste collection can be optimised with respect to[50]: (i) the type, number and location of facilities and bins, (ii) choice of the transportation means, (iii) choice of the transportation speed, (iv) choice of the transportation concept, (v) choice of the routing, and (vi) choice of the timing of collection (Dekker et al., 2012). Compared with other logistics operations, final load factors are usually high for waste collection vehicles, and there is not much choice of mode: 26-tonne collection trucks are typical (see also the BEMP on low-emission vehicles), though there may be opportunities to use smaller collection vehicles for some routes and fractions.

Waste collection round routes and schedules are typically developed over time based on driver knowledge and are revised periodically in response to changing collection requirements. For simplicity, collection rounds may be designed based on zoning for individual vehicles/crew, although this approach is likely to miss significant opportunities for optimisation (WRAP, 2010).

The modelling and optimisation of collection operations can be best performed by using a suite of commercially available software tools incorporating Computerised Vehicle Routing and Scheduling (CVRS) technology (Figure 4-47). This may be outsourced to specialist consultancies, or undertaken in-house following procurement of the necessary software and licenses. Information systems and data collection strategies may need to be upgraded to support CVRS.

[50] All the choices should take into account the local traffic conditions and the architecture of the examined area.



Source: WRAP (2010).

Figure 4-47. Schematic example of a Computerised Vehicle Routing and Scheduling (CVRS) software system

Waste collection optimisation involves the application of reverse logistics, defined as "planning, implementation and controlling the efficient, effective inbound flow and storage of secondary goods and related information opposite to the traditional supply chain directions for the purpose of recovering value and proper disposal" (Fleischmann et al., 1997, cited in Bing et al., 2014).

Alternative collection systems

In densely populated urban areas there is increasing interest in the use of alternative waste collection systems, such as pneumatic systems that use negative pressure (vacuum) to move waste along underground pipes from inlet points where citizens deposit waste fractions to waste collection points outside residential areas. These systems may also employ positive pressure to tackle blockages, and, although expensive to install, can considerably reduce operating costs (Waste Management World, 2009). Systems can be designed to accommodate multiple waste fractions, and can even be used to automatically empty litter bins (Envac, 2015). Such systems can considerably reduce traffic, noise and odours in urban centres, and may be particularly well suited to new-build residential districts. Additionally, there is less need for (i) waste storage space in households and (ii) accessibility of vehicles in the urban centre.

Finally, note that alternative road transport vehicles are described in the next BEMP.

Achieved environmental benefits

Pneumatic systems can lead to significant savings in fuel, as well as reducing noise, visual impact, odours and traffic associated with conventional waste collection systems. Installation of a pneumatic system in the Hammarby Sjöstad district of Stockholm is estimated to have reduced waste collection traffic (heavy waste collection vehicles) by 60 % (Envac, 2015). Whilst pneumatic systems may not generate environmental savings from a life-cycle perspective across the entire waste management chain, they are highly significant in the context of urban environmental quality.

The magnitude of fuel and environmental burden savings achieved through logistics optimisation is highly dependent on the pre-existing (in-)efficiency of waste collection operations.

WRAP (2010) reports on an example of CVRS application to optimise collection of MSW in the UK. The study found that CVRS could reduce transport distances and associated fuel consumption by 15 %, whilst increasing productivity by up to 9 %. This would lead to concomitant reductions in fossil resource depletion, GHG emissions, air-polluting emissions such as NO_x, PM and VOCs, and traffic.

Ricardo-AEA (2012) reports that active cruise control can reduce fuel use and GHG emissions by 1–2 % for regional delivery, which may apply to transport of waste fractions between depots (two- to four-month payback period). Telematic systems can reduce fuel consumption and associated emissions by approximately 5 % for long-distance transport, and up to 15 % for urban transport (Climate Change Corporation, 2008).

Owl Waste (2015) reports a trial with SITA UK in which they used telematics to target driver training; this allowed a reduction in fuel consumption of 12 %. Ricardo-AEA (2009) suggests that more efficient driving can reduce fuel consumption by up to 10 %.

Appropriate environmental indicators

In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are as follows:

- Fuel consumption per tonne of waste collected. However, depending on the waste collection system in place (e.g. vehicles and/or pneumatic collection, type of vehicles) and the data available, more useful alternatives to this indicator can be: primary energy consumption per tonne of waste collected, cumulative energy demand per tonne of waste collected, GHG emissions per tonne of waste collected.
- GHG emissions per tonne of waste and km (kg CO_2e/tkm).

Cross-media effects

All measures that reduce fuel consumption should reduce life-cycle fossil energy depletion and emissions of GHGs and substances affecting air quality.

Route and schedule optimisation based on economic data alone could lead to increases in fuel consumption and associated environmental burdens in some cases, especially where an environmental metric is not included in the optimisation algorithms.

In terms of network design, there may be a trade-off between minimisation of waste collection burdens and wider economic optimisation of the number of logistics hubs. Dekker et al. (2012) suggest that economic factors favour fewer, larger and more efficient waste treatment centres. This may or may not be congruent with logistics optimisation depending on the specific situation.

Implementation of logistics optimisation only after identification of the most efficient overall collection strategy should avoid potentially important trade-offs between minimisation of collection energy (e.g. via less frequent collection of separated fractions) and maximisation of waste separation (BEMP 4.5.1).

There is little published information on the energy consumption of pneumatic systems. Punkkinen et al. (2012) found that a hypothetical pneumatic collection system, modelled using patchy available data, generated considerably higher GHG and SO_x emissions per tonne of waste transported than road collection. However, NO_x emissions were lower, and air pollution largely arose upstream in power stations rather than in densely populated urban areas. Electricity consumption was the dominant source of emissions, but relied on uncertain data. ISWA (2013) claims that new systems using a combination of vacuum and positive pressure use up to 67 % less energy than vacuum-only systems. There is a need for better data to be reported on the electricity requirements of pneumatic systems.

Operational data

Network design

Variables affecting collection performance include household locations, collection day requirements, waste volumes, unloading locations and vehicle turnaround times / congestion (WRAP, 2010). These parameters are among those that can be inputted to routing software to produce "As Is" models that provide the basis for redesigning and optimising collection rounds using CVRS technology. Data generated by PAYT systems (BEMP 4.3.3) can provide a powerful basis for logistics optimisation. A case study of collaboration between PROMEDIO and Wellness Telecom in Badajoz, Spain, described under Reference organisations below, highlights the use of microchip sensors in bins to monitor bin fullness at the point of collection in order to inform optimisation of collection frequency and public collection point siting.

Ultimately, maximisation of waste separation and recycling rates is a priority to reduce the overall environmental burden of waste management from a life-cycle perspective. Logistics optimisation must therefore be constrained by priority parameters, such as the scale of waste treatment centres, that are set to maximise waste recycling rates.

WRAP (2010) notes that waste management organisations are sometimes sceptical of CVRS and similar technology, partly because information technology systems and record-keeping may not meet specifications required to implement it. There is a need for investment in information technology infrastructure and data to facilitate the use of CVRS.

WRAP (2010) reports on a trial with CVRS optimisation across three waste management organisations in the UK. Round data was supplied with postcode locations and collection sequencing, and used to map the individual days of work using the RoundManagerWM tool. Supporting data required included:

- daily vehicle weights;
- individual vehicle payloads;
- access and time restrictions for collections;
- start and finish times for the rounds;

- bin sizes and numbers;
- depot and reception location;
- driver breaks (legally required);
- reception facility turnaround time;
- average travel speeds (per round by tachographs).

A model representative of the pre-existing waste collection operation was devised based on further data provided in map and spreadsheet format, and reviewed by operational managers and supervisors at the waste management organisation. Spreadsheet data included:

- duration of the working day in hours;
- distance travelled in miles;
- bin numbers collected;
- number of loads tipped;
- tipping time;
- picking-up time;
- pick-up rate (number of bins collected per hour, excluding the travel time to the round (and return), and tipping time);
- total weight collected;
- yield per bin; and
- spare capacity on the vehicle.

WRAP concluded that the 15 % cost savings and 9 % productivity improvement demonstrated through application of CRVS support its adoption by organisations managing waste collection.

Harris et al. (2011) demonstrate the integration of both logistics costs and CO_2 emissions in logistics optimisation, ensuring that environmental efficiency is given more weighting within optimised network solutions.

Following network optimisation, there may be scope to implement route navigation for specific journeys. Route navigation indicates the route between two given points using sophisticated shortest-path algorithms to reduce the distance travelled, usually also reducing emissions (Dekker et al., 2012).

Multi-modal vehicles

One important aspect of the CVRS optimisation described in WRAP (2010) and referred to above is the use of multi-modal vehicles, which provides much greater flexibility in route scheduling and therefore greater potential to integrate multiple rounds during logistics optimisation. In Vienna, the waste management authority started a project to check the suitability of a special collection vehicle for various container sizes ("Mischzug") in 2010 (MA 48, 2014). The basic aim was to empty waste containers of different sizes within a collection area with only one collection vehicle. In the course of the project, that ended in 2013, approximately 95 200 properties and approximately 164 000 containers were involved in the planning and 126 new routes were designed. Through the project, the collection logistics were streamlined, and 10 waste collection routes were eliminated, leading to a reduction in truck traffic, and fuel savings, as well as a more efficient use of personnel and vehicles (higher productivity).

Alternative collection systems

There is increasing interest in pneumatic waste collection systems, replacing the use of outdoor bins and collection vehicles, in which users deposit their refuse directly into about 1.5 m high waste inlets at strategic locations, accessible 24 hours a day (Waste Management World, 2009). Radio frequency identification tags can be used to identify users of communal inlet points. There is one waste inlet for each type of refuse (e.g. mixed waste, organic waste and paper waste). Refuse is transported along pipelines using vacuum and/or over-pressure into containers at waste stations a few km away. Containers are then transported to processing plants using various modes of transport – potentially including existing underground networks in cities. The main network typically comprises 500 mm diameter steel pipes that are hermetically welded. Air-flushing of pipes between batches of waste reduces contamination between different waste types. The system is remotely monitored and controlled by operators at the waste station.

Pneumatic systems reduce fuel and personnel costs, and reduce noise, visual impact and traffic associated with conventional waste collection systems in cities. Such systems are best suited to densely populated metropolitan areas, and are expensive to install but are designed to last up to 60 years, and have payback periods of 10–12 years owing to lower operating costs and better use of waste storage spaces and pick-up points compared with conventional collection. Small-scale pneumatic waste systems are ideal for shopping centres, airports, hospitals and nursing homes, and can improve hygiene. The city of Helsinki, Finland, and the neighbouring city of Vantaa are planning to incorporate pneumatic waste collection systems into new urban development projects. The residential area of Jätkäsaari in Helsinki will be completed by 2023, and will house 16 000 residents and 6 000 workplaces. There will be 350 pneumatic collection points installed to handle 22 000 kg of waste per day (6 400 tonnes of residential waste plus 550 tonnes of commercial waste annually) (Waste Management World, 2009).

At Hammarby Sjöstad, a neighbourhood of Stockholm, four pneumatic systems have been installed since 1997 and operational since 2000. A total of 457 inlets and 12.5 km of pipes manage 11 tonnes of waste per day, split into four fractions:

- biowaste;
- paper;
- street litter;
- general waste.

The four systems serve 8 500 apartments, and approximately 20 000 inhabitants, and continue to expand, with self-emptying litter bins recently added. The system has reduced traffic from refuse collection vehicles by 60 % (Envac, 2015).

Figure 4-48. Pneumatic system inlets in Hammarby Sjöstad

Source: Envac (2015)

Economics

WRAP (2010) quotes costs in the range of GBP 5 000 to GBP 10 000 (EUR 7 042 to EUR 14 084) to model and optimise existing collection rounds for a waste management organisation running 12 collection vehicles. Adding alternative future scenarios costs GBP 2 000 to GBP 6 000 (EUR 2 200 to EUR 6 600) per scenario. In the case study example, WRAP (2010) estimates a fuel saving of up to GBP 36 200 (EUR 39 800) per year, indicating a short payback time. The study authors suggest that a return on investment can be made within one to two years, depending on the degree of change implemented and the size of the fleet (larger fleets are likely to realise greater savings).

The outsourcing of waste collection activities by waste management companies can reduce incentives for both separation efficacy and logistics optimisation, depending on how contracts are structured. In the absence of specific performance-related clauses, subcontracted collection companies may maximise revenue by maintaining high-frequency bin collections, justifying higher charges to the waste management companies. It is imperative that outsourcing of logistics operations sets clear performance objectives that avoid perverse incentives (TWG, 2015).

The installation cost of pneumatic systems is considerably greater than for conventional bin-collection systems. ISWA (2013) presents cost data for three case studies, indicating that, for apartment blocks, it can cost up to four times more to install a pneumatic system – up to EUR 15 million for 10 000 apartments. However, bin-collection systems require significant space for bin storage, which can be expensive in urban areas (estimated at over EUR 14 million for 10 000 apartments). Furthermore, collection costs for pneumatic systems are considerably lower: EUR 133 000 per year for 10 000 apartments, versus EUR 640 000 per year for conventional collection (ISWA, 2013). The economics of pneumatic systems therefore compare favourably where space (land) is expensive. Waste Management World (2009) reports that the estimated payback period for pneumatic systems is 10–12 years.

Applicability

All organisations involved in waste collection can implement some degree of logistics optimisation (e.g. planning the location of waste bins). However, the actions are limited in some cases by existing organisational structures (e.g. on-going contracts for outsourced waste collection services).

In terms of collection strategy optimisation, logistics optimisation is secondary to optimising recycling.

Pneumatic waste collection systems are more suitable for densely populated areas and are easier to install in new developments than in existing urban areas.

Driving force for implementation

Increasing collection costs associated with collection of separated waste fractions, alongside the long-term upwards trend in fuel prices, are major drivers for the optimisation of transport and logistics. This is driving increasing interest in collaborative agreements across waste management organisations (AMEC, no date).

Space restrictions and high land prices are a major factor favouring pneumatic systems that avoid the need for bin storage areas.

Reference organisations

Some municipalities which have improved the logistics of waste collection are: Sefton Metropolitan Borough Council (UK), Multi-council collaboration in Hampshire (UK).

A few examples of software providers for route optimisation are: http://www.webaspx.co.uk/ http://www.fleetroute.com/k1/e.php http://www.routesmart.co.uk/case_studies.php

Participants in the EC LIFE Ewas project, in which wireless sensors and GPS tracking are being employed to optimise waste collection timings and vehicle routings: http://life-ewas.eu/en/ See PROMEDIO case study below.

A number of case studies of pneumatic waste collection systems are available on the Envac website: http://www.envacgroup.com/references

Box 4.20. SITA UK telematics and driver training

In 2010, CMS SupaTrak began working with SITA UK to explore the potential benefits of implementing a telematics system throughout their fleet. An initial trial was carried out with "EcoTrak" fuel-saving technology on 12 municipal and recycling vehicles from the Warwick depot. EcoTrak is a telematics system which records driver behaviour in real time, measuring vehicle and driver performance against parameters including speed, idling time, harsh braking and accelerating, over-revving and excessive throttle use. This information can then be used to target remedial driver training to promote more fuel-efficient practices.

Following a two-week benchmarking period during which driver behaviour was covertly recorded and translated into summary reports, driver training and coaching was delivered by trainers with industrial experience and knowledge.

The trial resulted in fuel savings of 12 %, which were extrapolated up to an annual GHG emission reduction of 3 000 tonnes. Following on from the success of the trial, SITA UK has decided to roll out EcoTrak technology across 650 vehicles based around 32 sites, and the trial has been replicated across other SITA operations throughout Europe. The technology is compatible with all vehicle manufacturers.

Source: Owl Waste (2015).

Box 4.21. Optimisation of collection rounds for a new waste collection strategy by Sefton Council, UK

Sefton Metropolitan Borough Council (MBC) is a local authority covering 120 000 households. The council engaged a consultancy to develop optimised waste collection rounds following the development of a new strategic waste collection plan that involved changing to alternate week collection of refuse and garden waste in wheeled bins, replacing weekly collection of refuse sacks, and (for 80 % of households) garden waste sacks. A private contractor managed kerbside-sorted weekly dry recycling collection. Sefton Council required the new collection schedule to meet the following objectives:

- high levels of time and fuel efficiency;
- balance workloads across crews and vehicles;
- flexibility to accommodate different productivity rates and yields.

The consultants employed by Sefton MBC had worked with over 50 other local authorities, which enabled them to calibrate their models with regionally applicable productivity rates and yields for different types of households. The modelling identified the minimum number of vehicles and crews required to produce workable rounds to maximise productivity rates and yields. Feedback from the crews was used to refine the round optimisation, and designed rounds were tested for sensitivity to productivity rates and yields.

Sefton MBC said of the work: "The combination of AMEC and Webaspx's powerful optimisation technology, together with their experience of working with many authorities on round design, has helped us develop a solution of acceptable risk. We feel that the outcome has produced optimised and balanced workloads that will enable the new collection service to be introduced successfully."

Source: AMEC (no date).

Box 4.22. Logistics optimisation through multi-council collaboration and depot rationalisation in Hampshire, UK

Background

Project Integra is a partnership of the 15 parties (including waste collection, disposal authorities and Veolia) in Hampshire formed to find common, efficient waste collection solutions. Project Integra commissioned AMEC to evaluate the potential logistics benefits of joint refuse and recycling collections across six partner authorities (Basingstoke and Deane, East Hampshire, Hart, Havant, Portsmouth and Winchester).

Method

RoundManagerWM software was used, and a collection model parameterised using data provided by operational staff. An initial scenario maintained all existing depots and facilities across the six partner authorities, using a standardised set of design rules underpinned by the collection pick-up rates and yield data provided by each authority. A subsequent scenario modelled the impact of depot rationalisation, in which two depots were removed.

Results

Tactical models identified savings of nearly 400 000 km per year, 235 000 kg CO_2 and six vehicle equivalents (including drivers and loaders), resulting in financial savings of approximately GBP 1 million (EUR 1.4 million) per year. The potential logistics savings were slightly reduced in the depot rationalisation model, although closing down two depots could save GBP 250 000 (EUR 340 000) per year.

Source: AMEC (no date)

Box 4.23. PROMEDIO waste collection optimisation

Wellness Telecom and PROMEDIO implemented a project in the Spanish province of Badajoz to monitor 50 bins for 12 months, using electronic sensors to record bin weight at collection. The study was part of the EU LIFE-funded "Ewas" project, and revealed the following:

- only 20 % of bins have a fill rate high enough to require weekly collections;
- 18–20 % of bins are collected with a content below 40 % to 50 %;
- 75–80 % of bins are collected at least once a year with a content below 40–50 %.

From these findings, Wellness Telecom proposed the following measures to PROMEDIO:

- Identify a list of bins that need to be collected weekly due to a higher service demand. Reorganise collection site locations and enhance service availability, with additional bins in nearby locations.
- The rest of the bins should be collected every two weeks.

This will provide a basis from which to further optimise collection routes and frequency, saving in fuel and human resources. Continued monitoring of bin fill level through use of a simple electronic tool ("e-Garbage") is proposed to identify full bins requiring earlier collection. Expected savings in fuel are around 5 000 litres per year, whilst workforce savings are in estimated to be 40–50 %, switching from weekly to fortnightly collection.

Source: Wellness Smart Cities and Solutions (2015).

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4.5.5. Low-emission vehicles

Summary overview							
It is BEMP to improve the fuel consumption and emissions of waste collection vehicles. Priority technology options include: - stop/start and idle shut-off; - low rolling resistance tyres; - hybrid vehicles; - dedicated natural gas/biomethane vehicles or dual-fuel vehicles (diesel/gas); - electrically powered vehicles.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
This BEMP is broadly applicable. The presence of filling or recharging stations is less of an issue for refuse collection than other types of transport because vehicles are usually operated over a limited distance and the fleet is run from a centralised waste depot where refuelling can take place. Compressed natural gas (CNG) is available in all EU countries. Biomethane may not be available in many regions, but wet organic waste (e.g. food waste) can be used to produce biogas that can be upgraded to transport biomethane.							
Specific environmental performance indicators							
- Average fuel consumption of the waste collection vehicles (litres/100 km). - Share of vehicles that are Euro 6 in the total waste collection vehicle fleet (%). - Share of waste collection vehicles that are hybrid, electric, natural-gas- or biogas-powered (%).							
Benchmarks of excellence							
- All new refuse collection vehicles purchased or leased by the waste management organisation are Euro 6 and are fuelled by either compressed natural gas or biogas, or are hybrid-electric.							

Description

Municipal use of heavy goods vehicles (HGVs), primarily refuse collection trucks, accounts for approximately 4 % of HGV CO_2 emissions in the UK (Ricardo-AEA, 2012). A typical 26-tonne rigid HGV collection truck will consume between 57 L and 141 L per 100 km of diesel, reflecting inefficient low-speed and stop-start driving. Apart from the characteristics of the vehicles, fuel consumed for waste collection varies depending also on the levels of source separation (i.e. separate collection) achieved, since more separated fractions require more collection routes. Values can range from 3.3 L/tonne of waste (when source separation of waste is 25 %) to 3.8 L/tonne of waste (when source separation is 52 %) (Di Maria et al., 2013).

Priority measures identified by Ricardo-AEA to reduce GHG emissions from municipal HGV use are summarised in Table 4-18.

Table 4-18. Priority technology options to reduce greenhouse gas emissions from refuse truck operations proposed in Ricardo-AEA (2012)

Rank	Measure	Life-cycle CO_2e saving	Payback time*	Additional considerations
1	Stop-start and idle shut-off	5 %	<1–2.5 years	Small air quality and noise reduction benefits in congested urban areas. Marginal increase in life-cycle impact due to additional components.
2=	Hybrid electric vehicles	15–25 %	4–16 years	Air quality and noise reduction benefits particularly if able to run in electric-only mode. Life-cycle impacts of batteries need to be considered.
2=	Dedicated natural gas vehicles	5–16 % (CNG) 61–65 % (biomethane)	6–18 years	Significant particulate emission and noise reduction benefits; requires additional refuelling infrastructure. Substantially larger CO_2e reduction benefits with biomethane.
3	Electrically powered truck bodies	10–12 %	9+ years	Electrically powered refuse truck bodies can reduce noise and air pollution.
4	Low rolling resistance tyres	1–5 %		May have slightly shorter lifespan than standard tyres but CO_2 and fuel cost savings are expected to outweigh any negative environmental impact

*Based on current technology, marginal capital costs, fuel cost savings and low-high mileage sensitivities.

Source: Ricardo-AEA (2012).

Ricardo-AEA (2012) conclude: "The analysis indicates that one of the most effective strategies to achieve well to wheel CO_2e emission reduction in this [HGV] sector is to encourage a large scale shift to the use of gas as a fuel to replace diesel". Compressed natural gas (CNG) contains methane, which has a high hydrogen to carbon ratio, and therefore 20–25 % lower CO_2 emissions, per unit of lower heating value compared with petrol and diesel (Tassan et al., 2013). Perhaps more significantly, use of natural gas as a transport fuel significantly reduces air pollution emissions, such as NO_x and particulate matter (PM), compared with petrol and especially diesel. This effect is particularly beneficial in urban environments where refuse collection trucks operate, and where air quality is a major environmental and health concern. Biodiesel reduces GHG emissions but increases air pollutant emissions compared with diesel, whilst the climate change and air pollution performance is highly dependent on the method of electricity generation in the region of use.

Biomethane provides the same engine performance as CNG, but can reduce life-cycle GHG emissions by up to 180 % if a feedstock such as manure is used to produce the biogas. Greater than 100 % GHG avoidance can be achieved if emission credits associated with avoided counterfactual waste management are attributed to biogas uses including as biomethane transport fuel (the economic drivers for anaerobic digestion). Diverting food waste or manure to anaerobic digestion may avoid considerable GHG emissions that arise during composting and manure storage, respectively, depending on the prevailing alternative fate of those waste feedstocks. However, if accounting for upstream emission credits in this way, based on a consequential life-cycle assessment approach, it is imperative that double-counting is avoided – i.e. the waste management organisation accounts for the upstream emission savings from anaerobic digestion *either* in relation to waste treatment *or* transport fuelling (see BEMP on life-cycle assessment of waste management).

There are already over 1 million gas-powered vehicles on Europe's roads (Tassan et al., 2013). This BEMP therefore focuses on the use of CNG- and biogas-powered refuse collection trucks, or the use of hybrid-electric vehicles. Best environmental performance can be achieved by use of biomethane from organic waste, but where this is not yet available, converting collection fleets to run on CNG provides a useful step towards that goal. Alternatively, hybrid-electric vehicles significantly reduce transport impacts, and drive technological progress towards electrification of road transport which could lead to considerable future environmental benefits.

Dual-fuel vehicles

Typical 26-tonne refuse collection trucks run on diesel and can be readily converted to dual-fuel vehicles via simple modifications to the compression-ignition cycle via software remapping and injection modification. In dual-fuel vehicles, diesel is still required as a pilot fuel to initiate combustion under compression, but gas can then be injected as the main combustion fuel. The ratio of gas used in dual-fuel engines varies depending on the engine load and knocking issues under high compression, but can reach 90 % for integrated systems or 60 % for non-integrated systems.

Dedicated gas engines

Alternatively, HGVs can be selected with dedicated engine technology, such as Otto cycle stoichiometric combustion with a multipoint injection system, enabling 100 % gas fuelling and a superior overall environmental performance. Smaller petrol-driven collection vehicles can be converted to run on either 100 % gas, or as dual-fuel vehicles where the spark-ignition engine can switch between petrol or gas (Tassan et al., 2013).

Natural gas is becoming a relatively common transport fuel in Italy. In March 2015, there were more than 3 000 CNG stations in operation in Europe, most of them in Italy (1 054), Germany (920), Austria (178), Sweden (155), Switzerland (138), the Netherlands (134), Bulgaria (105) and the Czech Republic (82) (metanoauto.com, 2015).

Biomethane is becoming more common as a transport fuel in Germany and Sweden. The technology for the utilisation of gas for transport has been refined to a point where it is commercially viable. One main barrier to the use of gas in transport is the large storage volume required, or restricted range, compared with petrol and diesel engine vehicles. This is exacerbated by the fact that conversion of petrol and diesel engines (rather than ground-up design of dedicated gas engines) leads to suboptimal efficiency, and there remain relatively few gas filling stations in most countries (metanoauto.com, 2015). However, these barriers pose less of a challenge for refuse collection vehicles that travel limited distances around a central waste (refuelling) depot. Furthermore, biomethane may be produced within the waste management network, enabling an energy and carbon cycle in line with the concept of a circular economy. BSR, the public waste management company of Berlin, operates a fleet of 150 refuse collection vehicles running on biomethane produced from organic waste collected in the city (BSR, 2015a).

Hybrid-electric vehicles

Electric propulsion systems also have considerable potential to improve environmental efficiency, but are further from commercial application than gas fuels, although hybrid systems are becoming commercially available and can reduce environmental burdens significantly (Nehlsen, 2013).

Nehlsen (2013) reports on the testing of hybrid (Source: **Nehlsen, 2013**

Figure 4-49) and conventional diesel-powered refuse collection trucks in Bremen. In addition to the main diesel engine, the hybrid vehicles were fitted with a smaller (2 L) diesel engine that runs at optimum speed to charge high-power capacitors that in turn power electric motors for hydraulic operations.

Source: Nehlsen, 2013

Figure 4-49. A "Rotopress Dualpower" refuse collection truck during testing in Bremen, Germany

Maintenance costs are lower for hybrid vehicles because the hydraulic system is powered by low-maintenance electric motors, and because regenerative braking reduces brake pad friction.

Hybrid trucks tested in Bremen (Nehlsen, 2013) had the same total weight as conventional trucks (26 tonnes), but 1.5 tonnes less waste capacity owing to the weight of the hybrid system (especially batteries). The effect of additional journeys was considered in the fuel and GHG balance per mg of waste collected, as described above, although Nelsen (2013) notes that there may be routes where a truck's full capacity is not required and on which hybrid trucks would not require an additional refuelling stop. Carefully integrating hybrid vehicles into optimised collection rounds is therefore essential to obtain maximum efficiency savings.

Achieved environmental benefits

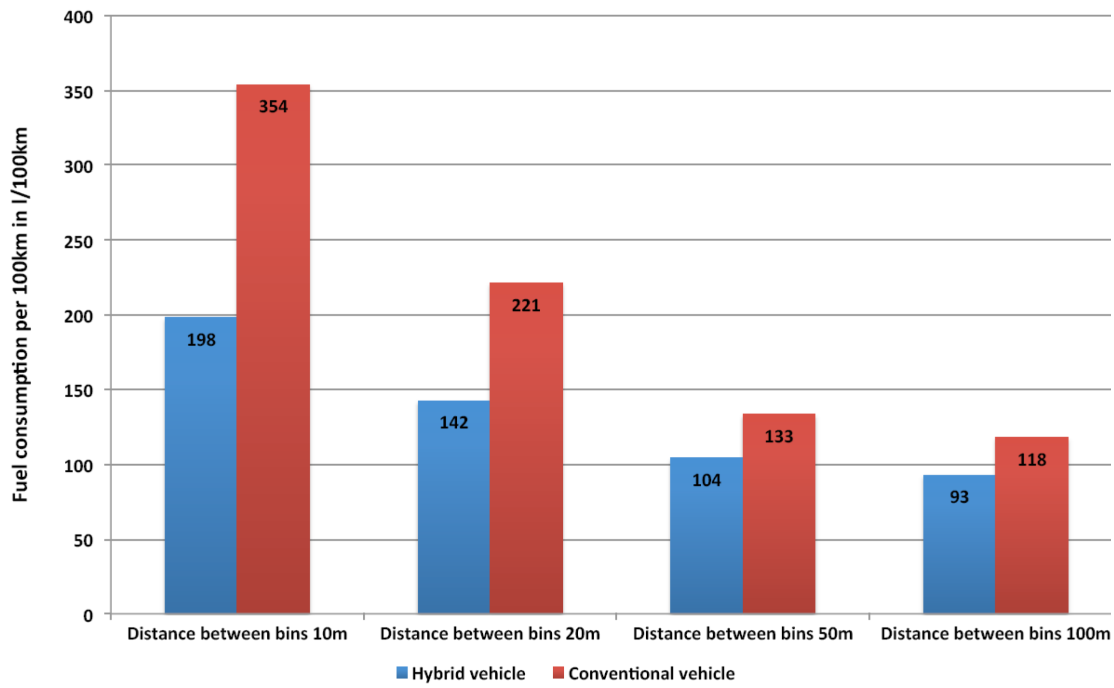
GHG emissions

Direct CO_2 emissions from combustion are significantly lower for CNG-powered trucks compared with diesel-powered ones, by up to 16 % (Ricardo-AEA, 2012). However, life-cycle GHG savings are somewhat lower than this owing to upstream burdens of CNG extraction, processing and transport, including leakage (CH_4 has a GWP 25 times higher than CO_2), and may in fact be negligible (Rose et al., 2013).

Biogas can achieve life-cycle GHG reductions of 65 % compared with diesel-powered vehicles (Ricardo-AEA, 2012), and up to 180 % if LCA boundaries are expanded to account for avoided counterfactual manure or food waste management (Tassan et al., 2013), as explained above.

Stop-start and idle shut-off can reduce GHG emissions by 5 %, and alternative-fuelled (electric) bodies can reduce GHG emissions by 10–12 % compared with conventional diesel refuse trucks (Ricardo-AEA, 2012).

Nehlsen (2013) reports that the overall fuel consumption per mg of waste collected decreases from 4.2 L to 3.5 L of diesel for the diesel-electric hybrid system, a 16 % saving, on average considering all factors (decreased load, transport to depot, etc.). However, the efficiency advantage of hybrid systems is strongly dependent on the route and collection characteristics, and is greatest during the stop-start collection stage of rounds, achieving reductions in fuel consumption of up to 40 % in the case of bin stops separated by short distances of 10 m (i.e. urban areas) (Figure 4-50).



Source: Nehlsen (2013).

Figure 4-50. Fuel consumption for a hybrid truck and a conventional 26-tonne refuse collection truck tested in Bremen, Germany

Emissions affecting air quality and health

Gas burns more cleanly than petrol or diesel, resulting in significantly lower emissions of particulate matter (PM), nitrogen and sulphur oxides (NO_x and SO_x), and volatile organic compounds (VOCs), amongst others (Table 4-19; Source: **LES (2011)**

Figure 4-51).

Table 4-19. Reductions in emissions affecting air quality for CNG vehicles compared with petrol- and diesel-powered vehicles

	SO_x	NO_x	VOCs	PM	Ozone promoters	Aromatic compounds
CNG vs. petrol*		52 %	92 %		96 %	99.9 %
CNG vs. diesel**	44 %	44 %	21 %	25 %		

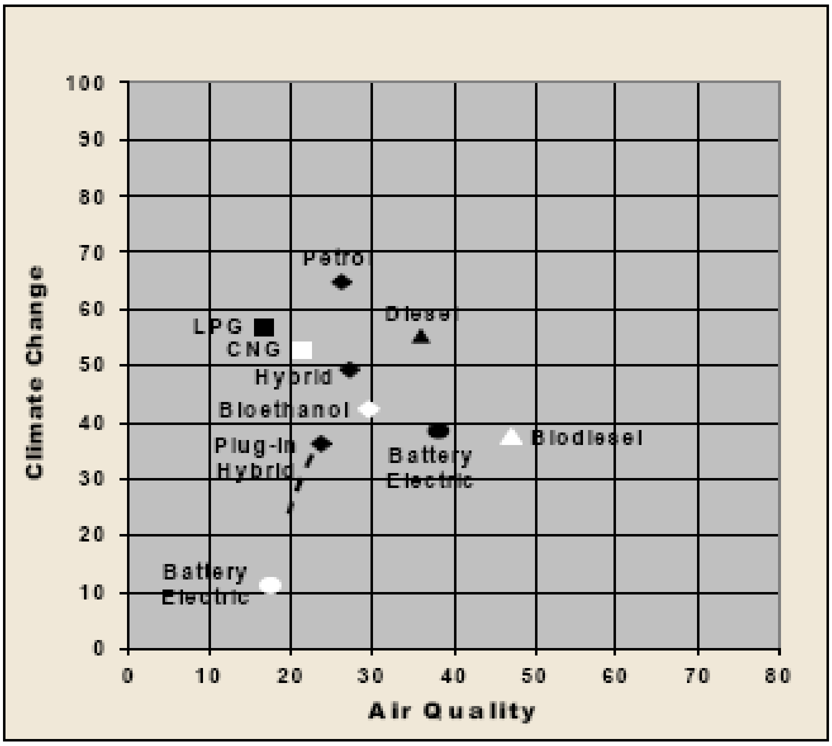
* Tassan et al. (2013).

** Rose et al. (2013), life-cycle reductions relative to diesel-powered refuse collection truck.

Rose et al. (2013) note that SO_x and PM emissions are mainly reduced at the feedstock and fuel production stages, while CO, NO_x, VOC and PM emissions are significantly reduced at the fuel dispensing and vehicle operation stages. At the location of vehicle deployment, a 54 % reduction in overall air pollutant emissions can be achieved, representing a significant benefit in urban areas.

Source: **LES (2011)**

Figure 4-51 shows that replacing petrol and diesel with alternative propulsion systems usually reduces both GHG emissions and air pollution, except in the case of biodiesel which leads to higher air pollution.



Source: LES (2011)

Figure 4-51. Performance of different vehicle propulsion options in terms of GHG emissions (y-axis) and emissions affecting air quality (x-axis).

Appropriate environmental indicators

The most appropriate indicators to monitor the implementation of this BEMP are:

- average fuel consumption of the waste collection vehicles (litres/100km);
- share of vehicles that are Euro 6 in the total waste collection vehicle fleet (%);
- share of waste collection vehicles that are hybrid, electric, natural-gas- or biogas-powered (%).

Cross-media effects

The life-cycle environmental balance of biogas produced from crops is much worse than biogas produced from waste, owing to nutrient losses during crop production (eutrophication), the need for agro-chemical inputs (multiple impacts) and possible indirect land use change incurred by agricultural land expansion (GHG emissions, but also biodiversity effects) (Boulamanti et al., 2013).

Biomethane upgrade of biogas is associated with methane leakage of around 1–2 %, which can have an important effect on the GHG balance of biomethane as a fuel (Ravina and Genon, 2015). Biomethane upgrade also requires significant electricity, which may be provided by an on-site combined heat and power plant fuelled by biogas, or imported from the grid. Cheshire (2014) reported electricity consumption of

1.06 kWh and 0.6 kWh per kg of methane, respectively, for biomethane upgrade and compression for use as a vehicle fuel, for a small-scale upgrade plant.

Abiotic resource depletion is associated with use of rare-earth metals in batteries for electrical and hybrid propulsion and alternative-fuelled bodies. This can be minimised through recycling of these metals. Whilst GHG emissions associated with vehicle manufacture are twice as high for a hybrid compared with a conventional diesel truck, significant GHG savings during operation mean that lifetime GHG emissions are 17 % lower for hybrid trucks (Nehlsen, 2013).

As the hybrid or CNG trucks cause less noise, they enable waste collection at times when there is less traffic (late evening, early morning), so they contribute to reductions in congestion and noise pollution.

Operational data

Biomethane may also be liquefied by cooling it to -160 °C, making liquid biomethane (LBM) which can be transported, stored and used in a more convenient, energy-dense form (Tassan et al., 2013). LBM may also be converted to compressed biomethane prior to use in vehicles. See the case study of transport biomethane production at the Västerås (Växtkraft) plant in Sweden (Monson et al., 2007).

Fuel quality

Biogas may be collected from (legacy) landfill or anaerobic digestion plants. Raw biogas contains various contaminants that need to be removed through a cleaning process, and CO_2 that needs to be removed via an upgrade process (Table 4-20).

Table 4-20. Typical compositions of landfill gas, biogas from anaerobic digestion (AD) and natural gas

Parameter	Unit	Landfill gas	Biogas from AD	Natural gas
Lower calorific value	MJ/Nm^3	16	23	39
Density	kg/m^3	1.3	1.1	0.82
Wobble Index, upper	MJ/Nm^3	18	27	55
Methane number		>130	>135	73
Methane, range	Vol-%	35–65	60–70	85–92
Heavy hydrocarbons	Vol-%	0	0	9
Carbon dioxide, range	Vol-%	15–40	30–40	0.2–1.5
Nitrogen, range	Vol-%	5–40	--	0.3–1.0
Hydrogen sulphide, range	ppm	0–100	0–4 000	1.1–5.9
Ammonia	ppm	5	100	--
Total chlorine, as Cl^-	mg/Nm^3	20–200	0-5	--
Source: SGC (2012).				

Concentrations of CO_2, hydrogen sulphide (H_2S) and chlorine in particular must be significantly reduced to achieve efficient combustion and to minimise engine corrosion and polluting emissions.

Table 4-21 shows specifications for biomethane if it is to be used in non-modified vehicle engines, from Tassan et al. (2013). Those authors note the low limit of hydrogen sulphide, set at a maximum concentration of 10 ppm, owing to the highly corrosive nature of this compound. They report that some national biomethane standards, such as Swedish standard SS 15 54 38, may allow significantly higher concentrations of H_2S.

Table 4-21. Biomethane specifications for use in engines without material or calibration modifications, from Tassan et al. (2013)

Methane content	> 83 % v/v
Other hydrocarbon content	< 13 % v/v
Carbon dioxide content	< 14 % v/v
Nitrogen content	< 14 % v/v
Hydrogen content	< 5 % v/v
Water content	< 55 mg/Nm^3
Methane number	> 70 according to Kubesh/King/Liss (AVL) method
Hydrogen sulphide content	< 10 ppm
Total sulphur content	< 10 mg/Nm^3 according to ISO 6326-5
Contaminants content	According to ISO TR 15403
Siloxane content	< 5 mg/Nm^3

Engine warranties may not be honoured by manufacturers if an engine fails when using an alternative fuel such as CNG or biomethane, unless it has been explicitly stated that the engine can run on that fuel (Tassan et al., 2013).

Dedicated engine technology

Natural gas dedicated engine technology (e.g. Otto cycle stoichiometric combustion with a multipoint injection system and three-way catalyst) is able to achieve the best environmental results, with drastic reductions in emissions of GHGs, substances contributing to photochemical smog, nitrogen oxides and particulate matter, and also a good economic performance (low cost and mature Original Equipment Manufacturing (OEM) technology). This is the preferred option for alternative-fuelled vehicles.

Dual fuel systems

Logistical or cost considerations may favour dual-fuel systems over dedicated alternative-fuelled systems. Fully-integrated, manufacturer-approved dual-fuel systems are available for some vehicle types and models, including (Tassan et al., 2013):

- Mercedes Hardstaff with oil ignition gas injection (OIGI) system;
- Volvo Clean Air Power Dual-Fuel system.

Meanwhile, some dual-fuel systems bypass the electronic Controller Area Network Bus system to control the diesel pilot ignition directly. Such semi-integrated systems do not perform as well as fully integrated systems. Integrated systems achieve diesel substitution rates of 85 % to 90 %, compared with 45 % to 60 % for non-integrated systems. In addition, while manufacturer warranties cover integrated systems, non-integrated systems require separate support warranties for the dual-fuel technology (Tassan et al., 2012).

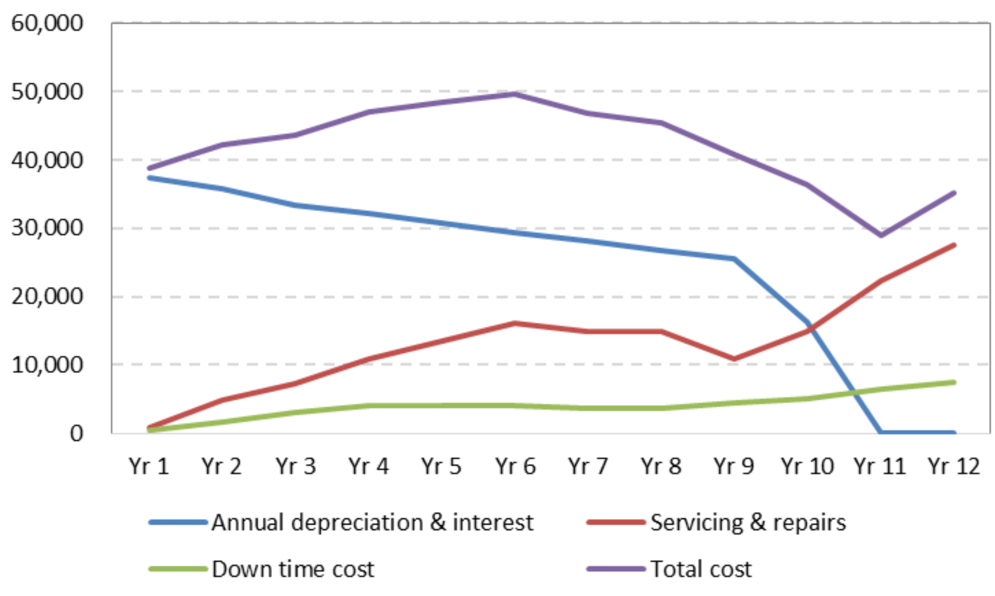
Applicability

This BEMP is broadly applicable. The presence of filling or recharging stations is less of an issue for refuse collection than other types of transport because vehicles are usually operated over a limited distance and the fleet is run from a centralised waste depot where refuelling can take place.

Compressed natural gas (CNG) is available in all EU countries. Biomethane may not be available in many regions, but wet organic waste (e.g. food waste) can be used to produce biogas that can be upgraded to transport biomethane.

Economics

National Grid (2014) quotes UK Department of Transport estimates that gas-powered trucks cost between GBP 15 000 and GBP 44 000 (EUR 21 000 and EUR 62 000) more than conventional diesel trucks. Private refuelling infrastructure can cost between GBP 400 000 (EUR 563 000) to GBP 1 million (EUR 1.41 million) to install, plus the cost of a grid connection. Safety considerations mean that CNG storage cylinders can be expensive to design and build, making a significant contribution to the additional costs of a gas vehicle (Tassan et al., 2013). Figure 4-52 shows average annual running costs, excluding fuel, for a fleet of 150 CNG refuse collection vehicles. BSR (2015b) notes that maintenance costs are only slightly higher for CNG compared with Euro 6 diesel trucks.



Source: BSR (2015b).

Figure 4-52. Average annual running costs, excluding fuel, for a CNG refuse collection vehicle

However, the retail prices of CNG and biogas are considerably lower than for petrol and diesel owing to reduced duties. National Grid (2014) reports that an articulated tractor unit doing an average of eight miles per gallon of diesel (8 mpg = 35 L/100 km) costs GBP 0.62 per mile (EUR 0.54 per km), while natural gas costs approximately GBP 0.39 per mile (EUR 0.34 per km). WRAP (2010) recorded a fuel efficiency of between 6 mpg and 10 mpg for a single-modal refuse collection vehicle (skip carrier), and 3.5 mpg to 4.5 mpg for a multi-modal refuse collection vehicle. Based on National Grid (2014) data, natural gas fuel cost savings for single- and multi-modal refuse collection vehicles could equate to EUR 40 000 and EUR 80 000 respectively over 200 000 km, at least offsetting the higher purchase cost.

Stricter vehicle emission standards are associated with higher operating and maintenance costs for HGVs. When converting an HGV to run on gas, the removal of parts of diesel system (including selective catalytic reduction) will save significant costs over the vehicle lifetime (Tassan et al., 2013). This may cancel out higher servicing costs for vehicles running on natural gas or biogas, as indicated by BSR (2015b).

Driving force for implementation

Stricter emission standards, currently Euro VI (European Regulation 595/2009 and European Regulation 582/2011), favour gas- over diesel-powered engines because of the increasingly complex and costly emission control technology required for diesel vehicles to comply with these standards.

Refuse collection trucks are well suited to CNG and biogas fuelling owing to relatively short routes and repeated returns to waste depots where they can be refuelled.

Alternatively (electric) fuelled bodies and hybrid refuse collection trucks generate significantly less noise during bin lifting operations owing to the use of electric motors rather than a revving engine. This is a major advantage, especially in urban areas.

Green procurement guidelines by municipalities may prioritise the purchase of low-emission vehicles directly for municipality-managed collections, or the subcontracting of waste management to companies that use low-emission vehicles to reduce their environmental footprint.

Reference organisations

Renova, Sweden. A total of 37 out of 180 heavy vehicles run entirely on natural gas, and 16 refuse collection vehicles use electric-hybrid technology (Renova, 2015).

Emterra, Winnipeg Canada. In 2012, Emterra committed to using CNG trucks in Winnipeg, Manitoba, and now have almost 60 natural-gas-powered, heavy-duty waste and recycling trucks in operation (Emterra, 2015).

Waste management organisations in the German cities of Munich, Nuremberg, Offenbach, Baden-Baden and Darmstadt have tested electro-diesel hybrid vehicles over the past 4–5 years (AWM, 2014).

Veolia is operating a landfill in Claye-Souilly collecting biomethane and converting it into biofuel (Bel, 2015).

SITA UK is a landfill operator in the UK which produces vehicle fuel from landfill gas.

Production is over 5 million litres of liquid biomethane each year from the landfill site at Albury, Surrey, which can be used alongside diesel in converted waste collection vehicles (SITA UK, 2017).

Box 4.24. BSR, Berlin, biomethane case study

BSR processes approximately 60 000 tonnes per year of organic waste from Berlin households in a biogas plant. The biogas produced is cleaned, processed, concentrated and fed into the city gas network as biomethane. A total of 150 biogas-powered refuse collection vehicles, about half of the BSR fleet collecting approximately 60 % of the city's MSW, are refuelled from this network via gas stations in three BSR depots. As a result, annual savings of around 2.5 million litres of diesel are achieved (BSR, 2015a).

Box 4.25. Courbevoie, Paris, electric vehicle case study (emerging best practice technology)

In 2011, SITA introduced the first fully electric domestic waste collection truck. A partnership between SITA, PVI, a leader in electrical traction for vehicles, SEMAT, a company specialising in collection and cleaning equipment, and Li-Ion, a battery manufacturer, developed this pioneering electric refuse collection truck. The vehicle benefits from zero direct emissions and extremely low noise levels, in addition to improved cab visibility enabled by the absence of a large combustion engine under the cab (Suez-environment.com, 2015). This technology represents an emerging best practice that may not yet be commercially applicable. If and when it becomes economically viable for commercially application, it may be regarded as best practice.

Source: Suez-environment.com (2015).

Box 4.26. Nehlsen GmbH & Co. KG electric-hybrid case study

Nehlsen GmbH & Co. KG, Bremen are participating in the *Electric Mobility* programme by testing one waste collection vehicle with diesel-electric drive and one with plug-in components. The usability and technical, environmental and economic performance of these vehicles are being monitored across a range of operating conditions, and will be compared with conventional refuse collection vehicles. The results will be used to evaluate hybrid vehicles and optimise route planning, workload, fuel consumption, CO_2 emissions, and noise performance (Schaufenster Elektromobilität, 2015). See the "Rotopress Dualpower" refuse collection truck under Description above.

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4.6. BEMPs for extended producer responsibility (EPR) schemes

4.6.1. Best use of incentives by producer responsibility organisations (PROs)

Summary overview
It is BEMP for producer responsibility organisations (PROs) to enhance the performance of their extended producer responsibility (EPR) scheme by setting up incentives (going beyond legal requirements) that drive increased separate collection, reuse and recycling rates for the waste collected under the EPR. Actions that PROs can implement include: - motivating citizens to source separate waste more and better through innovative communication actions, such as competitions among territories; - close cooperation (financial, technical and/or logistic) with public authorities at regional/local level; - cooperation with social economy actors for the collection and reuse of products; - incentivising producers to design more sustainable products (e.g. via "fee modulation"); - benchmarking environmental achievements of different areas covered by the EPR scheme, e.g. at the level of the territories of public authorities at a regional/local level.

Waste management area							
Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability
The actual leverage that a PRO has on the EPR depends on the national setup and legal allocation of roles and responsibilities. For the application of some incentives, proper allocation of finances is needed. For this, the governance structure of the PRO may play a role (owned by producers or not, for or not for profit, etc.).

Specific environmental performance indicators
- Recycling rate (% of waste that is actually recycled or sent for recycling out of the total waste covered by the EPR scheme); - Preparation for reuse rate (% of waste that is delivered as input to a centre for preparation for reuse out of the total waste covered by the EPR scheme). - (applicable at the local level for a specific local area where the EPR scheme is in place) Share of EPR-covered products found in residual waste based on composition analysis (% of the total quantity of mixed waste); - (applicable for a specific national, regional or local area where an EPR scheme for packaging waste is in place) Share of EPR-covered packaging that is

targeted by the selective separate collection system (% of the total quantity of EPR-covered packaging put on the market).

Description

Extended producer responsibility (EPR) reflects the idea that producers who put products on the market should assume responsibility for their products beyond the commercialisation stage and in particular for their end-of-life treatment (Lindhqvist, 2000). The rationale behind this is that, when faced with the obligation to treat their products at the post-consumption phase, producers will have an incentive to reconsider their products' design up front – thus promoting environmental improvements of product systems in the long run. Rather than implying one single policy design, EPR can take various forms and can be applied through a combination of policies and instruments in order to adapt to differing local contexts, legislative climates, economic situations or legal constraints (OECD, 2016). Today, EPR is applied globally to manage waste from different product types.

The application of EPR very often involves a requirement for the producers[51] to establish systems for the collection (and recycling) of their waste with a view to achieving certain targets (i.e. a "take-back mandate"). For reasons of practicality, they usually join a producer responsibility organisation (PRO) in order to comply with this requirement. A PRO is a collective body operating nationally which takes charge of meeting the legislative requirements of producers on their behalf and against a financial contribution on their part. As a result, once a producer has joined a PRO (typically through the payment of a fee corresponding to the type/quantity and characteristics of the products they put on the market), the PRO becomes the entity which is legally responsible and thus needs to ensure that the legislative targets and requirements are fulfilled. Legal requirements usually include reaching specific rates of collection and/or recycling (depending on the type of product), bearing the costs of end-of-life management, contributing to communication and awareness-raising actions. In the future it is expected that requirements on producers will increase, together with increasing ambitions for recycling and the trend towards more cost coverage by producers (e.g. for cleaning of littering).

A PRO collaborates/interacts with a whole range of different stakeholders – producers, waste management operators, public authorities, citizens. Each of them has corresponding roles and responsibilities and can accordingly (and only to a certain extent) have an influence on the whole system and other participating actors.

The resulting system of contractual arrangements and operational solutions is referred to as an EPR system or scheme. In EU countries, such EPR schemes are set up at national level[52] and cover a wide range of products[53]. Their implementation and set-up

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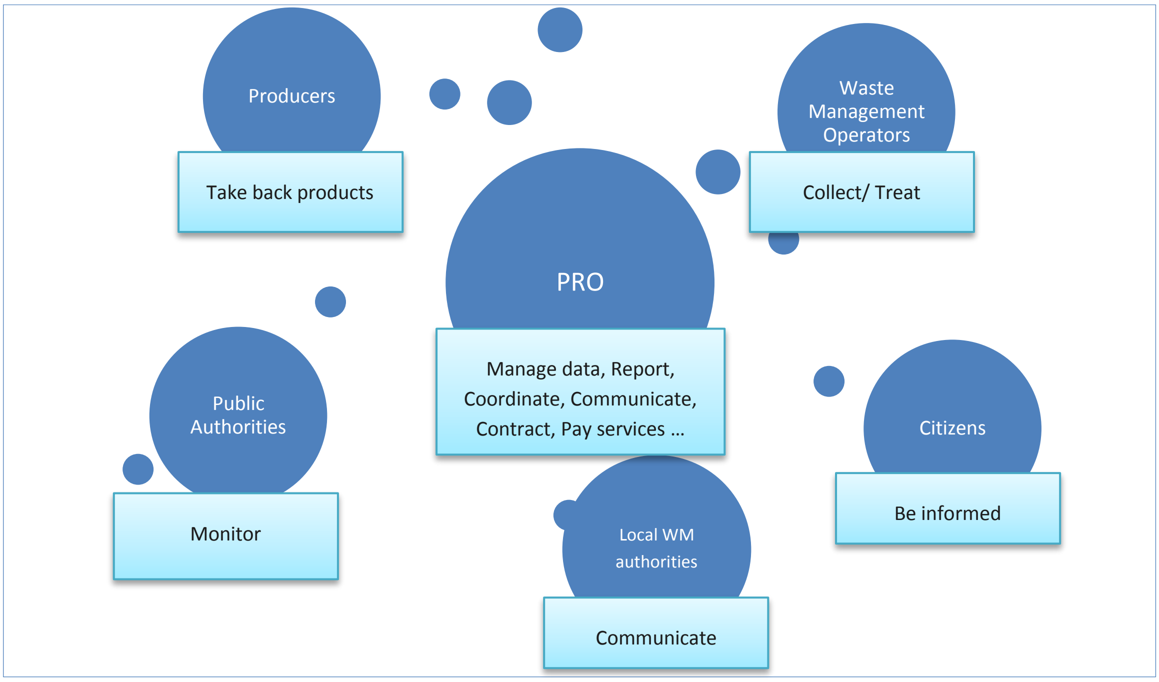
[51] In practice, the take-back obligation applies to manufacturers, importers, intermediaries and retailers of different products. In this report for the sake of simplicity, the party on which the EPR obligation falls is referred to as the "producer".

[52] Globally, there are cases of EPR schemes at a subnational level (ex. individual States in the US or Canada). In Europe there are different forms of implementation of the EPR principle at a local/ regional level (ex. as a voluntary agreement with some producers). However, up to this moment there are no records of full-scale implementation of classic EPR schemes at other level than the national.

varies a lot across the EU Member States (actors involved, their roles and responsibilities, and performances).



Across the different setups of EPR schemes, a PRO usually has three main functions (BIO by Deloitte, 2014):

- to finance the collection and treatment of the product at the end of its life;
- to manage corresponding data, including financial (membership, contractual) and technical (recovery options) information;
- to organise and/or supervise these activities.

Consequently, the PRO plays a central role in the system and acts as an intermediary between the rest of the stakeholders. Therefore, there is scope for the PRO to enhance the performance of the EPR scheme – **in addition to legal requirements** – by setting a number of incentives for the different stakeholders involved.

At a local level, a PRO can interact with a number of actors in order to improve performance in the territory, as shown by the examples below:

- Motivation of **citizens** to source separate waste more and better through innovative communication actions. It is often foreseen in the national legislation that the PRO contributes (financially) to communication with and awareness-raising of citizens. This is often done in close cooperation with public waste management (WM) authorities since they are typically in charge of communicating environmental / waste management issues to the citizens. However, the PRO can take some additional actions to boost the involvement of citizens. Direct motivation

[53] Three EU waste Directives – on waste electrical and electronic equipment (WEEE), batteries and end-of-life vehicles – explicitly mandate or encourage the application of EPR. EPR is further widely used to transpose the European Packaging and Packaging Waste Directive although the Directive itself does not impose EPR.

for individual citizens can be especially effective, for example through the organisation of competitions with prizes awarded for the highest collection.

- Close cooperation with **public authorities at a regional/local level**. This can be the case when the PRO provides support to the local waste management authority for the collection, recycling and/or reuse of different products in the specific territory. The support can be financial (in addition to what is required by law), technical (benefitting from the expertise accumulated), logistical (provision of materials) or other. In addition, open dialogue with national and local stakeholders is important to exchange information and cooperate to find common solutions.
- Cooperation with **social economy actors** for collection and reuse of products in a certain territory since reuse activities are typically local initiatives (Step, 2016). This is especially relevant for products with high reuse potential such as WEEE.

Examples on a national level include the following:

- Incentives to **producers**, mainly concerning the design of their products. This happens typically via "modulation" of the producer fees, i.e. setting varying fees reflecting the real end-of-life management costs of products with a view to rewarding those producers who make eco-design efforts. In addition, by centralising knowledge and expertise in the area, the PRO can also provide consultancy services and guidance in order to support producers in improving the design of their products. It can benefit from its central position to have access to the data and best practices of its members and serve as a platform for exchange and dissemination to the others.
- The PRO can play a role in benchmarking environmental achievements in its network of public authorities at regional/local level – by pointing out the best performers in order to motivate others to follow.

Achieved environmental benefits

Setting EPR schemes is regarded as a relevant way to **decrease disposal** and improve the **recycled** quantities of a given type of product/waste. This entails an increase in environmental benefits linked to the production of recycled materials which substitute virgin materials and allow energy savings during production processes. Additionally, thanks to the rise in recycled materials, the amount of residual waste to be disposed of is reduced. For instance, in the last two decades, following the introduction of an EPR system for household packaging in Germany, the national recycling quota of sales packaging (from households and small businesses) rose from 37.3 % to almost 80 % (GVM, 2014).

Schemes boosting **reuse**[54] also have a potentially significant environmental benefit. Reuse of products conserves embodied energy and material and thus avoids the extraction of resources necessary for the production of new products. In particular for WEEE, reuse provides environmental benefits that are much higher than the benefits from recycling. This is because recycling of electronic equipment with presently available technologies implies the partial destruction of the embodied value of

[54] This BEMP focuses on the improving traditional or centralised reuse of products, e.g. diverting fractions towards preparation for reuse. Therefore, packaging reuse through deposit-refund systems is out of the scope of the study.

materials (through shredding) and still going through a manufacturing process afterwards, with the associated environmental impacts (as compared to the environmental impacts of reuse due to refurbishing and manufacture of replacement parts) (CM consulting, 2014). Furthermore, reuse activities have social benefits since they create employment opportunities (often in social economy enterprises) and can potentially make products more affordable to low-income households and institutions.

In the longer term, it is expected that EPR will provide producers with incentives to reconsider the **design** of their products and will thus trigger overall environmental improvements in the global product systems. For simple products such as lightweight packaging, there has been an effect mostly in terms of weight reduction (GVM, 2016; Pro-Europe, n.a). However, the evidence of the past 25 years of existence of EPR systems worldwide only confirms this effect to a limited extent (OECD, 2016). In any case, it is difficult to make a direct link between product design and the contribution of EPR and of other factors (such as the financial considerations of using less material resources). As for more complex products such as WEEE, the eco-design effect of EPR schemes has been even more limited (Tojo, 2004; OECD, 2016).

Appropriate environmental indicators

The following indicators allow monitoring the implementation of the BEMP:

- Recycling rate (% of waste that is actually recycled or sent for recycling out of the total waste covered by the EPR scheme);
- Preparation for reuse rate (% of waste that is delivered as input to a centre for preparation for reuse out of the total waste covered by the EPR scheme).
- (applicable at the local level for a specific local area where the EPR scheme is in place) Share of EPR-covered products found in residual waste based on composition analysis (% of the total quantity of mixed waste);
- (applicable for a specific national, regional or local area where an EPR scheme for packaging waste is in place) Share of EPR-covered packaging that is targeted by the selective separate collection system (% of the total quantity of EPR-covered packaging put on the market).

Cross-media effects

Improving the management of one or several waste fractions in relation with an EPR scheme can lead to various cross-media effects, e.g. related to the increasing number of collection schemes that might lead to more emissions linked with collection and transport.

Other cross-media effects, such as energy use, linked with the processing (sorting, dismantling and pretreatment) of waste might also occur but will vary depending on the waste streams considered.

However, the cross-media effects are likely to be outweighed by the environmental benefits if the waste streams are diverted from disposal to recycling.

Operational data

To monitor the implementation and practical operation of this practice, great attention is paid to analysing in particular the following aspects:

- The allocation of roles and responsibilities of the different stakeholders involved.

- The mechanisms of interactions between the different stakeholders, be they institutionalised or informal/voluntary.
- The various instruments in use for the BEMP: legal instruments (legal targets, mandatory separate collection), financial instruments (subsidies and taxes), and communication. Specific collection and sorting schemes will also be detailed.

PROs creating incentives for stakeholders at local/regional level

Bebat and Recupel (Belgium)

Bebat – the national PRO for batteries in Belgium – has set up competitions among schools and secondary education institutions for the selective collection of this waste stream. At least once a year, it organises collection campaigns throughout the country in order to collect as many used batteries as possible over a certain period. A prize for the winning school is for example the organisation of a live concert of a popular musician in the school itself (this is done in cooperation with a major media company). Furthermore, Bebat has developed a special "saving programme" where schools can accumulate value points corresponding to quantities of batteries collected. These value points can later be exchanged against different types of goods or services (a reduction on public transport tickets for instance).

For WEEE, the national PRO Recupel has a similar initiative, but targeting entire municipalities. So far, two rounds of the competition have been organised and the prize has been a "gigantic picnic" for all the inhabitants of the winning municipality. In the second edition, in 2015, more than half of all the Belgian municipalities took part, collecting 8 830 tonnes of WEEE during the duration of the campaign (one week). For comparison, the annually collected quantities of WEEE amount to 111 357 tonnes (equivalent to a weekly average of 2 100 tonnes) (Recupel, 2016). One of the key objectives of the initiative was to make citizens more aware of the fact that they have the possibility to discard their small WEEE and lamps not only via the most common and well-known channels (civic amenity sites and reuse centres) but also via drop-off points at retailers.

This latter objective is further enhanced by a joint initiative of both PROs. It is called "Recycling Point" and aims to promote an even more efficient collection of batteries, lamps and small WEEE (smaller than 25 cm) through the instalment of collection points at retail stores. This has a number of advantages:

- increased convenience for the consumer: they are able to dispose of unused WEEE in the same place where they do their regular shopping;
- better reputation and more traffic for the retailer: the collection points are very visible and can attract more consumers to their premises;
- increasing collected quantities for the PROs: this is the ultimate objective of the PROs in order to fulfil the legislative requirements.

Figure 4-53: "Recycling Point" (Source: Bebat-Recupel, 2017)

In fact, every store selling batteries and/or EEE is obliged to also take back used products, even if they have not been originally bought at the store. When becoming a host to a "Recycling Point", the PROs take on this take-back obligation from the store free of charge – they provide the necessary collection material, ensure timely collections and guarantee the subsequent proper treatment of the waste. In addition, Bebat offers this service to other big generators of battery waste (such as companies or schools).

Partly because of regional legislation, Recupel also has some initiatives to promote reuse of EEE through the work of social economy actors. The conditions are put down in a cooperation agreement between Recupel and two federations of social economy companies active in reuse, repair and recycling (Ressources for the territory of the Brussels-Capital and Walloon regions and Komosie for the region of Flanders). In 2015, this led to the placing of about 788 tonnes of used EEE on the Belgian second-hand market, contributing to the creation of around 350 jobs in the sector (RTBF, 2016).

Ecofolio (France)

In France, the PRO for graphic paper Ecofolio[55] dedicates a minimum of EUR 5 million every year to financially support projects which aim to improve the performances of municipalities in terms of paper waste management. Three types of projects are eligible for financing of up to 75 % of their total costs (up to a limit of EUR 800 000):

- projects that put an emphasis on paper collection from households;
- projects aiming to improve the process and organisation related to sorting of paper;
- projects aiming to capture new sources of generated waste paper, including paper from offices.

Commonly in France, graphic paper is collected along with other recyclable materials in a co-mingled waste stream, which renders the subsequent sorting process more complicated and costly. Therefore the projects financed by Ecofolio aim to allow for better recycling of paper waste through a significant increase in the collection and sorting performances of a municipality. For this, the PRO promotes for instance more efficient bring bank systems (via which graphic paper is typically collected), optimisation of collection rounds or the putting in place of a selective collection of graphic paper from offices and administrations (generators of bigger quantities of this material).

In the period between 2013 and 2016 Ecofolio supported more than 130 projects by French municipalities, amounting to a total of around EUR 30 million. The beneficiary municipalities estimate an average reduction of 20 % of the related management costs and an average increase of 20 % in revenues resulting from the additional collected quantities (Ecofolio, 2016).

Ecofolio has also set up a team of three Project Officers who are in charge of services to municipalities. Their role is to interact directly with them in order to better identify their specific needs and to inform and facilitate the implementation of projects with the objective of boosting paper recycling. These meetings are a possibility for municipalities to deepen their knowledge of the paper value chain and to find solutions

[55] In France currently there are two distinct PROs for packaging (including paper and cardboard paper) and for graphic paper.

to the specific problems they face in their territories. In 2015, 36 bigger meetings were organised in which 330 municipalities took part, representing around 38 % of the total population (Ecofolio, 2016).

CONAI/Comieco (Italy)

In Italy, the umbrella PRO for household packaging CONAI provides support and co-financing to public authorities at different levels. The aim is to promote the introduction of appropriate management models for packaging waste or to ensure that the already existing ones are optimised and efficient. The conditions are specified in a Framework Agreement between CONAI and the Association of Italian Municipalities (ANCI). The Framework Agreement is the instrument through which the EPR system guarantees municipalities the coverage of the additional costs incurred for introducing separate collection systems for packaging waste.

In addition to the obligations stated in the Framework Agreement, CONAI provides different kinds of support to local authorities:

- Training courses mainly addressed to public administration staff (at administrative and managerial levels) and to employees of waste management companies (public or private). Specific training projects can be related to dissemination of best practices, explanation of relevant environmental legislation, presentation of the PRO system including its activities and operations, or organisation of communication activities.
- Support for local communication campaigns including guidance, materials and necessary information.
- Support of local projects in a specific territory where performances are lagging behind. In such cases, CONAI can provide support to local governments, municipalities or groups of municipalities in a range of areas, starting with the planning of the waste management services. The support can take different forms such as feasibility studies, providing expertise on the design of the waste management system, legal and technical assistance for drafting tenders for collection services, and co-financing of the communication and awareness-raising campaigns towards citizens on separate collection.
- Collaboration with regions and provinces for the analysis of the status quo, definition of waste management plans and strategies, and identification of guidelines for collection, information and public awareness.
- Database and Observatory of local performances: these are two specific tools which have the objective to gather the most important data related to municipal waste management of Italian municipalities. This provides a useful database for further processing – for example available to municipalities to track and monitor their own progress and performances. The Observatory builds on the database to provide a benchmarking tool for local administrations to get knowledge and support to improve their strategies.

Comieco – the PRO for paper packaging, part of the CONAI consortium – has several initiatives for promoting better paper collection at local and regional levels.

One such initiative is the annual Cartoniadi competition which encourages the involvement of all citizens. Therefore this initiative enhances the commitment of the community towards a common goal of better waste management. It is a competition between districts of the same city or towns of the same region that, for a month, compete to collect paper and cardboard better (higher quality, less impurities) and in higher quantities. At the end, a "recycling champion" is announced based on the achieved results. The aims are:

- to increase, during and after the competition, the quantity and quality (less contamination that impairs recycling) of the separately collected paper and cardboard;
- to gather and consolidate data on quantity and quality in the months following the competition;
- to raise awareness among citizens on environmental issues, in particular on separate collection, and to give them the opportunity to intervene with concrete actions;
- to reassure citizens of the proper functioning of paper and cardboard recycling (all that is collected is actually sent for recycling);
- to raise awareness and remind citizens of a number of "golden rules" and mistakes that affect the quality of the collection and therefore the recycling performance.

For the actual implementation at local level, no specific training or technical means are needed, but focus is put on promotion and effective communication. During Cartoniadi, the collection service is in operation in the usual way and at the end of the month the prize is given to the city that has achieved the highest increase compared to its initial level. The prize is in cash but has to be compulsorily spent for projects with a public benefit (e.g. redevelopment of a public area, books or computers for schools, new photovoltaic lighting).

As a result, it is observed that the municipalities participating in Cartoniadi achieve on average the following improvements:

- an increase of 30% in the collection of paper and cardboard during the competition period;
- during the following months, collection rates stabilise to a level 15 % higher compared to the period before the competition.

Additionally, Comieco provides co-financing to medium-sized municipalities (of approximately 1.3 million inhabitants) with performance levels below the national average in order to increase the quality and quantity of paper collection in their territories. The grants are used to purchase new equipment required to perform or improve separate collection of paper and cardboard (boxes, bells, drums, paper bags, etc.).

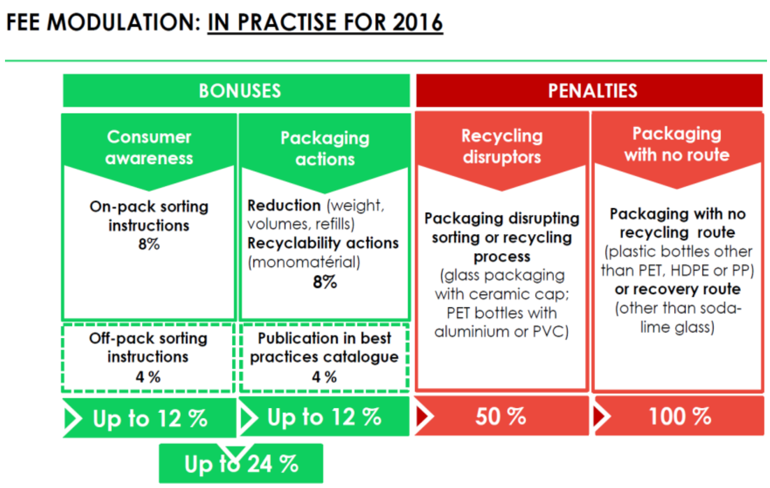


Figure 4-54: Eco-Emballages fee modulation (Source: Eco-Emballages, 2016)

Eco-Embes (Spain)

The Spanish PRO for packaging, in collaboration with the national Federation of Municipalities and Provinces, has developed a Handbook on effective communication on waste for local entities. The Handbook was produced in response to a study carried out by the Organisation of Consumers and Users (OCU) which revealed the need for the citizens to receive clearer information on waste from the local administrations responsible. It aims to provide municipal technicians with tools that allow them to systematise information on waste management and offer it effectively on any web platform in a way that is transparent and easy to understand.

The project includes:

- a handbook in both paper and digital format;
- online workshops and training for municipal technicians;
- an online template for the development of waste content on the municipal website;
- a video on the importance of communication to generate responsible habits among citizens.

In 2016, 235 municipal technicians were trained, four technical studies carried out and more than 240 technical consultations organised.

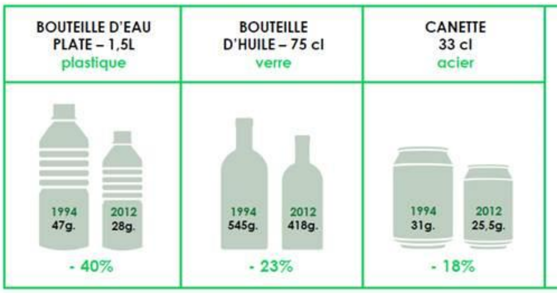
PROs creating incentives for stakeholders at national level

Eco-Emballages (France)

The French national EPR scheme for packaging has designed a comprehensive system of incentives for producers obliged to improve the design of their packaging in order to facilitate recycling. The incentives are twofold:

- On the one hand, fees are modulated in accordance with the quantities and properties of the packaging (see Figure 4-55). The fees are set per weight for each material (as an incentive to reduce packaging weight) and per packaging unit (as an incentive to remove over-packaging). In addition, a modulated grid of fees is applied according to more specific characteristics of the packaging. As a result, producers can receive a deduction of up to 24 % of the amount due or, in contrast, be penalised by paying more than 100 % of the basic fee.
- On the other hand, producers are provided with a number of tools and services to support them in changing their packaging in line with these modulations. This includes joint research and development projects of Eco-Emballages together with producers, an online catalogue promoting best practices, various guides and training, specific eco-design tools to assess the environmental impact and the recyclability of a certain type of packaging.

The weight reduction effect for several types of packaging is well documented. In particular, it was estimated that between 2007 and 2012 packaging waste put on the market was reduced by 106 000 tonnes (surpassing the national target of 100 000 tonnes). Moreover, since the start of the EPR scheme in 1994, the weight for packaging made out of glass, steel and certain plastics has been reduced (as shown in Figure 3-24).



Source: Eco-Emballages, 2017

Figure 4-56: Reduction in packaging weight from 1994 till 2016

In addition, the penalty system has proven to be especially effective in certain cases (e.g. ceramic cap on glass bottles). Nevertheless, the optimal way remains to support the company on R&D so as to avoid packaging that disturbs recycling. Furthermore, the EPR fee is part of the global cost

of packaging, and, as such, of a global decision-making process including industrial costs and product/brand market position. It is therefore difficult to precisely evaluate the impact of fee modulation on packaging design. However, it is the case that eco-modulation is one of the elements producers take into account (along with other business costs).
PROs for other products in France apply a similar system of bonuses and penalties according to a set of environmental criteria such as reusability, recyclability, lifetime, presence of hazardous substances, etc.

Applicability
The actual leverage that a PRO has on the EPR depends on the national setup and legal allocation of roles and responsibilities. For the application of some incentives, proper allocation of finances is needed. For this, the governance structure of the PRO may play a role (owned by producers or not, for or not for profit, etc.).

Economics
The economic indications will vary greatly depending on the way the BEMP is implemented and the exact setup of the EPR scheme itself.

Generally, any financing by PROs originates from the fees that producers pay into the system (if it is assumed that the PRO does not make any losses). Therefore, any decision to engage in additional activities not foreseen by legislation can potentially impact the amount of these fees. However, since the aim of these activities is to improve performance, the rationale is that new incomes will also be generated (linked to the sales of reused products and the increase in recycled quantities) which would in turn compensate for incurred costs.

The funding by PROs is usually linked to results – this can be used as a guarantee for their initial investment.

Driving force for implementation
The first objective of a PRO is to reach the waste targets set in national legislation. For one-off annual reporting and when taking a short-term perspective, these could in many cases be easily achieved (e.g. through cherry-picking techniques focusing on easy-to-reach waste). However, in order to be able to comply with those targets in the long term and to ensure the sustainability of the system, PROs will (often) have to address more problematic situations such as those occurring at a more local level or concerning fundamental elements (such as eco-design). Therefore, there are a number of reasons why a PRO would want to do something more than is required by the law, like the below for instance:

- Given the wide array of EPR schemes in Europe, engaging in additional activities in order to improve results can be a way to promote its own model (e.g. non-competing, not for profit).
- In countries where performances are not that good or there is PRO accreditation from public authorities, it is important that the PRO demonstrates its long-term commitment and wish to improve.
- In countries where there is competition between PROs, they might want to differentiate themselves with a positive image towards both their "customers" (producers) and public authorities.
- By supporting local authorities, the PRO reaches its own objectives of improving separate collection of the waste they are in charge of (both in

quantitative and qualitative terms). Consequently, also the quantity and quality of waste going to recycling/reuse is also improved, as well as the associated revenues for the PRO.
- Boosting local employment especially with reuse. Indeed the reuse sector has a significantly higher job creation potential than recycling, incineration and landfill (Reuse, 2015). US figures show that for 10 000 tonnes of waste products and materials, 1 job would be created if incineration was used compared to 6 jobs in landfill, 36 jobs in recycling, and up to 296 in refurbishment and reuse. Similarly, data from Belgium suggests even greater potential for reuse, at 800 jobs for 10 000 tonnes. While PROs might not be directly concerned by the creation of new jobs, this is an argument they can use to promote their sustainable image and acceptance by other stakeholders.
- Engaging in additional actions is a way for PROs to respond to possible public pressure (e.g. from NGOs, civil society).

Reference organisations

Belgium

- Ressources: a federation of social economy companies active in reuse, repair and recycling in the territory of the Brussels-Capital and Walloon region.
- Bebat: Belgian PRO for waste batteries. It has several initiatives to increase collection of batteries, in particular targeting schools: http://www.bebat.be/fr/programmescolaire and http://www.rtl.be/plugrtl/page/la-grande-recolte-inter-ecoles-avec-bebat-et-plug-rtl/1158.aspx#concept. It also provides collection infrastructure and service to bigger generators of waste batteries (stores, companies, schools): http://www.bebat.be/fr/dois-je-devenir-un-point-de-collecte
- Recupel: Belgian PRO for WEEE. It organises competitions for municipalities: http://www.recyclonsensemble.be/ and has a programme for separate collection of WEEE at retailer stores (together with Bebat): http://www.pointderecyclage.be/fr

France

- Eco-Emballages: French PRO for packaging. Provides a system of incentives to producers to improve the design of their products: http://www.ecoemballages.fr/bienvenue-dans-votre-espace-entreprises
- Ecofolio: French PRO for graphic paper. It supports municipalities in order to improve separate collection performances locally: http://www.ecofolio.fr/collectivites/accompagner-changement
- Eco-Systèmes: French PRO for WEEE. It applies a system of eco-modulated fees according to a set of environmental criteria: http://www.eco-systemes.fr/partenaires-et-professionnels/producteurs/comprendre-la-modulation

Italy

- CONAI: the Italian National Consortium for packaging recycling (umbrella PRO for packaging) provides support and co-financing to municipalities for improving the separate collection of packaging waste: http://www.conai.org/enti-locali/sostegno-alla-raccolta-e-al-riciclo
- Comieco: PRO for paper and cardboard packaging organises several initiatives for promoting paper collection in municipalities – such as an annual prize: http://www.comieco.org/cartoniadi/ and co-financing of municipalities for

improving paper collection: http://www.comieco.org/il-nostro-ruolo/l-attivita-dei-convenzionati/news/bando-comieco--anci-2016.aspx#.WAhs0cm9Q5x

Spain

- Eco-Embes: the PRO for packaging recycling provides guidance on effective communication for municipal technicians: https://www.ecoembes.com/es/planeta-recicla/tag/manual-tecnico-de-comunicacion-efectiva

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4.7. BEMPs on waste treatment

4.7.1. Sorting of co-mingled light packaging waste to maximise recycling yields for high-quality output

Summary overview

When light packaging waste (i.e. packaging made of plastics, composites, aluminium and steel, sometimes also including fibres (paper and cardboard)) is collected together (co-mingled), it is BEMP to implement advanced sorting of the co-mingled packaging waste in materials recovery facilities (MRF).

A typical state-of-the-art plant has five main technical sections:

- Feeding and preconditioning: this includes opening bags and feeding a constant flow of input material.
- Pre-sorting: this involves removing unsuitable items.
- Sorting: this includes several steps, e.g. separating fibre from containers; sorting fibre; sorting metal containers by using magnets, eddy currents or X-ray; first sorting of plastic containers by polymer (e.g. separation of PET bottles from other plastic containers).
- Refining: this consists of additional sorting steps, such as further sorting of polymers by type (e.g. HDPE, PP) and colour in order for the material output quality to meet market requirements. Quality control is performed by automatic or manual sorting.
- Product handling: this section consists of the baling processes and product storage as bales, loose material or in containers; product handling can also include loading operations for further downstream processes.

As MRFs tend to receive and sort materials from different local collection schemes, with varying compositions, a state-of-the-art MRF must have the flexibility to efficiently accommodate these variations.

Waste management area

Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability

In principle, there are no barriers to building and operating a packaging waste sorting plant. However, careful planning (especially considering the collection schemes in place, the plant capacity and the availability of markets for the sorted materials) is required as part of an integrated waste management concept. An important factor that needs to be determined is the optimal plant capacity. Finally, the impurity rates of co-mingled light packaging waste delivered to the plant affect its operations, performance (e.g. plant sorting rate) and economics (e.g. processing costs, revenues from recyclable fractions).

Specific environmental performance indicators

- Plant sorting rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of co-mingled packaging waste processed.
- Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of co-mingled packaging waste processed.
- GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (Scope 1 and 2) of the plant divided by the quantity of co-mingled packaging waste processed.

Benchmark of excellence

- Material recovery facilities sorting co-mingled light packaging waste have a plant sorting rate of at least 88 %.

Description

In many parts of Europe, packaging waste (i.e. packaging made of plastic, composites, aluminium and steel, sometimes also including paper and cardboard) is collected together in order to ease the waste separation task for consumers and to reduce collection costs.

When that is the case, in order to enable a high level of recycling, an advanced sorting of the co-mingled packaging waste in a material recovery facility (MRF) can be considered best practice. This BEMP deals with the sorting of co-mingled recyclables, including or excluding paper/cardboard. A number of technologies (e.g. NIR (near-infrared), multi-sensor systems, ultrasonic or VIS-camera, magnetic and/or air separation) are used for sorting and achieving the high level of segregation that allows recycling of a very high share of the mixed packaging waste collected from households.

There is a large variation in MRF plant design and process configurations owing to regional differences such as inflowing waste compositions, plant size, availability and cost of manual labour and regulatory frameworks. Moreover, relevant differences exist based on the inclusion or exclusion of fibres (paper and cardboard) and the types of plastics managed. In general, it is observed that large plants with treatment capacities of more than 75 000 t/year are the best performing ones, as they reach the economies of scale needed for investing in the most advanced sorting technologies (Cimpan et al., 2015, 2016; WRAP, 2007).

Despite the significantly different process layouts, different sections or modules in the plants, that have a standard main function, can be identified (Cimpan et al., 2016; WRAP, 2006). On the basis of this main function, five main technical sections can be identified in a typical state-of-the-art plant:

- *Feeding and preconditioning*: this section consists of reception (unloading) and storage of input materials and input feeding and preconditioning processes, such as bag opening and metering the flow of materials. The objective of this stage is to open and empty bags, loosen up recyclables and produce a constant and even flow of material into the process.

- *Pre-sorting*: this section consists of removing those products not intended for recycling, such us oversized items, unrecyclable contaminants, recyclable materials which the sorting system is not designed to segregate, or other items that might otherwise hinder sorting activities downstream, such as plastic film or oversized cardboard.
- *Sorting*: this section consists of primary sorting processes, which first separate the material flow per groups or types (two-dimensional fibre streams from three-dimensional container streams), followed by advanced sorting steps that continue the sorting process by further size segregation, isolating in each flow the different valuable fractions (paper by fibre grade, containers by material type, etc.).

  Typical sorting steps and related equipment are (WRAP, 2006; WRAP, 2007; Titech, 2011):

  - separating fibre streams (i.e. paper, card, cardboard) from container streams (i.e. cans, plastic bottles and other containers, etc.) using disc screens or trommel screens;
  - sorting fibre into its various grades (old corrugated cardboard, newspapers and magazines, mixed papers) using disc screens or more advanced optical scanners (NIR (near-infrared) sensors);
  - sorting metal containers using magnets for sorting steel or eddy current separators for sorting aluminium or X-ray sorting technologies to distinguish metals based on their density;
  - sorting plastic containers into a wider range of polymers (typically HDPE and PET) using optical scanners (NIR sensors).
- *Refining*: this section consists of additional sorting steps, such as sorting polymers by type (e.g. PP, LDPE) and colour using optical sensors (VIS-camera, NIR sensors), which aim to bring the material output quality to market requirements. Quality control is performed by automatic or manual sorting.
- *Product handling*: this section consists of the baling processes and product storage as bales, loose material (sorting residues) or in containers (metals). This section includes loading operations for products and residue streams to be delivered to downstream processes.

A detailed description of the plant design and process configuration in the most advanced and efficient sorting plants is provided in the operational data section of this BEMP.

As MRFs tend to receive and sort materials from a variety of different local collection programmes, which can collect different materials or the same materials in a different manner, a state-of-the-art MRF must have sufficient flexibility to efficiently accommodate these variations. This can be achieved by having adequate in-feed lines, i.e. different points in the overall sorting process where various materials may enter the system. This avoids the costs of passing the materials already sorted prior to delivery to the MRF through unnecessary sorting stations (WRAP, 2006). The plant flexibility is also important because the composition of collected co-mingled packaging waste is continuously changing due to evolving production and consumption patterns (e.g. reduction of paper use), new material use (e.g. bio-plastics) and even changes in regulation frameworks (addition of new materials or products admitted in the co-mingled streams), which also requires continuous development in sorting technologies.

Achieved environmental benefits

The sorting of co-mingled packaging enables the recycling of plastic, paper/cardboard, ferrous metals and non-ferrous metals. Thus, the material cycle can be closed, with significant savings in terms of primary raw materials and energy consumption and CO_2 emissions.

There is a lot of literature about the evaluation of the environmental benefits of recycling, mainly based on the application of LCA methods (Hogg D. et al., 2015; Bianchi D., 2012), but there is a lack of comprehensive studies focused only on the environmental benefits of material recovery facilities, considering the different types of existing MRFs and comparing their environmental benefits with those of other treatment alternatives.

Some scientific and grey literature exist related to the analysis of specific case studies with an LCA approach (Palm D., 2009; Carré A., 2015; Krones J. et al., 2012), and a more comprehensive study, although focused only on Portugal, Belgium and Italy (Lombardy), was developed by the European Investment Bank (2014) as part of the EIMPack – Economic Impact of the Packaging and Packaging Waste Directive. In order to provide some figures about the environmental benefits of this BEMP, below the results observed in this last reference study are described.

The LCA methodology applied in the European Investment Bank study was developed according to the ISO 14040:2006 requirements and was carried out focusing on the end-of-life of packaging, considering within the system boundaries waste packaging collection, sorting in MRFs, transport of waste to recyclers, the recycling process itself and the savings in terms of consumption of energy and raw materials from the recycling process (expanded boundaries). The functional unit of the LCA study is one tonne of municipal packaging waste managed by each Green Dot Company (i.e. SPV, Portugal; Fost Plus, Belgium; and Conai, Italy) in the year 2010. The assumptions adopted include the following aspects:

- The secondary materials produced through the recycling of packaging waste replace the corresponding primary materials (i.e. those produced from virgin raw materials), assuming a substitution ratio of 1:1 for all packaging materials except paper and cardboard packaging, for which a substitution ratio of 1:0.83 was assumed, because the paper fibres degrade in the recycling process, so they cannot be reused indefinitely. The savings in energy, raw materials and emissions released from the avoided production were considered in the recycling process.
- The electricity produced from the landfill gas (LFG) for the Portuguese case and from the waste incineration for the three case studies is supposed to substitute the same amount of electricity produced in each country (considering the different energy sources). This energy corresponds to the real energy mix production in 2010 (average approach).
- The sorting processes were modelled considering the main collection and sorting schemes in place in the different countries. In Portugal, the sorting processes were modelled for paper/cardboard, plastic, metal and drink packaging. For glass packaging, only separation efficiency was taken into account. In Belgium, the

sorting process was only considered for the mixed flow (plastic, metal and drink packaging) since paper/cardboard and glass packaging waste is sent directly to the recyclers/reprocessors. In Italy, only the sorting of ferrous metals and the separation of the multi-material fraction were modelled. The sorting processes were modelled based on the main consumption levels (electricity, diesel, etc.) related to the operation and considering the rejected material (quantities and final disposal).

In the study, besides the CO_2 emissions (Climate change indicator), other impact categories of the LCA method were considered: Photochemical oxidant formation, Eutrophication, Human toxicity and Acidification. Two different scenarios were analysed and compared:

- The real scenario in 2010 (hereinafter called "Recycling scenario"), where packaging waste was selectively collected, sorted and sent for recycling (i.e. in this scenario, they considered the 2010 recycling level in each country).
- A hypothetical scenario (hereinafter called "Non-Recycling scenario"), where packaging waste would be collected as residual waste (in the refuse collection circuit) and sent for incineration and/or landfill. Note that in Belgium and Italy (more specifically the region of Lombardy) only incineration was considered in this alternative scenario.

The total environmental impacts resulting from each scenario for the three countries are shown in Table 4-22. As expected, the current "Recycling scenario" proved to be more environmentally friendly than the "Non-Recycling scenario" for the three case studies. Regarding the GHG emissions, in 2010, the "Recycling scenario" saved between 14.3 Mt/year of CO_2e in Portugal, 516 Mt/year in Belgium and 643 Mt/year in Lombardy. It should be noted that in Lombardy the "Non-Recycling scenario" showed good results for the environment in contrast to the other two countries due to the incineration process with energy recovery. The large difference observed between Portugal and the two other countries was due to the recycling of paper/cardboard. In Portugal, the primary pulp production (replaced by the recycling fibres) generates electricity from by-products (biomass, black liquor, etc.) of the process. The pulp and paper production is self-sustainable in terms of energy with a surplus that is introduced into the National Grid. This surplus of electricity is accounted for as a benefit lost with recycling since this activity only consumes energy. In Belgium and Italy, the primary pulp is imported and information about the quantity of electricity generated during the pulp production process is not available. The pulp production process figures existing in the Ecoinvent 2.2 database of SimaPro was assumed as the avoided product in the paper/cardboard recycling process. The surplus of electricity generated in the avoided product was excluded as a simplification of the problem.

Table 4-22. Total environmental impacts of each scenario for the three case studies, considering the LCIA methods used for the Eco-costs 2012 valuation[56]

[56] The study use different Life Cycle Impact Assessment (LCIA) techniques and different weighting sets for the several environmental impact categories analysed which has an important impact on the overall results. Here we only refer to the results obtained applying the Eco-costs2012 method.

Impact category	Unit	Portugal	Belgium	Italy (Lombardia region)
Recycling scenario				
Climate change	kg CO_2e	-1.43E+07	-5.16E+08	-6.43E+08
Human toxicity	CTUh	-2.10E-01	-1.05E+01	9.53E+01
Photochemical oxidant formation	kg NMVOC	-4.07E+05	-2.82E+06	-4.25E+05
Eutrophication	kg Peq	-4.91E+03	-7.12E+04	-3.59E+05
Acidification	kg SO_2e	-1.11E+06	-4.57E+06	-3.23E+06
Non-Recycling scenario				
Climate change	kg CO_2e	7.46E+08	8.35E+08	-2.38E+08
Human toxicity	CTUh	1.04E+00	3.93E+00	3.73E+01
Photochemical oxidant formation	kg NMVOC	1.84E+05	3.39E+05	-2.11E+05
Eutrophication	kg Peq	3.72E+04	6.55E+02	-1.77E+05
Acidification	kg SO_2e	1.13E+05	1.60E+05	-1.65E+06

Source: European Investment Bank, 2014

Appropriate environmental indicators

The sorting rate achieved at the plant scale by the MRF is a good indicator for evaluating the environmental performance of the sorting process of co-mingled packaging. The indicator must be calculated at the plant scale and could be simply expressed in terms of percentage of recovered materials sent for recycling from the co-mingled packaging input flow over the year, calculated as follows:

$$\text{Plant sorting rate (weight \%)} = \frac{\text{recovered materials sent for recycling (total weight per year)}}{\text{co-mingled packaging waste processed (total weight per year)}} (\%)$$

The environmental performance of the sorting process also depends on the quality of the materials recovered, which determines whether they are suitable for reprocessors to produce higher quality recyclate. The more material grades are recovered with the minimum of contaminants, the higher the product quality that can be achieved in the recycling industry. Therefore the sorting rate indicator must also be complemented at least by the percentage breakdown of the materials sent for recycling per material grade.

For each material recovery facility, the material recycling indicators can thus be expressed as in the example provided in Table 4-23.

Table 4-23. Example of sorting rate indicators for co-mingled packaging in a MRF

Main flows	Tonnes per year	Sorting rate (%)	Recovered material grades	% in recovered material
Co-mingled packaging waste	100 000	90	Cardboard	22.25
			Newspaper & magazines	19.95
			Mixed paper	27.27
			HDPE clear bottles	2.58
			HDPE coloured bottles	1.72

Main flows	Tonnes per year	Sorting rate (%)	Recovered material grades	% in recovered material
			LDPE film	1.44
Sorted materials sent for recycling	90 000		PET clear	3.01
			PET coloured	2.15
			Polypropylene	1.15
			Mixed plastic	3.59
			Ferrous metal	3.59
			Aluminium	1.29

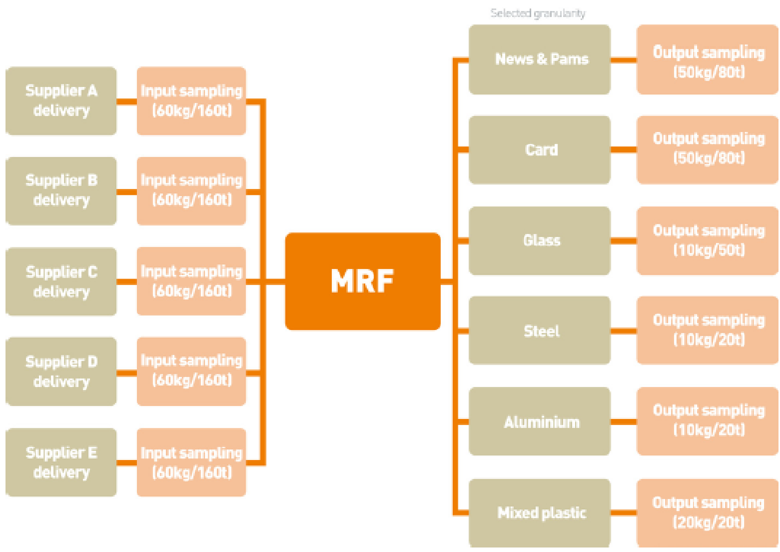
Source: own elaboration

The sorting rate achieved by the MRF also depends on the quality of input materials coming from the dry recyclable collections and on the material-specific recovery rates (i.e. the percentage of contaminants in each specific recovered material grade). The performance of the sorting processes would be better described using more detailed data provided by the MRFs by applying specific sampling and testing procedures for assessing the composition of their input and output materials, which would also allow the sorting efficiency to be assessed by type of material (paper, metal, plastic). This requirement is also regulated by law or by the extended producer responsibility frameworks in place for packaging waste in the EU Member States, and reference guidelines are also available for defining adequate sampling methods. For instance, in UK guidance (WRAP, 2014; SEPA, 2015), minimum requirements for sampling and reporting the MRF sorting efficiency are defined as follows:

- Input material (co-mingled packaging waste received for sorting) must be sampled to identify the types of target[57], non-target and unrecyclable materials. Target material must, as a minimum, be separately identified by reference to glass, paper, metal and plastic.
- Output material must be sampled with consideration given to the grade of material output from the MRF (e.g. paper grades may be cardboard, newspapers and magazines and mixed paper; metal grades may be steel and aluminium; glass grades may be mixed coloured glass containers, mixed coloured glass aggregate, green glass, clear glass, brown glass; plastic grades may be HDPE bottles, PET bottles, polypropylene, mixed plastic, etc.).
- The number of samples a MRF needs to take depends on the overall weight of input material delivered by each supplier and how much output material is produced. In order that an appropriate sampling schedule can be established, the weight of input material (by supplier) and weight of output material need to be monitored and recorded. The specific sample weights and sampling frequencies required are also defined according to the output material's grade (see example of sampling requirements defined by the Scottish Environment Protection Agency (SEPA) in Figure 4-57).
- Suitable minimum methods and adequate equipment must also be applied by MRFs for testing (i.e. sorting and weighing) sampled materials. For each sample, the

[57] A material that is specifically targeted by the MRF licence or permit holder as destined to be separated out from other materials to facilitate its recycling (SEPA, 2015)

percentage breakdown of the input flow per type (target, non-target and unrecyclable materials) and of the output flow per material grade must be registered. These data are then processed by calculating as key statistics the mean and standard deviation of the sampled target materials.



Source: SEPA, 2015

Figure 4-57. MRF sampling requirements set by SEPA until 1 October 2016

Based on this sampling and testing procedure, the sorting rate achieved by a MRF sorting lightweight packaging waste can be expressed as in the example provided in the next table.

Table 4-24. Example of indicators describing the sorting efficiency of MRFs based on the implementation of sampling and testing procedures

Input material	
Material type	**% of waste received**
Target material in input flow	88
of which Paper	*65*
of which Plastics	*17*
of which Metals	*6*
Non-target recyclable material	8
Unrecyclable material	4

Output material		
Material type	**Mean average % targeted material in output**	**Output material grades sorted**
Paper	97.7	- Cardboard - Newspapers and magazines - Mixed paper
Plastics	95.3	- HDPE clear bottles - HDPE coloured bottles - LDPE film - PET clear bottles - PET coloured bottles - Polypropylene (PP)

		- Mixed plastic
Metals	96.2	- Ferrous metal - Aluminium

Source: own elaboration

Considering that the sorting process requires significant energy consumption, and also taking into account EMAS core environmental performance indicators, other appropriate indicators for describing the MRF performance are energy efficiency and GHG emissions.

As for energy efficiency, the indicator can be expressed in terms of the annual specific consumption of energy (kJ per tonne of input waste) and can be calculated as follows:

$$\text{Energy efficiency (kJ/t)} = \frac{\text{total energy consumption (kJ per year)}}{\text{co-mingled packaging waste processed (total weight per year)}}$$

This can be complemented by also considering the total energy consumption (%) of energy produced by the organisation from renewable energy sources.

As for GHG emissions, the indicator can be expressed in terms of annual emissions of greenhouse gases, expressed in tonnes of CO_2 equivalent per tonne of input waste as follows:

$$\text{GHG emissions (t } CO_2\text{e/t)} = \frac{\text{total } CO_2 \text{ equivalent emissions (total weight per year)}}{\text{co-mingled packaging waste processed (total weight per year)}}$$

The CO_2 equivalent emissions are calculated according to the GHG Protocol Corporate Standard (World Resources Institute and World Business Council for Sustainable Development, 2004), adopted as the basis of ISO 14064, and refer both to the direct greenhouse gas emissions produced by the plant operation (scope 1 according to the reference methodology) and to the indirect emission savings related to the substitution of raw materials with secondary material (scope 2 according to the reference methodology, in order to measure the achieved environmental benefits as described in the previous section).

It must be noted, finally, that other environmental performance parameters will also be measured at the plant scale, as defined in the plant permits and in the related monitoring plan, according to the national and regional regulations.

Cross-media effects

As stated in the previous section, the operation of the MRFs is associated with energy consumption. Reference figures for this impact category are provided in the article by Cimpan et al. (2016) who developed a model simulating the technical and economic performance for MRFs sorting lightweight packaging waste, i.e. a material mixture with a high content of plastics (around 50 %) consisting of a mix of different packaging polymers, ferrous and non-ferrous metals, a paper and cardboard packaging fraction, beverage cartons and other composite packaging. Four different plants were modelled in the study, as shown in Table 4-25, reflecting clearly the MRFs operating in Germany but representative also of the plant operating conditions in other EU countries.

Table 4-25. Main MRF process parameters for the plants modelled in the reference study

Specification	Basic	Medium	Medium plus	Advanced
Planned processing capacity (t/year)	25 000	50 000	75 000	100 000
Working days (days/year)	250	250	250	250
Shifts and hours per shift (shift/day; hours/shift)	2;8	2;8	3;8	3;8
Operational hours (hours/year)	4 000	4 000	6 000	6 000
Plastic sorting – products	Plastic film, Mixed hard plastics	Plastic film, PE, PP, PET	Plastic film, PE, PP, PET, PS	Plastic film, PE, PP, PET, PS, PET bottles
Processing technology	Only essential material conditioning steps (sieving and air classification), heavily reliant on manual sorting	Comprehensive conditioning (several sieving steps, air classification and ballistic separation), both automatic and manual sorting, mostly manual product quality control	Almost identical to the medium plant with more extensive plastic sorting	State-of-the-art process design and technology, almost entirely based on automatic sorting, both automatic and manual product quality control

Source: Cimpan C. et al., 2016

Modelling results indicate that the average consumption of electricity to process one tonne of lightweight packaging waste amounts to about 100 kWh and more than two thirds of this amount is connected to sorting and refining steps. The estimated consumption levels for the four MRF types, in the different processing steps, are shown in Table 4-26.

Table 4-26. Yearly total and specific (per tonne) consumption of electricity and diesel for the plants modelled in the reference study

	Basic		Medium		Medium plus		Advanced	
	Electr. (MWh)	**Diesel** (10^3 L)	**Electr.** (MWh)	**Diesel** (10^3 L)	**Electr.** (MWh)	**Diesel** (10^3 L)	**Electr.** (MWh)	**Diesel** (10^3 L)
Per tonne input	**102.4 (kWh)**	**3.7 (L)**	**89.8 (kWh)**	**2.2 (L)**	**91.3 (kWh)**	**2.2 (L)**	**96.5 (kWh)**	**2.2 (L)**
Total per year	**2 560**	**91.9**	**4 488**	**110.3**	**6 847**	**165.4**	**9 649**	**222.7**
Feeding and preconditioning	95	35.4	206	42.4	309	63.6	1013	106.0
Conditioning	201	0	389	0	585	0	790	0
Sorting	924	0	1 319	0	2 094	0	2 794	0
Refining	874	0	1 950	0	2 925	0	3 383	0
Product handling	217	56.6	289	67.9	435	101.8	863	116.7
Unassigned	250	0	333	0	499	0	807	0

Source: Cimpan C. et al., 2016

The lowest specific consumption levels are observed in the "medium" plant, while they increase in the "medium plus" and "advanced" configurations as one of the processes that contributes most to electricity consumption is the production of compressed air for the NIR sorters, which are especially large for the "advanced" plant. A consumption level of around 3 L of diesel per tonne of input waste is also identified in connection with mobile equipment in the plants, i.e. the equipment used to move waste on the tipping floor and bales of recovered material (e.g. front-end loaders, forklifts, polyp excavators). Diesel consumption levels are a bit higher (3.7 L/t) in the "basic" plant than in the more automated ones (2.2 L/t).

Besides energy consumption, emissions of dust and odour can also occur but do not appear to be significant for MRFs, as only dry recyclables are sorted in these facilities. In any case, adequate emission abatement technologies must be considered given the potential presence of biowaste residues or dusty materials. Drainage infrastructure of the tipping floor and storage areas and adequate treatment of the collected waste water must also be foreseen. The safety and health of workers performing manual sorting have to be assured, with special regard to their exposure against airborne fungi, bacteria and other biological agents.

Operational data

As already highlighted, the MRF design and process configuration can vary significantly owing to multiple factors. Considering state-of-the-art lightweight packaging MRFs, the most relevant differences are in any case determined by the waste input flow, with particular reference to the inclusion or exclusion of fibres.

Operational data about lightweight packaging MRFs are mainly derived from the scientific articles by Cimpan et al. (2015, 2016), focused on the performances of this type of sorting plant with different plant configurations, as described in Table 4-25. Data and information reported have also been verified and complemented through direct contacts and information related to the reference organisations.

Besides the differences in the process flow and in the related plant layout, all MRFs apply, in different configurations, similar equipment and processes, briefly described in Table 4-27.

Table 4-27. Description of the typical equipment and processes applied in MRFs

Equipment/ Process	Description
Rolling stock/Mobile equipment	Non-stationary equipment typically used to move waste on the tipping floor and bales of recovered material (e.g. front-end loader, forklift, polyp excavator)
Drum feeder	Opens bags and puts material on initial conveyor at a nearly constant rate
Disc screen	An inclined plane filled with a series of parallel rods with discs spread along each rod such that large materials travel over the top while smaller materials fall between the discs
Trommel screen	Removes smaller materials via a rotating cylindrical screen
Air (or aeraulic) classifier	Separates light materials from heavy materials via high-pressure air; it can be vertical or horizontal (also called air knife)
Ballistic separator	Separates materials having different physical characteristics (weight, shape, surface) that assume different trajectories passing through a series of parallel paddles, following the orbital movement of the paddles
Eddy current separator	Uses magnetic fields to remove aluminium and other non-ferrous metals
Magnetic separator	Separates ferrous metals by means of a magnet
Optical sorter	Identifies predetermined material(s) using optical technology (e.g. cameras, lasers, NIR sensors) and removes the identified material from the stream using a burst of compressed air
Baler (one-way)	Compresses material (typically fibre) in one direction

Equipment/ Process	Description
Baler (two-way)	Compresses material (typically containers) in two directions
Positive sort	Sorting of recyclable materials (the intended materials in the output flow)
Negative sort	Removal of undesirable materials (contaminants)

Source: Pressley P.N. et al., 2014; Cimpan C. et al., 2016; JRC, 2015; WRAP, 2006

Regarding the lightweight packaging MRFs, one main characteristic of these plants, as they have evolved to handle large volumes of very light waste, is that early in the process the input stream is split into selected size intervals, which are subsequently processed on individual sorting lines. State-of-the-art plants have a total of up to 20 NIR sorters and multi-sensor systems which are commonly used for specific tasks (combining NIR, colour or induction sensors). Some of these plants use additional sensing equipment for material and process surveillance. For this purpose, ultrasonic or VIS-camera-based volume flow measurement devices are in use, which helps the plant operator to react to changes of the volumetric flow in the plant setup. Despite the high level of automation, these installations need to be complemented with some manual quality control in order to correct systematic sorting errors and carry out some refining tasks before products are ready for the market. A detailed process flow diagram for this MRF type is provided in Figure 4-58

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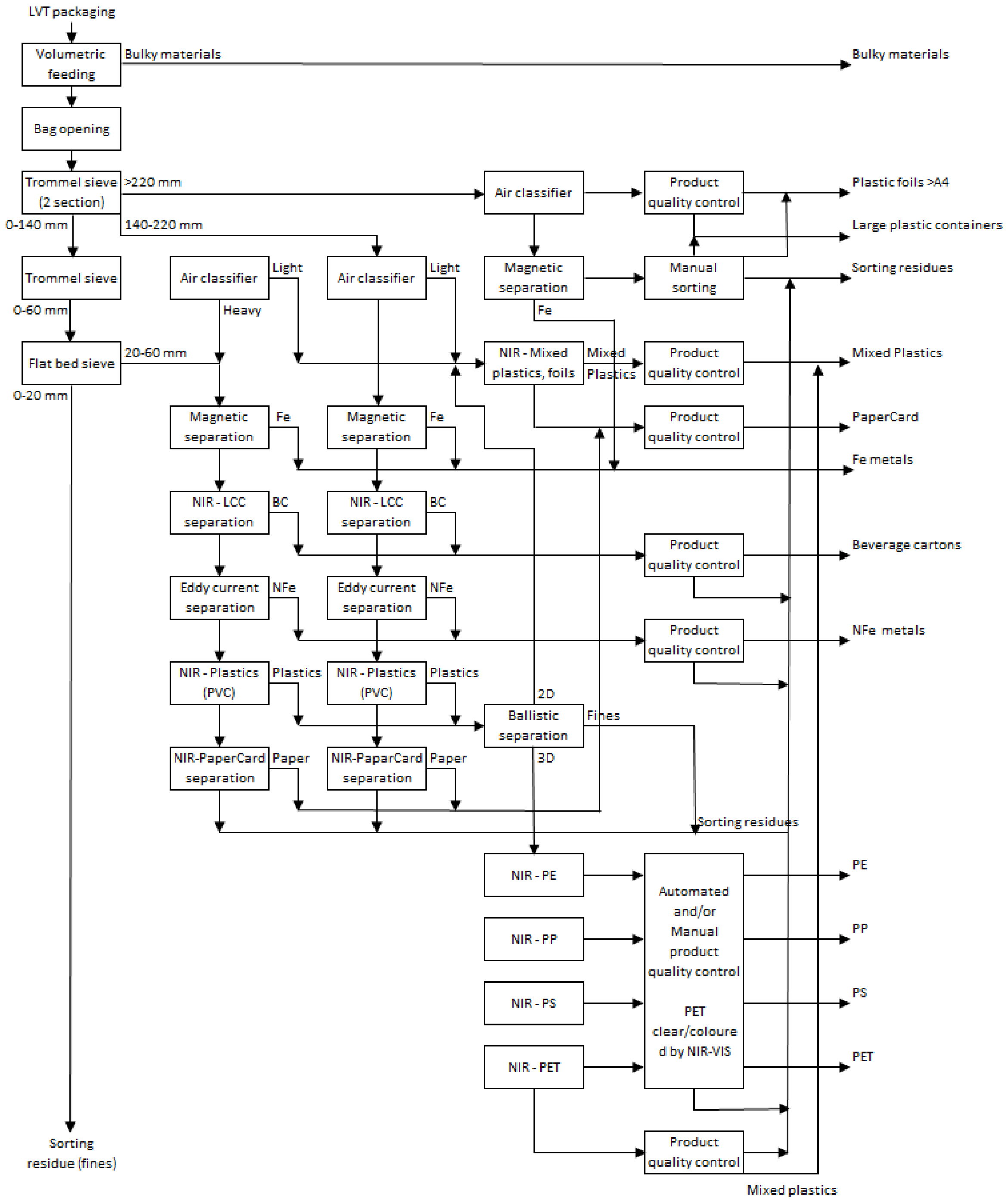

Source: Cimpan et al., 2016

Figure 4-58. Process flow diagram of a state-of-the-art lightweight packaging MRF

The first processing step is always bag opening coupled with volumetric dosage feeding. The materials then undergo conditioning steps for size classification, performed with drum screens (trommels) with one or two functional separation cuts, which are then sent to different air classifiers, followed by magnets, optical sorters and eddy current separators. The mixed plastics stream can be further conditioned, typically by using ballistic separators to remove fines and any remaining two-dimensionsal material, before it enters the polymer sorting block. Here, plastics are sorted in a cascade by polymer type in the four standard packaging polymers, i.e. PE, PP, PET and PS. Individual sorted polymers can undergo a second automatic "cleaning" step, or be refined by automatic colour sorting (PET and HDPE). The leftover plastics, after polymer sorting, will typically constitute a mixed polymer product. However, another sensor unit can be used to pick out remaining/missed valuable polymers and recirculate them to the start of the polymer sorting process, thus increasing recovery rates. After the automated sorting steps, some manual product quality control and sorting steps are also needed in order to correct systematic sorting errors and carry out some refining tasks before products are ready for the market

The degree of automation of the MRF is linked to its capacity, as discussed in detail in the Economics section. The processing lines, type and number of pieces of processing equipment per MRF section in lightweight packaging MRFs with different plant capacities (as modelled by Cimpan C. et al., 2016, and described in Table 4-25) are reported in Table 4-28.

Table 4-28. Processing lines, type and number of pieces of processing equipment per MRF section

Plant section	Equipment category	Equipment type	Basic	Medium/ Medium plus	Advanced
Number of processing lines			3	4	5
Feeding and preconditioning	Processing equipment	Feeding equipment	Screw feeder (including bunker)	Screw feeder (including bunker)	Shredder and screw feeder
Conditioning	Mobile	Conveyors	1	3	3
		Wheel loader	1	1	1
	Processing equipment	Screening	1	3 (2 trommel screens + flat bed sieve)	3 (2 trommel screens + flat bed sieve)
		Air classifier	1	2	3
		Ballistic separator	-	1	1
		Conveyors	5	12	16
Sorting	Processing equipment	Magnetic separation	2	3	4
		Eddy current	2	2	3
		NIR sorter	2	8/9	11
		Conveyors	14	26/28	37
	Manual sorting	-	4	2	2
Refining	Processing equipment	Air classifier	-	-13	2
		NIR sorter	-	-	5
		Conveyors	2	4	12
	Manual quality control		5	12/13	7
Product handling	Processing equipment	Balers	1	1	2
		Conveyors	7	9/10	14
	Mobile equipment	Polyp excavator	1	1	1
		Forklift	1	1	2

Source: Cimpan C. et al., 2016

Reference data about the performance of lightweight packaging sorting plants, in terms of material recovery, quality of products and related reprocessing routes, are not well documented in the scientific and technical literature. Generic recovery efficiencies reported by Cimpan C. et al. (2015) are presented in Table 4-29.

Table 4-29. Material recovery rate in state-of-the-art lightweight packaging MRFs

Product	Sorting technology	Recovery yield (%)	Reprocessing route
Bulky materials (buckets/large cans)	Manual	-	Mechanical recycling
Ferrous metals	Magnetic separation	> 95	Steel industry
NF metals (Al)	Eddy current	60–90 (typically 80)	Pyrolysis and Al industry
Beverage cartons	NIR	90	Paper industry
Plastic foils > A4	Air separation, NIR, foil grabber	> 70	Mechanical recycling
Hard plastics (PE, PP, PS, PET)	NIR	70–90	Mechanical recycling
Mixed plastics	NIR	> 85	Mechanical recycling or energy recovery
Residues	-	-	Energy recovery

Source: Cimpan C. et al., 2015, based on Bünemann et al., 2011

Applicability

In principle, there are no barriers to building and operating a packaging waste sorting plant. However, careful planning (especially considering the collection schemes in place in the surrounding area, the plant capacity and the availability of markets for sorted materials) is required as part of an integrated waste management concept, including awareness-raising and information campaigns for citizens and efficient waste collection.

In this respect, an important issue that needs to be considered is related to the optimal plant capacity. This factor affects the overall MRF efficiency, as well as the specific processing costs (as explained in the Economics section), and must be carefully considered case by case, given the region/site-specific framework conditions. The following needs have to be considered in particular:

- the transport distances from collection areas to the sorting facilities: this would suggest keeping the treatment capacity low so that the facility serves a relatively small geographical area, thus allowing collection vehicles easy access to unload their materials during collection rounds;
- the economies of scale: this, on the other hand, would suggest keeping the treatment capacity high, so that investments in advanced sorting technologies are more feasible and allow the achievement of higher recovery rates with lower specific processing costs (EUR/t of input waste);
- the availability of manual labour or the will to create local jobs: this would influence the choice of manual versus automated sorting (i.e. low-capacity versus high-capacity plants respectively), although it should also be considered that manual sorting jobs imply difficult working conditions (noise, risk of injuries and infections, ergonomy);
- the need to avoid plants operating in overload capacity conditions: this would significantly reduce the sorting efficiency and increase the specific processing costs.

Economics

The economics of the different sorting systems vary widely depending on system specifics, such as location, size, whether they serve urban or rural communities and many other factors (Cimpan C. et al., 2015). Based on the available scientific and grey literature (ADEME, 2013; Cimpan C. et al., 2016; WRAP, 2007), an overview of the costs associated with co-mingled waste packaging sorting in MRFs is provided below, focusing in particular on the economy of scale benefits.

A study from ADEME (2013) shows that the average sorting cost in the MRFs operating in France, based on the costs observed in 112 sites, is EUR 163/t, with a high dispersion of values, ranging between EUR 100/t and EUR 220/t. Analysis of changes in sorting costs shows that many elements simultaneously influence such variations. The region where the MRF is located appears to be an important factor, related to the level of urbanisation and the resulting land pressure, with higher costs in the most urbanised regions. The collection scheme is also a factor influencing the sorting cost, which is lower for double-stream than single stream collections. The increase in the reject rate also appears to be correlated with the increase in the cost of sorting, while the simultaneous treatment in the same plant of municipal and commercial waste appears to be correlated with a decrease in the cost of sorting.

Valuable reference figures as to the economies of scale are provided by Cimpan C. et al. (2016), who evaluated the economic performance for MRFs sorting lightweight packaging (LWP) waste by modelling four plants of progressively higher capacity and technological level, as described above in Table 4-25. The method used was budget-based economic analysis, whereby only direct financial costs and benefits were counted. The analysis precludes taxes, subsidies and revenues from gate fees (based on contracts with Dual Systems Deutschland Gmbh). The cost categories included were: (1) specific processing costs (these relate only to the facility); (2) costs of output management (revenues/disposal cost); and (3) transfer and long-distance transport for the supply of LWP. The results obtained for the four MRFs modelled are shown in Table 4-30, Figure 4-59, Figure 4-60 and Figure 4-61.

Table 4-30. Model results: total capital and operational costs

Specification	Basic	Medium	Medium plus	Advanced
Capital investment				
Construction/building costs (EUR)	2 947 000	4 785 000	4 863 000	6 843 000
Processing equipment (EUR)	3 153 000	6 634 000	6 987 000	12 616 000
Mobile equipment (EUR)	638 000	693 000	693 000	1 067 000
Project costs (EUR)	203 000	364 000	377 000	616 000
Total capital investment (EUR)	**6 939 000**	**12 475 000**	**12 919 000**	**21 141 000**
Annualised capital expenditure (Capex)				
Construction/building costs (EUR/year)	237 000	384 000	391 000	550 000
Processing equipment (EUR/year)	409 000	860 000	1 074 000	1 681 000
Mobile equipment (EUR/year)	148 000	161 000	161 000	247 000
Project costs (EUR/year))	17 000	30 000	31 000	50 000
Total Capex (EUR/year)	**809 000**	**1 433 000**	**1 654 000**	**2 526 000**
Operational expenditure (Opex)				
Costs for repairs/maintenance (EUR/year)	138 000	236 000	245 000	392 000
Costs for resource consumption (EUR/year)	525 000	856 000	1 303 000	1 810 000
Costs for personnel (EUR/year)	1 297 000	1 838 000	2 732 000	2 379 000
Insurance (EUR/year)	43 000	78 000	80 000	120 000
Total Opex (EUR/year)	**2 003 000**	**3 006 000**	**4 358 000**	**4 700 000**

Specification	Basic	Medium	Medium plus	Advanced
Capex + Opex	**2 815 000**	**4 439 000**	**6 012 000**	**7 226 000**

Source: Cimpan C. et al., 2016

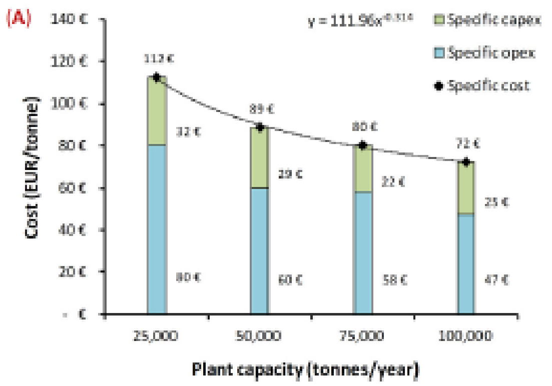

Figure 4-59. Specific processing costs, excluding costs for LWP waste supply and output management

Source: Cimpan et al., 2016

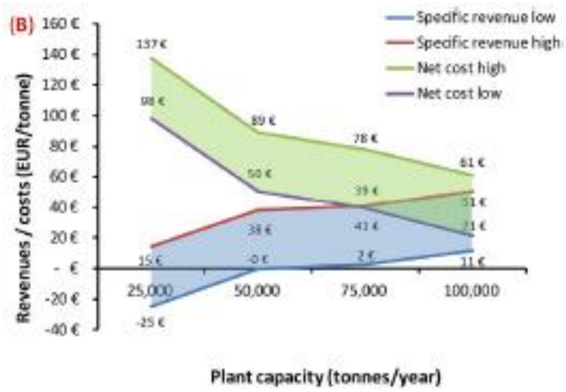

Figure 4-60. Net processing costs (Specific processing costs - costs of output management), considering low and high revenue values

Source: Cimpan C. et al., 2016

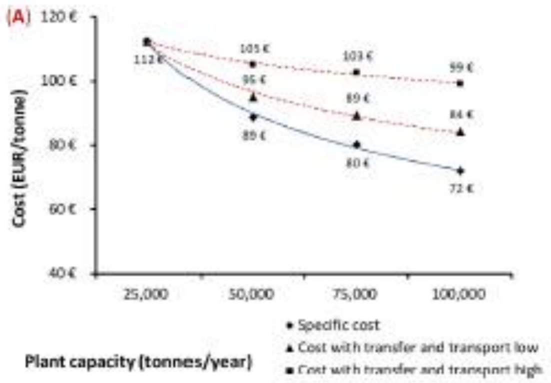

Figure 4-61. Specific processing costs with the addition of transfer and long-distance transport

Source: Cimpan C. et al., 2016

The results for <u>specific processing costs</u> illustrated in Figure 4-59 suggest that economy of scale effects do materialise in LWP MRFs, as shown by the cost of sorting one tonne of LWP which decreases from EUR 110 in the small-capacity basic plant to EUR 70 in the large-capacity advanced plant.

The effect on the specific processing <u>costs of the revenues from material sales and disposal costs</u> (costs of operational management) is illustrated in Figure 4-60. The analysis was carried out considering the range of market prices and waste disposal costs reported in Table 4-31 for each plant output. The lower/blue band in Figure 4-60 illustrates the interval of variation for costs/benefits pertaining to output management, with the lower and higher border lines reflecting the low and high price levels. The higher/green band illustrates the interval of variation induced by the output management on the calculated net costs. The conclusion that can be drawn is that LWP MRFs always incur net costs when solely the income of material sales is considered. This net cost then has to be balanced by the income from gate fees.

Table 4-31. Market values assumed for the MRFs outputs.

Output	Low level prices (EUR/t)	High level prices (EUR/t)
Plastic foils > A4	50	150
Large plastic containers/HDPE coloured	190	240
Paper/Card and composites	30	60
Ferrous metals	140	175
Non-ferrous metals	300	470
Beverage cartons	0	0
PET bottles	120	180
Standard packaging polymers (PP, PE, PS, PET)	100	120
Mixed plastics	-30	0
Sorting residues	-90	-50

Source: Cimpan C. et al., 2016

The possible increase in specific processing costs considering additional costs of transfer and long-distance transport is illustrated in Figure 4-61. These cost items assume relevance in determining the plant capacity, as it is estimated that a catchment area with around 800 000 inhabitants is adequate to provide the LWP waste input for the basic plant, whereas for the advanced plant, a catchment area of over 3 million inhabitants is required. For high-capacity plants, this means that additional costs relating to transfer stations and long-distance transport become important factors in the economics of sorting. As in the case of output management costs, for the transfer and transport costs different cost ranges have also been considered in the study, as shown in Table 4-32. The new cost curves in Figure 4-61 still indicate economies of scale with increasing plant size, although these appear to become very small with the high cost for transfer. This result emphasises the importance of accounting for necessary transfer and transport when large plants are planned. Although costs related to these operations are not necessarily incurred by the sorting plants, they do contribute to the overall waste management system costs.

Table 4-32. Costs associated with LWP waste transfer and long distance transport

Specification	Basic	Medium	Medium plus	Advanced
Transfer cost low (EUR/year)	0	125 000	250 000	375 000
Transfer cost low (EUR/year)	0	625 000	1 250 000	1 875 000
Transport >25 000 and < 50 000 t/year (EUR/year)	0	190 972	190 972	190 972
Transport > 50 000 and < 75 000 t/year (EUR/year)	0	0	254 630	254 630
Transport > 50 000 and < 75 000 t/year (EUR/year)	0	0	0	381 944
Total cost increase low (EUR/t)	**0**	**6**	**9**	**12**
Total cost increase high (EUR/t)	**0**	**16**	**23**	**27**

Source: Cimpan C. et al., 2016

In brief, the analysis carried out by Cimpan C. et al. corroborated the fact that LWP MRFs operate at an overall net cost, which has to be covered by the gate fees or sorting fees under any plant configuration, as the revenues from sales of recovered materials cannot fully cover the processing costs. The analysis also showed that strong capacity-related economies of scale occur with regard to processing costs and that the practical optimal capacity level is achieved at around 50 000 t/year, while optimal process efficiency, measured as total material recovery, is realised in large plants with high degrees of automation (>75 000 t/year), but is in all cases significantly dependent on operational practice.

These main results are also confirmed by other reference studies. In particular, WRAP (2006, 2007) has developed a MRF cost model that provides representative capital and operating costs involved in setting up and operating a MRF. A sample cost curve for MRF operations in the case of a single-stream MRF is presented in Figure 4-62.

The curve shows the variation in the unit cost per tonne for MRFs of different design capacities. It shows that the unit cost per tonne begins to level out at higher throughput tonnages (80 000–100 000 t/year) but rises significantly at lower throughput tonnages. Besides the differences in the specific processing costs simulated in the previous study, due to the different assumptions in the cost models, this result confirms that economies of scale can be realised by processing more recyclables at larger MRFs. The WRAP MRF cost model suggests indeed that MRFs below an annual capacity of 80 000–100 000 t will not achieve optimal operating costs. Facilities of this scale are needed to achieve economies of scale but also to justify investment in more automated and sophisticated sorting equipment that will help maximise the value of the recovered materials.

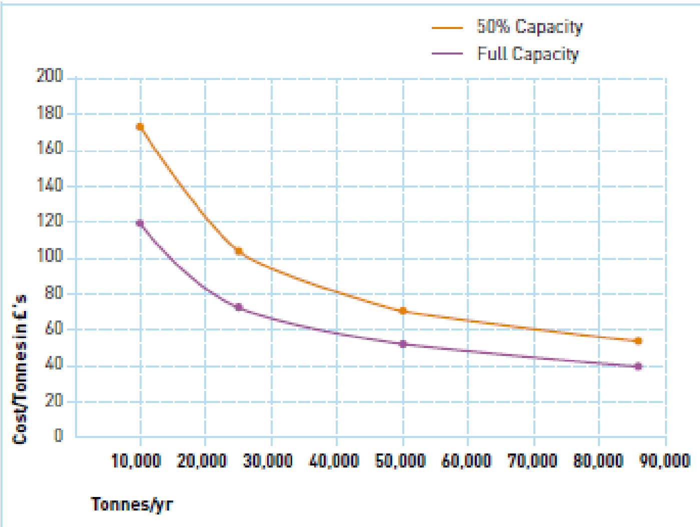

Figure 4-62. Specific processing costs in a fully co-mingled MRF

Source: WRAP, 2007

The cost curve in Figure 4-62 also shows the cost implications of operating a MRF at 50 % capacity (i.e. on a single-shift basis) compared to full capacity (i.e. a two-shift basis). The lowest cost curve for any particular MRF is that produced when the MRF is operating at full design capacity. Any reduction in throughput tonnage below that level increases the unit cost of processing.

Driving force for implementation

The European Packaging Directive (1994/62/EC; 2004/12/EC amended) has been the most important driving force for the implementation of this BEMP, as it introduced binding targets to collect, recover and recycle all materials used in packaging, including paper and cardboard, plastic, composites, aluminium and steel. Since then, most Member States have made major investments in packaging recycling systems. This has led to extended producer responsibility (EPR) regulations that ensure that manufacturers are responsible and have to bear the costs for the adequate treatment and recycling of packaging waste.

More recently, the implementation of the BEMP has been reinforced by the recycling target set by the EU's Waste Framework Directive (2008/98/EC), requiring that "by 2020, the preparing for reuse and the recycling of waste materials such as at least paper, metal, plastic and glass from households and possibly from other origins as far as these waste streams are similar to waste from households, shall be increased to a minimum of overall 50 % by weight". The new Circular Economy Package, which includes revised legislative proposals on waste, further reinforces this target introducing a common EU target for recycling 65 % of municipal waste and 75 % of packaging waste by 2030. Also, the targets for limiting waste landfilling and related increasing of landfilling costs are relevant drivers.

Other important drivers for the implementation of this BEMP, which in any case need to be substantially reinforced in EU countries, are the pull mechanisms for the creation of fully functional secondary raw material markets, such as economic instruments (i.e. tax reduction for companies producing recyclates, or lower taxes on products with recycled contents) or Green Public Procurement Policies (Plastics Recycling Europe, 2016).

Reference organisations

Based on available literature, in the UK (WRAP, 2009) there were 93 MRFs in operation in 2009, including both single-stream and dual-stream installations. In France (ADEME, 2013) 253 plants operate, 7 % of which (17 plants) are equipped with the most advanced sorting technologies. In Germany, the number of MRFs in 2011 was 92, but almost 90 % of the lightweight packaging collected was processed in less than 50 plants and, of these, 7 large plants were advanced sorting plants equipped with automated sorting of mixed plastics by resin type (Cimpan et al., 2015). Advanced MRFs are also operative in many other EU countries.

Examples of advanced MRFs identified are briefly described below:

- SUEZ MRF in Rotterdam, Netherlands: a technologically advanced sorting plant for co-mingled lightweight packaging (including plastics, metals and beverage cartons), with a treatment capacity > 17 t/hour, achieving a sorting rate of 89 % (3 % metals, 4 % beverage cartons, 45 % rigid PE, rigid PP, rigid PS, PET, PET film, 37 % mixed plastics). A video produced by the company is available at: https://www.youtube.com/watch?v=Xjot6NpySac.
- Alba MRF in Walldürn and Berlin, Germany: technologically advanced sorting plants for co-mingled lightweight packaging with respective treatment capacities of 170 000 and 130 000 t/year. A video of the Berlin plant produced by the company is available at: https://www.youtube.com/watch?v=CDGAhVb4r1w.
- Veolia MRF in Portsmouth, UK: a technologically advanced sorting plant for co-mingled lightweight packaging, applying an innovative sorting technology called

"magpie" which separates mixed plastic into different waste streams. A video produced by the company is available at https://www.youtube.com/watch?v=iKuiyY6x0cc.

- Invader MRF in Willebroek, Belgium: a technologically advanced sorting plant for co-mingled lightweight packaging (including plastics, metals and beverage cartons), with a treatment capacity > 10 t/hour, achieving a sorting rate of 86 % (27 % metals, 12 % beverage cartons, 47 % clear PET, blue PET, green PET, rigid PE, rigid PP). A video produced by the company is available at: https://www.youtube.com/watch?v=AfP32IyqBak.
- Hera Ambiente MRF in Granarolo, Italy: a technologically advanced sorting plant for co-mingled lightweight packaging, equipped with two different treatment lines (one for paper and cardboard and one for plastics and metals), with an overall treatment capacity of 100 000 t/year. More information is available on the company website: http://ha.gruppohera.it/plants/main_plants/page105-082.html.

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4.7.2. Processing of mixed plastic packaging waste to maximise recycling yields for high-quality output

<table>
<tr><th colspan="8"><u>Summary overview</u></th></tr>
<tr><td colspan="8">It is BEMP to process separately collected mixed plastic packaging waste in individual material streams that can be transformed into valuable high-quality secondary raw materials and recycled products. The process encompasses the following steps:
- sorting flexible plastic packaging waste from the rigid items (film sorting) by using film grabbers, air drum or ballistic separators followed by a manual quality assurance step;
- sorting plastic bottles and other rigid items by polymer and colour with optical sorting systems;
- reducing sorted film and residual rigid items (as separate flows) in flakes by using granulators;
- cleaning flaked plastic packaging using friction cleaning (dry or wet grinding systems);
- separating and washing flaked plastic packaging by polymer and colour by using optical sorting systems or density separation technologies;
- extruding flaked material into pellets.</td></tr>
<tr><th colspan="8"><u>Waste management area</u></th></tr>
<tr><td><u>Cross-cutting</u></td><td><u>MSW - strategy</u></td><td><u>MSW - prevention</u></td><td><u>MSW - collection</u></td><td><u>MSW - EPR</u></td><td><u>MSW - treatment</u></td><td><u>CDW</u></td><td><u>HCW</u></td></tr>
<tr><th colspan="8"><u>Applicability</u></th></tr>
<tr><td colspan="8">Good waste collection systems and the good quality of the collected materials need to be assured in order for the recycled output to be suitable for the market. Current market trends towards more complex multi-layer and multi-material plastic products also make mixed plastics sorting and reprocessing much more difficult. As with the previous BEMP, there are no general barriers to building and operating such a plant. However, careful planning and determination of the optimal plant capacity are important.</td></tr>
<tr><th colspan="8"><u>Specific environmental performance indicators</u></th></tr>
<tr><td colspan="8">- Plant processing rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of mixed plastic packaging waste processed.
- Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of mixed plastic packaging waste processed.
- GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (scope 1 and 2) of the plant divided by the quantity of mixed plastic packaging waste processed.</td></tr>
</table>

- Water use (m^3/t), calculated as the annual total water used on site divided by the quantity of mixed plastic packaging waste processed
Benchmark of excellence
- Plastic recovery facilities processing mixed plastic packaging waste have a plant processing rate of at least 60 %.

Description

The term "mixed plastic packaging waste" covers all plastic packaging waste sourced from the domestic waste stream and includes rigid and flexible plastic items of various polymer types and colours that are typically generated by households. It usually excludes non-packaging items, even though in some EU countries some non-packaging items are starting to be admitted in the recyclables collection bin. Based on available domestic waste assessments, the share of plastic in the household waste ranges from 8 % to 12 % by weight, and, assuming an average value of 9 %, bottle plastics account for 2.4 %, non-bottle plastics including films, bags and other packaging comprise 5.2 %, with the remaining 1.4 % represented by the non-packaging fraction (WRAP, 2008). When this collected waste stream enters material recovery facilities (MRFs), plastic bottles and containers of the different polymer types can be efficiently separated, but the rest of this flow (about 6.6 % of the total) largely remains as a residual component that is commonly sent to final disposal or energy recovery or, in the best case, at least partly down-cycled.

The major challenge in producing valuable recyclate (i.e. materials resulting from the processing of plastic waste, such as pellets, granules and flakes) from mixed plastic packaging waste is that most plastic types are inherently immiscible at the molecular level and have different processing requirements. Furthermore, when plastic is contaminated, or of a limited quantity or with a varied composition, recycling is more difficult. Therefore, to achieve efficient mechanical recycling, mixed plastic packaging waste should be processed as far as possible into clean single types (Plastics Recyclers Europe, 2013).

The route that household plastic waste takes for separation and reprocessing depends on how it is collected (WRAP & Zero Waste Scotland, 2012):

- Plastic packaging waste from multi-stream systems is typically baled at a transfer station or depot and sent directly to a so-called plastics recovery facility (PRF), i.e. a facility set up specifically to sort plastics by polymer type and/or colour.
- Materials collected co-mingled usually pass through a material recovery facility (MRF), where the co-mingled stream is separated into material types. Some MRFs also separate one or more of the more abundant and higher value plastic bottle streams, typically PET and HDPE, but most of them concentrate on separating mixed plastic bottles for further sorting at a specialist PRF.

In both cases, the different plastic packaging polymers sorted at the PRFs are sent to plastics reprocessors that convert them, through mechanical recycling steps, into raw materials (pellets, granules, flakes) that can be used to manufacture new plastic products.

There are six main polymer types in the household waste stream, which account for around 75 % of the demand from converters. Table 4-33 shows these main types and the related recycled products with well-established markets (Plastics Recyclers Europe, 2013).

Table 4-33. Main polymer types in the household waste stream and related recycled products

Polymer type		Recycled products
LDPE	Low-density polyethylene	Bin liners, carrier bags, agricultural film mulch, agricultural film sheet, construction film, tubes, cling film, flexible packaging, heavy-duty sacks
HDPE	High-density polyethylene	Tubes, sewer pipes, pallets, boxes, buckets, bottles for detergents, construction, food product packaging, toys, cable insulation
PP	Polypropylene	Pipes, pallets, boxes, buckets, furniture, car parts, pots of yoghurt/butter/margarine, fibres, milk crates
PS	Polystyrene	Clothes hangers
PVC	Polyvinyl chloride	Sewer pipes, window frames, construction, flooring, wallpaper, bottles, car interiors, medical products packaging
PET	Polyethylene terephthalate	Bottles, sheets, strapping (e.g. carpets, clothing automotive parts), food and non-food packaging, films and fibres

Source: Plastics Recyclers Europe, 2013

To recover high-quality recyclable materials from the household mixed plastic packaging waste and transform them into valuable raw materials and recycled products, the following sorting and reprocessing steps are needed (WRAP, 2008; Reclay StewardEdge, 2013):

1. sorting flexible plastic packaging waste from the whole rigid items (film sorting) by using film grabbers, air drum or ballistic separators followed by a manual quality assurance step;
2. sorting plastic bottles and whole rigid items by polymer and colour by using optical sorting systems;
3. reducing sorted film and residual rigid items (as separate flows) in flakes by using appropriate granulators;
4. cleaning flaked plastic packaging using friction cleaning (dry or wet grinding systems);
5. separating and washing flaked plastic packaging by polymer and colour by using optical sorting systems or density separation technologies;
6. extruding flaked material into pellets.

The first two sorting steps are generally performed at PRFs, but the most advanced MRFs also include mixed plastics sorting lines featuring partial sorting steps, as described in the BEMP related to the sorting of co-mingled light packaging waste. The other steps are commonly featured at plastic reprocessing plants, but some PRF operators have invested in downstream reprocessing capacity in order to produce high-grade recycled polymers (WRAP & Zero Waste Scotland, 2012). This BEMP deals with plastics sorting and processing facilities able to maximise the recovery of the different plastic materials collected from municipal solid waste, by polymer and colour, including films and flaked plastics, thus enhancing the overall recycling rates from the mixed plastic packaging waste flow.

Given the current state-of-the-art of mixed plastics sorting and reprocessing technologies, even in the most advanced PRFs some challenges still remain, mainly regarding two critical plastic fractions: black polymers and biodegradable plastics. Black or dark plastic packaging wastes are not recognised by optical sorters and generally end up in the residual waste fraction of the PRFs (WRAP, 2008a; Chacón et al., 2016), while the presence of biodegradable plastics in the sorted polymer flows could lower the quality of the recyclates (Plastics Recyclers Europe, 2013). Recent research has demonstrated that innovative technologies and equipment can effectively recognise and also sort these critical fractions (Hollstein F. et al., 2013; BP sorting, 2014; Filmsort, 2015; Chacón et al., 2016) and specific sorting machines are starting to become commercially available. Future perspectives as to the advanced sorting techniques are also offered by robotic technologies; these use multiple sensors and artificial intelligence software to monitor and analyse the waste stream in real time and industrial robotic arms to pick up waste fractions of various shapes and sizes quickly and accurately. Interesting robotic waste sorting solutions are already applied at the pilot scale (Sadako Technologies, 2016) or at the full industrial scale for particular waste flows (Zenrobotics, 2016).

Achieved environmental benefits

As in the case of the previous BEMP about sorting of co-mingled lightweight packaging, the main environmental benefit of this BEMP is related to the substitution of virgin plastic materials with those recovered through the sorting and reprocessing of the collected mixed plastic packaging waste.

Lots of studies based on life cycle assessments have demonstrated that recycling is generally an environmentally preferable option compared to other waste management alternatives. In the case of plastic recycling, the energy savings with respect to plastic production from virgin materials are in the order of 80–90 % (Plastics Recyclers Europe, 2013). A major review of LCAs carried out by WRAP in 2006 and updated in 2010 (WRAP, 2010) concluded that mechanical recycling is the best alternative regarding the climate change potential, depletion of natural resources and energy demand and that the environmental benefits are mainly derived from the avoided material production. In order to maximise the benefits, emphasis should be put on recovering good-quality material with a high purity (to limit the rejected fraction) that, once recycled, can replace virgin plastics at a high ratio (1 to 1).

Some figures showing the relative benefits of recycling the different plastic polymers, taken from a recent study carried out by Eunomia (2015), are provided in Table 4-34.

Table 4-34. Selected values – Impacts of recycling dense plastic

Data source	Impacts (tonnes CO2 equivalent per tonne of aluminium recycled)
Association of Plastics Manufacturers in Europe (in WRATE)	Mixed plastic -1.04
	Bottle plastics -1.15
US EPA (2002/6)	HDPE -1.40
	LDPE -1.71
	PET -1.55
AEA (2001)	HDPE -0.53
	PET -1.80
APME (2005)	HDPE -1.90
WRAP (2006)	Average -1.08 (of landfill scenarios)1
ERM (2006 a)	-2.32
ERM (2006 b)	1.82 (lumber) / -0.85 closed loop
Prognos / IFEU (2008)	-0.16 – -1.72
SCM (2013)	-0.578
Franklin Associates (2010)	PET -1.98 HDPE -1.2
Notes: 1. Unlike the other studies referenced above, WRAP's values included the benefits associated with avoided residual treatment; these are, however, likely to be minimal for landfilled plastic. 2. Depending on production process and polymer mix	

Sources: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001; ERM (2006 a) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006; ERM (2006 b) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006; APME data cited here from http://www.plasticseurope.org ; USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002; Prognos / IFEU / INFU (2008) Resource Savings and CO2 Reduction Potential in Waste Management in Europe and the Possible Contribution to the CO2 Reduction Targets in 2020, October 2008; WRATE database; Zero Waste Scotland (2013) The Scottish Carbon Metric: Technical Report, October 2013; Franklin Associates (2010) life cycle inventory of 100% postconsumer HDPE and pet recycled resin from postconsumer containers and packaging, Report for The plastics division of the American chemistry council, inc., July 2010

Considering the processing of mixed plastic packaging waste in advanced PRFs in more detail, an in-depth analysis is provided in the WRAP report "LCA of Management Options for Mixed Waste Plastics" (2008b), which shows the results of an environmental life-cycle assessment (LCA) study of a range of recycling technologies and includes a comparison with a selection of alternative disposal options for domestic mixed plastic waste. The basis for the comparison between the various recycling technologies and alternative disposal routes is the recycling, reprocessing or disposal of one tonne of mixed plastic arising as waste from a typical UK material recovery facility (MRF). For each recycling scenario, the boundaries of the LCA study range from the point at which this mixed plastic waste leaves the MRF through to the production of granulate material ready to be made into "new" products. Non-recycled fractions are modelled up to the point at which the material is considered to be disposed of (e.g. in landfill) or to the point where it can substitute a primary material (e.g. after the agglomeration process for producing a redox agent for blast furnace injection). In the case of recycled/recovered products the assessment also includes the avoided production of material or energy from primary sources. It should be noted that the chosen study boundaries mean that the process of collecting the mixed plastic waste is not included in the assessment.

The following impact categories have been assessed:

- global warming potential (GWP);
- solid waste arisings (solid waste);
- primary energy consumption (energy);

- photochemical ozone creation potential (POCP);
- eutrophication potential (EP);
- acidification potential (AP);
- human toxicity potential (HTP);
- ozone layer depletion potential (OLDP);
- abiotic depletion potential (ADP).

The alternatives analysed are shown in Table 4-35 and the results of the comparison are illustrated in Source: WRAP, 2008b

Figure 4-63.

Table 4-35. Key processes included in the modelled mixed plastics sorting facilities

Scenario	Key processes
A	■ Landfill (all materials)
B	■ Municipal incineration with energy recovery (all materials)
C	■ Near infra-red (NIR) sorting (Titech) ■ Conversion to solid recovered fuel (SRF) for cement kilns (non-PVC fraction) ■ Mechanical recycling of PVC fraction
D	■ Film removal (Stadler) ■ NIR sorting of rigids (Titech) ■ Pyrolysis of PP and PE fractions (BP polymer cracking process) ■ Mechanical recycling of PVC and PET fractions
E	■ Film removal (Stadler) ■ NIR sorting of rigids (Titech) ■ Pyrolysis of PP, PE and PS fractions (Ozmotech process) ■ Mechanical recycling of PVC and PET fractions
F	■ Film removal (Stadler) ■ NIR sorting of rigids (Titech) ■ Conversion of PE and PP fractions for use as redox agent in blast furnace ■ Mechanical recycling of PVC and PET fractions
G	■ Film removal (Stadler) ■ NIR sorting of rigids (Titech) ■ Mechanical recycling of PE, PP, PET and PVC fractions
H	■ Film removal (Stadler) ■ NIR sorting of rigids (Pellenc) ■ Mechanical recycling of PE, PP, PET and PVC fractions
I	■ Film removal (Stadler) ■ NIR sorting of rigids (Qinetiq) ■ Mechanical recycling of PE, PP, PET and PVC fractions
J	■ Film removal (Stadler) ■ NIR sorting of rigids (Sims) ■ Mechanical recycling of PE, PP, PET and PVC fractions
K	■ Film removal (KME) ■ NIR sorting of rigids (Titech) ■ Mechanical recycling of PE, PP, PET and PVC fractions
L	■ Film removal (Stadler) ■ Density separation (TLT) ■ Mechanical recycling of PE and PP fractions
M	■ Sorting and cleaning PE and PP fractions (Swiss Polymera) ■ Mechanical recycling of PE and PP fractions (Swiss Polymera)
N	■ Sorting and cleaning PE and PP fractions (B+B) ■ Mechanical recycling of PE and PP fractions
O	■ Film removal (Stadler) ■ Density separation (Herbold) ■ Mechanical recycling of PE and PP fractions
P	■ Film removal (Flottweg) ■ Density separation (TLT) ■ Mechanical recycling of PE and PP fractions

Source: WRAP, 2008b

	High priority			←	→				Low priority
Scenario	Global Warming Potential	Solid Waste	Energy	Human Toxicity Potential	EUtrophication Potential	Photochemical Ozone Creation Potential	Acidification Potential	Abiotic Depletion Potential	Ozone Layer Depletion Potential
A (Landfill)	15	16	16	16	16	16	16	16	16
B (Incineration)	16	1	8	15	10	15	15	15	2
C (SRF)	11	2	1	14	2	12	11	1	10
D (BP pyrolysis)	14	12	4	2	8	13	13	14	3
E (Ozmotech pyrolysis)	13	15	3	3	1	11	12	13	1
F (Redox agent)	12	4	2	4	13	14	14	5	9
G (Stadler & Titech)	1	5	5	5	3	6	4	3	6
H (Stadler & Pellenc)	4	7	7	11	5	8	8	7	4
I (Stadler & Qinetiq)	7	14	10	13	7	10	10	12	5
J (Stadler & Sims)	2	6	6	6	4	7	5	4	7
K (KME & Titech)	5	8	9	12	6	9	9	9	8
L (Stadler & TLT)	6	10	12	8	11	3	2	6	11
M (Swiss Polymera)	3	3	11	1	9	1	1	2	13
N (B+B)	9	13	14	10	14	5	6	10	14
O (Stadler & Herbold)	10	11	15	9	15	4	7	11	15
P (Stadler & Flottweg)	8	9	13	7	12	2	3	8	12

rank 1=best,
rank 16=worst
green=top 25%,
red=bottom 25%

Source: WRAP, 2008b

Figure 4-63. Results of LCA comparison of alternative disposal or sorting options for mixed plastic waste, *showing the relative ranking of the scenarios against each impact category*

From Source: WRAP, 2008b
Figure 4-63 it is clear that scenario A (landfill) is the option with the least favourable environmental performance followed by B (incineration) – although interestingly incineration has the best performance for solid waste arisings, the impact category ranked second. The recycling scenarios (G to P) tend to have the best environmental performance if all impact categories are taken into account, but if only global warming potential, primary energy consumption and solid waste arisings are studied then C (conversion of mixed plastics in solid refuse-derived fuel) ranks in the middle of the recycling options.

Appropriate environmental indicators

The first appropriate environmental indicator for this BEMP refers to the processing rate of mixed plastic packaging waste. The indicator represents the material recovery efficiency of the plant, expressed as the percentage of recovered plastic materials from the mixed plastic packaging waste input flow sent for recycling over the year, calculated at the plant scale as follows:

$$\text{Plant processing rate (weight \%)} = \frac{\text{recovered materials sent for recycling (total weight per year)}}{\text{mixed plastic packaging waste processed (total weight per year)}} (\%)$$

Considering the importance of processing mixed plastic packaging waste as far as possible into clean single polymer types to produce higher quality recyclate, the processing rate indicator must also be complemented by at least the percentage breakdown of the recovered materials sent for recycling per plastic polymer. And specific sampling and testing procedures should be applied at the PRFs as well, to assess the composition of their input and output materials and consequently the sorting efficiency by polymer type. The sampling and reporting methods for PRFs can be the same as those described for MRFs in the previous BEMP.

Bearing in mind that the sorting process requires significant energy consumption, other appropriate indicators for describing the PRF's environmental performance are energy efficiency and GHG emissions.

As for energy efficiency, the indicator can be expressed in terms of the annual specific consumption of energy (kJ per tonne of input waste), which can be calculated as follows:

$$\text{Energy efficiency (kJ/t)} = \frac{\text{total energy consumption (kJ per year)}}{\text{mixed plastic packaging waste processed (total weight per year)}}$$

This can be complemented by also considering the total consumption (%) of energy produced by the organisation from renewable sources.

As for GHG emissions, the indicator can be expressed in terms of the annual emissions of greenhouse gases, expressed in tonnes of CO_2 equivalent per tonne of input waste as follows:

$$\text{GHG emissions (t CO2e/t)} = \frac{\text{total CO2 equivalent emissions (total weight per year)}}{\text{mixed plastic packaging waste processed (total weight per year)}}$$

The CO_2 equivalent emissions are calculated according to the GHG Protocol Corporate Standard (World Resources Institute and World Business Council for Sustainable Development, 2004), adopted as the basis for ISO 14064, and refer both to the direct greenhouse gas emissions produced by the plant operation (scope 1 according to the reference methodology) and the indirect emission savings related to the substitution of raw materials with secondary material (scope 2 according to the reference methodology, in order to measure the achieved environmental benefits as described in the previous BEMP).

For PRFs applying wet grinding systems and density separation technologies for cleaning and sorting plastic flakes, water consumption could be a significant environmental impact as well and the appropriate environmental indicator for describing it is the annual specific consumption of water (m^3 per tonne of input waste):

$$\text{Water use (m3/t)} = \frac{\text{total water consumption (m3 per year)}}{\text{mixed plastic packaging waste processed (total weight per year)}}$$

As with CO_2 emissions, for calculating water use, it would be ideal to include not only the water used in the processing plant, but also the indirect water savings achieved thanks to the substitution of raw materials with secondary materials. This calculation is normally difficult since reliable figures on the water savings from the substitution of raw materials are not easily available to the plant operator. it is therefore meaningful to calculate the water use only accounting for the water used on site.

The indicator on water use can be complemented by other useful information such as the source of the water (e.g. surface water, groundwater), the amount of waste water, waste water treated and reused, rainwater and grey-water recycling.

It must be noted, finally, that other environmental performance parameters will also be measured at the plant scale, as defined in the plant permits and in the related monitoring plans, according to the national and regional regulations. But for comparing plant performances, within the scope of this document, the processing rate and the energy and water consumption efficiency of the PRFs are considered the most suitable parameters.

Cross-media effects

The cross-media effects are fundamentally the same as in the case of the previous BEMP related to sorting of co-mingled lightweight packaging: electricity consumption, emissions of dust and odour, and safety and health of workers performing manual sorting. For PRFs using wet grinding systems and density separation technologies for cleaning and sorting plastic flakes, water consumption and discharges are a significant environmental impact as well. When this is the case, closed-loop water use is recommended for reducing water consumption and the need for effluent treatment.

As for the energy consumption in the sorting processes, i.e. film separation and sorting of plastic bottles and whole rigid items per polymer and colour, it can be assumed that the figures are the same as those reported for MRF consumption in Table 4-26 within the previous BEMP about sorting of co-mingled lightweight packaging.

As for the energy and water consumption in the reprocessing steps, i.e. plastic flakes cleaning, separating and washing, some reference figures are provided in Table 4-36 with reference to the secondary production of PET and HDPE. These figures are related to the whole process for producing one kg of recycled PET or HDPE (Rigamonti L. et al., 2014).

Table 4-36. Input material for the secondary production of PET and HDPE (expressed per kg of recycled PET or HDPE)

Consumption	Unit	Average	Std. dev.	Min.	Max.
RECYCLED PET					
Electricity	kWh	0.32	0.10	0.24	0.47
Methane	MJ	2.56	0.31	2.29	2.90
Water	kg	2.96	-	-	-
RECYCLED HDPE					
Electricity	kWh	0.44	0.17	0.20	0.56
Methane	MJ	0.51	0.21	0.27	0.65
Water	kg	1.78	-	-	-

Source: Rigamonti L. et al., 2014

Operational data

As pointed out in the BEMP description, plastic waste generated by households ranges from 8 % to 12 % by weight of the total waste. Considering an average of 9 %, and after the plastic bottles are removed, the mixed plastic waste amounts to about 6.6 % (WRAP, 2008a). There is limited data on the detailed composition, by polymer type, of this mixed plastic fraction and of the household mixed plastic packaging waste in general. Moreover, the composition varies significantly from place to place, depending on the collection system in place, and, over time, depending on the evolving trends in product packaging. Some reference figures are provided in Table 4-37 and Figure 4-64.

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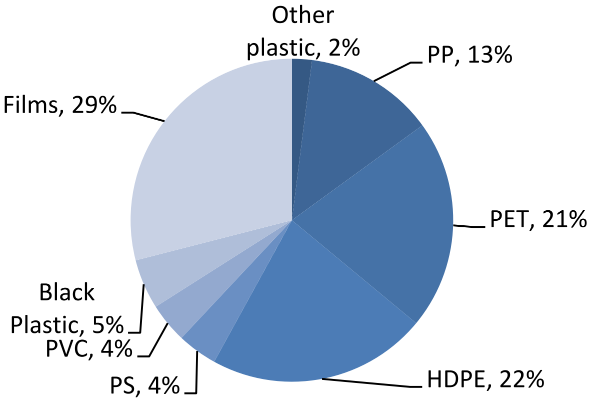
Table 4-37. Polymer composition of plastic fractions in municipal solid waste in Italy (% weight)

Material plastic fraction	Average in waste	PET	PE		MIX
			LDPE	HDPE	
Bottles	33	25	-	8	-
Soft plastic	39	-	39	-	-
Hard plastic	5	-	-	5	-
Mixed plastics and residues*	23	-	-	-	23
Total	**100**	**25**	**39**	**13**	**23**

*of which 57 % are mixed plastics.

Source: Rigamonti L. et al., 2014



Source: WRAP, 2008a

Figure 4-64. Household kerbside plastic sample composition from UK

As for the composition of the mixed plastic packaging waste, after the removal of plastic bottles, reference values are shown in Table 4-38, corresponding to the composition assumed by WRAP in its study on "Domestic Mixed Plastics Packaging Waste Management Options" (2008a).

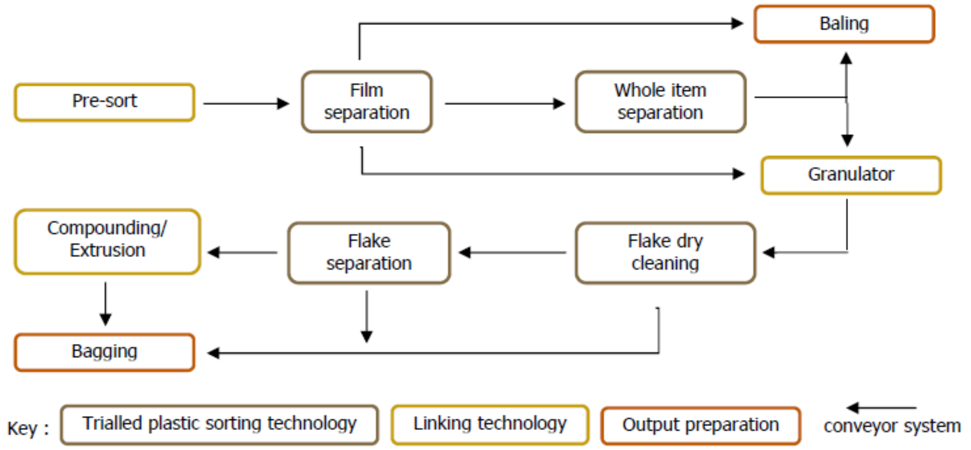
Table 4-38. Reference composition of mixed plastic waste (%)

Flexible		Rigid					Residual	Total
PE	PP	PP	PE	PET	PVC	PS		
25	5	17.22	13.44	15.33	3.5	3.99	16.52	100

Source: WRAP, 2008a

The analysis of the mixed plastic packaging waste composition by polymer type provides important operational data, as it determines whether it is convenient to recover a specific polymer type in the PRF. Below, the technological state-of-the-art of an advanced integrated PRF, including plastic reprocessing steps, is described with reference to the average composition provided in Table 4-38 and based on the results of the WRAP study (2008a) that modelled the process designs required for effectively processing mixed plastic packaging sourced from the household waste stream. The results were obtained by analysing in detail the technical, environmental and economic performance of different PRF process designs based on practical sorting trials, applying available sorting and reprocessing equipment. The comparison of such process designs with identified frontrunners (see Reference organisations) confirms that the PRFs modelled in the study are representative of the most advanced state-of-the-art facilities currently operating in Europe and worldwide.

The process designs modelled by WRAP include four key sorting stages (as shown in Figure 4-65): film separation, whole item NIR separation, flake dry cleaning, and flake separation. A range of linking technologies are required such as pre-sorting, shredding and compounding to complete the process. Output material preparation is also required through the baling of whole items or bagging of rigid items.



Source: WRAP, 2008a

Figure 4-65. Generic mixed plastics sorting/processing process

The designed process incorporates an initial pre-sorting stage, as the prior removal of contamination would allow separation systems to operate more efficiently. In this stage, operatives manually remove non-packaging plastics and non-plastic items, paper and cans.

Film separation, using film grabbers, air drum or ballistic separators followed by a manual quality assurance step, is a standard first separation step when sorting mixed plastic packaging to avoid downstream issues. The significant presence of film in a NIR input stream can cover the rigid packaging which reduces the ability to accurately identify and eject individual rigid packaging items. The output fraction is baled and sold directly on the market or can also be shredded and processed through flake sorting systems to add market value.

The whole item separation sorts the plastic flow by polymer type. The number of sorting units and the polymers sorted vary depending on the required output streams. The most common polymer types, such as PP, are sorted first with each NIR system running at no more than three tonnes per hour. NIR sorters can be also programmed to identify PLA (polylactic acid, a biodegradable plastic) (Hollstein F. et al., 2013; Environment Australia, 2002).

After optical sorting, an appropriate granulator reduces the size of whole items to approximately 15 mm flakes, which is the size acceptable for all the flake sorting technologies tested. Dry cleaning (or wet cleaning) is used after the shredding stage to remove excess dirt and labels. A density separation unit (float-sink flake separation) is then used to clean and further separate the polyolefin (PP and PE) flakes. These are fed into a compounding and bagging line to maximise market value. The sink fractions from these processes are sent for residual waste management.

The inclusion of colour-sorting steps increases the market value of recovered materials, but in the WRAP study it was not tested. In any case, colour sorting of whole items can be achieved using the NIR technology.

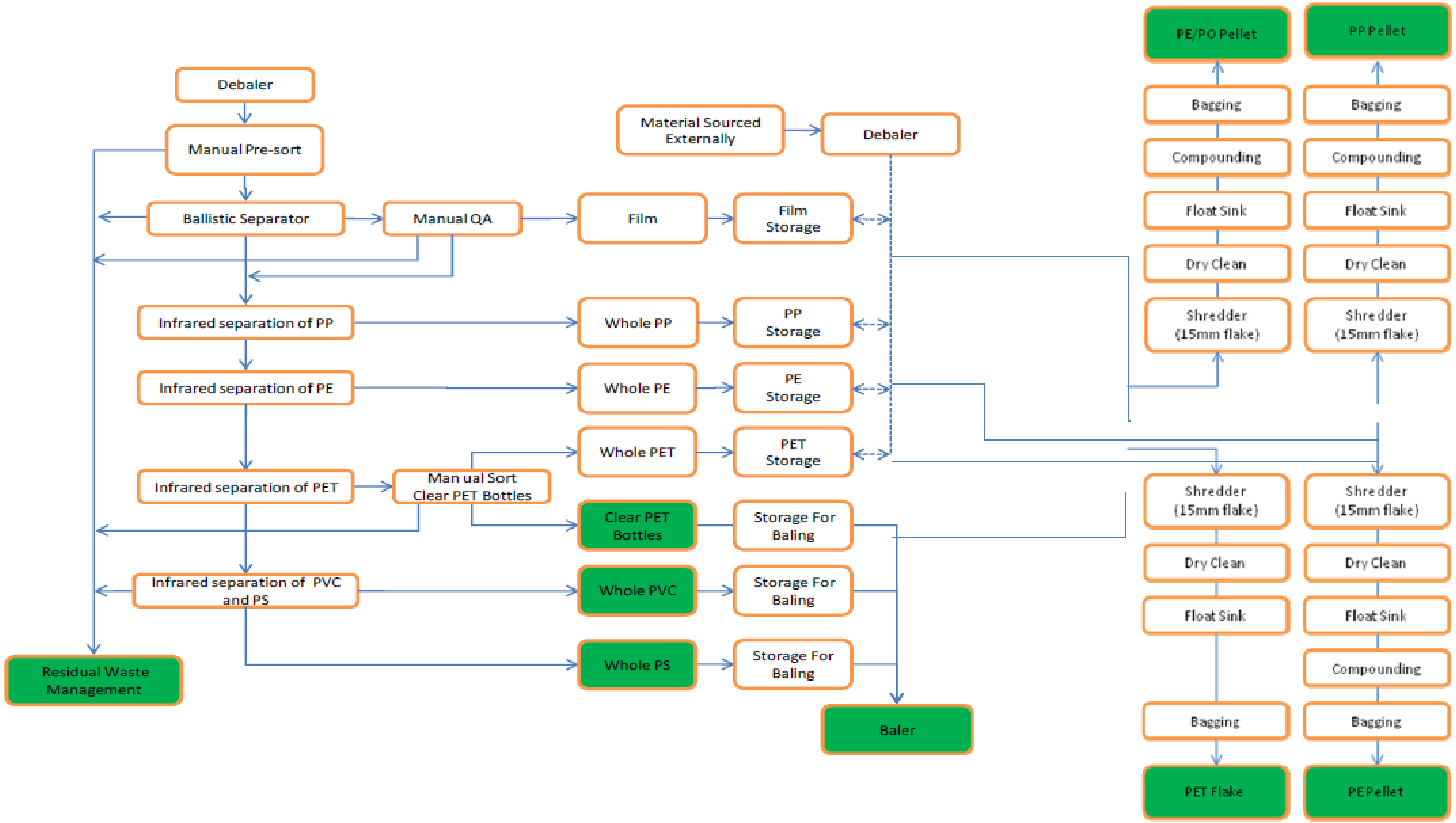
Despite the inclusion of a pre-sorting step, quality assurance functions must be included across the process to ensure material quality is not compromised and equipment is protected against damage from processing unsuitable items.

Different process designs, combining differently the process steps and processing technologies described above, were also reviewed in the WRAP study, based on practical, economic and environmental performance, with reference to a Hypothetical PRF with a potential capacity of 40 000 tonnes per year of input material. The flow diagrams of two process designs (processes A and B) that were shown to be technically, environmentally and economically feasible are shown in Figure 4-66 and Figure 4-67.

The process design A facility incorporates the whole item sorting technologies to produce polymer-sorted fractions. The PP, PE and PET are then shredded and separated using flake sorting technologies. The process is also designed to accept baled, mixed or sorted plastic material from third-party facilities as an additional stream. The outputs include whole baled PS and PVC, cleaned flaked PET and compounded PP, PE and film. It recovers 67 % of mixed plastic packaging from the input stream for recycling. It incurs a capital cost of ~ EUR 18 million (GBP 15.4 million)[58].

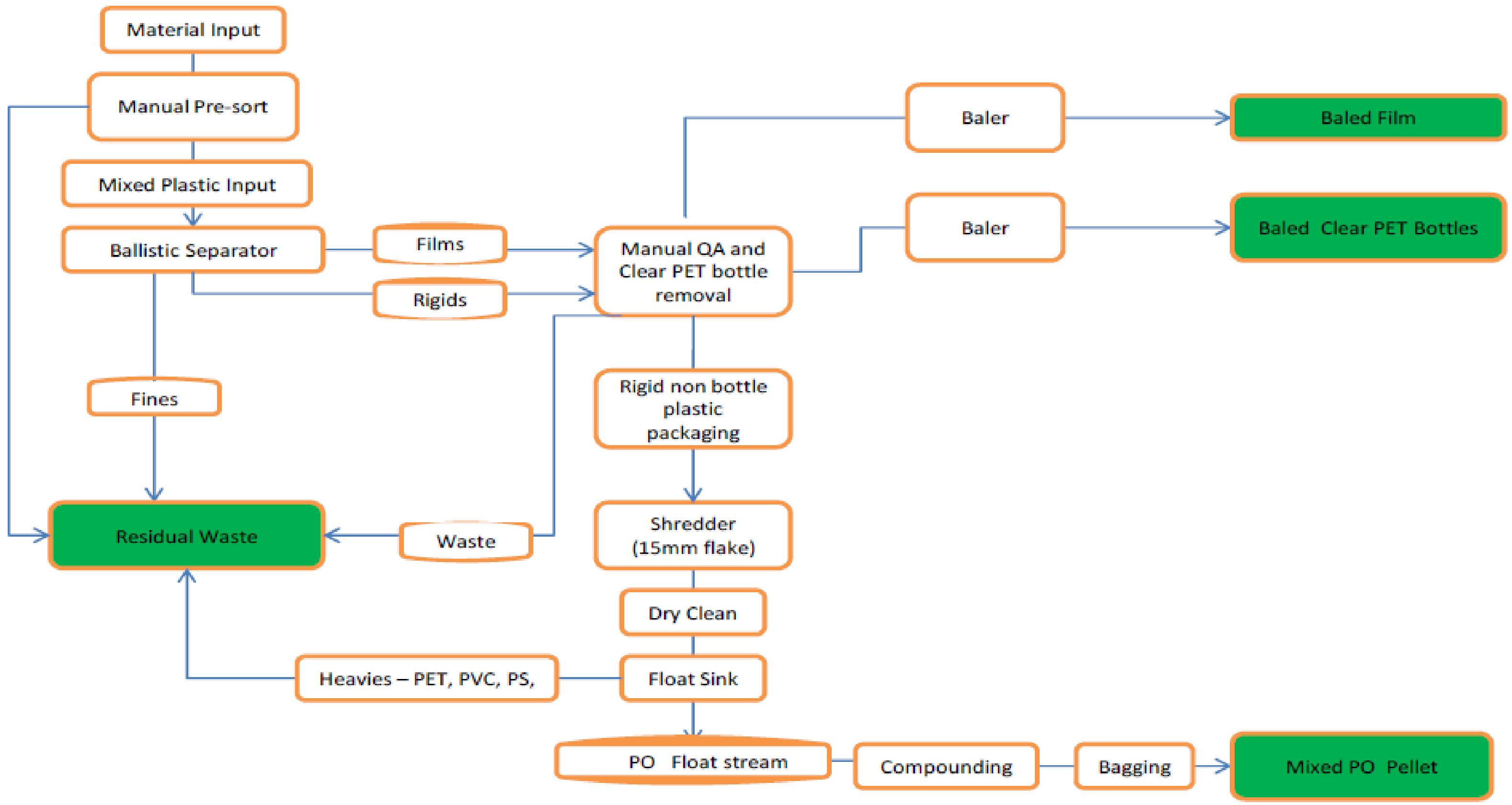
Process design B utilises flake sorting technology as the primary separation technique, but still includes the initial removal of film. The mixed polyolefin output is compounded, bagged and sold. The heavy fraction is stored for residual waste management. The facility recovers 62 % of mixed plastic packaging from the input stream. It incurs a capital cost of ~ EUR 3.5 million (GBP 2.6 million)[59].

[58] The capital costs provided in the WRAP study have to be assumed just as an indicative reference, considering that the study has been developed in 2008.
[59] See previous footnote (n.3).



Source: WRAP, 2008b

Figure 4-66. Process design A, including film sorting, NIR sorting of whole items followed by flake sorting and compounding



Source: WRAP, 2008b

Figure 4-67. Process design B, using flake sorting technology as the primary separation technique for whole film and other plastic items

As pointed out in the BEMP description, most of the PRFs operating in Europe generally feature only the first sorting steps described above (pre-sorting, film separation and NIR plastic-type separation) and separate PET, HDPE, PP and PS into coloured and clear streams, as clear plastics have a higher market value. The sorted materials are baled and sent to plastic reprocessors, where sorted plastics are flaked, washed and/or extruded into pellets. But as plastic product values increase rapidly at the reprocessing stage, some PRF operators have invested in downstream reprocessing to make high-grade finished recycled polymers, applying process designs of the kind described in Figure 4-66 and Figure 4-67 (WRAP & Zero Waste Scotland, 2012).

This is the case, for example, of the mixed plastic sorting and washing facility opened by Biffa in 2011 at Redcar (UK), which applies a process similar to that described in Figure 4-67 (process design B), or of the post-consumer plastic packaging sorting and reprocessing plant operated by Montello SpA at Montello (Italy), which applies a process similar to that described in Figure 4-66 (process design A). These and other best performing PRFs are briefly described in Reference organisations.

Applicability

As to the applicability of this BEMP, there are no legal or country-specific barriers to building and operating integrated mixed plastic sorting and reprocessing plants, but some constraints exists with reference to the economic feasibility and related optimal plant capacity and feedstock availability.

The challenge for integrated PRF operators is that their customers (plastics converters) demand large quantities of recycled plastics, manufactured to strict specifications at a price that has to be competitive with virgin plastic. Technical requirements can vary greatly depending on the end use required by the buyer. Meanwhile, the quantities available to PRF operators and tonnages of recyclate produced can be of varying quality as there is no EU-wide certification in place. The market for recovered plastics is still small in comparison with virgin plastics, and subject to the broader economic climate as well as several other factors that can be volatile in nature. Since recyclates aim to partly replace virgin polymers in existing applications, their market value is directly linked to virgin plastic prices, which depend heavily on volatile oil prices (Plastics Recyclers Europe, 2013).

In this context, the first prerequisite for the applicability of the BEMP is that good waste collection systems for household post-consumer plastic packaging are put in place and the good quality of the collected materials is assured through effective household communications. And going back in the plastic value chain, it is also important to somehow limit, as far as possible, the market trends in plastic applications towards more complex multi-layer and multi-material products, which make mixed plastics sorting and reprocessing much more difficult.

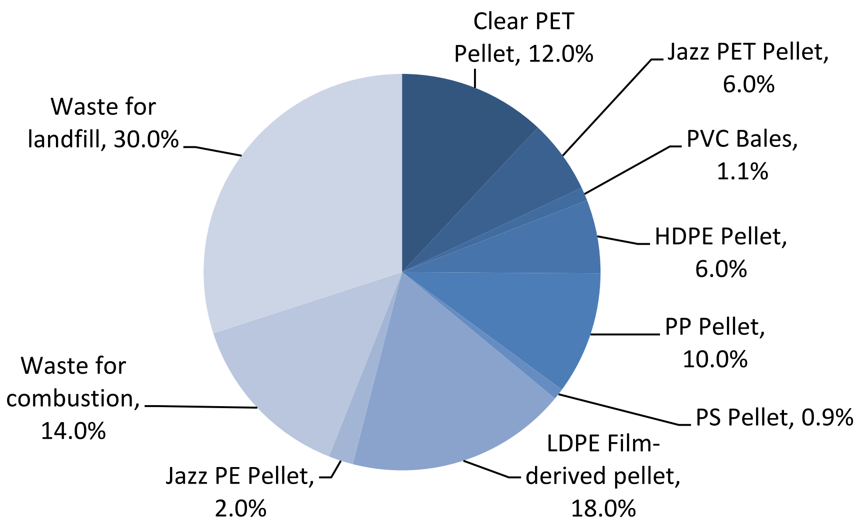
Given these framework conditions, the critical issue regarding the BEMP applicability is related to the optimal plant capacity, as in the case of sorting co-mingled packaging waste in MRFs. The elements that need to be taken into account are exactly the same as for MRFs. As explained in detail in Economics below, an integrated mixed plastic sorting and reprocessing plant should generate a profit at a throughput of around 80 000 t/year.

Economics

The costs and benefits of this BEMP are highly dependent on the quality of the materials recovered and on the market value of such materials. As in the case of MRFs, economies of scale play a relevant role in determining the economic feasibility of PRFs: the financial assessment of recycling mixed plastics realised by WRAP (2009) points out that a fully integrated plastics recovery and reprocessing facility producing high-grade clear PET and natural HDPE and industrial-grade PE, LDPE, coloured PET, PS and PP appears to be able to generate adequate investor returns at a scale of about 80 000 t/year. The financial assessment developed by WRAP provides a detailed analysis of the expected economic viability of a PRF, based on the financial modelling of an integrated mixed plastics sorting and reprocessing facility with a capacity of 80 000 t/year (24 000 t/year of plastic film and 56 000 t/year of rigid plastic) including the following elements:

- A semi-automated PRF section, where plastic films are removed by hand at the start of the sorting process and then rigid PET, HDPE, PP, PS and PVC containers are identified and separated automatically by NIR sorting technology. The PRF section produces separated rigid container fractions (including bottles) and a film fraction for further processing within the integrated PRF and reprocessing facility, plus a baled PVC fraction for sale.
- A flake washing plant which dry-cleans and granulates segregated rigid plastics from the PRF section and then washes and separates the material further to produce clean washed single polymer flake fractions for further processing.
- An extrusion section which melts and vacuum degasses (where appropriate) the clean washed flake to produce food-grade natural PET, food-grade natural HDPE, and non-food-grade but high-quality PP, PS, mixed colour (Jazz) PET and Jazz PE pellet products for sale.
- A NIR film sorting section which separates the film fraction produced by MRFs.
- A film washing, agglomeration and extrusion section which produces clean low-density polyethylene (LDPE) pellets for sale.

The product yields foreseen for the plant are shown in Figure 4-68. The recovery rate of the plant is 56 % of the mixed plastics input flow.



Source: WRAP, 2009

Figure 4-68. Product yields from the modelled PRF

As the financial model was created by WRAP in 2009, the assumptions about costs and revenues for input and output materials, capital costs and labour costs are not up-to-date and must be considered with caution. But, besides the exact values for the PRFs' expected cash flow, the results of the financial assessment and sensitivity analysis provide the following reliable conclusions:

- A stand-alone mixed plastics sorting and reprocessing plant should generate a profit at a throughput of around 80 000 t/year (24 000 t/year of mixed films plus around 56 000 t/year of other mixed rigid plastic and bottles).
- The business would be significantly more robust to increases in feed costs or reductions in selling prices if the plant could be built with a capacity of at least 100 000 t/year (30 000 t/year of films plus 70 000 t/year of rigid plastics).
- The commercial viability of the facility is particularly sensitive to the price of recycled pellets and to the yield of useful plastic that is extracted from the feed material.
- Variations in labour and utility cost have less impact on commercial viability than product price and yield factors.
- Variations in capital costs have a significant impact on investor returns. Grant support to reduce effective capital costs or use of second-hand equipment where feasible will improve project viability.
- A mixed plastics sorting and reprocessing plant processing only other rigid plastics where clear PET and natural HDPE have already been removed should be commercially viable as a stand-alone venture or as an addition to an existing reprocessing facility for HDPE and PET, provided the additional facility is built at a scale of at least 80 000 t/year.
- The assumptions used in this model indicate that film processing should be commercially viable, both as part of an integrated facility and as a stand-alone business. The viability of this option is sensitive to the cost of the delivered mixed film feed material.

Besides the results of this financial analysis, in its previous report about "Domestic mixed plastics packaging waste management options", WRAP (2008a) also developed a preliminary assessment of the economic viability of the process designs defined in the study (see process design A and process design B in Figure 4-66 and Figure 4-67, considering an input flow of 6 tonnes of mixed plastics waste per hour (about 35 000 t/year). On the basis of the analysis carried out and given the model assumptions, process design A and process design A both appeared to generate attractive internal rates of return. The sensitivity analysis carried out suggested that process design B could represent a more robust option, albeit at a significantly higher capital cost, as it allows the recovery of more valuable plastic grades. Indeed, plastic product values increase more rapidly at the reprocessing stage and this determines the better economic performance of integrated PRFs also featuring plastics reprocessing operations, even if higher capital costs are required.

Driving force for implementation

As in the case of sorting of co-mingled lightweight packaging waste, the most important driving force for sorting of collected mixed plastic packaging waste has been the European Packaging Directive (1994/62/EC; 2004/12/EC amended) and the related extended producer responsibility (EPR) regulations introduced by Member States, subsequently reinforced by the recycling targets and landfill bans set by the EU's Waste Framework Directive (2008/98/EC), and more recently by the proposal for reinforcing these targets introduced in the EU's Circular Economy Package. The pull mechanisms for the creation of well-functioning secondary raw materials markets (i.e. economic incentives, Green Public Procurement) are also relevant drivers, but still need to be reinforced.

In the case of plastic sorting, a further driving force consists of the need to sort plastics as far as possible into single polymer types to obtain efficient mechanical recycling and thus a higher economic value from the plastic materials recovered. This need becomes particularly relevant in the territories where the collection schemes include all plastic packaging or all plastic waste in the sorting guidelines, in order to ease sorting for inhabitants. Indeed, most plastic types are inherently immiscible at the molecular level and have different processing requirements. For example, a small amount of PVC contaminant present in a PET stream will degrade the recycled PET resin and vice versa. The cleaner and the fewer different types of plastic, the less mechanical treatment is required and the higher the quality of the recycled plastic products (Plastics Recyclers Europe, 2013).

Reference organisations

Advanced sorting of mixed plastic packaging waste is much less developed than advanced co-mingled sorting and currently there are only a few integrated PRFs sorting and reprocessing mixed plastics, as described in the previous paragraphs, operating in Europe. Examples of best performing integrated PRFs are provided below:

- Montello SPA, Italy: advanced integrated PRF located in Montello (province of Bergamo, Italy), with a capacity of 150 000 t/year, performing advanced sorting of mixed post-consumer packaging waste followed by reprocessing steps for some of the recovered plastic waste flows (production of PET flakes; LDPE, HDPE and PP granules, polyolefin granules and Geomont® dimpled sheet. Further information on the company website http://joomla.montello-spa.it/en/index.php. The company also produced a video describing the sorting and recycling process, available at https://www.youtube.com/watch?v=w8_CvCM-85Y.
- Biffa Polymers, UK: mixed plastic sorting and washing facility, applying a flake separation technology, opened by Biffa in 2011 at Redcar and upgraded in 2013. The PRF currently sorts and reprocesses a pre-sorted polypropylene (PP) packaging input stream, consisting of pots, tubs and trays from the household waste stream, producing high-quality washed PP flake outputs. Some of the output is processed through Biffa’s food-grade HDPE recycling facility at the same site and go back into the manufacture of new milk bottles. A video describing the sorting process is available at https://www.youtube.com/watch?v=z_dIogMIz5A.
- ALBA Group, Germany: in Eisenhüttenstadt, ALBA operates a plant that mechanically recycles used plastic packaging. Here, as well as state-of-the-art sorting and washing processes, the used packaging undergoes a treatment process specially developed in-house. Using fusing and compression, the plastics are converted into an innovative resin that can be used to manufacture brand new plastic products. The company also produced a video describing the recycling process, available at https://www.youtube.com/watch?v=0rsidi-2gnk.
- SUEZ, France: in its plant in Rochy-Condé, in 2015, SUEZ inaugurated a new plastics processing line capable of separating and grinding all types of plastic resins into flakes or aggregate, transforming plastic waste into high-quality granulate that is reused in the composition of make-up palettes, vehicle headlights, gutter ducts or even textiles. The site has a total capacity of 105 000

t/year and 4 000 t/year for the plastic processing including 1 500 t/year for the grinding. The company also produced a video describing the sorting and recycling process, available at https://www.youtube.com/watch?v=_n_VMt9UZw8.

It is more common for advanced PRFs to just perform the sorting steps of the household plastic packaging waste fraction, recovering a high number of plastic grades (by polymer type and colour), as in the case of Veolia's UK Rainham plant, known as the Parrot POLY-mer separation facility, which has the capability to separate up to nine different grades of plastic, ranging from bottles, yoghurt tubs and trays, with a sorting capacity of 50 000 t/year.

As described in the previous BEMP about sorting of co-mingled lightweight packaging waste, some advanced MRFs are also equipped with plastic sorting units for polymer type and colour., The SUEZ MRF plant located in Rotterdam is particularly interesting, as it efficiently sorts from a co-mingled packaging input, including plastic and metals, different grades of plastics that are then reprocessed by QCP (Quality Circular Polymers), a new plastics recycling facility realised in partnership with SUEZ (see related videos at https://www.youtube.com/watch?v=Xjot6NpySac - https://www.youtube.com/watch?v=gfVHQ9EvU4Q).

It is finally worth noting, considering the innovation trends, that some promising technological solutions are emerging for sorting black polymers, film plastics including biodegradable films, or for applying robotic arms in the sorting process. Reference organisations working on these solutions are as follows:

- SADAKO Technologies, Spain: a Barcelona-based start-up that has developed a high-speed industrial robotic arm (Wall-B) with a grasping-by-suction system plus a state-of-the-art computer vision system that can overhang conveyor belts. The robotic arm is already used at two processing plants near Barcelona (https://www.youtube.com/watch?v=-BN18Re0g00).
- ZenRobotics, Finland: a high-tech company specialised in Artificial Intelligence (AI)-controlled robotic systems that has developed robotic solutions for sorting waste streams; these are already commercially available for sorting CDW and under development for other waste streams (http://zenrobotics.com/).
- Steinert, Germany: Steinert recently developed the UniSort BlackEye technology, a sensor-based sorting machine that has the ability to classify plastics according to their polymer group categorisation, enabling also the recovery of black or dark plastics (http://www.steinertglobal.com/fileadmin/user_upload/global/download-area/EN/UNI_blackeye_EN.pdf), and the Unisort Film machine, for sorting buoyant objects such as conventional PVC film, bio-based film or biodegradable film (http://www.steinertglobal.com/de/en/products/unisort/unisort-film/).

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4.7.3. Treatment of mattresses for improved recycling of materials

Summary overview

It is BEMP to sanitise and disassemble end-of-life mattresses, separating and sorting the different materials by type.

Five main technical operations can be identified in a best performing end-of-life mattress treatment facility:

- feeding and storage: reception (unloading) and dry storage to avoid contamination, sorting by type;
- sanitising: applying chemical or heat treatments for sterilisation;
- filleting: cutting the mattress' outer fabric cover and the binding flanges;
- disassemble and sorting: separating and sorting the different materials by type;
- handling materials: baling processes, product storage as bales, loose material (sorting residues) or in containers (metals), before delivery to downstream processes (e.g. recycling of metals).

The disassembling and sorting operations can be carried out mechanically or (more commonly) manually.

Waste management area

Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability

There are no main technical barriers to the applicability of this BEMP. The simplicity of the treatment process does not require significant investments, even for the most automated processes.

The most important obstacles for mattress recycling are identified as follows:

- economic factors, notably the low cost of landfilling and the low quality of the materials arising from mattresses, linked to the need to store end-of-life mattresses in a clean and dry place and current mattress designs preventing easy disassembly;
- the low treatment capacity of the facilities, limited by the end-of-life mattress flow collectable in the area surrounding the plant at affordable transport costs.

Specific environmental performance indicators

- Plant sorting rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of waste mattresses processed.
- Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of waste mattresses processed.

- GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (scope 1 and 2) of the plant divided by the quantity of waste mattresses processed.
<u>Benchmark of excellence</u>
- Facilities treating waste mattresses have a plant sorting rate of at least 91 %.

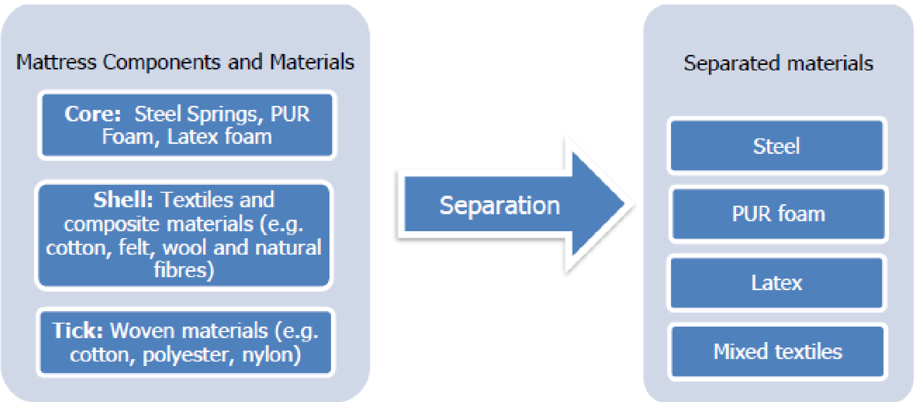
Description

In Europe, up to 30 million mattresses annually reach their end of life and it is estimated that 60 % go to landfill and 40 % are incinerated (EBIA, 2014). However, at least 85 % of their mass can be readily recycled through simple disassembly (CalRecycle, 2012). Their bulkiness makes them difficult to handle during waste pickup and transport, their low density makes them undesirable landfill material (an average mattress takes up 650 litres of landfill space as compression is difficult) and their springs have a tendency to damage landfill and transfer station compacting or shredding equipment. On the positive side, many municipalities throughout the EU already have in place effective collection schemes for bulky items, including mattresses, which can conveniently transport end-of-life mattresses to the treatment facilities (CalRecycle, 2012; ISPA, 2004; WRAP, 2013).

This <u>BEMP tackles the treatment of end-of-life mattresses</u>, consisting of sanitising and fully deconstructing them, separating and sorting the different materials by class and supplying these materials to relevant end markets for recycling.

The composition of mattresses varies greatly, but they are usually categorised based on their main core material, which falls into three common types: steel springs, polyurethane foam and latex foam. Mattresses may also contain other shell materials surrounding the core and ticking which contain and protect the internal sections of the mattress (Zero Waste Scotland, 2013). In the mattress treatment process, these different material types are separated to achieve maximum value in end markets (Source: Zero Waste Scotland, 2013

Figure 4-69).



Source: Zero Waste Scotland, 2013

Figure 4-69. Composition of mattresses and types of separated materials

Despite the different mattress types and process methods applied, five main technical operations can be identified in a best performing treatment facility:

- Feeding and storage: this operation consists of reception (unloading) and storage of the end-of-life mattresses in a dry and covered area. Mattresses are stored off the ground, to prevent contamination and damage from water and dirt, and are stacked efficiently to maximise the number of units loaded in the storage containers. Best management practices are applied to check the mattresses for suitability, in order to keep unacceptable items out of the facility, and to prevent the spread of bedbugs. In most cases, mattresses are already stored according to type at this stage, as their type (and in particular the presence of inner springs) significantly influences the dismantling process.
- Sanitising: this operation is carried out by applying chemical or heat treatments for mattress sterilisation, in order to guarantee healthy working conditions and the hygienisation of the recovered materials.
- Filleting: this operation consists of cutting the mattress' outer fabric cover and the binding flanges.
- Deconstruction and sorting: this operation consists of separating and sorting the different materials composing the mattress by type; for innerspring mattresses, the first operation consists of separating the metal innerspring unit and the wooden box spring foundation from the other components; for the other mattress types or the other components of innerspring mattresses, the mattress is then dismantled in its different layers, separating and sorting the cotton and other textile fibres and the cushioning materials (including mainly polyurethane foam, memory foam, latex rubber foams and natural fibres).
- Handling materials: this operation consists of baling processes, product storage as bales, loose material (sorting residues) or in containers (metals), and includes loading operations for products and residue streams to be delivered to downstream processes.

The deconstruction and sorting operations can be carried out in different ways:

- manual processing, with the first filleting operations carried out using non-power box cutters or disc grinders and all the operations of removing, separating and baling the different materials done by manual labour, supported by simple equipment like forklift trucks with bale clamps, pallet trucks, pallet racking, workbenches, low-speed shredders/granulators and material balers;
- automated processing, by using specific equipment for the removal of the innerspring units[60] or even fully automated lines using metal detectors to separate non-metal and metal-based mattresses, machines for cutting mattress edges, peeling rolls for removing the outer textiles, magnets for removing the steel springs, cutting machines for reducing foams into manageable pieces[61]; in most cases, this method first requires manual operations for the filleting of the mattress;
- a combination of the two methods.

[60] This is the case, for example, of Recyc-Matelas Europe in France, described as specific case study in operational data.
[61] This is the case of Retour Matras in the Netherlands, that has designed on its own the fully automated processing system. This case study is also described more in detail in operational data.

Currently, the most commonly applied option is manual deconstruction because of the high costs of automated sorting equipment and the low revenues for the recovered materials, but the most advanced treatment options based on the use of automated equipment, at least for some phases of the dismantling process, are also applied. Detailed operational data for the best performing case studies are included in the section on Operational data below.

Most of the recovered mattress' components can be recycled and made into new useful products (PSI, 2011; WRAP, 2013; ADEME, 2014; Zero Waste Scotland, 2013; Innortex, 2016):

- textile fibre components are reprocessed into a variety of products including geotextiles, industrial oil filters, construction and automotive insulation materials;
- foam is reprocessed as carpet underlay, gym mats, animal bed stuffing, cushioning material for upholstered furniture and even new mattresses;
- springs are recycled as metal scrap;
- clean wood is reprocessed for chipboard, mulch or animal bedding.

As already highlighted, the composition of mattresses varies significantly, but currently steel and polyurethane foam tend to be the main contributors to the weight of the materials recovered, as well as to the revenues from selling the materials to their existing end markets, as they have a positive market value. They are followed by textile fibres which are usually grouped together, as they are difficult to separate into the different materials due to the construction of the mattress, and are sold on to mixed textiles markets as low-quality fibres (short fibre length), often in the form of shredded mixture (Zero Waste Scotland, 2013). Latex foam can be used in small amounts when combined with other materials for carpet underlay while pure polyester layers have a high value and high recyclability (WRAP, 2013). Reference figures for the average mattress composition are provided in Table 4-39.

Table 4-39. Average mattress material composition

Material	Average mattress composition (kg)	Average mattress composition (%)
Steel	6.2	29%
PUR foam	5.3	25%
Cotton, non-woven	3.3	15%
Natural Fibres (e.g. coconut, sisal, jute)	1.6	7%
Felt	1.6	7%
Cotton, woven	1.4	6%
Wool	0.8	4%
Polyester, non-woven	0.8	4%
Latex foam	0.6	3%
Total	**21.4**	**100%**

Source: Zero Waste Scotland, 2015

In order to add value to the recovered materials, mattress treatment facilities can also directly reprocess them by producing secondary products. This is the case, for example, of ECOVAL and VALORMAT in France (see case studies description in Operational data).

Some of the mattress' components, i.e. the innerspring units and the cushioning materials, could also be reused for rebuilding new mattresses, but as no appropriate standards or labelling requirement currently exist at European level for this practice, and considering that it can pose a risk to consumers for hygienic (the potential presence of bedbugs, dust mites and their droppings and other allergens) or safety reasons (the compliance with flammability standards) (PSI, 2011), this option can be considered within the scope of the BEMP only as a future option.

Achieved environmental benefits

Recycling end-of-life mattresses can produce several environmental benefits:

- reduction of reliance on landfill disposal;
- recovery of valuable materials to make other products, thus reducing the need for virgin materials to be extracted; and
- reduction of greenhouse gas emissions and energy use by decreasing the energy-intensive production of new mattresses or other products.

As for the greenhouse gas emissions and energy implications of using different end-of life management methods for mattresses and box springs, a reference study that provides detailed and comprehensive data was developed by the California Department of Resources Recycling and Recovery in 2012 (CalRecycle, 2012). The study uses LCA methodology to estimate the greenhouse gas emission reductions that could be achieved through increased reuse and recycling of end-of-life products. The greenhouse gas emission reductions from reuse and recycling are calculated as the greenhouse gas savings from avoided landfill and avoided primary production minus the added greenhouse gas emissions from reverse logistics and reprocessing (extended boundary approach).

In more detail, the study estimates the greenhouse gas emissions considering product manufacturing (including all supply chain activities), forward logistics and product end-of-life management for an average mattress and box spring set characterised by the material composition reported in Table 4-40.

Table 4-40. Material composition of an average mattress and box spring set

	Mass (kg)	Mass (%)
Entire mattress and box spring	54.4	100
Steel	27.2	50
Wood	5.44	10
Foam	5.44	10
Cover (toppers)	5.44	10
Cotton	2.72	5
Unspecified	8.16	15

Source: CalRecycle, 2012

The analysis allows the comparison of the CO_2e emissions from landfilling or recycling of the mattress at its end-of-life. In both scenarios, the CO_2 emissions come from the product manufacturing, forward logistics, reverse logistics (i.e. the transportation of the mattress and box spring from their pickup location to the treatment/final disposal site) and from the end-of-life management steps described below:

- <u>Landfilling scenario</u>: chemical and biological degradation process of the mattress and box spring materials in the landfill and construction, maintenance and operation of the landfill itself;
- <u>Recycling scenario</u>: reprocessing at the treatment facility via manual disassembly assisted by some basic equipment such as forklifts and balers, and recycling of secondary materials with the following assumptions:
    - the steel of the innerspring unit is used for steel making;
    - the polyurethane foam is used for making rebound carpet cushion;

- the cotton fibres and covers are reprocessed via mechanical recycling into a variety of products;
- the recycled materials displace their virgin counterparts;
- no recycling benefit is calculated for the wood;
- the unspecified material is landfilled.

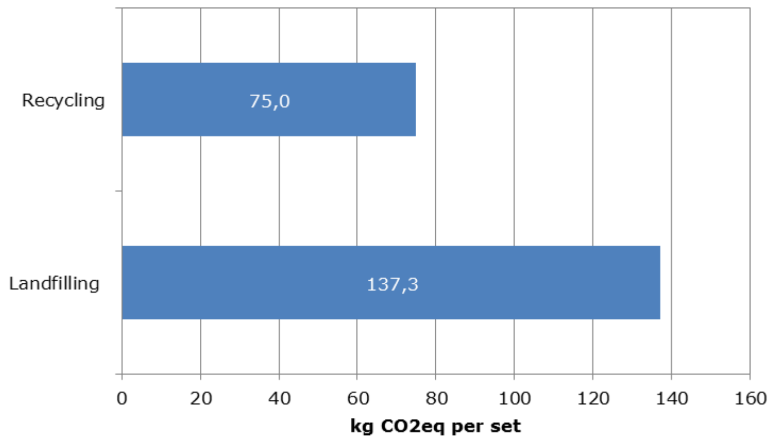
The greenhouse gas emissions estimated in the two scenarios for one mattress and box spring set are reported in Table 4-41 and in Source: own elaboration based on CalRecycle, 2012

Figure 4-70.

Table 4-41. Greenhouse gas emissions estimate per mattress and box spring set

Process phase	GHG emissions (kg CO_2e/set)
Landfilling scenario	
Production of mattress and box spring	129
Final disposal in landfill	8.3
Total CO_2e in landfilling scenario	**137.3**
Recycling scenario	
Production of mattress and box spring	129
Landfilling of treatment scraps (unspecified components)	1.2
Reverse logistics	4.1
Reprocessing	0.6
Net avoided burdens from secondary materials recycling	-59.9
Total CO_2e in recycling scenario	**75.0**

Source: CalRecycle, 2012



Source: own elaboration based on CalRecycle, 2012

Figure 4-70. Comparison of greenhouse gas emissions in the different scenarios

Comparing the two scenarios, it can be observed that recycling rather than landfilling allows a significant environmental benefit, reducing GHG emissions by 62.3 kg CO_2e (45 %).

Nonetheless, it should be noted that the results obtained in the study consider a mattress treatment process without any sterilisation treatment (which would in fact be recommended for the health of workers) and based on manual disassembly, while more automated processes would imply higher energy consumption and GHG emissions. Unfortunately LCAs for mattresses are limited in number and none analysing a more automated treatment process has been found. In any case, considering the relevant net margin observed in the case of manual deconstruction, it can be expected that the GHG savings of recycling versus landfilling would remain relevant in the case of automated processes, which could also improve the process productivity.

Appropriate environmental indicators

The material sorting rate at the plant scale is a good indicator for evaluating the environmental performance of the treatment processes of end-of-life mattresses. The indicator can be expressed in terms of the percentage of materials sent for recycling from the end-of-life mattresses input flow over the year, calculated at plant level as follows:

$$\text{Plant sorting rate (weight \%)} = \frac{\text{recovered materials sent for recycling (total weight per year)}}{\text{waste mattresses processed (total weight per year)}} (\%)$$

The environmental performance of the treatment process also depends on the final destination of the recovered materials, i.e. if they are up-cycled or down-cycled. Therefore the sorting rate indicator must also be complemented by the percentage breakdown of the materials sent for recycling by material type (steel, foam, textile fibres, wood), specifying also the final material or product into which the material flow is transformed.

For each mattress treatment facility, the material sorting indicators can thus be expressed as in the example provided in Table 4-42.

Table 4-42. Example of sorting rate indicators

Main flows	Tonnes per year	Sorting rate (%)	Recovered materials types	Recovered material (%)	Final material/product obtained
Mattresses treated	3 000	93	Steel	31	Recycled metal
			PUR foam	25	Carpet underlay
			Latex foam	5	Carpet underlay
Recovered materials sent for recycling	2 790		Mixed textiles	30	Automotive felts
			Wood	2	Mulching

Source: own elaboration

With regards to other potential environmental impacts, if we consider a treatment process mainly based on manual disassembly and sorting operations, no significant pressures are expected in terms of energy and water consumption and emissions to air and water. However, if automated processes are considered, in particular those with fully automated processing systems, energy consumption can become relevant. In this case, also taking into account EMAS' core environmental performance indicators, other appropriate indicators for describing the mattress treatment performance are energy efficiency and GHG emissions.

For energy efficiency, the indicator can be expressed in terms of the annual specific consumption of energy (kJ per tonne of input waste), which can be calculated as follows:

$$\text{Energy efficiency (kJ/t)} = \frac{\text{total energy consumption (kJ per year)}}{\text{waste mattresses processed (total weight per year)}}$$

This can be complemented by also considering the total consumption (%) of energy produced by the organisation from renewable sources.

For GHG emissions, the indicator can be expressed in terms of the annual emissions of greenhouse gases, expressed in tonnes of CO_2 equivalent per tonne of input waste as follows:

$$\text{GHG emissions (t CO2e/t)} = \frac{\text{total CO2 equivalent emissions (total weight per year)}}{\text{waste mattresses processed (total weight per year)}}$$

The CO_2 equivalent emissions are calculated according to the GHG Protocol Corporate Standard (World Resources Institute and World Business Council for Sustainable Development, 2004), adopted as the basis for ISO 14064, and refer both to the direct greenhouse gas emissions produced by the plant operation (scope 1 according to the reference methodology) and the indirect emission savings related to the substitution of raw materials with secondary material (scope 2 according to the reference methodology, in order to measure the achieved environmental benefits as described previously). As for the indirect emission savings, they could also be relevant in the case of manual treatment processes.

It must be noted, finally, that other environmental performance parameters will also be measured at the plant scale, as defined in the plant permits and in the related monitoring plans, according to the national and regional regulations. But, for comparing plant performances, within the scope of this document, the sorting rate and the energy efficiency and GHG emissions of the mattress treatment facilities are considered the most suitable parameters.

Cross-media effects

It is expected that the only relevant environmental impact of the treatment process is associated with energy consumption in the case of mostly automated treatment processes. The safety and health of workers, in particular when performing manual deconstruction and sorting, have to be assured, with special regard to the risk of injuries during the cutting operations and to their exposure to dust and bedbugs, mites and other allergens.

Operational data

For the mattress treatment operations, a light industrial covered warehouse with truck access for unloading mattresses and loading recovered materials is required. Considering the operations that characterise the mattress treatment process (see Description section), independently of the process method applied (manual versus automated), space is required for four main functions: storage of incoming mattresses, processing of mattresses on workbenches or using automated equipment, storage of mattress materials, office and amenities (Zero Waste Scotland, 2013).

The space needed for the plant operation varies considerably depending on the processing methods (manual versus automated) and on the related treatment capacity. Based on the information acquired about the operating facilities analysed, it transpires that plant capacities range between about 400 tonnes of mattresses treated per year up to 5 000 t/year, where the lower values are typically associated with manual treatment processes, average values (around 1 500–2 500 t/year) are typical of partially automated processes, while the higher values are reached only in the most automated ones. These values are reported in Table 4-43, where they are also expressed in terms of the number of mattresses treated, assuming an average weight per mattress of 25 kg. The table also shows the average productivity levels observed for the different processing methods, expressed in terms of mattresses treated per worker per year. Based on available data, in the manual processes a single worker takes about 5–6 minutes to dismantle one mattress, which, over the year, means that one worker can process about 10 000–12 500 mattresses. If at least some phases of the treatment process are supported by specific equipment, the productivity can increase up to 15 000 mattresses per worker per year, while in the fully automated facilities the productivity can be even higher than 30 000 mattresses per worker per year (Zero Waste Scotland, 2013; Halifax C&D Recycling, 2009).

Table 4-43. Typical plant capacity ranges of mattress treatment facilities

	Manual process	Partially automated process	Fully automated process
Plant capacity (t/year)	400–800	1 500–2 500	3 000–5 000
Plant capacity (mattresses/year)	16 000–32 000	60 000–100 000	120 000–200 000
Processing productivity (mattresses treated per worker per year)	10 000–12 500	12 500–15 000	> 30 000
Number of workers needed for the deconstruction operations	2–4	4–8	4–6

Source: own elaboration

It can thus be observed that these facilities never reach high treatment capacities, even if the process automation allows the treatment of higher volumes of mattresses, due to the high transport costs related to the bulky nature of mattresses. Regarding this issue, available data show that the yearly discard rate for end-of-life mattresses can vary between one mattress for every 6 to 21 persons, with an average of one discarded mattress for every 12 persons per year. This means that the collection of 200 000 mattresses per year can be reached in an area populated by about 2.4 million people, which, excluding big cities, corresponds to a wide territory where transport costs become a limiting factor.

As for the sorting rates, they seem to be independent of the automation level of the process. The values reported for the analysed case studies range between 80 % up to 98 %, where the higher values are reported both for manual and for automated processes, with an average value of 93 % of materials recycled from the end-of-life mattresses input flow over the year. The percentage breakdown of the materials sent for recycling by material type is rarely reported and varies a lot, as does the mattress composition from place to place. It is thus not possible to provide average reference data for this process parameter.

The equipment used in the treatment process is of course dependent on the automation level of the process. In the manual deconstruction processes, the main piece of equipment used by operatives is a strong utility knife with replaceable blades or other handheld power cutting tools, as well as safety equipment that usually includes heavy-duty safety gloves, safety boots, high-visibility vests, goggles and dust masks. Other pieces of equipment are also required: forklift trucks, pallet trucks, pallet racking, wheeled storage cages, workbenches and balers (at least for textiles). Other mechanical equipment generally used in more automated processes can be steel compactors, which can potentially increase the scrap value of the metal by three times, and shredders for processing steel, wood and textiles.

More specific mechanical equipment is generally produced in-house by the mattress treatment operators and currently is not commercially available, with the exception of the fully automated system that deconstructs mattresses and strips and sorts materials, developed by the Dutch mattress treatment operator RetourMatras (see case study 3) (Zero Waste Scotland, 2013). To provide more technical specifications about the operational implementation of the most automated processes and related equipment, three selected case studies, representative of different best performing processes, are described below.

Case study 1 – Mattress recycling at Langon, by SUEZ and Recyc-Matelas Europe

SUEZ, a key world player in the sustainable management of resources and the production of secondary raw material, and Recyc-Matelas Europe, a French company specialised in the treatment of end-of-life mattresses, partnered in order to realise and operate innovative facilities to recycle and recover up to 90 % of mattress components, in France (Langon) and Belgium.

Here it is described the treatment facility operating in Langon (RM Sud-Ouest), inaugurated in June 2015, which is one of the most advanced in Europe (Suez and Recyc-Matelas Europe, 2015). The facility has a treatment capacity of 7 000 tonnes of mattresses per year (30 000 units per month), employs 10 operators and has been realised with a total investment of EUR 1 million. The treatment process is a combination of manual and automated operations. At the reception, mattresses are classified and stored according to their type. Materials not in a suitable condition are rejected. The processing method applied is different for innerspring and other types of mattresses. The former are filleted manually and are then processed in specific equipment that separates the innerspring unit from the other materials, which are then separated manually by a couple of operators and sorted by type of material. The innerspring unit is then fed into a steel shredder. The other mattresses are also filleted manually and are then sent to other processing equipment that cuts and sorts the materials. In both cases, after filleting, the mattresses are hygienised by passing them into thermal treatment equipment. All the sorted materials (wool, cotton, felt, PU foam, latex and mixed textiles) are pressed into bales by specific equipment and are then sent for recycling, reaching an average recovery rate of 90 %.



The materials recovered are mostly recycled in the following products:

- textile fibres are reprocessed into construction insulation materials and automotive car mats;
- foams are reprocessed as carpet underlay and cushioning material for the automotive sector;
- springs are reprocessed as metal scrap.

In partnership with the French company Innortex and with the financial support of Ademe, Recyc-Matelas has developed an industrial pilot for producing recycled raw materials from PU foams and textile fibres recovered by dismantling used bedding products, with a capacity of 4 500 t/year (Valormat project, Ademe factsheet, 2013).

Some videos describing the deconstruction process are available at:

- http://www.recyc-matelas.fr/actualite-medias.html
- https://www.youtube.com/watch?v=brXvoIFkHNQ
- https://www.youtube.com/watch?v=JjGVQ8bEIU4
- http://www.actu-environnement.com/ae/news/process-recyclage-matelas-langon-usine-24760.php4

Further information on SUEZ and Recyc-Matelas Europe is available at:
http://www.suez.com
http://www.recyc-matelas.fr/index.html

Case Study 2 – ECOVAL mattress recycling plant

The ECOVAL mattress recycling plant has been realised by the industrial group Cauval (mattress producers), in cooperation with the local authorities and with the support of the French Government, in order to requalify their mattress manufacturing facility located in Flaviac, which was closing (ADEME, 2013). The facility has recently been acquired and is currently managed by Secondly Sud-Est.

In the recycling plant, launched in 2012, the following operations are carried out:

- sanitisation and disassembly of end-of-life mattresses, obtaining cushioning materials (latex, foam), textiles, metals and wood;
- shredding of textiles, which are transformed into fibrous material, and chipping of wood;
- reprocessing of the cushioning materials by applying an airlay technology developed by the Italy-based company Cormatex.

The main components of the airlay processing line are: a waste material shredder, a granulator, a thermo bonding oven, a lap formair (vertical or horizontal) and a longitudinal and cross cutter. In this processing line, the recovered foam and latex are granulated and homogeneously mixed within feeding silos. Thermal bonding components (i.e. bico fibres) are added to the granulated mix, which enters the thermo bonding oven and subsequently the lap formair, producing a panel that can have different heights and material densities. This output material is then used for new mattresses, but in minor quantities is also employed for producing insulating panels and cushioning materials for vehicles or packaging.

<u>Treatment capacity and sorting rates of the plant</u>:

- Mattresses recycled: about 180 000 mattresses per year, foreseen 470 000 at full capacity (~ 10 000 t/year).
- Recovered materials: 4 500 t/year (93 % sorting rate).

<u>Technical performance</u>:

- Sanitisation process: 100 mattresses per cycle; 2.5 hours sanitisation time per cycle.
- Deconstruction process: from 400 to 600 mattresses per day (~ 10–12 t/day).
- Cutting and granulation of foam and latex: 600 kg/hour.
- Shredding of textiles (2 machines): 800 kg/hour.
- Chipping wood: 2 t/hour.
- Airlay line: 1 t/hour.



Case Study 3 – RetourMatras plant

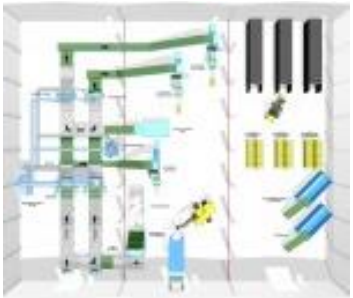
RetourMatras is a Dutch company that has built a fully automated mattress treatment plant in Lelystad, in the province of Flevoland. This fully automated processing line has been designed and built internally by RetourMatras in partnership with specialised suppliers (RetourMatras, 2016, Zero Waste Scotland, 2013).

All mattresses collected from mattress containers are deposited directly in the storage hall, ensuring that they stay dry at all times. An electric crane is used to move the individual mattresses from the deposit point onto a moving conveyor, after which they are cut open. The predominantly cotton cover is removed using a specially developed 'peeling roll' and is stored separately, to be given a new life as a duster or used in new textile products.

The mattresses are then sorted by passing through a metal detector; 100 % foam mattresses go to one side of the line and mattresses with steel springs are sent to the magnet. The magnet removes the steel springs, which are melted and used again as secondary material. What remains is the foam part of the mattress. This is reduced in size and pressed into bales. The foam, polyurethane foam and latex are used in carpet underlay, judo mats, filling in the car industry, etc. RetourMatras does not make new products, but ensures that the materials obtained from the old mattresses are sent to specialised recyclers.

The plant has the capacity to process 190 000 mattresses per year (about 4 750 t/year), with a productivity of about 30 000 mattresses per worker per year. Only four members of staff are required per shift.





RetourMatras produced a video to present their technology. It is available at:
https://www.youtube.com/watch?v=epxWhT7e4kg

Applicability

Mattress treatment facilities can be set up in any EU Member State as this BEMP is perfectly in line with the EU waste legal framework and the simplicity of the treatment process, which does not require significant investments even in the case of the most automated processes, allows for EU-wide implementation.

Nevertheless, several barriers to the development of mattress treatment infrastructure exist at the European level (Zero Waste Scotland, 2013; WRAP, 2013). Of these, the following are identified as the most important:

- The economic factors, which is the main limiting factor for the need to rely on a gate fee much higher than the average landfilling one to sustain the business, due also to the low and uncertain value of materials to end markets. At present, landfill is cheaper than recycling in every EU country; therefore, other drivers such as the need for environmental performance, landfill bans or extended producer responsibility schemes for mattresses, as in the case of France, are required to promote the applicability of mattress recycling. Also, the development of stable and receptive end markets for the recovered materials is an important issue. Currently the end markets are limited and provide low revenues, due to the poor quality of materials, perceptions of their cleanliness and saturation of the markets. Considering for example PU foam, mechanical recycling at present provides materials whose properties are inferior to those of virgin material for a slightly lower price (10–20 %) and there is a need to develop different outlets to the uses of virgin polyurethane (ADEME, 2014). Promoting research and innovation initiatives for the recycling of the recovered materials, with particular reference to foam and textiles, into new valuable products would provide greater options and increase the market value of the materials.

- The low quality of the materials arising from mattresses is another relevant barrier, linked both to the collection methods and to the mattress manufacturers who pay little attention to the product disassemblability at its end-of-life.

As for the first issue, for the applicability of the BEMP it is important to ensure a suitable supply of mattresses that are clean and dry. While the operators of mattress treatment facilities know how mattresses must be stored and handled to make recycling an option, many end-of-life mattress generators and collectors may not. For mattress recycling to be successful, it is important to publicise best practice methods of removal, collection, storage and handling to households as well as to major commercial and institutional mattress purchasers, such as hotels, universities and colleges, and healthcare facilities. Emphasis should be placed on the need to keep mattresses clean and dry, especially those that originate from locations that generate significant volumes of used mattresses (PSI, 2015). Specific requirements can be also set for the acceptance of mattresses in a treatment facility, as in the case of the Mattress Recycling Council Program, which in California has set specific guidelines that mattress collection providers must respect for the acceptance of their mattresses to the treatment facilities (Mattress Recycling Council, 2015).

As for the second issue, i.e. the improvements in mattress design, it must be noted that at present the design of mattresses prevents easy deconstruction and separation of materials. Encouraging design principles which align with end-of-life processing could be encouraged through mechanisms such as standards or Ecolabel schemes. However, this would likely take a long time to have an impact as mattresses have a lifetime of several years (Zero Waste Scotland, 2015).

- The low treatment capacity of these facilities, which is limited by the end-of-life mattress flow collectable in the area surrounding the plant at affordable transport costs. This factor determines poor economies of scale for the processing operations, limiting the scope for large-scale automated processing that would allow the achievement of much higher treatment capacities. It also requires the development of effective mattress collection plans that minimise the handling and transport of the products and allow for a consistent product flow (ISPA, 2004), not only relying on the collection schemes for bulky items usually offered to households by local authorities, but also setting up agreements with mattress retailers for the implementation of take-back schemes and offering specific collection services to hospitals, hotels, colleges and other institutions that produce significant numbers of discarded mattresses.

Economics

Reference literature on the economics of end-of-life mattress treatment or actual economic data from operating plants are limited (ISPA, 2004; Halifax C&D Recycling, 2009; Zero Waste Scotland, 2013 WRAP factsheets, 2013) and have been complemented with some specific calculations to provide an evaluation of the economies of scale for the processing operations.

The economic viability of a mattress treatment process has thus been evaluated estimating the potential expenditures and income sources considering different technical solutions and framework conditions. The estimation has been based on the following assumptions:

- Three different case studies have been considered: a facility applying manual deconstruction with a low treatment capacity (800 t/year or about 32 000 mattresses per year); a facility with a partially automated processing method with a medium treatment capacity (2 000 t/year or about 80 000 mattresses per year); a fully automated treatment facility with a high treatment capacity (4 500 t/year or about 180 000 mattresses per year).
- Gate fees for the incoming end-of-life mattresses ranging from a minimum value of EUR 240/t (EUR 6 per mattress), and a medium value of EUR 270/t (EUR 6.75 per mattress), to a maximum value of EUR 300/t (EUR 7.50 per mattress).
- Revenues from the main recovered materials also ranging from minimum to maximum unitary values, and calculated assuming a recovery rate of 93 % with the following percentage breakdown for the materials: 22 % steel, 36 % foam, 32 % textiles, 3 % wood.

These assumptions are summarised in Table 4-44.

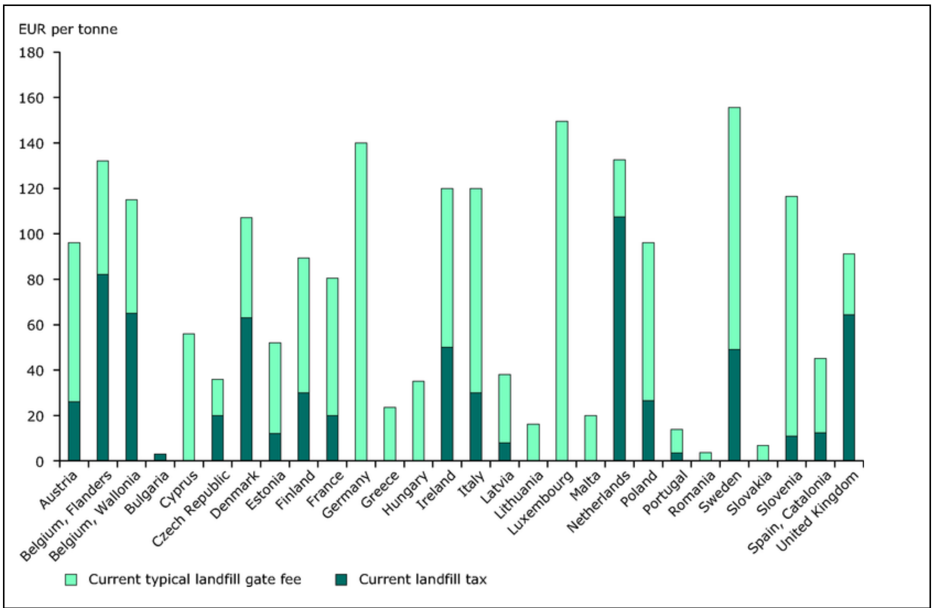
Table 4-44. Assumptions for the basis of the economic viability evaluation of mattress treatment processes

Mattresses treated per year			
	Low capacity Manual	Medium capacity Partially automated	High capacity Fully automated
Number of mattress/year	32 000	80 000	180 000
t/year	800	2,000	4 500
Range for gate fees - unitary values			
	Min.	**Med.**	**Max.**
Gate fee (EUR/t)	240.00	270.00	300.00
Gate fee (EUR/mattress)	6.00	6.75	7.50
Range for revenues from recovered materials - unitary values			
	Min.	**Med.**	**Max.**
Revenues for steel (EUR/t)	60.00	100.00	130.00
Revenues for foam (EUR/t)	100.00	120.00	140.00
Revenues for textiles (EUR/t)	60.00	80.00	100.00
Revenues for wood (EUR/t)	-	1.00	5.00
Recovery rates for the different materials			
Recovery rate for steel (%)	22		
Recovery rate for foam (%)	36		
Recovery rate for textiles (%)	32		

Recovery rate for wood (%)	3
Total recovery rate (%)	**93**

Source: own elaboration

The gate fees and revenues have been defined from information gathered during this study from literature review and information provided by operators. As for the gate fees, it was found that the charge per mattress applied by actual mattress treatment operators varies a lot, with lower values around EUR 5.5–6 and higher values up to EUR 16 per mattress, which correspond respectively to EUR 220–240 per tonne up to EUR 650 per tonne. Comparing these values with the current typical landfill gate fees and taxes, it is possible to see that even the lower values for mattress treatment gate fees are much higher than the highest landfilling costs per municipal waste, which are lower than EUR 160 per tonne of waste. As is shown in the following figure, such values cannot sustain a mattress disposal operation, but as an acceptable compromise, in our simulation we have assumed gate fees per tonne of waste ranging between EUR 240 and EUR 300. This requires specific intervention by national or regional governments in EU countries for promoting the applicability of mattress recycling, as highlighted in the Applicability section.



Source: EEA, 2013

Figure 4-71. Current typical landfill gate fees and taxes in EU countries

Also, the increase of the potential income from recovered materials is an important issue. The range of values assumed in the simulation are based on the reference literature and data provided by some interviewed operators, but currently the end markets for the materials recovered are highly volatile and more valuable products need to be developed to increase the market value of the materials, as highlighted as well in the Applicability section.

Based on the assumptions described above, the profit and loss accounts for the three different case studies have been simulated, calculating the expected annual revenues and expenditures, assuming investment costs ranging between EUR 30 000 for the low-capacity manual facility, EUR 400 000 for the medium-capacity and partially automated facility and EUR 2 million for the high-capacity fully automated facility, with amortisation periods of 5, 7 and 10 years respectively. The results of the simulation are shown in Table 4-45.

Table 4-45. Estimation of incomes and expenditures for the three case studies

Type of plant	Low capacity Manual	Medium capacity Partially automated	High capacity Fully automated
Plant capacity (t/year)	800	2 000	4 500
Plant capacity (mattresses/year)	32 000	80 000	180 000
Revenues from gate fees - Annual values (EUR/year)			
Min. gate fee (assuming EUR 6/mattress)	192 000.00	480 000.00	1 080 000.00
Med. gate fee (assuming EUR 6.75/mattress)	216 000.00	540 000.00	1 215 000.00
Max. gate fee (assuming EUR 7.5/mattress)	240 000.00	600 000.00	1 350 000.00
Revenues from recovered materials - Annual values (EUR/year)			
Min. revenue values	54 720.00	136 800.00	307 800.00
Med. revenue values	72 664.00	181 660.00	408 735.00
Max. revenue values	88 920.00	222 300.00	500 175.00
Investment and operating costs			
Investment costs (EUR)	30 000.00	400 000.00	2 000 000.00
Annual amortisation (%)	20	14	10
Total annual amortisation (EUR/year)	6 000.00	57 142.86	200 000.00
Annual operating expenditures (EUR/year)	280 000.00	650 000.00	1 300 000.00
Total annual expenditure (EUR/year)	**286 000.00**	**707 142.86**	**1 650 000.00**

Source: own elaboration

As can be observed, the majority of income arises from the gate fee, while the revenues from recovered materials assume a certain relevance in the scenario with the maximum values. As for the expenditures, major costs are mainly determined by the plant operation, as the process is labour-intensive and does not require relevant investments, except in the case of the fully automated facility.

Based on the incomes and expenditures estimated, the net cash flow before taxes has been calculated for the three case studies, considering all possible combinations of the framework conditions. The results are shown in Table 4-46.

Table 4-46. Net cash flow before taxes estimated for the three case studies

LOW CAPACITY - MANUAL TREATMENT PLANT				
Net cash flow before taxes		**Gate fee revenues**		
		Min.	**Med.**	**Max.**
Material revenues	**Min.**	-EUR 39 280.00	-EUR 15 280.00	EUR 8 720.00
	Med.	-EUR 21 336.00	EUR 2 664.00	EUR 26 664.00
	Max.	-EUR 5 080.00	EUR 18 920.00	EUR 42 920.00

MEDIUM CAPACITY – PARTIALLY AUTOMATED TREATMENT PLANT				
Net cash flow before taxes		**Gate fee revenues**		
		Min.	**Med.**	**Max.**
Material revenues	**Min.**	-EUR 90 342.86	-EUR 30 342.86	EUR 29 657.14
	Med.	-EUR 31 838.00	EUR 14 517.14	EUR 74 517.14
	Max.	-EUR 4 842.86	EUR 55 157.14	EUR 115 157.14
HIGH CAPACITY – FULLY AUTOMATED TREATMENT PLANT				
Net cash flow before taxes		**Gate fee revenues**		
		min	**med**	**max**
Material revenues	**Min.**	-EUR 112 200.00	EUR 22 800.00	EUR 157 800.00
	Med.	-EUR 11 265.00	EUR 123 735.00	EUR 258 735.00
	Max.	EUR 80 175.00	EUR 215 175.00	EUR 350 175.00

Source: own elaboration

The net cash flow before taxes is always positive in the event that the maximum or medium value for the gate fee is applied (with the only exception being for the medium-capacity plant with minimum values of the revenues from recovered materials), while it is always negative or very low in the event that the minimum value for the gate fee is applied. Considering the best case for the three case studies (maximum gate fee and maximum material revenues) and assuming that the taxes would be about 30 % of the net cash flow, the following payback periods for the treatment plants can be estimated:

- low-capacity manual treatment plant: one-year payback period, as the investment costs are very low, with earnings after taxes of about EUR 30 000 per year;
- medium-capacity partially automated plant: five-year payback period, with earnings after taxes of about EUR 80 000 per year;
- high-capacity fully automated plant: eight-year payback period, with earnings after taxes of about EUR 245 000 per year.

These evaluations show that, in the best case scenario, all the types of facilities are financially viable. Sustainable economic performances are also achieved in two of the intermediate scenarios (maximum gate fee and medium material revenues; medium gate fee and maximum material revenues), with the most sustainable performance achieved in the case of the manual treatment plant, thanks to the very low investments required. This peculiarity, associated with the labour-intensive and unskilled work required, make this business particularly interesting for social economy networks, including charities or social economy companies.

As for the most technologically advanced plant solutions, the following conclusion can be drawn: the investments required for fully automated plants can be considered financially viable only where a relevant flow of end-of-life mattresses and a good price for the recovered materials can be assured. One possible way for the operators of the waste treatment facilities to achieve this can be the direct reprocessing of the recovered materials for producing secondary products, thus enhancing their added value.

Driving force for implementation

The main driving forces for the implementation of this BEMP are as follows:

- The problems caused by the high volume and difficult handling of mattresses in landfill sites, which are pushing landfill operators to impose bans, limitations or high gate fees for the acceptance of mattresses and box springs (ISPA, 2004, PSI, 2011).
- EU legislation targets for municipal waste recycling (> 50 % by 2020 according to the Waste Framework Directive, > 65 % by 2030 according to the proposal for the revision of the Waste Framework Directive introduced by the Circular Economy Package) and diversion away from landfilling (targets for reducing the amount of biodegradable municipal waste landfilled introduced by the EU Landfill Directive (1999/31/EC) and proposal for a binding target to reduce landfill to maximum of 10 % of municipal waste by 2030 introduced by the Circular Economy Package), and in some Member States also bans on the landfilling of high caloric waste that have been already introduced in Austria, Belgium, Denmark, Germany, Italy, Norway, Sweden and Switzerland (EBIA, 2014).
- The introduction of EPR schemes for mattresses, like the one introduced in France in 2009 for furniture, including mattresses, that set a target for increasing the recycling of furniture waste up to 45 % by 2015 (Des Abbayes C., 2015). In this EPR scheme, producers are responsible for organising and financing the system; for this purpose, 24 companies (12 retailers and 12 manufacturers) in 2011 founded Eco-mobilier[62], a state-approved, non-profit private cooperative, financed by a visible recycling fee, added to the price of products and clearly shown as a separate charge at the in-store point of sale and paid for by the consumer. The scheme recognises the eco-contribution for recycling (the fees recognised for mattresses in 2014 are shown in Source: *Éco-mobilier, 2014*
- **Figure 4-72**), which helps make recycling a viable solution, and provides incentives to producers to redesign their products in order to improve the operational, economic and environmental performance of their end-of-life management.

Eco-contribution before tax per furnishing item*

PRODUCT TYPE	<= 120cm	> 120cm	Other
Slatted frame	€ 1,25	€ 2,08	Pair : € 2,50
Slatted bed base	€ 2,08	€ 3,33	Pair : € 4,17
Box springs	€ 2,50	€ 3,33	Pair : € 5,00
Mechanical/electric relaxation mattress	€ 4,17		Pair : € 8,33
All mattresses, including, folding mattresses	€ 1,67	€ 3,33	
Baby bed base, futon bed base, bundle bed base, mattress topper and baby mattress			€ 0,83

* The eco-contribution scale is applicable for the year 2014

Source: Éco-mobilier, 2014

Figure 4-72. Recycling fees for bedding recognised by Eco-mobilier in 2014

The materials used in mattresses have value when separated and political drivers are pushing towards minimising waste sent to landfill and increasing recycling rates. These

[62] More information at: http://www.eco-mobilier.fr/

factors together provide a stimulus for this potential opportunity. But given the low economic margins of these activities and their limited economies of scale, it would be important to reinforce the driving conditions by setting adequate economic incentives in each EU country.

Reference organisations

Mattress treatment facilities are operative mainly in France, Belgium, the UK, the Netherlands and Spain. Reference organisations are identified as follows:

- Recyc-Matelas Europe (France and Belgium): they are industry leaders within the mattress recycling market sector in France (detailed case study description provided in Operational data).
- Ecoval plant(France): a mattress recycling facility that reprocesses the dismantled materials into new mattresses (detailed case study description provided in Operational data).
- RetourMatras plant (Netherlands): a fully automated mattress treatment plant in Lelystad, in the province of Flevoland (detailed case study description provided in Operational data).
- JBS Fibre Recovery (UK): JBS operates in the mattress recycling sector in the UK, from sites in Telford, Trowbridge and Bridgend. In May 2013, it was acknowledged at the Let's Recycle Awards for Excellence in Recycling, winning the Recycling Business of the Year award. More information is available on the company website: http://jbsrecyclingcouk.fatcow.com/new/?page_id=35.
- Furniture Recycling Group (UK): UK-based company that operates mattress recycling plants based in Lancashire and Derbyshire. More information is available on the company website: http://www.tfrgroup.co.uk/.
- Mattress treatment plants in Spain: mattress recycling facilities operated in Spain by Comsermancha (http://www.comsermancha.es/), Recicolchon (http://www.recicolchon.com/english/), and the German company Sutco Recyclingtechnik (http://www.sutco.de/en/plant-technology/sorting-of-mattresses/).

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4.7.4. Treatment of absorbent hygiene products for improved recycling of materials

Summary overview							
It is BEMP to treat separately collected absorbent hygiene products (AHP) waste for recycling. The core process is a thermal treatment in an autoclave, an horizontal cylindrical vessel where the AHP waste is sanitized and opened. The output solid stream is then shredded and separated through a mechanical process into the two AHP components: polypropylene and polyethylene plastics and cellulose fibres, which can be sent for recycling.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
This BEMP is broadly applicable as no particular geographical or technical barriers exist. However, some specific conditions can influence the technical and economic viability of this treatment solution: - implementation of a selective collection scheme for AHP waste as a prerequisite; - minimum plant treatment capacity (based on treatment techniques and economics) of 8 000 t/year; - transport distance from collection areas to the plant and costs for landfilling and incineration; - population density in the collection area; - criteria and rules for recognising the end-of-waste and local market for recovered materials (plastic and cellulose).							
Specific environmental performance indicators							
- Plant sorting rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of AHP waste processed. - Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of AHP waste processed. - GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (scope 1 and 2) of the plant divided by the quantity of AHP waste processed. - Water use (m^3/t), calculated as the annual total water used on site divided by							

the quantity of AHP waste processed.
Benchmark of excellence
- Facilities treating absorbent hygiene products waste have a plant sorting rate of at least 90 %.

Description

Absorbent hygiene products (AHPs) is the category name for baby nappies, sanitary protection pads, tampons, adult incontinence products and personal care wipes, and nowadays represents one of the most challenging types of post-consumer waste. Today post-consumer AHP waste represents about 2–3 % of total municipal solid waste. Currently, AHP waste is mainly not recycled and belongs to "unrecyclable" municipal solid waste. It is typically disposed of via either landfill or incineration, thus causing loss of valuable material resources and high economic and societal costs. As the EU moves towards its recycling targets, AHP waste has quickly risen to already represent up to 15–25 % of the residual waste in some facilities, where selective collection rates above 70 % are in place. As a consequence, and due to its potential for contamination and infection, consumers and stakeholders alike perceive AHP waste management as a growing environmental sustainability issue that needs to be addressed in an integrated way (Recall, 2015).

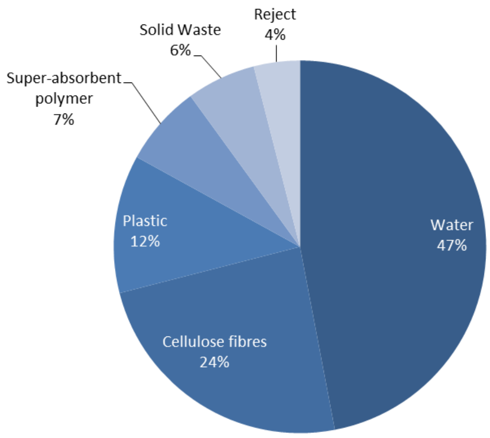
Public authorities and waste management companies are aware of this issue and are starting to take action to support the recycling of this waste stream. The BEMP on this issue is thus focused on the treatment techniques available for the separation of the AHP waste into its different material components, which can be recycled into secondary raw materials (e.g. plastic, cellulose).

At present, there is still a lack of large-scale treatment facilities for such disposables in EU countries, but in recent years some innovative techniques have been developed and are now ready to market and the first plants are starting to operate in EU Member States.

Considering the treatment techniques available and the existing plants, a treatment process that allows the sanitisation of AHP waste and the recovery of cellulose and PE/PP plastics, which are the basic components of AHP, can be considered a BEMP. The core process in the available techniques is represented by a thermal treatment in an autoclave, a horizontal cylindrical vessel where the AHP wastes, through the effect of steam at high temperature and pressure and continuous mixing by the rotation and alternative oscillation of the vessel, are sanitised and opened. The output streams from the autoclave are a water discharge, which is sent to a waste water treatment plant[63], and a solid stream that is then shredded and separated through a mechanical process into the two AHP components: plastics and cellulose fibres.

Based on available literature (Deloitte, 2011), the average composition of AHP waste can be considered to be as shown in the following figure.

[63] The water discharge can be treated in a municipal wastewater treatment plant



Source: Deloitte, 2011

Figure 4-73. Average breakdown of AHP waste (% weight)

Available treatment technologies are able to recover 100 % of the original products of disposable nappies, i.e. 100 % of the plastics and 100 % of the cellulose fibres plus the super absorbent polymers (SAP) that are present in the AHP waste flow, considering the output of the nappy's recycling process but not considering the residues resulting from the further sorting and industrial processing of these recovered materials. As cellulose and plastics used as raw materials for the production of AHP are high-quality materials, their recovery through treatment technologies provides the market with valuable secondary raw materials with multiple potential uses. The plastic is granulated and can be reused in several production cycles to make high-quality plastic goods or as an ingredient in composite materials replacing concrete and steel. The cellulose fibres can be used for the production of pet litter, pet care absorbent products, concrete and tarmac additives, brick manufacture, paper and cardboard, insulation materials and agricultural nutrients (Recall, 2015; Knowaste, 2016).

Achieved environmental benefits

The main environmental benefit of AHP waste recycling consists of the recovery of valuable materials (plastic, cellulose fibres) to make other products, thus reducing the need for virgin materials to be extracted and reducing greenhouse gas emissions and energy use.

As for the greenhouse gas emissions and energy implications of recycling AHP waste, reference studies, related to two of the available technologies identified (Deloitte, 2011; RECALL, 2015), analysed the environmental benefit based on a life-cycle assessment (LCA) approach, consistent with the ISO 14040 international standard. The system boundaries assumed in the two studies are different, so a direct comparison of their results is not possible, but both studies point out that nappy recycling, compared to a "business-as-usual" scenario (BAU) based on landfilling and incineration of this waste flow, allows the achievement of relevant CO_2 savings.

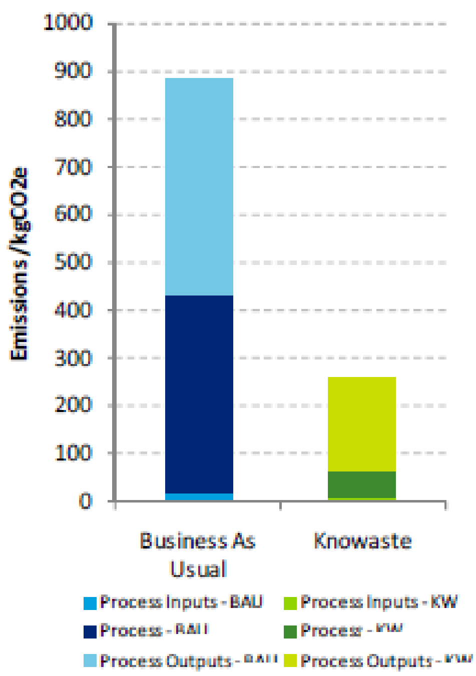
The study carried out by Deloitte (2011), related to the nappy recycling technology developed by Knowaste Ltd as described in Operational data, compares the performance of the recycling process to standard UK disposal practice, as outlined below:

- BAU consists of a standard UK waste scenario based on waste collection, landfill (81 %) and incineration (19 %), including useful energy recovery from landfill (as a result of methane capture) and incineration processes. Finally, since no recyclate materials (such as plastic) are recovered from this process, the extraction and manufacture of additional virgin materials are included.

- The "Knowaste" scenario covers the collection and processing of AHP waste, using a "two-stream" process that generates its own energy from gasification of organic fibres produced by the process and produces useful recyclates for the UK market (mainly plastics, but also some metals and other process rejects). Since the AHP waste is not landfilled or incinerated, this scenario includes the extraction and combustion of additional fuels for energy that would have been produced in a BAU situation.

Based on these assumptions, compared to landfill and incineration, the Knowaste recycling process emits up to 71 % less carbon emissions, which based on an annual capacity of 36 000 tonnes of AHP waste, means 22 536 tonnes of greenhouse gas emissions saved per year (equivalent to the annual carbon emissions of 2 064 UK citizens). Source: Deloitte, 2011

Figure 4-74 shows the global warming results per tonne of AHP waste, highlighting that the largest carbon impact for BAU and Knowaste is from the "outputs stage". For BAU, the outputs stage refers mainly to carbon emissions from the manufacture of virgin plastics, as all material that could potentially be reclaimed or recycled via Knowaste processing is effectively lost under BAU. For BAU, the outputs stage also refers to the generation of grid electricity from landfill gas and energy from waste incineration which is effectively lost under Knowaste. A number of alternative scenarios were created to analyse the sensitivity of the results, and, even in the worst case (the Knowaste process using grid electricity rather than generating its own electricity from gasification), the carbon emissions are reduced by 50 % with respect to the BAU scenario.



Source: Deloitte, 2011

Figure 4-74. Comparison of CO_2 emissions per tonne of AHP waste for the BAU and Knowaste scenario

The study realised within the RECALL project (2015), related to the nappy recycling technology developed by FATER SpA as described in Operational data, compares the performances of the recycling process to standard Italian disposal practice, as outlined below:

- BAU consists of a standard Italian waste scenario consisting of waste collection, landfill (65 %) and incineration (35 %), including useful energy recovery from incineration processes and emission offsets due to the storage of carbon in landfill processes (carbon sink)
- The RECALL scenario covers the collection and processing of AHP waste, with the recovery of two flows of materials (plastics and cellulose), the sorting and grinding of plastics which replace propylene production from virgin materials, the sorting and refining of the cellulose fraction for the production of pulp which replaces pulp production from virgin fibres.

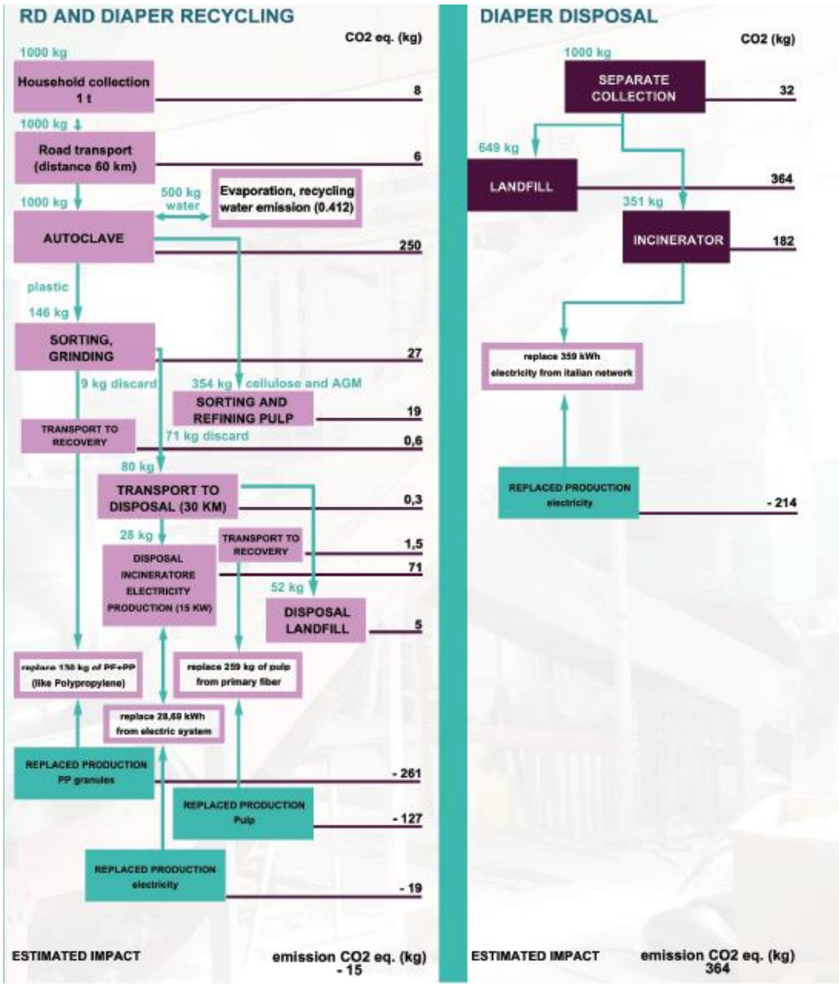
Source: Recall, 2015

Figure 4-75 shows in detail the results of the comparison, considering the different process steps, with reference to one tonne of AHP waste. The main results are also summarised in Table 4-47, which shows how the end-of-life of nappies in the RECALL scenario becomes carbon negative, i.e. the recycling process recovers all greenhouse gas emissions generated by the collection and processing of AHP waste and even saves 14.8 kg of CO_2e per tonne, thanks to the replaced production of polypropylene and pulp for cardboard production from virgin raw materials.

Table 4-47. Main results of the comparison of CO_2 emissions per tonne of AHP waste for the BAU and RECALL scenarios

	RECALL scenario	BAU scenario
kg of CO_2e generated/t	375.4	577.6
kg of CO_2e avoided/t	-390.2	-213.7
CO_2e balance	**-14.8**	**363.9**

Source: Recall, 2015



Source: Recall, 2015

Figure 4-75. Detailed results of the comparison of CO_2 emissions per tonne of AHP waste for the BAU and RECALL scenarios

Both studies also allow the provision of some reference figures related to other environmental benefits.

With reference to the Knowaste process, the study by Deloitte also compared the process performance to that of the BAU scenario considering also, besides global warming, the other environmental indicators defined in LCA methodology. The results show that the Knowaste process has reduced impacts for all other environmental impacts assessed in the study, namely: toxicity impacts for humans reduced by up to 97 %, toxicity impacts for animals and plants reduced by up to 99 %; acid rain impacts reduced by up to 48 %, resource depletion reduced by up to 54 %, eutrophication reduced by up to 93 %.

With reference to the RECALL process, the studies carried out within the Ecoinnovation project show that, for a plant with an annual capacity of 8 000 t/year of AHP waste, the following environmental benefits can be expected:

- recovery of raw materials: 4 000 t/year;
- air quality: 27 kg/year less particulate, 432 MJ/year less nitrogen oxides and 368 kg/year less carbon monoxide (compared with incineration);

- primary energy consumption savings: 18 574 MJ/year (equivalent to the annual electricity consumption of more than 800 families).

Appropriate environmental indicators

As in the case of the other waste treatment techniques, the material sorting rate calculated at the plant scale could be a good indicator for evaluating the environmental performance of the treatment processes of AHP waste. In this case, defining the calculation method is more complex, as the outputs of the AHP waste treatment process are the two flows of recovered materials, corresponding to almost 100 % of the original products making up disposable nappies, and a water discharge, ranging between 50 % and 60 % of the input flow (depending on the composition of the input waste stream), which is sent to a water treatment plant and can be considered a process loss. Given this process mass balance, the resulting plant sorting rate, calculated on the basis of the tonnage after the primary sorting and processing stage and not at the reprocessing stage of the recovered materials, would be 100 % in each treatment plant.

A more precise calculation of the sorting rate defined in this way would require more detailed data provided by the treatment plants by applying specific sampling and testing procedures for assessing the composition of their AHP input waste and output materials, which would also allow the sorting efficiency to be assessed by type of material (plastic, cellulose fibres). In any case, based on the data relating to the input waste flow and water discharge, the sorting rate at the plant scale could be expressed as follows:

$$\text{Plant sorting rate (weight \%)} = \frac{\text{recovered materials sent for recycling (total weight per year)}}{\text{(AHP waste processed - water content in input flow) (total weight per year)}} (\%)$$

where:

water content in input flow = total water discharge – sewage sludge sent to final disposal

The sorting rate calculated in this way would most probably result in a value lower than 100 % (considering the average breakdown of AHP waste provided in Source: Deloitte, 2011

Figure 4-73, it would result in a 90 % sorting rate). Anyhow, this indicator can be calculated only if the AHP waste treatment facility has a dedicated waste water treatment plant, which would allow the quantification of the "solid waste" of the AHP treatment process which will match the volume of the plant sewage sludge. Otherwise, the only data available for calculating the sorting rate would be the total water discharge. If this is the case, the sorting rate at the plant scale could be expressed as follows:

$$\text{Plant sorting rate (weight \%)} = \frac{\text{recovered materials sent for recycling (total weight per year)}}{\text{(AHP waste processed - total water discharge) (total weight per year)}} (\%)$$

which would result in a value very close to 100 %, based on the information available from existing treatment techniques. It should be considered, in any case, that the AHP waste treatment process will also generate a sewage sludge from the water treatment process.

As in the case of the other waste treatment techniques, the environmental performance of the treatment process also depends on the final destination of the recovered materials, i.e. if they are up-cycled or down-cycled. Therefore the sorting rate indicator must also be complemented by the percentage breakdown of the materials sent for recycling by material type (plastic, cellulose fibre), specifying also the final material or product into which the material flow is transformed.

Additional indicators that can be useful for evaluating the environmental performance of the AHP waste treatment techniques are also the energy and water consumption rates (respectively kWh of electricity, Nm^3 of natural gas and m^3 per tonne of input waste), GHG emissions, the quantitative/qualitative emissions to air (odour in particular) and water, that can be significant for this treatment technique. Taking into account also the EMAS core environmental performance indicators, the aforementioned indicators can be expressed as follows.

$$\text{Energy efficiency (kJ/t)} = \frac{\text{total energy consumption (kJ per year)}}{\text{AHP waste processed (total weight per year)}}$$

This can be complemented by also considering the total consumption (%) of energy produced by the organisation from renewable sources.

$$\text{GHG emissions (t CO2e/t)} = \frac{\text{total CO2 equivalent emissions (total weight per year)}}{\text{AHP waste processed (total weight per year)}}$$

The CO_2 equivalent emissions are calculated according to the GHG Protocol Corporate Standard (World Resources Institute and World Business Council for Sustainable Development, 2004), adopted as the basis of ISO 14064, and refer both to the direct greenhouse gas emissions produced by the plant operation (scope 1 according to the reference methodology) and the indirect emission savings related to the substitution of raw materials with secondary material (scope 2 according to the reference methodology, in order to measure the achieved environmental benefits).

$$\text{Water use (m3/t)} = \frac{\text{total water consumption (m3 per year)}}{\text{AHP waste processed (total weight per year)}}$$

As with CO2 emissions, for calculating water use, it would be ideal to include not only the water used in the processing plant, but also the indirect water savings achieved thanks to the substitution of raw materials with secondary materials. This calculation is normally difficult since reliable figures on the water savings from the substitution of raw materials are not easily available to the plant operator. it is therefore meaningful to calculate the water use only accounting for the water used on site.

The indicator on water use can be complemented by other useful information as the source of the water (e.g. surface water, groundwater), the amount of waste water, waste water treated and reused, rainwater and grey-water recycling.

Odour concentration (OUE/m^3) = number of odour units present in one metre cubed at standard conditions, calculated according to the Standard "CEN - EN 13725:2003 Air quality - Determination of odour concentration by dynamic olfactometry", obtained by using specific detection instruments and sampling methods, through periodical (at least biannual) odour monitoring campaigns. This indicator, despite its importance for workers and the neighbours, is less practical and easy to calculate than the previous ones.

As in the case of the other waste treatment techniques, it must be considered that other environmental performance parameters will be also measured at the plant scale, as defined in the plant permits and in the related monitoring plans, according to the national and regional regulations. But for comparing plant performances, within the scope of this document, the aforementioned indicators are considered the most suitable parameters.

Cross-media effects

The operation of AHP waste treatment plants is associated with energy consumption (both electricity and natural gas). Emissions of odour and water discharges are also significant environmental effects. Precise figures about these impacts are not available because of confidentiality issues of the operating plants.

Another cross-media effect associated to the implementation of this BEMP may be the increase in GHG and air emissions due to the need to introduce additional collection routes for this specific waste flow. The relevance of such emissions largely depends on the collection schemes adopted, with a very low potential impact where AHP waste is collected at waste collection centres or via door-to-door collections combined with other waste flows, while it can be more significant in the case of collection schemes specifically for AHP waste. In any case, the analysis of GHG emissions reported in the Achieved environmental benefits section, which includes the waste collection phase in the impact assessment, shows that the potential additional impacts due to such collection are largely offset by the environmental benefits offered by the recycling of the recovered materials.

Operational data

As highlighted in the description of the BEMP, the treatment of AHP waste for improved recycling of materials is an innovative technique which is currently applied in a few small-scale plants, while new ones are already planned in different locations in Europe (in Italy, the Netherlands and the UK).

The operational data provided in this section refer specifically to two treatment technologies that are currently on the market for the treatment of post-consumer AHP waste:

- the Knowaste technology, developed by the UK-based company Knowaste Ltd;
- the RECALL technology, developed by the Italy-based company FATER SpA in the framework of the CIP Ecoinnovation RECALL project.

These are presented in the form of two case studies.

Case study 1 – Knowaste's nappy recycling technology

Knowaste (Knowaste, 2016) is a UK company that has been researching and developing technologies for recycling absorbent hygiene products since the 1990s. In 2011, Knowaste opened a treatment facility in West Bromwich, Midlands (UK), that was operative from 2011 to 2013. It had a capacity of 36 000 t/year and accepted waste from commercial collectors and local authorities. This plant was closed in May 2013 for financial reasons, but the company is now planning to develop a new nappy recycling facility in Hayes, West London, which should be launched in late 2017, as well as further facilities.

The Knowaste technology is a process in which shredded nappies are stirred and washed, separating the nappy fibres and gel from the shell plastics. Within an air-controlled working environment, AHP waste is delivered in a dedicated receiving bay. The waste is shredded, separated and, using advanced thermal treatment technology (autoclave), sterilised. At this stage, the super absorbent polymers are collapsed via a chemical treatment, rendered inert and the moisture is released.

There follows further sorting and separation of plastics and fibres and removal of contaminants. The plastics continue through granulation and multiple washing stages before being pelletised to be used in new products such as plastic components or as an ingredient in composite materials replacing concrete and steel. The fibres are washed, dried and processed for use in pet litter, concrete and tarmac additive, brick manufacture and insulation materials. Of the original components of the AHPs, 100 % can be recovered and sent for recycling, with the remaining solids and liquids sent for treatment.

Knowaste produced a video to present their technology. It is available at: https://www.youtube.com/watch?v=etz5DFY-HfQ

Case study 2 – RECALL's nappy recycling technology

The RECALL nappy recycling technology has been developed by the Italian Company FATER SpA in the framework of the CIP Ecoinnovation RECALL project (RECALL, 2015; http://www.recall-ecoinnovation.eu/).

The post-consumer AHP waste is treated in a simple process: through the effect of steam and pressure in an autoclave, post-consumer AHP waste is sanitised, decomposed and dried, leading to the recovery of its valuable components (cellulose and plastic). The autoclave, which is the core of the plant, is a horizontal cylindrical vessel, which is heated by means of a jacket of saturated “no-contact” steam. The injection of another stream of “contact” steam provides the necessary sterilisation of the AHP waste, which is continuously mixed by the rotation and alternative oscillation of the vessel. The treatment scheme utilises methane from the grid to produce the steam necessary for the sterilisation process (Arena et al., 2016). Prior to entering the autoclave, AHP waste is stored within an air-controlled receiving bay, equipped with a biofilter and a leachate basin. The sanitised flow exiting the autoclave is shredded and mechanically treated for the recovery of cellulose fibre and plastic.

The recovery rate is almost equal to 100 % of the original fractions in the AHP, and the recovered fraction is of a high quality, because high-quality plastic and cellulose are used to manufacture nappies. From the evidence available, the recovery of about 146 kg of plastics and 345 kg of cellulose plus super absorbent polymers per tonne of AHP waste is observed, representing respectively 15 % and 35 % of the input flow, the rest being discharged process water sent to the treatment plant.

The outputs of the RECALL process are two high-quality reusable materials:

- plastic, which is extruded and takes the form of small beads and can be reused in several production cycles in a wider range of processes with no colouring issues;
- cellulose, which can be used for the production of absorbent products, pet litter, high-quality paper, chemical building blocks, agricultural nutrients and energy production via gasification.

Outlets from Recall recycled plastic – Source: Recall, 2015

Outlets for RECALL recycled cellulose

Source: Recall, 2015

The first RECALL treatment plant (8 000 t/year treatment capacity) was installed in 2015 within the recycling site of the public waste management company Contarina, located in Lovadina di Spresiano (TV), in the North-East of Italy. Currently it is authorised to operate as an experimental plant (Veneto Region, 2016) at a maximum capacity of 1 500 t/year. Further improvements to the recycling process are being implemented in order to optimise the plant efficiency (i.e. reduce energy consumption) and the quality of the recovered materials.

AHP Recycling plant in Lovadina di Spresiano, operated by Contarina

Source: Recall, 2015

Applicability

This BEMP is broadly applicable as no particular geographical or technical barriers exist and the implementation of the BEMP is perfectly in line with the EU waste legal framework.

However, some specific framework conditions can influence the technical and economic viability of this treatment option (Recall, 2015):

- The plant operation requires the implementation of a selective collection scheme for AHP waste, from households or other relevant producers of this waste fraction (retirement homes, hospitals, kindergartens), in order to secure a stable input flow to the plant. Such schemes are already in place in only a limited number of municipalities across the EU, in particular in areas with door-to-door waste collection schemes, especially if accompanied by PAYT charging systems[64].
- The plant treatment capacity, based on the available treatment techniques and considering the economics of the treatment process, cannot be lower than 8 000 t/year.

[64] The collection of AHP waste is particularly developed in Italy, where a selective collection for this waste stream is already implemented in almost 600 municipalities covering a population of over 8 million inhabitants. In the UK the AHP collection systems in place are run mainly by private companies specialized in the healthcare waste sector (OCS/Cannon Hygiene, Initial, SRCL, PHS), but in the last years also some Local Authorities are implementing separate collection of AHP waste produced by households (e.g. Cardiff and Monmouthshire, in south east Wastes, or some local authorities in Scotland) (Recall, 2015).

- The transport distance from collection areas to the plant, AHP waste collection schemes as well as local costs for landfilling and incineration must be carefully taken into account when planning an AHP waste recycling facility, as they can significantly influence its economic feasibility.
- Population density in the collection area is also an important parameter to be considered, as it can significantly influence the maximum AHP input flow to the plant and consequently its economic sustainability. From literature estimates (Recall, 2015), it can be cautiously assumed that territorial areas of about 1 million inhabitants can generate at least 10 000 t/year of AHP post-consumer waste. If these inhabitants are spread over a large territory (low population density), transport costs from collection areas to the treatment plant can become too high for the plant to be economically sustainable. This aspect then needs careful evaluation during the planning phase.
- All the aforementioned factors affect the AHP waste plant capacity.
- Clear criteria and rules for recognising the end-of-waste and local market for recovered materials (plastic and cellulose) are needed. This condition guarantees the presence of a receptive market for the secondary raw materials produced by the recycling plant and consequently its environmental and economic sustainability.

Economics

In the absence of reference literature on the economics of AHP waste treatment or actual economic data from operating plants, some evaluations have been produced considering different scenarios and applying specific estimations, based on information provided by experts.

The economic viability of an AHP waste treatment process has been evaluated estimating the potential expenditures and income sources, considering treatment plants operating at different throughputs and under different framework conditions. The estimations have been based on the following assumptions:

- three different case studies have been considered, corresponding to treatment facilities with treatment capacities respectively of 10 000 t/year, 25 000 t/year and 40 000 t/year;
- gate fees for the incoming AHP waste range from a minimum value of EUR 90 per tonne, and a medium value of EUR 110 per tonne, to a maximum value of EUR 130 per tonne;
- revenues from the two recovered materials also range from minimum to maximum unitary values, and are calculated assuming a recovery rate of 100 % of the original AHP components (estimated as 40 % of the AHP waste input flow), with the following percentage breakdown for the materials: 28 % cellulose fibres and 12 % plastics.

These assumptions are summarised in Table 4-48 and Table 4-51.

Table 4-48. Assumptions for the basis of the economic viability evaluation of AHP waste treatment processes

Treatment capacity			
Type of plant	**Low capacity**	**Medium capacity**	**High capacity**
Plant capacity (t/year)	10 000	25 000	40 000

Range for gate fees - unitary values			
	Min.	Med.	Max.
Gate fee (EUR/t)	90.00	110.00	130.00
Range for revenues from recovered materials - unitary values			
	Min.	Med.	Max.
Revenues for plastic (EUR/t)	200.00	250.00	300.00
Revenues for cellulose fibres (EUR/t)	100.00	200.00	300.00
Recovery rates for the different materials (% weight with respect to input flow)			
Recovery rate for plastic (%)	12		
Recovery rate for cellulose fibres (%)	28		

Source: own elaboration

The gate fees and revenues have been defined from information gathered during this study by interviews with plant operators. As for the gate fees, the unitary values have been set considering also the need to keep them of the same order, or at least not much higher, than landfilling[65] or incineration fees. As for the potential income from recovered materials, the range of values assumed is affected by many uncertainties, as the market value of recyclates is very volatile and highly dependent on their quality, and at present – given the innovative nature of nappy recycling processes – reliable data related to actual case studies are not yet available. The market value for recycled plastic can be defined based on current reference values for high-quality plastics recovered from packaging/plastic sorting, but the market value of the cellulose fibres is more difficult to determine given the different potential uses and the lack of reference data for similar secondary raw materials. The results of the estimates provided must therefore be considered a general reference, affected by substantial uncertainties.

Based on the assumptions described above, the profit and loss accounts for the three different case studies have been simulated, calculating the expected annual revenues and expenditures, assuming investment costs ranging between EUR 4.5 million for the low-capacity facility, EUR 9 million for the medium-capacity facility and EUR 13 million for the high-capacity facility, with amortisation periods of 10 years. The results of the simulation are shown in Table 4-49.

Table 4-49. Estimation of incomes and expenditures for the three case studies

Type of plant	Low capacity	Medium capacity	High capacity
Plant capacity (t/year)	10 000	25 000	40 000
Revenues from gate fees - Annual values (EUR/year)			
Min. gate fee	900 000.00	2 250 000.00	3 600 000.00
Med. gate fee	1 100 000.00	2 750 000.00	4 400 000.00
Max. gate fee	1 300 000.00	3 250 000.00	5 200 000.00

[65] Indeed, landfilling fees are much lower in some EU Countries, as shown in Source: EEA, 2013 **Figure 4-71** reported in the BEMP about mattresses treatment, but it is assumed that in future they should increase because of landfill phasing out policies

Type of plant	Low capacity	Medium capacity	High capacity
Revenues from recovered materials - Annual values (EUR/year)			
Min. revenue values	520 000.00	1 300 000.00	2 080 000.00
Med. revenue values	860 000.00	2 150 000.00	3 440 000.00
Max. revenue values	1 200 000.00	3 000 000.00	4 800 000.00
Investment and operating costs			
Investment costs (EUR)	4 500 000.00	9 000 000.00	13 000 000.00
Annual amortisation (%)	10	10	10
Total annual amortisation (EUR/year)	450 000.00	900 000.00	1 300 000.00
Annual operating expenditures (EUR/year)	850 000.00	2 125 000.00	3 400 000.00
Annual labour costs (EUR/year)	374 400.00	561 600.00	748 800.00
Other annual general costs (EUR/year)	167 440.00	358 660.00	544 880.00
Total annual expenditure (EUR/year)	**1 841 840.00**	**3 945 260.00**	**5 993 680.00**

Source: own elaboration

As can be observed, the majority of income arises from the gate fee, but the revenues are also relevant, in particular in the scenario with the maximum values. As for the expenditures, major costs are mainly determined by the plant operation (raw materials, energy, waste water treatment, maintenance, etc.), as the process is not labour-intensive. Investment costs are also significant, in particular in the case of the high-capacity facility.

Based on the incomes and expenditures estimated, the net cash flow before taxes has been calculated for the three case studies, considering all possible combinations of the framework conditions. The results are shown in Table 4-50.

Table 4-50. Net cash flow before taxes estimated for the three case studies

LOW-CAPACITY TREATMENT PLANT				
Net cash flow before taxes		**Gate fee revenues**		
		Min.	**Med.**	**Max.**
Material revenues	**Min.**	-EUR 421 840.00	-EUR 221 840.00	-EUR 21 840.00
	Med.	-EUR 81 840.00	EUR 118 160.00	EUR 318 160.00
	Max.	EUR 258 160.00	EUR 458 160.00	EUR 658 160.00
MEDIUM-CAPACITY TREATMENT PLANT				
Net cash flow before taxes		**Gate fee revenues**		
		Min.	**Med.**	**Max.**
Material revenues	**Min.**	-EUR 395 260.00	EUR 104 740.00	EUR 604 740.00
	Med.	EUR 454 740.00	EUR 954 740.00	EUR 1 454 740.00
	Max.	EUR 1 304 740.00	EUR 1 804 740.00	EUR 2 304 740.00
HIGH-CAPACITY TREATMENT PLANT				
Net cash flow before taxes		**Gate fee revenues**		
		Min.	**Med.**	**Max.**
Material revenues	**Min.**	-EUR 313 680.00	EUR 486 320.00	EUR 1 286 320.00
	Med.	EUR 1 046 320.00	EUR 1 846 320.00	EUR 2 646 320.00
	Max.	EUR 2 406 320.00	EUR 3 206 320.00	EUR 4 006 320.00

Source: own elaboration

The net cash flow before taxes is always positive in the event that the maximum or medium value for the gate fee is applied (with the only exception being for the low-capacity plant with minimum values of the revenues from recovered materials), while it is negative or very low in the event that the minimum value for the gate fee is applied with minimum or medium values of the material revenues or in any case when the minimum material revenues are assumed. In general, the financial performance of the high-capacity plant is better than that of the medium- or low-capacity plant, under each set of framework conditions.

Considering the best case for the three case studies (maximum gate fee and maximum material revenues) and assuming that the taxes would be about 30 % of the net cash flow, we can estimate the following payback periods for the treatment plants:

- low-capacity treatment plant: 10-year payback period, with earnings after taxes of about EUR 460 000 per year;
- medium-capacity treatment plant: 6-year payback period, with earnings after taxes of about EUR 1 600 000 per year;
- high-capacity treatment plant: 5-year payback period, with earnings after taxes of about EUR 2 800 000 per year.

These evaluations show that, in the best case scenario, all the types of facilities are financially viable. Sustainable economic performances are also achieved for the medium- and high-capacity facilities in three of the intermediate scenarios (maximum gate fee and medium material revenues; medium gate fee and maximum material revenues; minimum gate fee and maximum material revenues), with the most sustainable performance achieved in the case of the high-capacity plant, which can be also financially viable in the case of a medium gate fee and medium material revenue.

Based on this analysis, the following conclusion can be drawn: the financial viability of an AHP waste treatment plant can be considered more robust where a relevant flow of AHP waste and a good market value for the recovered materials can be assured. It must be noted that a treatment facility with a high or even medium capacity (> 20 000 t/year) could face feeding constraints, as it is estimated that to collect about 10 000 tonnes per year of AHP waste an area of about 1 million inhabitants is required (Recall, 2015).

Given these data, in the assessment of the economic viability of the treatment plant it is also important to consider the collection and transport costs of AHP waste, i.e. the economic viability of the BEMP implementation from the point of view of local authorities or private waste collection operators, which would provide the necessary input flow to the treatment facilities. The costs of waste collection and transport have thus also been estimated, under different scenarios characterised by the following parameters:

- collection costs for four different collection schemes: door-to-door (specific and combined with other waste fractions), street bins (opened by key provided to households requiring the collection service), waste collection centre;
- distance from the plants ranging between 50 km and 100 km and 150 km.

The costs under the different scenarios are simulated considering a small town of about 10 000 inhabitants in a flat territory with a medium population density, where 10 % of the households need a nappy waste collection service.

Based on these assumptions and considering reference data regarding the productivity of the different collection systems, the following collection costs are estimated.

Table 4-51. AHP waste collection unitary costs under different collection systems

COSTS	Door-to-door collection (specific for AHP waste)	Door-to-door collection (combined with another waste flow)	Street bin collection	Waste collection centre
Collection cost per tonne of AHP waste (EUR/t)	119.00	63.00	55.00	14.00

Source: own elaboration

Transport costs are estimated considering that most of the territories that send their AHP waste to the recycling plant need to collect them in a transfer station in order to optimise the transport costs, and that weekly transfers are realised from each transfer station to the recycling plant, by using a dump track. Transport costs in the simulated case study are then estimated considering the following unitary costs:

- transfer with average distance to the plant of about 50 km: EUR 200/container;
- transfer with average distance to the plant of about 100 km: EUR 350/container;
- transfer with average distance to the plant of about 150 km: EUR 425/container.

Based on these assumptions, the following transport costs are estimated.

Table 4-52. AHP waste transport costs considering different average distances from the transfer station to the recycling plant

Average distance between transfer station and recycling plant	Cost per trip (EUR)	Number of trips per year	Annual cost (EUR)	Unitary cost (EUR/t)
50 km	200.00	52	10,400.00	45.22
100 km	350.00	52	18,200.00	79.13
150 km	425.00	52	22,100.00	96.09

Source: own elaboration

To complete the analysis of the economic viability of the BEMP implementation from the point of view of local authorities, the treatment costs for sending AHP waste to the treatment plants must be also considered, based on the gate fees applied by the plants. Three different scenarios have been thus assumed (considering gate fee ranges as defined in Table 4-48):

- <u>best scenario</u>: average transport distance = 50 km; gate fee = EUR 90/t;
- <u>medium scenario</u>: average transport distance = 100 km; gate fee = EUR 110/t;
- <u>worst scenario</u>: average transport distance = 150 km; gate fee = EUR 130/t.

The final results of the cost analysis, under the three different scenarios, are reported in Table 4-53. The costs include AHP waste collection, transport to the treatment plant and treatment.

Table 4-53. Costs for AHP waste collection, transport and treatment under three different scenarios

EUR/tonne	Best scenario	Medium scenario	Worst scenario
Door-to-door collection (specific for AHP waste)	254	308	345
Door-to-door collection (combined with another waste flow)	198	252	289

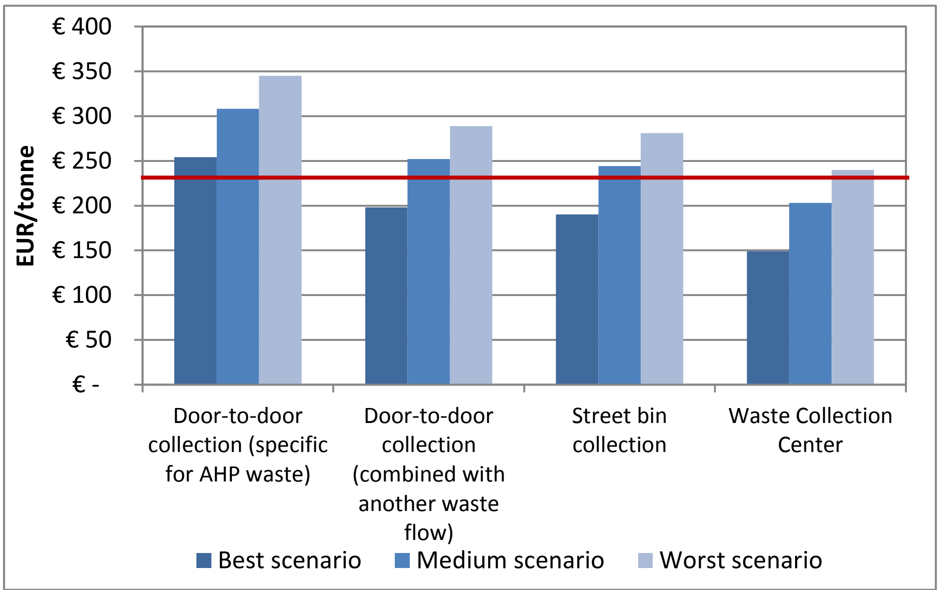
EUR/tonne	Best scenario	Medium scenario	Worst scenario
Street bin collection	190	244	281
Waste collection centre	149	203	240

Source: own elaboration

The economic viability of the BEMP implementation, from the point of view of local authorities, is highly dependent on the costs of the municipal waste management services in each specific context, which largely depend on the costs of landfilling and/or incineration (gate fees and taxes).

Assuming a reference context where the average waste collection and transport costs are ~ EUR 130/t and landfilling/incineration costs are on average ~ EUR 110/t (which is the case, for example, of Italy, according to Utilitalia, 2016), the total cost for the municipal waste management service is ~ EUR 240/t. Under these conditions, the best scenario for separate collection and recycling of AHP waste appears convenient, except in the case where a door-to-door collection specifically for AHP waste is implemented. And it also appears convenient or at least comparable with the status quo in the medium scenario, except in the case where door-to-door collection services are implemented, as shown in Source: own elaboration

Figure 4-76.



Source: own elaboration

Figure 4-76. Cost analysis of AHP waste collection, transport and treatment under different scenarios

Driving force for implementation

The main driver for the implementation of this BEMP in Europe is the EU waste legal framework, and in more detail:

- the binding targets for municipal waste recycling (> 50 % by 2020 according to the Waste Framework Directive, 60 % by 2030 according to the proposal for the revision of the Waste Framework Directive introduced by the Circular Economy Package);

- the binding targets for reducing waste landfilling (targets for reducing the amount of biodegradable municipal waste landfilled introduced by the EU Landfill Directive (1999/31/EC); proposal for a binding target to reduce landfill to maximum of 10 % of municipal waste by 2030 introduced by the Circular Economy Package).

Since the introduction of these targets, in many EU countries a lot of effort has been devoted to achieving higher recycling rates and in some regions ambitious targets have already been achieved (recycling > 70 % and residual waste sent to landfill almost zero). Under these conditions, the AHP waste can represent up to 15–20 % of the residual waste, and its diversion from landfills or incinerators can be the best option for further reducing the residual waste.

Another driving force that is being observed in the territories with door-to-door (or kerbside) waste collection schemes, especially if accompanied by PAYT charging systems, is the operative and social problem encountered by households producing AHP waste. In these contexts, considering that the residual waste is usually collected no more than once a week, many municipalities are forced to introduce the selective collection of AHP waste, independently from the existence of recycling solutions, because households producing AHP waste need a more frequent service for this waste stream. Moreover, this household category produces a higher amount of residual waste, which is generally the only waste fraction that is charged when PAYT schemes are in place. It is therefore the case that these households pay high fees for the waste collection service, for a waste fraction that they can hardly reduce. The introduction of selective collection systems for AHP waste, which would remove this waste fraction from residual waste, also becomes the solution to this social problem.

These factors create favourable conditions for the implementation of this BEMP. Indeed, the construction and operation of AHP waste treatment plants becomes the solution by providing the opportunity to recover materials from this waste stream that otherwise would be sent to final disposal, even if separately collected.

Reference organisations

The reference organisations identified for this BEMP are the following:

- Contarina SpA: a public waste management company that operates the first recycling plant based on the RECALL recycling technology developed by FATER SpA. More details can be found at: http://www.contarina.it/chi-siamo/impianti/riciclo-prodotti-assorbenti
- Fater SpA: an Italian company, and an industrial leader in the production of absorbent hygiene products, that has developed the RECALL Recycling technology in the framework of the CIP Ecoinnovation RECALL project (Recall, 2015). More details can be found at: http://www.fatergroup.com/it/news/progetti/progetto-riciclo
- Knowaste Ltd: a UK company that has developed, since the 1990s, a nappy recycling technology and has built the first operating plants in Europe. More details can be found at: http://www.knowaste.com/

Besides these reference organisations that are already implementing AHP waste treatment techniques for material recycling, it is worth also mentioning Diaper Recycling Technology Pte. Ltd, a Singapore-based company that has developed a treatment technology that currently converts absorbent hygiene factory scraps back into the original raw material streams (including cellulose pulp, SAP and PE/PP). Research and development efforts are under way to expand the applicability of this newly developed technology to post-consumer AHP waste (Nonwovens Industry, 2016). More details can be found at: http://www.diaperrecycling.technology/.

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5. Construction and demolition waste (CDW)

5.1. Introduction

Construction and demolition waste (CDW) is a waste stream characterised by its very high volume and weight (34 % of the total waste in Europe), but with probably the lowest environmental burden and the highest inert fraction. However, the management of construction and demolition waste is still the main focus of many environmental programmes around the world, especially in Europe in recent years (e.g. FP7-funded projects such as C2CA and IRCOW[66] and Horizon-2020-funded projects such as FISSAC, Hiser and BAMB[67]), where a recycling rate of 70 % for construction and demolition waste was established in the Waste Framework Directive and included in the proposal for an amended Directive (EC, 2015). The industry, however, has pointed out that national circumstances are heterogeneous in European Member States and that the Waste Framework Directive is no longer an incentive for the industry of those countries or regions where the 70 % recycling rate benchmark was superseded a long time ago (Craven, 2015).

The management of waste from construction and demolition sites, and the technological options for its treatment and recycling are well defined and described in the Technical Report on Best Environmental Management Practice for the Building and Construction Sector (EC, 2012). Most of those techniques were oriented to construction site managers, although developers, public administration, waste managers and all the actors involved in the end-of-life stages of buildings are also part of the target audience of that document.

This chapter focuses on the involvement of waste authorities and waste organisations directly or indirectly responsible for the main environmental aspects of CDW. Since part of the logistical aspects, on-site management and treatment operations are already covered in the building and construction sector document, this chapter is simplified and oriented to fill the gaps and extend the scope of the treatment options described in that document. The main gaps identified for public administration, and waste management and treatment organisations are listed below:

- formulation of specific local, county and regional plans for construction and demolition waste, including the quantification of waste generated, the required treatments and the integration with final users;
- the management of CDW for the production of secondary raw materials (e.g. recycled concrete aggregates);
- the management of hazardous substances, with a specific focus on PCB- and asbestos-containing wastes, where new approaches are being developed.

[66] More information on these two FP7 projects are available at: http://www.c2ca.eu/ and http://www.ircow.eu/
[67] More information on these there H2020 projects are available at: - https://fissacproject.eu/en/, http://www.hiserproject.eu/ and http://www.bamb2020.eu/

These issues are addressed in five BEMPs presented in the next sections of this chapter.

5.2. Technique portfolio

A full list of best practices presented in this document and already described in the technical report on best environmental management practice for the building and construction sector (EC, 2012) are provided in Table 5-1.

Table 5-1. Techniques portfolio for the management of construction and demolition waste

Management aspect	Techniques in this document	Section in the Building and Construction document (EC, 2012)
Strategy	1. Integrated Construction and Demolition Waste Plans 2. Avoidance of polychlorinated biphenyls (PCBs) contamination of construction and demolition waste	Section on site waste management plans (5.6.2.1)
Prevention	-	Section on designing out waste (3.4.7), Section on site waste prevention and management (5.6.2.1) Section on material use efficiency (5.6.2.2)
Collection	3. Local schemes for proper management of waste asbestos removed by residents	Section on site waste prevention and management (5.6.2.1) Section on selective deconstruction of buildings (7.3.1) Section on selection of environmentally friendly deconstruction / demolition techniques (7.3.2)
Reuse	-	Section on reuse of materials (5.6.2.3)
Treatment	4. Processing waste plasterboard to foster recycling 5. Processing of CDW for the production of recycled aggregates	Section on construction and demolition waste sorting and processing (7.3.3) Section on use of recycled materials (5.6.2.4)

Reference literature

Craven, P. (2015). Are current EU C&D waste recycling targets an obstacle to growth? Waste Management World, Feb 2015. Available at www.waste-management-world.com, last access September 2017.

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European Commission, EC (2015). Proposal for a Directive of the European Parliament and of the Council amending Directive 2008/98/EC on waste. Available at www.ec.europa.eu, last access December 2015.

5.3. BEMPs about waste in the Technical Report on Best Environmental Management Practice for the Building and Construction Sector

The Technical Report on Best Environmental Management Practice for the Building and Construction Sector (EC, 2012) gathers a set of BEMPs for the whole value chain of the construction sector, from inception to execution of construction projects, and for the whole life cycle of buildings, from raw materials to end-of-life of buildings.

Within the many aspects covered in the document, an important number of BEMPs actually cover waste-related techniques. A summary is provided in the table below (Table 5-2).

Table 5-2. Best Environmental Management Practice related to waste from the Sectoral Reference Document for the Building and Construction Sector

Section	BEMP	Summary
Building design	Designing out waste	Preventive design (or designing out waste, as defined by WRAP) consists of minimising waste at every stage of the life cycle of a building construction during its design. The identification of opportunities for waste prevention during design activities and the implementation during its construction or use are considered best practices. The most common preventive measures would consist of the use of prefabricated elements, modern methods of construction, rental and reuse of auxiliaries (e.g. scaffolds, formworks), reduced requirement of cuttings through smart design, etc.
	Design for deconstruction	Design for deconstruction is a technique that considers the implementation of key design features for the easy disassembly of construction elements and the planning for possible reuse of construction elements. Some key concepts are followed in the implementation of this BEMP: transparency (all elements are visible), regularity (same materials are used for the same applications), simplicity, limited number of materials and components and easy-to-separate materials.
Building construction	Waste prevention and management	This BEMP is an overarching technique that gathers all possible practice in the management of waste on site and its prevention. The establishment of waste management plans for sites (which is mandatory in several European Member States), the monitoring of waste generation, and the establishment of waste separation and collection strategies are the main features of this BEMP.
	Material use efficiency	Regarding the important loss of materials during construction due to inefficiencies in handling, this BEMP is oriented to techniques for the improvement of the logistics of materials, management of remains and storage and handling practices. Consolidation centres for material delivery (and in some cases for waste handling) are also considered under this BEMP.

Table 5-2. Best Environmental Management Practice related to waste from the Sectoral Reference Document for the Building and Construction Sector

Section	BEMP	Summary
	Reuse of materials	This is a BEMP that can be performed with materials, products or auxiliary materials that are harvested on site. In the case of construction materials, it refers to bricks, tiles, slaps, beams, etc., and for auxiliary materials the technique can be easily applied to pallets, formworks, auxiliary structures, etc.
Building end-of-life	Selective deconstruction	This is a technique oriented to the economical optimisation of the systematic disassembly of buildings in order to maximise the reuse and recycling of recovered materials. This technique should consider building reuse as a priority before deconstruction and the reclamation of materials should also be oriented for *in situ* practices, e.g. recycling, in order to avoid the impact from its transport.
	Selection of environmentally friendly deconstruction and demolition techniques	Best recovery rates are usually achieved through manual stripping and using light machinery; however, the economic balance is usually against slow stripping and reclamation processes. The description focuses on all techniques, from manual to explosive demolition,their well-known economic performance and the environmental benefit of material reclamation achieved by each one.
	Construction and demolition (deconstruction) waste sorting and processing	The main focus of this BEMP is the separation and processing of separated mono-fractional waste streams, both at mobile or stationary plants. Separation, processing techniques (e.g. screening, crushing) and quality assurance of materials made from recycled materials are described in this BEMP.

The target group of the building and construction document differs substantially from the target group of this report. The first document is oriented to all construction stakeholders (designers, developers, contractors, etc.). However, there are wide overlaps; while waste management organisations will manage wastes derived from construction activities, the waste management practice on site is key for its recovery. Then, well-sorted waste, with a minimum level of impurities, can be fully recycled. A poor performance of on-site management practices directly affects the recovery of materials and their quality. A good example is the separation of plasterboard and gypsum, which, if not separated, are extremely detrimental to the application of recycled aggregates in new construction.

The different levels of interaction between waste management organisations, construction contractors and public authorities, and the availability of natural materials and economic instruments has developed a relatively heterogeneous map of practices for CDW in Europe. With that in mind, the best practices shown in this document and in the Technical Report on Best Environmental Management Practice for the Building and Construction Sector (EC, 2012) try to draw a general picture of frontrunners' achievements.

5.4. BEMPs for construction and demolition waste

5.4.1. Integrated construction and demolition waste plans

Summary overview							
It is BEMP for local authorities to develop and implement integrated CDW plans that: - Involve stakeholders from the local construction industry, representatives of residents, local business associations, and relevant public actors; - Prioritise waste prevention in construction projects through instruments oriented to the industry and public administration, such as a demolition code of practice and promotion of appropriate green public procurement provisions; - Establish minimum requirements for waste sorting and management in construction sites of a certain size, e.g. requirements for a site waste management plan (SWMP), or required fractions to be separated; - Identify and quantify future flows of waste, ensure the local urban development plan allocates sufficient areas for collection and treatment of CDW; - Calculate the total costs and the impact of implementation; - Establish more ambitious objectives than the EU or national CDW recycling targets as well as appropriate monitoring and enforcement mechanisms; - Include measures to avoid illegal dumping and provide clear guidance (e.g. for SMEs, residents and producers of very small quantities of CDW) on correct CDW management practices.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
The formulation and implementation of local waste management plans for CDW is a commonly used instrument by regions and large municipalities.							
Specific environmental performance indicators							
- Share of total collected CDW that is correctly segregated and managed towards reuse, recycling or recovery (%). - Provision for pre-demolition audits aimed at reuse (y/n).							
Benchmark of excellence							
- An integrated CDW management plan is implemented with a target CDW recycling rate in 2020 of at least 80 % and provisions for monitoring and enforcement mechanisms.							

Description

The drawing up of integrated waste management plans or strategies is a common approach in local, county, and regional governments. However, waste authorities are not the only ones responsible for their implementation through mandatory or voluntary approaches. In many locations in Europe, recycling of construction and demolition waste (CDW) has become a privately driven activity. Its performance is dependent on the existence of certain drivers, e.g. taxes or levies on natural materials, regulations, standards, enforcement practices and awareness. All these elements need to be considered in an integrated plan for construction and demolition waste at national level, as CDW is the most important waste in terms of volume. At national level, plans should identify recycling opportunities and provide realistic frameworks for the industry for its implementation. For instance, the use of recycled aggregates from CDW is encouraged through the natural aggregates levy or tax, which has proven effective if both a legal and normalised standardised approach exists, e.g. mandatory (Netherlands) or voluntary (Germany).

In addition, a regional plan, which implements those policies, identifies and quantifies the collection and treatment needs required to achieve national objectives. Such identification and quantification of CDW needs then to be considered and reflected in the urban development plan at local level, ensuring sufficient areas are allocated for collection and treatment of CDW.

After waste is transported, the main facilities involve sorting, crushing, screening and, in some cases, disposal. The optimisation of the size of these treatment plants and the use of mobile plants at local level increase the amount of waste diverted from landfills, while reducing the costs of transport and management. In this way, some regions in Europe have provided the industry with tools for the safe use of recycled aggregates. The most important is the existence of quality assurance schemes for secondary materials, which have demonstrated their ability to open markets to recycled materials in the construction sector.

The drafting of a local plan or a strategy to manage CDW is not a best practice per se but a necessity and a very common approach. At local level, the main focus of this BEMP, a specific approach should be defined for the minimisation and management of CDW by the local waste authority. However, it is recognised that it will be dependent on the provisions at regional level, e.g. waste authorities can establish minimum sorting requirements through their permits if the infrastructure for its transport and treatment exists at regional level.

Regarding best practice CDW plans, a local authority does the following:

- It Involve stakeholders from the local construction industry, representatives of residents, local business associations, and relevant public actors.
- It prioritises waste prevention in construction projects through several instruments, oriented both to the industry and public administration. For instance, through green policies of public procurement (see GPP case studies at http://ec.europa.eu/environment/gpp/ for Vienna, Hamburg, etc.), municipal building reuse schemes (ICE, 2008), and other tools oriented to the avoidance of construction waste at source. When the main focus of the construction activity is demolition the strategies are similar, but it involves a higher volume of wastes. An example of an integrated plan for demolition is the Dutch demolition code of practice developed by VERAS (VERAS, 2015). The Code for Responsible Commissioning and Contracting during the Tendering and Execution of Demolition works describes best practice for professional clients, contractors and other

stakeholders during the tendering of demolition projects. It includes all types of criteria, with a special emphasis on the availability of information and transparency. From the performance point of view, environmentally friendly demolition practices are also encouraged, among other activities related to safety and corporate social responsibility policies in the preparation and execution of demolition projects.

- It establishes minimum waste sorting and management requirements in construction sites of a certain size. The most popular measure is the site waste management plan (SWMP), which is mandatory in several regions of Europe for works over a certain size. However, best practice performance has not been achieved in those countries with mandatory SWMP, but its implementation has increased the amount of waste diverted from landfill (e.g. UK, Spain, Italy) along with other measures. Even in regions without a legal requirement, local government, through their permitting activities for construction sites, can enforce the implementation of waste management plans for sites (e.g. see section 5.4.2). For example, Frankfurt includes an extended range of construction waste separation requirements for new municipal buildings: mineral mixed construction waste, metals, synthetic foam, foam insulation, plastic foils, solid wood and untreated timber, and hazardous wood materials (such as sound absorbers, medium-density fibre boards, and glued laminated timber) (Frankfurt, 2013).
- It defines a performance baseline, based on actual quantifiable data and empirical observations.
- It identifies and quantifies future flows of wastes, ensuring the urban development plan includes sufficient areas for the collection and treatment of current and future CDW. There are no common approaches for CDW quantification. Wu et al. (2014) identified several waste quantification methods: Per-capita multipliers, financial value extrapolation, area-based calculation, building lifetime analysis, materials lifetime analysis, classification system accumulation, variables modelling method. Most of these methodologies are site-oriented (i.e. they identify waste flows within a site) but have helped the development of regional-oriented approaches through the application of combined approaches. For instance, per-capita multipliers are used for national-level forecasts, and financial value extrapolations or area-based calculations are frequently used at regional and county levels.
- It calculates the total costs and the impact of implementation.
- It establishes more ambitious objectives than the EU or national CDW recycling targets as well as appropriate monitoring and enforcement mechanisms. Two examples were identified by Gradman et al. (2013) for the Committee of Regions: Wales, with a recycling target of 90 %, and the city of Copenhagen with an achieved recycling rate of 88 %.
- It includes measures to avoid illegal dumping and provide clear guidance (e.g. for SMEs, residents and producers of very small quantities of CDW) on correct CDW management practices.

Planning of CDW management should consider all the stages in a construction project (WRAP, 2011a). In Figure 5-1, the relationship between waste minimisation and management strategies and the construction activity is shown.

Source: Adapted from WRAP, 2011a

Figure 5-1. Construction and demolition waste strategies in relation to construction projects' life cycle

As stated in the EMAS Sectoral Reference Document on Best Environmental Management Practice for the Building and Construction Sector (EC, 2012), the best opportunities for waste prevention and minimisation are provided during the initial stages (pre-design, tendering and design) along with the use of recycled materials, while waste management activities are focused on the on-site construction activity. Construction companies usually manage and transport an important amount of wastes, and usually need a waste manager permit to operate their own sites.

Achieved environmental benefits

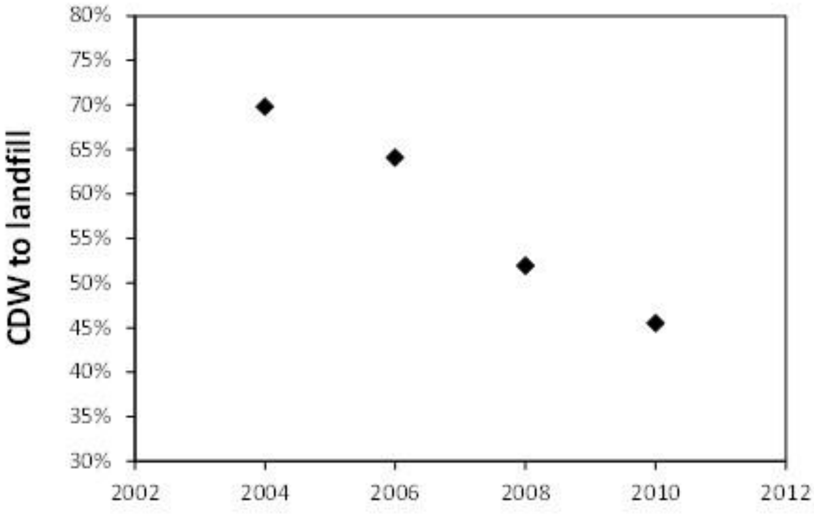
The impact of plans is not easily quantifiable, as they enable a number of techniques, which are applied with different degrees of success, and the influence of those plans on their application is uncertain. As an example, the avoided impact by the proper application of waste sorting techniques, through site waste management plans, in a case study in the UK during the building of a commercial centre (project value GBP 150 million) is shown in Table 5-3.

Table 5-3. Waste diverted from landfill in a best environmental management case in the UK

Material	Recovery rate (%)	Tonnes diverted from landfill / GBP 100K	Avoided GHG emissions kg CO_2e / GBP 100K
Concrete	100	0.5–0.6	0.25–0.3 (avoided aggregate only)
Timber	90	0.1–0.15	40–60 (non-biogenic emissions)
Metal	100	0.1–0.15	150–250 (assumed to be reinforcement steel)

Source: Own estimations, carbon footprint of materials from ICE (2012)

As an example of the impact of several policy instruments, a big change over the last decade in the UK in the amount of CDW going to landfill can be observed in Figure 5-2 (see *Implementation of national strategies at local level* in Operational data for details of the applied instruments).



Source: Data from Defra (2011)

Figure 5-2. Construction and demolition waste going to landfill in the UK

However, it is challenging to differentiate the impact of isolated waste management plans of single cities or communities, since the statistics are usually generated at treatment centres, without any differentiation of the origin of wastes. The application of certain policies, partially developed through these management plans, has been reported in a case study on Westmeath County Council. There, the green public procurement of city infrastructure, a civic amenity centre, considered the use of recycled aggregates from CDW treatment plants, using 4 200 m^3 of recycled concrete aggregate for the concrete formulations, plus smaller amounts of recycled rubber and asphalt for landscaping purposes. One of the main benefits, however, was considered the increased environmental awareness and the availability of more sustainable materials in the local construction centre (Environcentre, 2015).

Appropriate environmental indicators

Several indicators can be used to monitor the performance of CDW strategies. The most relevant is the construction waste diverted from landfill. This indicator is expressed as:

- Share of total collected CDW that is correctly segregated and managed towards reuse, recycling or recovery (%).

The calculation of this indicator includes the real amount of waste for the calculation and not estimations. The efficiency of material recovery at plants should be considered (e.g. rejects from recycling plants are not considered to be diverted). Incineration of certain wastes may be preferred and its inclusion in this indicator may be considered, depending on the final monitoring objectives of the CDW strategy in place, i.e. more priority is given to diversion from landfill, or material recycling is encouraged. It is, however, challenging for local authorities to monitor this, except for their own procured buildings. There, the estimation can easily be performed through the documentation for the site permit, including, if available, a site waste management plan.

For estimations, the main indicator is the amount of waste per built m^2, which can be measured in tonnes or per m^3. The volume unit tends to be more accurate, as monitoring by waste managers typically takes into account the volume of the means of transportation used (trucks, lorries, skips, etc.). Table 5-4 shows reference values calculated by BRE in its SMART Waste model (BRE, 2010).

Table 5-4. Environmental performance indicator: reference value volumes of construction waste arising per type of construction project

Construction project	m^3 waste / 100 m^2 floor area	m^3 waste / GBP 100K project value
Residential	17.3	12.8
Commercial offices	19.9	9.6
Commercial other	12.5	9.3
Commercial retail	20.8	17.3
Education	21.3	10.5
Healthcare	15.8	9.6
Industrial buildings	17.2	11.9
Leisure	15.8	9.0
Public buildings	24.8	12.8

Source: BRE (2010)

Additionally, another relevant indicator to monitor the implementation of this BEMP is:

- provision for pre-demolition audits aimed at reuse (y/n).

Cross-media effects

An important observed fact is the increase in illegal dumping of CDW as a consequence of (i) the increase in waste management fees and other economic instruments, especially in the case of small producers, and (ii) the increased requirement for waste sorting. Although better regulation enforcement is required at the local level, awareness is the best action against illegal dumping and landfills in the long-term.

Operational data

Estimation methods and monitoring at regional level

Several methodologies are available for estimating CDW flows:

- **Per-capita multiplier.** This is a methodology based on assigning a CDW generation rate to a region, county or municipality based on its population and on the demographic growth forecast. In Europe, the average CDW generation is around 1 tonne per person per year (McBean and Fortin, 1993; BioIS, 2011).
- **Financial value extrapolation.** It is proven that, given the specific value of buildings or construction projects, certain waste streams can be accurately estimated. For instance, gypsum plasterboard waste can be accurately calculated in projects from the construction project value (EUR/m^2) as the

generation rate is fairly constant (Yost and Halstead, 1996). However, the methodology is region-specific and requires previous surveys.

- **Area-based calculations.** This is the most frequently used methodology (EC, 2012; Llatas, 2011). As a rule of thumb, construction projects generate around 100–200 kg/m^2 of built area and demolition projects 1 000–1 500 kg/m^2 of demolished area. Table 5-5 shows average CDW generation rates.

Table 5-5. Average CDW generation rates in kg/m^2 of built, rehabilitated or demolished area

Activity	Heavyweight construction		Lightweight construction and use of modern methods of construction	
	Residential	**Non-residential**	**Residential**	**Non-residential**
New buildings	120–140	100–120	20–22	18–20
Rehabilitation	300–400	250–350	90–120	80–90
Demolition	800–1 000	1 000–1 200	500–700	700–800

Source: Llatas (2011)

The monitoring mechanism should involve the main CDW facilities at regional level, as it is mandatory for waste managers to keep a record of quantities, waste type and treatment. However, data retrieval can become an endless procedure prone to inaccuracies. The types of waste to be reported should correspond to category 17 of the European Waste List (EWL). However, this accounting system has been revealed to be inefficient, and needs to include other categories, such as the generation of MSW-like waste or packaging.

Stakeholders' involvement

After identifying the main stakeholders, it is important to establish mechanisms for their mobilisation and participation in the planning process, not only as a reactive process (complaints, opinions, etc.) but also actively through, for example, data provision, early participation in committees, etc. This provides a self-correcting mechanism to the planning activity. It is important that the role of each stakeholder is clear and well defined, so duplication of work is avoided. According to ISWA, 2012, the best-functioning SWMP systems should involve all the stakeholders in planning, implementing and monitoring the changes. In this sense, it is crucial that the waste authority demonstrates a range of good practices in issues such as the following:

- **Consultation, communication and involvement of users**. Usually achieved through information campaigns, targeted letters, social media, etc.
- **Participatory and inclusive planning**. Those stakeholders that express interest become part of a local steering committee that meets regularly to establish the performance of the system (initial state), define objectives for the future and establish the measures and benchmarks.
- **Inclusivity at all levels**. The waste authority should establish similar mechanisms of involvement during the implementation, monitoring and redefinition of the plan. For that, the creation of a local waste platform that meets regularly and takes decisions is a recommended practice (ISWA, 2012).

WRAP can be considered a frontrunner in the implementation of best practices in stakeholder management. A good example is the involvement of stakeholders at UK level in the "Halving Waste to Landfill Commitment". This inclusivity was replicated at local level in signing parties, e.g. Dumfries and Galloway Councils involved local stakeholders in the implementation of CDW prevention and minimisation policies derived from such a commitment (WRAP, 2011bc).

On a more practical level, Copenhagen developed an exemplary brick reuse system, still in the pilot phase, with the help of local collection centres, 'recycling hubs', which the city manages, involving construction companies, builders and other stakeholders for the reuse of bricks from construction sites (Copenhagen, 2014). Also, bricks can be sold in local stores (second-hand or construction material suppliers).

Implementation of national strategies at local level

As stated in the description, plans at national level also include the implementation of voluntary agreements with the industry. These agreements have a huge impact on the performance at local level, especially on recovery rates. One of the most important agreements for CDW is the "Halving Waste to Landfill Commitment" in the UK (WRAP, 2011b). It was encouraged by public authorities for waste and was considered a best practice by the European Commission (EC, 2009). However, it failed to achieve its main objective, to reduce CDW going to landfill by half (CPA, 2012) due to an unexpected increase in excavated materials. But the other inert fractions from CDW were effectively reduced by half or more in 2012. The commitment consisted of the signature of a very simple paragraph (WRAP, 2011b):

> *"We commit to playing our part in halving the amount of construction, demolition and excavation waste going to landfill by 2012. We will work to adopt and implement standards for good practice in reducing waste, recycling more, and increasing the use of recycled and recovered materials."*

This was implemented with the involvement of more than 750 companies (100 of whom were big players in construction) from the whole construction supply chain, including waste managers and public authorities (e.g. WRAP, 2011c), and through these basic actions:

- procurement includes WRAP's recommendations for waste prevention and reduction from the early stages of the project;
- waste is designed out by suppliers, architects and designers;
- waste management contractors optimise waste management on site along with contractors to maximise recovery;
- site waste management plans are implemented and waste and its treatment monitored.

Case study: the Basque Country

The Basque Country regulated by law its own regional CDW plan (Basque Country Government, 2012). The first article establishes the objectives of encouraging prevention and reuse, and other environmentally sound recovery operations, minimising the need for landfill and treatment of CDW also linked to sustainable building practices. For any work that requires a permit, a study or estimation of the amount of wastes has to be provided along with the project description in the licensing phase, which is managed and implemented at local level. If demolition of an existing building is required, a study of the materials and recycling possibilities of the building should also be added to the project. In the case of using secondary materials, these should be highlighted in the bill of materials of the new building. Segregation is mandatory when the predicted amount of wastes is higher than these values:

- concrete: 10 tonnes;
- masonry: 10 tonnes;
- metal: always;
- wood: always;
- glass: 250 kg;
- plastic: always;
- paper and board: 250 kg;
- plasterboard: always;
- hazardous waste: always.

Waste management studies for licensing should, as a minimum, contain:

- estimation of the amount of waste;
- measures for waste prevention;
- planned recovery and disposal operations;
- segregation practices on site;
- description of installations for storage, handling and separation of waste;
- cost of management;
- inventory of potential hazardous waste.

Also, the plan provides several ratios that are applicable to the construction and demolition of several types of buildings (see example in Table 5-6).

Table 5-6. Ratio of waste generation, total and per material, assumed for permitting purposes in the Basque Country

	Construction of residential building	Construction of industrial building	Demolition of residential building	Demolition of industrial building
Total waste (t/m^2)	0.0841	0.0841	1.13	0.71
Concrete	23 %	33.1 %	20.5 %	7 %
Masonry	37.6 %	30 %	54 %	54 %
Gypsum-based	7.35 %	2 %	3.7 %	3.2 %
Wood	9.5 %	9.5 %	4 %	8.5 %
Glass	0.25 %	0.25 %	0.5 %	0.5 %
Plastic	2.75 %	2.75 %	1.5 %	1.5 %
Bituminous	1.50 %	1.5 %	2.8 %	2.8 %
Metals	5.15 %	8 %	5 %	3 %
Others	7.6 %	7.6 %	5 %	16.5 %
Paper and board	2 %	2 %	-	-
MSW-like waste	1 %	1 %	0.5 %	0.5 %
Hazardous waste	2.3 %	2.3 %	2.5 %	2.5 %

Source: Basque Country Government (2012)

Applicability

The formulation and implementation of local waste management plans for CDW is a commonly used instrument by regions and large municipalities. A good example of a waste management plan for CDW at county level can be found in Hastings Borough Council (UK), which establishes clear objectives for CDW, since it observed that half of the total waste going to landfill was actually CDW (HBC, 2015).

Economics

Some examples of waste management fees applied by waste management companies are shown in the Technical report on Best Environmental Management Practice in the Building and Construction Sector (EC, 2012). Prices range between EUR 6/tonne (minimum management fee observed for clean concrete) up to EUR 75–100 per unsorted or polluted tonne of waste (observed in Germany). It can be deduced that, from a purely economic point of view, waste minimisation always reduces costs. The use of economic instruments, e.g. levies on natural aggregates and landfill taxes, has been extensively included in national CDW strategy plans, but this is outside the scope of this document.

Driving force for implementation

Given the small economic savings of best practice in waste management, driving forces for implementation are regulations, mandatory schemes, and green credentials through enhanced environmental performance and awareness.

Reference organisations

Organisations providing best practice guidance on CDW management strategies are: WRAP (UK), BRBL Recycling (NL), GERD (ES), RUMBA Guidelines (AT), The Vereniging voor Aannemers in de Sloop (VERAS (NL)), Bundesverband der Deutschen Recycling-Baustoff-Industrie resp. Kreislaufwirtschaft Bau (2015) (DE), International Solid Waste Association (ISWA, 2012).

In addition to the case study of the Basque Country presented in Operational data, the region of Paris (Île-de-France) in 2015 adopted a state-of-the-art regional plan for the management of construction and demolition waste (Île de France, 2015)

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5.4.2. Avoidance of polychlorinated biphenyl (PCB) contamination of construction and demolition waste

Summary overview							
In the case of demolition or deconstruction as well as refurbishment of buildings, bridges and structures from the 1950s, 1960s and 1970s, there is a risk that CDW materials may be contaminated with polychlorinated biphenyls (PCBs) which prevent its recycling. It is BEMP for waste authorities to introduce provisions in the CDW plan (see secion 5.4.1) that include: - pre-auditing and mapping of the building, bridge or structure to be demolished, deconstructed or refurbished in order to identify any PCB-containing material (e.g. sealants); - separate removal of the PCB-containing materials from the rest of the CDW; - separate collection and appropriate disposal of the removed PCB-containing materials.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
This BEMP is broadly applicable for waste authorities responsible for CDW. Small works, producing less than 1 tonne of CDW or affecting less than 10 m2 of the surface area of the building, can be excluded from the provisions on identifying and separating PCBs in the CDW plan.							
Specific environmental performance indicators							
- Inclusion of provisions for the mapping and separate removal and collection of PCB-containing materials in the CDW plan (y/n)							

Description

In order to ensure appropriate treatment and enable recycling of construction and demolition waste (CDW), the presence of hazardous substances in the CDW needs to be limited and avoided as much as possible. Hazardous substances are often present in buildings of certain ages and the procedures for their screening, identification, removal and separation have always been an issue during demolition. This is the case for asbestos-containing materials used from the early 20th century until their carcinogenic character revealed the need to ban them and to establish specific procedures for their removal and waste management. The same also happens for (Lend Lease, 2012) fluorescent lamps, lead, certain paints, chlorofluorocarbons (CFCs), hydrochlorofluorocarbons (HCFCs), halons, pentachlorophenol-treated timber, lindane, tributyltin, polychlorinated terphenyls (PCTs) as well as materials containing polychlorinated biphenyls (PCBs), the focus of this BEMP.

PCBs are a group of organic chemical compounds consisting of two benzene rings with 1 to 10 chlorine atoms bound to the carbon atoms of the benzene rings, with a total of 209 configurations. They were used frequently in the construction industry, e.g. in sealants, until PCBs were banned in the 1970s due to their environmental toxicity and their classification as a persistent organic pollutant.

A rise in the content of PCBs in the inert fractions of CDW was recently detected and this is a consequence of the increased rate of demolition and refurbishment of buildings from the 1950s, 1960s and 1970s (Butera et al., 2014), generating CDW with PCB-containing sealants. These waste streams, if exceeding a limit concentration of PCBs, are considered hazardous waste and need to be properly disposed of, preventing any possibility of recycling CDW (e.g. as recycled concrete aggregate).Therefore, it is BEMP for waste authorities to introduce provisions, while developing or updating the CDW plan (see Section 5.4.1) that include:

- pre-auditing and mapping of the building, bridge or structure to be demolished, deconstructed or refurbished in order to identify any PCB-containing material (e.g. sealants);
- separate removal of the PCB-containing materials from the rest of the CDW;
- separate collection and appropriate disposal of the removed PCB-containing materials.

If these steps are included in the CDW plan by the waste authority, the CDW generated will not be contaminated by PCBs and will be suitable for recycling.

Achieved environmental benefits

This technique corresponds to an Environmentally Sound Management technique (ESM) according to the PCB elimination network established at the Conference of the Parties of the Stockholm Convention on Persistent Organic Pollutants (UNEP, 2009). The benefits of PCB control and appropriate management have to be considered a priority. The control of PCB releases from CDW is, in any case, extremely important. In 2006, high levels of PCBs were detected in San Francisco Bay in the US, which were linked to demolition activity and the consequent landfill of a huge amount of PCB-containing waste. PCB was washed away with storm water run-off. As a consequence, marine species from the bay accumulated PCBs, resulting in an increased cancer risk for those eating fish (Lee et al., 2010).

Appropriate environmental indicators

The most appropriate environmental indicator is qualitative and assesses if the waste authority has considered PCBs in the CDW plan and included provisions for its mapping and correct separate removal and collection:

- Inclusion of provisions for the mapping and separate removal and collection of PCB-containing materials in the CDW plan (y/n).

Cross-media effects

In some cases, improving the understanding of PCB contamination of CDW can lead, in the short term, to a reduction in the amount of CDW that can be sent for recycling. However, once effective segregation and separate processing of PCB-containing materials is in place, the remaining CDW is suitable for recycling and therefore CDW recycling rates increase again after an adaptation period (BioIS, 2015).

Operational data

Danish example for avoiding PCBs in CDW

In Denmark, there has been a great concern recently regarding the presence of PCBs in CDW. It is required that, when demolishing or refurbishing a building from the period 1950 to 1977, a screening for the presence of PCBs is performed, especially in those parts where it is expected to be found (e.g. double-glazed windows). If the PCB_7 (2,4-dichlorobiphenyl) concentration is higher than 50 mg per kg, the waste has to be considered hazardous and disposed of safely. If the concentration is lower it may be considered non-hazardous, but still not suitable for recycling. Local authorities assess the suitability of CDW from the buildings concerned, and use a limit concentration of 100 μg/kg (PCB total) as a reference value (BioIS, 2015). The Danish government has also published guidelines on the management of PCB-containing waste, available at: www.pcb-guiden.org.

Study on Danish construction sites

Butera et al. (2014) conducted a study on the presence of inorganic elements (due to leaching) and organic compounds in CDW. They determined the concentration of different PCBs in CDW from different sites and from different segregation practices. Table 5-7 below shows the results obtained. Butera et al. (2014) analysed those PCBs according to the EN 15308:2008 standard.

A statistical analysis indicates that the PCB total content does not vary significantly between sites and that the only relevant variation is observed between mixed and clean concrete. "New concrete" CDW is waste from buildings built after 1977, where PCB-containing sealants were not used.

Table 5-8 and Table 5-9 show the statistics and the comparison, which is only of significance for mixed aggregates and clean concrete. The lack of a significant difference between clean concrete and new concrete led the research team to conclude that a background level of PCBs in is present construction raw material. In any case, the average PCB total of sampled waste is still lower than the benchmark used by Danish authorities (100 μg/kg of CDW).

Table 5-7. Analysis of 33 samples of CDW from different sites, in μg per kg of CDW

Source: Butera et al., 2015

Sample composition	Site	PCB total	PCB_7	PCB-28	PCB-52	PCB-101	PCB-118	PCB-138	PCB-153	PCB-180
Clean concrete	A	16	3.2	0.375	0.898	0.724	0.314	0.347	0.36	0.2
Clean concrete	A	34	6.8	0.682	0.617	1.07	0.54	1.51	1.51	0.917
Clean concrete	A	26	5.3	1.4	1.2	0.881	0.386	0.536	0.562	0.33
Clean concrete	A	23	4.7	0.983	0.701	0.706	0.402	0.683	0.754	0.452
Clean concrete	A	6.5	1.3	0.142	0.28	0.309	0.173	0.166	0.158	0.0729
Clean concrete	A	8	1.6	0.219	0.372	0.35	0.206	0.192	0.191	0.0782
Clean concrete	B	6.3	1.3	0.181	0.378	0.33	0.14	0.106	0.0903	n.d.
Clean concrete	B	3.6	0.73	0.0787	0.163	0.205	0.101	0.0789	0.0741	n.d.
Mixed aggregates	C	37	7.5	0.255	0.677	1.2	0.499	1.68	1.82	1.36
Mixed aggregates	C	30	6	0.245	0.405	0.848	0.346	1.48	1.48	1.2

Table 5-7. Analysis of 33 samples of CDW from different sites, in μg per kg of CDW

Source: Butera et al., 2015

Sample composition	Site	PCB total	PCB_7	PCB-28	PCB-52	PCB-101	PCB-118	PCB-138	PCB-153	PCB-180
Mixed aggregates	C	5.4	1.1	0.0827	0.145	0.221	0.106	0.193	0.196	0.138
Mixed aggregates	C	25	5	1.07	0.388	0.555	0.303	0.972	0.969	0.781
Mixed aggregates	C	27	5.4	0.173	0.368	0.808	0.514	1.39	1.24	0.951
Mixed aggregates	C	1.7	0.33	n.d.	0.0746	0.0864	0.0493	n.d.	n.d.	n.d.
Clean concrete	D	4.5	0.9	0.073	0.155	0.195	0.123	0.151	0.132	0.0662
Clean concrete	D	5.3	1.1	0.108	0.202	0.249	0.142	0.154	0.142	0.0644
Clean concrete	D	3	0.59	n.d.	0.116	0.171	0.0667	0.0853	0.0943	n.d.
Clean concrete	D	6.7	1.3	0.101	0.206	0.311	0.195	0.226	0.201	0.108
Mixed aggregates	E	27	5.3	0.283	0.441	0.992	0.508	1.2	1.18	0.713
Mixed aggregates	E	41	8.2	0.173	0.394	1.32	0.512	1.96	2.25	1.62
Mixed aggregates	E	69	14	1.17	2.51	3.23	1.97	2.18	1.98	0.786
Mixed aggregates	E	21	4.3	0.107	0.429	0.799	0.406	0.927	1	0.615
Mixed aggregates	F	12	2.4	0.218	0.327	0.452	0.368	0.416	0.418	0.194
Mixed aggregates	F	24	4.8	0.416	0.616	0.809	0.342	0.878	0.972	0.73
Clean asphalt	F	38	7.6	0.442	1.21	1.43	1.09	1.15	1.42	0.821
Clean concrete	G	9.5	1.9	n.d.	0.304	0.414	0.22	0.335	0.369	0.232
Clean concrete	G	6.1	1.2	n.d.	0.17	0.253	0.131	0.232	0.253	0.144
Clean concrete	H	2.3	0.46	n.d.	0.0889	0.0996	0.0484	n.d.	0.0831	0.0688
Clean concrete	H	4.6	0.93	n.d.	0.111	0.168	0.0779	0.155	0.201	0.183
New concrete	I	11	2.3	1.067	0.618	0.236	n.d.	n.d.	n.d.	n.d.
New concrete	J	17	3.3	1.462	0.964	0.448	0.196	n.d.	n.d.	n.d.
New concrete	J	7.6	1.5	0.658	0.441	n.d.	n.d.	n.d.	n.d.	n.d.
New concrete	K	7.1	1.4	0.621	0.382	n.d.	n.d.	n.d.	n.d.	n.d.

Table 5-8. Statistics for PCB total by nature of waste

Source: Butera et al., 2014

Waste nature	Average (μg/kg of CDW)	Standard deviation (μg/kg of CDW)
Clean concrete	10.3	9.4
Mixed aggregates	26.7	17.7
New concrete	10.7	4.6

Table 5-9. Statistical comparison of the nature of the waste for PCB total

Source: Butera et al., 2014

Comparison	Difference of means (μg/kg of CDW)	t-student	p-value
Mixed aggregates vs. Clean concrete	16.337	3.314	0.007
Mixed aggregates vs. New concrete	16.000	2.147	0.079

Clean concrete vs. New concrete	0.338	0.0468	0.963

Butera et al. (2014) could not link the nature of the PCBs, as shown in Table 5-9, to real sealants used in the construction industry, probably due to the different degradation kinetics of different PCBs. In any case, the higher presence of PCBs in mixed aggregates (composed mainly of concrete with some bricks and tiles) indicates that a lower segregation quality may also have an impact on the use and applicability of recycled aggregates.

Sources of PCBs in CDW

PCBs are not only present in sealants but also in other main components of buildings, for instance mineral-oil filled electrical equipment, capacitors, plastics, paints, adhesives, some fluorescent ballasts, etc. Table 5-10 shows primary sources (materials manufactured with PCBs), secondary sources (not manufactured with PCBs but easily contaminated due to their physical characteristics, e.g. porosity), non-porous surfaces and concentrations in exposure media.

Table 5-10. PCB-containing building materials and exposure media

Material	Range maximum concentrations measured from buildings (mg/kg)
	Primary sources
Sealant	960–752 000
Adhesives	3.9–3 100
Surface coatings	140–255
Paint	0.7–89 000
Ceiling tiles	57–51 000
Glazing	Up to 100 % liquid PCB
Light ballast	1 200 000
Electric wiring	14
	Secondary sources
Insulation materials	0.2–310
Blacker rod	99 000
Gaskets	4 300
Cove base	170
Polyurethane foam	47–50
Wood	380
Bricks and similar	2.8–1 100
Asphalt	140
Stone	130
Concrete	53–17 000
	Non-porous materials
Door frame	102
Railing	70
	Exposure media
Soil	0.1–581
Indoor air	35–24 000 ng/m^3
Dust	1.5–190

Source: EHE, 2012

Assessment of PCB concentration in materials

Concentration levels of PCBs in materials are usually determined according to EN 15308:2008. Values are calculated for PCB total and for the seven selected congeners to be included and multiplied by 5. The standard congeners are PCB-28, PCB-52, PCB-101, PCB-118, PCB-138, PCB-153 and PCB-180 (CEN, 2008). The number of the PCB indicates the number of the congener, which is defined as each of the existing chlorinated biphenyls, numbered from 1 to 209. The concentration of PCBs in materials, according to the standard, is reported in ng, μg, or mg per kg of material.

Applicability

This BEMP is broadly applicable for waste authorities responsible for CDW. Small works, producing less than 1 tonne of CDW or affecting less than 10 m^2 of the surface area of the building, can be excluded from the provisions on identifying and separating PCBs in the CDW plan.

Economics

The producer of the PCB-containing CDW, usually the developer and/or the construction contractor, covers the costs of screening, identification, removal and separation of the PCB-containing materials. These costs are then passed on to the (new) building owner. However, separating the PCB-containing material reduces the costs for the treatment of the non-contaminated CDW.

Driving force for implementation

The hazardous character and the health risks associated with PCBs are the main priority for proper management of PCBs in CDW.

Reference organisations

Stockholm Convention (UN, UNEP), http://chm.pops.int

Danish EPA, http://eng.mst.dk/

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5.4.3. Local schemes for proper management of waste asbestos removed by residents

Summary overview							
It is BEMP for waste authorities and waste management companies to ensure the proper management of the small quantities of asbestos-containing construction and demolition waste removed from private buildings by residents without the intervention of a specialised company. To do so, they can provide: - clear instructions on the condition required (e.g. no risk of powder dispersion) in order for the asbestos material to be removed by the private owner and on how to prepare the construction site for asbestos removal; - guidance on the rules that the private owner has to follow in order to ensure the health and safety of nearby residents during removal; - a list of certified companies or information on collection points for asbestos-containing waste; - sealable double-coated bags (for collection/disposal) available to residents undertaking the removal; - either proper collection points (e.g. at civic amenity sites) or free home collection services. Frontrunner local authorities go one step further and set a strategy for assessing the presence of asbestos in their territory, helping private owners plan proper action and keeping track of all asbestos in buildings even before it is removed.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
This BEMP is applicable only to certain cement-bonded asbestos (such as asbestos cement roofs, wall and ceiling cladding; asbestos down pipes and gutters, etc.) in good condition (no risk of powder dispersion) and in case of very small amounts. Cement-bonded asbestos at risk of powder dispersion, as well as other asbestos applications, especially those of lower density (or crumbly/flaky) such as insulating boards, lagging, or sprayed asbestos, are always required to be removed and disposed of by a specialist contractor.							
Specific environmental performance indicators							
- Number of collection points for asbestos waste per 100 000 residents. - Total amount of asbestos collected through the scheme, expressed in weight (tonnes) or surface area (m²). - Number of sealable bags for collection/disposal of asbestos used by residents.							
Benchmark of excellence							

- There is at least one collection point per 100 000 residents or free home collection for waste asbestos removed by residents.

Description

Asbestos-containing construction and demolition waste is considered a serious threat to public health and the environment that needs to be mitigated with proper management practices all the way from removal from buildings to final disposal.

Asbestos is present in a large number of existing buildings[68], including buildings undergoing renovation and/or buildings specifically needing the removal of asbestos due to its degradation releasing hazardous asbestos fibres.

When asbestos is found in public or commercial buildings or when substantial quantities arise (e.g. from a demolition), specialised companies for the safe removal and disposal of asbestos are appointed for this task. However, for small renovations or where a limited amount of asbestos is removed, private owners encounter high costs just for the intervention of a specialised company and some of them dispose of the removed asbestos incorrectly (e.g. mixed with other construction and demolition waste (CDW) in sites for the deposit of general CDW or even illegal dumping).

Local authorities have recognised this issue as a threat to the environment and to public health. In fact, asbestos removed from private properties must not be included, even in small quantities, in the general CDW, in order to allow the recycling of the general CDW. In the case of illegal dumping of waste asbestos, the threat to public health is even higher since the dispersion of asbestos fibres into the environment is not prevented.

In order to ensure the proper management of the small quantities of asbestos removed from private buildings without the intervention of a specialised company, it is BEMP for municipalities and waste management companies to offer a number of services to the population:

- clear instructions on the condition required (e.g. no risk of powder dispersion) in order for the asbestos material to be removed by the private owner and on how to prepare the construction site for asbestos removal;
- guidance on the rules that the private owner has to follow in order to ensure the health and safety of nearby residents during removal;
- a list of certified companies or information on collection points for asbestos-containing waste;
- sealable double-coated bags (for collection/disposal) available to residents undertaking the removal;
- either proper collection points (e.g. at civic amenity sites) or free home collection services.

[68] For its insulating properties, both for heat and electricity, asbestos was extensively used in the construction sector between the 1950s and the 1970s, especially in workshops, warehouses, schools, gyms, hospitals and train stations, but also in residential buildings. The most frequent use was as covering material, where it consists of a complex matrix product which may become hazardous only if it releases fibres into the environment due to the degradation of the cement matrix. Asbestos is also found in bins, tanks and water pipes, where ingestion of fibres could occur when it is degraded.

Frontrunner local authorities go one step further and set a strategy for assessing the presence of asbestos in their territory, helping private owners plan proper action and keeping track of all asbestos in buildings even before it is removed. Such strategy can include:

- an initial assessment of the presence of asbestos in the whole territory, based on existing data and field tests;
- the establishment of an operational plan, including assessment and characterisation of any asbestos present and facilitation of its voluntary remediation by the owner, if needed; the type of remediation depends on the condition of the material and can consist of confinement or removal;
- a follow-up of the actions: monitoring and keeping track of data on the amount of asbestos still in buildings, and removed and disposed of.

Achieved environmental benefits

The environmental benefits related to the proper management of asbestos removal mainly concern public health and air pollution (particulate matter), as amosite and crocidolite are considered to be the two most dangerous and hazardous fibre types (UK Health Protection Agency). Moreover, asbestos-containing material is replaced by new building (roofing, flooring, insulating) material that is more environmental friendly and also better for public health.

There are no benefits to reusing these asbestos-containing materials as most countries have forbidden manufacturing, importing, extracting or simply putting on the market asbestos-containing products including reused previously dismantled elements containing asbestos. Therefore this BEMP does not have major achievements in terms of recycling and reuse performance.

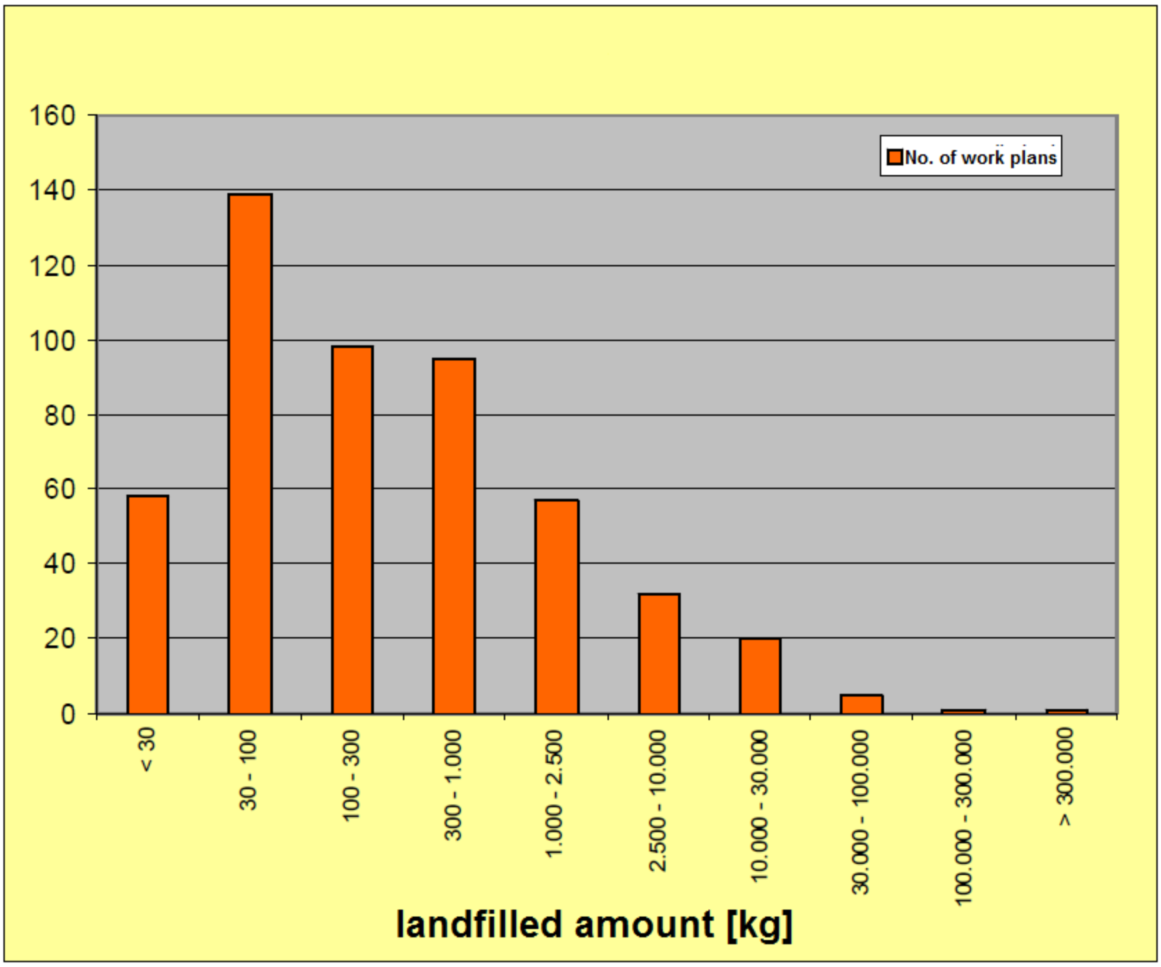
As for the environmental benefits more specifically related to this BEMP, which aims at promoting and supporting the proper management of asbestos removed from private buildings by residents, they consist mainly of an increase in the amount of asbestos removed accompanied by a decrease in its illegal dumping, as can be observed in the frontrunner case study of Bologna (Municipality of Bologna, 2014). Bologna has been tracking the amount of asbestos removed since 2011 and an increasing number of interventions can be observed in Table 5-11.

Table 5-11: Number of work plans submitted for asbestos removal in Bologna

Year	Number of interventions
2011	318
2012	173
2013	446
2014	521

Source: Piano comunale di bonifica dall'amianto; Relazione generale, Municipality of Bologna, 2014)

In the territory of the Municipality of Bologna, between 751 tonnes and 896 tonnes of asbestos are removed each year (Trevisani, pers. comm. 2017). The figure below gives an overview of the amount of landfilled asbestos and the respective number of work plans.



Source: Municipality of Bologna, 2014

Figure 5-3: Landfilled amounts of asbestos and the number of respective work plans submitted to the Municipality of Bologna

The success of the Municipal Plan can be seen by comparing this performance to other examples in this study. For instance, in the financial year 2015/16, Cambridgeshire County Council disposed of 123 tonnes of cement-bonded asbestos. And the Council does not have any data on how much of this material is in households in Cambridgeshire (Pratt, Cambridgeshire County Council, pers. comm. 2016)

The amount of asbestos removed and properly landfilled can be assessed for its efficiency and success by comparing it to the amounts inappropriately disposed of. Inappropriate disposal causes both non-compliance and, furthermore, a potential dispersion of fibres. In the case of the Municipality of Bologna, together with the increase in asbestos removed and reported to the local authority, the amount of asbestos dumped illegally dropped significantly, although the decrease has not been continuous but instead occasional, as seen in Table 5-12.

Table 5-12: Cases of illegal dumping of asbestos on public or private properties in the Municipality of Bologna

Year	Reported illegal dump sites at demolition sites	Reported illegal dump sites at private properties	Amount of asbestos (kg)
2009	0	1	5 130

2010	0	2	5 720
2011	3	4	9 438
2012	1	4	2 332
2013	1	3	2 200
2014	0	3	6 538
2015	0	3	1 212

Source: Piano comunale di bonifica dall'amianto; Relazione generale, Municipality of Bologna, 2014

Asbestos is one of the common materials which can be found in illegal dump sites, with a share of up to 5 % (Environmental Agency, UK, 2013). Illegal dumping usually comes as a consequence of improper asbestos collection and treatment or high prices for its disposal which property owners try to avoid. Applying this BEMP has reduced the amount of asbestos disposed of in illegal dump sites and the number of these sites.

When an individual asbestos removal plan is being developed, something that is worth exploring as an option and an added value is extending it to a wider energy upgrading project, namely installation of photovoltaic systems and simultaneous thermal insulation of roofs where the removal work is being undertaken. In the Italian case, until July 2013, in the Energy Bill – the photovoltaic energy production incentive was higher for those who installed a photovoltaic systems on a roof after the removal of asbestos. Thanks to this opportunity, 26 000 roofs were freed of asbestos. This represents a surface of 20 km^2 with a total installed power of 2.5 GW, mainly in northern and central Italy. Emilia Romagna had a share of around 740 000 m^2. These incentives are now over; however, the considerable reductions in the costs of installation of photovoltaic systems can still be a way to recover the removal costs over time.

Appropriate environmental indicators

The success of the BEMP is reflected in the total amount of removed asbestos, whether expressed by weight or surface (t or m^2). In order to assess the results achieved thanks to the implementation of the BEMP, these indicators can be compared to the situation before the introduction of the scheme for the removal of asbestos or with data from existing studies.

According to the census in 2011, Bologna had a surface of 474 000 m^2 covered in asbestos (Municipality of Bologna, 2014). With the conversion factor provided by the same Municipal Plan, it is assumed that 1 m^2 of asbestos equals 12.5 kg of this material. Therefore, in terms of weight, the total presence of asbestos in Bologna in 2011 was 5 925 tonnes.

The survey carried out by aerial photography of asbestos cement roofing identified 1 624 buildings that probably had such coatings. To this number 210 covers are added too, for which limited dimensions or other detection difficulties exist, which leads to a somewhat uncertain number.

The data presented in the previous section (Table 2.4-1) can give an indicative overview of the trend of the disposal of asbestos. Additionally (see Table 2.4-2), the avoided amounts of asbestos dumped illegally or stored inappropriately, whether at private properties or in public areas during demolition and construction projects, can be extended to other environmental performance indicators such as inappropriate disposal of asbestos in civic amenity sites or in non-hazardous landfill sites.

As the ultimate goal of the BEMP is to eliminate asbestos from private properties by removal and organised collection, the quantities that are removed and collected give a good insight into the BEMP performance over time. The timeline approach also gives an opportunity to look at and compare the state-of-the-art before and after implementing the BEMP, in terms of the influence of removed and collected amounts of asbestos on air quality and particulate matter.

Other relevant indicators to monitor the implementation of the BEMP are:

- number of collection points for asbestos waste per 100 000 inhabitants;
- number of sealable bags for collection/disposal of asbestos distributed to residents.

Cross-media effects

Cross-media effects appear once the asbestos leaves its original source and becomes the responsibility of the local waste management company or operator. This means that in certain cases the local municipality has to identify the closest available landfill which might not be conveniently close. Such distances contribute to air pollution caused by transporting this waste, noise produced by trucks and above all potential release of fibres in the event that the asbestos-containing waste is not well immobilised and confined.

Once asbestos is removed, further cross-media effects also appear to be the following:

- in the case of certain materials that require energy-intensive production, the pressure on resources increases;
- if no reusable material that is applicable for the replacement of asbestos is available, the removal of asbestos cannot be characterised as an example of material efficiency or circular economy in general;
- certain materials that could be a good replacement on the other hand, according to certain studies, have similar health risks and further limited treatment options (e.g. PVC).

What is important to mention as a cross-media effect, namely on emissions and health in general, is that many other health- and safety-related rules, laws and requirements must be taken into consideration in order to avoid any risks of release and inhalation of asbestos particles and fibres. This applies to the entire handling chain, from the property owner, contractor or whoever dismantles the elements containing asbestos, to the collection service, disposal and finally to whoever accepts it for further treatment.

Operational data

The BEMP is oriented towards the safe removal of asbestos from private properties and its processing in appropriate treatment facilities. However, certain health risks still exist in the event that the removal is left in the hands of the private owners. Below are examples of services that best performing local authorities can offer to the population for promoting the removal of asbestos by residents while at the same time minimising these risks.

Guidance for residents and property owners

The Brussels region provides residents with a set of guidelines and an explanation of how to apply do-it-yourself practices for asbestos removal. At the same time, a list of certified contractors is provided for undertaking necessary work. However, until recently, asbestos could not be disposed of at any Brussels waste collection points by the owners, only by a certified contractor (Brussels Environnement, 2004). From 2017, Brussels will offer (at a cost) its residents the possibility to dispose of asbestos themselves at one collection point in the region. The Flemish waste authority, OVAM, on the other hand is exploring an alternative option within a new project framework with several local partners and pilot projects. The objective is to set up local initiatives, bringing together inter-municipal waste companies and local authorities. Personal protection materials and equipment would be purchased in large quantities, so they could later be sold to citizens as a toolkit/safety pack at a lower price than can be found in various do-it-yourself stores at the moment. The project would also see these inter-municipal waste companies collecting asbestos at source with appropriate bags (Sherrier, Brussels Environment and Verheyen, OVAM Flanders, pers. comm. September 2016).

Free take-over of removed asbestos

This is implemented by local authorities based on a set of steps which a private property owner has to follow. Cambridgeshire County Council has specified the following procedure for safe asbestos removal and collection. Apart from a set of advice and precautions, it describes and defines the permitted amount of asbestos for collection, timeframe of the service and necessary data and paperwork. Once the *in situ* removal of asbestos is carried out, the property owner can get in touch with the Council. By providing proof of residency, e.g. utility bill or council tax bill, the property owner can request the number of bags required for the appropriate disposal of the asbestos. At the same time, a drop-off site has to be chosen from a list of provided locations on the website (three in Cambridgeshire), whilst providing also the vehicle type, registration number and the date of planned disposal. The property owner will be given a unique code by the District Council which has to be written on the box provided on each bag using a permanent marker. Once the free bags are delivered (within 10 days) together with the permit, the removed asbestos is placed into them and sealed, so no asbestos dust can escape. The private property owner has six weeks to complete this work. It is requested not to put more than two sheets of asbestos in each bag as this can make them too heavy to handle, and it will not be permitted to dispose of the asbestos at the chosen drop-off site. Once the bags are taken to the drop-off site they are placed in the container provided. This service has not foreseen any staff at the drop-off site to help unloading the bags. Once the disposal is done, the resident is provided with a receipt. Medway Council has, however, secured staff on site for manipulating the asbestos. Site staff are required to assess the waste and how secure the bagging/wrapping is in order to minimise contamination for other site users.

Facilitation of asbestos removal on a larger scale

While the previous practices facilitate asbestos removal only to some extent, the Municipality of Bologna has tackled this issue with a more holistic approach.

The local authority of Bologna has come up with a detailed Municipal Plan for Asbestos Removal which contains strategic objectives for the 2014-2016 three-year period. It clearly defines the types of fibrous silicates which are covered and which this Plan applies to. The Plan has listed the following minerals as the ones of concern:

- asbestos actinolite;
- grunerite asbestos (amosite);

- asbestos anthophyllite;
- chrysotile;
- crocidolite;
- tremolite asbestos.

These six types of asbestos are the same as defined by the European Council's Directive of 19 March 1987 on the prevention and reduction of environmental pollution by asbestos. Therefore, this strategy can also be taken as a good example of complementing existing European legislation and its implementation at a local level.

The Plan comes as a follow-up activity and fulfils the obligations laid down in Italian legislation and the framework law no. 257 from March 1992. This framework agreement banned asbestos with a ban on the extraction, importation, marketing and production of all products containing asbestos.

Initial assessment of the affected and the targeted area, as well as the stakeholders in the process

The assessment of the presence of asbestos before the introduction of the Municipal Plan in Bologna was carried out thanks to monitoring activities based on measurements, ground surveys and campaigns that required considerable organisational effort, time and resources, which were not always available. To overcome these problems, the Municipality of Bologna used remote sensing, i.e. the use of aerial photographs which can provide almost instantly and on a large scale detailed information of the buildings in the territory. This exercise was composed of three phases:

1. observation of the city from aerial photos;
2. geo-referencing;
3. field test and verification.

The management and execution of the Plan required coordination, both political at the governmental level and technical with the presence of representatives of the concerned ministries (health, economic development, labour and social policy, as well as environment), certain technical institutes of the different administrations involved, representatives of regions and public administrations, trade unions and associations of victims and those exposed to asbestos. When it came to the objectives, the Plan identified three main areas that would eventually benefit from such a plan:

- health protection;
- environmental protection;
- safety aspects of labour, social security and public health.

Definition of the steps for further facilitation of asbestos removal

The Municipality of Bologna later defined an operational plan including the activities to be carried out during the remediation process. The set of activities and operations is distributed over a large time span and takes place in different places or different parts of sets of buildings. The Plan defined the following organisation for proper verification, assessment and necessary actions, from the initial stages to the final remediation.

1. first phase of verification of the potential presence of asbestos on a private property;
2. second round of assessment of the quality, state of the asbestos application and damage on the property;

3. third phase of intervention and remediation planning, as well as health and environmental protection for the remediation phases;
4. remediation-related phase with or without the removal of asbestos-containing material;
5. in the event that removal is not required, a monitoring plan, including maintenance and control of the asbestos-containing material, has to be ensured until removal is required.

The timeline and individual actions defined by the Plan

By following this order of actions, the property owner can ensure that full attention is given to the remediation project and that the most suitable and feasible option will be applied. Under current Italian legislation, any individual ownership of any type of property is accountable for performing the verification and evaluation of asbestos cement roofing materials and other material containing asbestos at its own expense and through a competent technical authority. The Italian Ministerial Decree from 1994 and further provisions and indications laid down in Emilia-Romagna Region's Guidelines for assessing the conservation status of asbestos cement roofing and health risk assessment from 2002 closely define and describe the evaluation of the quality of asbestos-containing material. Three possible outcomes are possible:

- tolerable state;
- poor state (remediation required);
- very bad state (remediation required).

When it comes to other applications, such as flues, chimneys, etc., the material can be classified as:

- intact material not susceptible to damage;
- intact material susceptible to damage;
- damaged material in non-extended area;
- damaged material in extended area (remediation required).

The verification itself consists of several tasks which range from the identification of the roofing area and its identification and inspection of the type of material used, by going through the technical documentation of the building (also trying to trace the construction company that was contracted) or by direct inspection of the materials and trying to identify the potential presence of asbestos fibres and the state of degradation of the asbestos-containing material. The identified material and its state should reveal whether the potential release of fibres into the environment could occur under certain circumstances. The material could furthermore be forwarded to a laboratory for final confirmation of asbestos presence. In the event that asbestos is discovered, that building unit and the area are mapped and all other accompanying information during the verification phase is collected in order to better define the remediation plan, if necessary.

Two types of intervention are possible:

- decontamination, with or without removal;
- maintenance and continuous control and monitoring of the material.

To conduct the remediation, the property owner should contract a company registered as a waste manager and together with the company establish a work plan which would be submitted to the authorities at least 30 days prior to the work. The remediation can include the following actions:

- surface encapsulation with special paints;

- confinement;
- removal.

For choosing a suitable contractor, it is requested that the property owner make sure that the company has skilled technical managers and instruments and adequate financial resources, in order to ensure the security and health of workers and the environment. There is a list of companies included in the Environmental managers' registry at the Chamber of Commerce.

Upon the completion of the work, the property owner should keep the documentation which proves the work has been done, especially if disposal of the removed asbestos is required. The documents include:

- certificate of completion with a declaration of compliance with the disposal of asbestos (proper labelling, transport and transfer to a storage facility or licenced landfill);
- waste identification form.

In the event that maintenance and monitoring is needed, it must be performed by a qualified technician. The property owner and/or the technician must perform the following tasks:

- appoint a person responsible for the monitoring tasks and coordination of all the maintenance activities that may take place;
- keep proper documentation showing the location of asbestos-containing materials;
- place a suitable warning on affected units (e.g. boiler and pipes) in order to prevent asbestos being inadvertently disturbed;
- ensure compliance with effective safety measures during cleaning activities, maintenance work and during any event which might cause the disruption of the materials;
- provide correct information to the occupants of the building on the presence of asbestos in the building, on the potential risks and how they should behave;
- if there are in friable asbestos materials in place, proceed to have the building inspected at least once a year.

To monitor the effectiveness of the remediation operation with encapsulation that might have taken place, the following tasks have to be performed:

- check that no delamination occurred, chipping and cracking of the layer coating the surface of the unit;
- check for faded colours of the last layer.

Follow-up of the facilitation and the final disposal of asbestos

When it comes to the practices in Bologna, the removed and collected asbestos becomes the responsibility of the waste management operator or the local authority. Initially, according to the European Waste Catalogue, six types of asbestos-containing waste were identified and two of them were included in the list of hazardous waste. Subsequently, with the modifications and additions made by Commission Decisions Nos EC/118/2001 and EC/119/2001 and Council Decision No EC/573/2001, implemented in Italy by the Legislative Decree No 152/2006, the number of asbestos-containing wastes increased to eight and all have been classified as hazardous waste. The Plan defines two pathways for this waste:

- hazardous waste landfill, with a dedicated asbestos cell provided;
- non-hazardous waste landfill, with a mono-cell provided for asbestos.

Only the asbestos which appears as a compact matrix can be disposed of in landfills for non-hazardous waste, while the other remaining categories, usually as a friable matrix, can only be disposed of in hazardous waste landfills. Concerning the recovery of such waste, the Ministerial Decree 248/2004 defines two types of treatment processes:

- treatments that reduce the release of the fibres without changing the structure of asbestos or modifying it to some extent before sending it to the landfill;
- treatments that completely change the structure of asbestos and invalidate the dangers linked to asbestos content; such treated asbestos could have reuse as secondary material as its final purpose.

From the research conducted on the basis of required performance levels and availability of treatment options, 19 landfills were identified in Italy in June 2013 and another 6 awaiting authorisation (Municipality of Bologna, 2014).

Applicability

Geographically speaking, the BEMP is applicable to many parts of the EU as asbestos-containing products were widely used everywhere throughout the 20th century. However, since asbestos was a versatile material and its applications were very diverse, removal varies depending on its use. Therefore, this BEMP can cover certain cement-bonded asbestos in good condition (e.g. no risk of powder dispersion), such as:

- asbestos cement roofs;
- asbestos cement wall and ceiling cladding;
- asbestos down pipes and gutters;
- asbestos cement flues;
- asbestos promenade tiles (used in walkways).

Cement-bonded asbestos at risk of powder dispersion as well as other asbestos applications, especially those of a lower density, such as insulating boards, asbestos lagging, sprayed asbestos and other low-density (crumbly/flaky) asbestos, are required to be removed and disposed of by a specialist contractor.

In terms of limitations, certain practices limit the quantity of asbestos permitted to be removed, such as free collection or disposal schemes. For instance, Cambridgeshire County Council has foreseen to supply a maximum of nine bags within a six-month period. The bags come in two different sizes - 2.59 m x 1.37 m and 1.37 m x 0.9 m. In the case of larger quantities of asbestos to dispose of, requiring more than nine bags, the Council can arrange a home visit to see if the resident qualifies for more bags.

Medway Council in the UK also offers a free disposal service for its residents. It covers certain types of asbestos applications, such as asbestos cement roofing, asbestos down pipes and gutters, asbestos flues and asbestos tiles.

When it comes to the final disposal, one limitation that exists in many EU Member States is the lack of proper sanitary landfills with a cell dedicated to asbestos waste. disposing of asbestos waste in inappropriate landfills can have severe counter-effects as, even when removed and transported away from private properties, fibres can be released into air due to the potential breaking up of the waste and the pressure in a landfill. Civic amenity sites are usually another feature of countries and regions with a good waste management system. The absence of collection points in general (whether civic amenity sites or equipped landfill gates) can impact the effectiveness of the BEMP. Lack of such points or large distances and the absence of trained staff can prevent residents from disposing of asbestos properly and eventually lead to illegal dumping.

Economics

This section presents a summary assessment of costs for asbestos removal works in order to better understand and quantify the necessary work and funds.

The Municipality of Bologna reports costs which vary between EUR 6 and EUR 15 per m^2 of removed surface, including the work on encapsulating the material to be removed, the removal of the sheets, their handling, packaging, transport and disposal at the final destination. This amount also includes the further transport and landfill disposal. It does not cover costs related to temporary structures (scaffolding, safety etc.). These costs are borne by the property owner. Experience shows that disposal can account for 25 % to 40 % of the total costs. In cases of very complex operations for removal, the charges related to the safety of the construction site and the preparation may increase significantly.

Since the most common destination for asbestos waste is landfill, a gate fee applies to its disposal. WRAP reports on the gate fees in the UK which range between EUR 35 and EUR 105. Gate fees for bonded asbestos materials (EUR 35–79) are significantly lower than for unbonded/fibrous insulation materials containing asbestos (EUR 46–105). This is because unbonded insulation materials take up a significant amount of space, are difficult to compress, and frequently require special management measures (WRAP, Gate Fees Report, 2013).

These costs are also applicable to other cases where the local authorities have decided to take on the costs of the collection and disposal of asbestos, like in the case of Birmingham, UK.

When applying this BEMP, the local authority has to take into consideration and provide an appropriate collection facility with necessary treatment options or ensure a further contractor for the treatment of asbestos. Apart from this, some municipalities have also foreseen the provision of private property owners with firm plastic bags for safe asbestos disposal. In the UK, the price range for 100 sealable bags for asbestos disposal is between EUR 55.40 (Polybags.co.uk, February 2017) and EUR 75.40 (Arco.co.uk, February 2017).

Support measures from the local authority

The Municipality of Bologna has foreseen certain support tools and other activities to ensure the proper deployment of the Plan. The information system that has been set up helps to store and organise the information related to every step and phase of the process, including the census and the preliminary identification of asbestos presence all the way to the verification, individual project design and execution of the remediation. This process of computerisation is required to manage an otherwise unmanageable flow of information and documents, so that the various parties and stakeholders are able to find the personal data, photographs, site visits, etc. in a holistic way and to manage the deployment and the flow of data and information with maximum efficiency. Further possible activities for a better deployment directly targeting residents are the following in particular:

- Lower costs for remediation procedures: the Municipality has two options:
  - providing a reference price list for various services provided by private companies for asbestos removal and remediation operations in order to provide the residents with an overview and enable them to find an affordable contractor;
  - cost reductions for disposal of asbestos.
- Planning of suitable disposal methods and criteria, including costs: development of local disposal facilities for decreasing the related costs.
- Reactivation of the Energy Bill or other forms of contribution, for contribution in the event that the asbestos removal is followed by the installation of photovoltaic panels; where this Bill was in place in some Italian regions like Lazio and Tuscany a large increase in asbestos remediation was recorded.
- Simplification of procedures.
- Renewed legislative framework.

Potential added value of removing asbestos

For a better economic viability and the overall feasibility of a removal project, the Municipal Plan suggests expanding the project with a wider energy upgrade project, namely installation of a photovoltaic system and simultaneous thermal insulation of roofs affected by the intervention.

The Plan has run a simulation presented in Table 5-13, where investments in asbestos removal and further installations of photovoltaic (PV) systems can be understood in terms of costs and financial gains.

This simulation is run and based on the presumptions that the removed surface is insulated with technology that gives the right to the application of the tax deduction of 65 % (up to the end of 2014) and that a PV system is installed for self-consumption of the energy produced.

Table 5-13: Costs and savings for an asbestos removal project and its replacement with a PV system, based on 25 years of life ()

Installed power (kW)	Surface covered in asbestos (m^2)	Surface covered in PV panels (m^2)	Overall costs (EUR)	Total savings during the PV system's life (EUR)	Gain during the PV system's life (EUR)
3	60	30	13 650.00	22 241.55	8 066.55
6	120	60	24 230.00	42 565.61	17 294.01
10	200	00	36 500.00	68 938.51	30 758.51
20	400	200	58 350.00	127 569.08	66 699.08
50	1000	500	126 800.00	316 273.96	184 223.96
100	2000	1000	240 350.00	625 207.92	375 057.92
200	4000	2000	556 750.00	1 190 415.85	608 465.85

Source: Municipality of Bologna, 2014

What can be observed when looking at the table above is that the higher the investment, the higher the eventual saving and gain from the installed PV system.

Driving force for implementation

Asbestos removal and replacement with energy-efficient and healthy material is an ongoing process in many public facilities across Europe. However, private properties are often lower on the priority list, yet their owners are encouraged to remove asbestos through various schemes. The BEMP should allow them to get rid of asbestos at a low cost through certain services provided free of charge. The rationale behind this BEMP is to improve public health and also lower the risk of cancer among the local population.

The fact that asbestos is taken over from private property owners leads to the avoidance of inappropriate storage of asbestos, potential damage and its reuse. Taking over asbestos from private properties by the local authorities also prevents the creation of illegal dump sites and further spreading of this hazardous material.

Three key driving forces for implementation can be highlighted in the case of asbestos-containing waste, also recognised by the Plan of the Municipality of Bologna:

- Health protection: the European Parliament urges the EU in general to develop models for monitoring existing asbestos in public and private buildings, workplace, residential areas and landfills. Furthermore, the EU is requested to carry out a cost-benefit analysis in relation to the possibility of establishing an action plan for the removal of asbestos by 2020. The establishment of public registries containing information on asbestos-related risks during its manipulation is favourable as well.
- Environmental protection: the European Parliament was further requested by the European Commission to integrate the issue of asbestos into other policy areas, such as energy and waste.
- Job security and social protection: the European Parliament also calls on the EU in general to develop programmes and outreach activities that focus on the risks posed by asbestos and the need for appropriate training for all workers who may come into contact with asbestos-containing waste, as well as labour inspectors and occupational physicians.

Further national legislation can set the framework and priorities for acting on asbestos removal. National legislation can mandate local authorities to take appropriate steps to identify and localise asbestos presence by running a mapping and census exercise. Additionally national legislation can explore the possibilities of setting up financial backing schemes for intervention and remediation activities and promote research on new technologies for the disposal of asbestos, to ensure better cost-effectiveness than current methods. These drivers are further implemented and adapted to regional plans and strategies, like the one in Bologna.

Reference organisations

The organisations, platforms and services listed below are already running the previously mentioned practices, providing examples of good practices and models that can be replicated in other parts of Europe:

- Municipality of Bologna (IT): has a very thorough and holistic approach to asbestos removal in its territory in the form of a Plan.
- Birmingham City Council (UK): provides free asbestos collection.
- Brussels region and Flanders (BE): provide free disposal for small quantities of asbestos and detailed guidelines on asbestos removal; Flanders is starting a pilot project looking at possibilities to provide residents with affordable safety packs.

- Lille Metropole (FR): provides plastic films for asbestos disposal and allows citizens to bring the collected asbestos to the dedicated collection points.
- Cambridgeshire County Council (UK): provides free plastic film bags for asbestos collection and a free disposal service at their waste management facilities.
- European Asbestos Risk Association (EU): a European association founded in order to promote important initiatives to inform public opinion, without unnecessary alarmism, and to raise awareness of the dangers and diseases caused by the use of asbestos.
- European Demolition Association (EU): leading platform for national demolition associations, demolition contractors and suppliers.

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5.4.4. Processing of waste plasterboard to foster recycling

Summary overview							
It is BEMP for waste management companies processing waste plasterboard to recover gypsum. Processing of waste plasterboard for the recovery of gypsum usually consists of the following steps (for well-segregated waste plasterboard): reception, visual check and classification, separation of unsuitable materials (e.g. metals), (if required) grouping of the panels according to size, paper and gypsum separation (through a grinding and sieving process) and sieving of gypsum. Recovered gypsum can then be used (usually up to 25 % of the total content) for the production of new plasterboard.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
There are no technical barriers to the applicability of this BEMP. However, there are significant economic barriers: the recyclability of the waste plasterboard depends on the level of segregation at the site where it is generated[69] and poor segregation leads to cost-inefficient situations. Moreover, transport costs of waste plasterboard over long distances may also affect the economic viability.							
Specific environmental performance indicators							
- Efficiency of material recovery at the waste plasterboard processing plant (%).							

Description

Plasterboard (also known as drywall, gypsum board, wallboard, etc.) consists of kiln-dried panels made of gypsum plaster pressed between two thick sheets of paper. The gypsum plasterboard life cycle has become an example of how a circular economy can work effectively. In Europe, 2.35 million tonnes of waste plasterboard are produced from construction and demolition projects per year and an extra 0.6 million tonnes are produced during its manufacture and installation (GTG, 2015). However, almost all the waste plasterboard can be successfully fed into the manufacture of new plasterboard or as raw material for other uses. Moreover, plasterboard itself can incorporate waste from other industrial processes. Plasterboard produced with 89 % recycled material was achieved by Knauf, a manufacturer of building materials, in 2013 (Knauf, 2013).

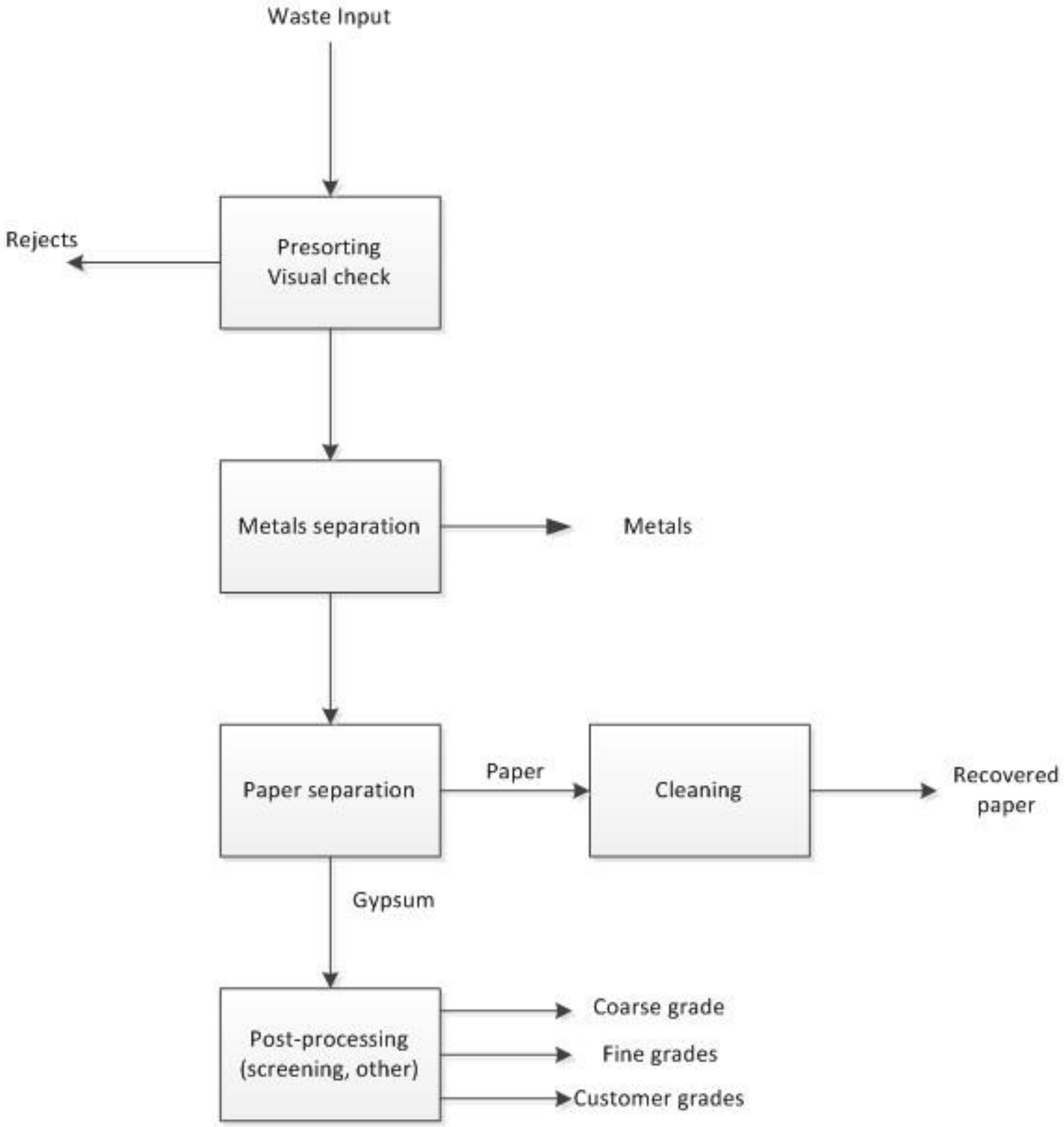
[69] In some cases, segregation at the construction site may not be possible due to space constraints. In such situations, the waste plasterboard can be pretreated and segregated at different locations before being processed.

In order to realise the circular economy potential of plasterboard as much as possible, it is BEMP to recover gypsum from waste plasterboard so that this can be fed into the production of new plasterboard. The process described below is of informative use for waste authorities and of practical use for waste management companies treating waste plasterboard. The recyclability of plasterboard depends on the level of segregation at the site where the CDW is generated or, when there is insufficient space at the site, at different locations where CDW is pretreated and segregated. Poor segregation leads to cost-inefficient situations for waste management companies processing waste plasterboard.

Chemically, the production of gypsum consists of a dehydration-rehydration process. Natural or synthetic calcium sulphate dihydrate ($CaSO_42H_2O$) is dehydrated at 150–200 °C under a steam atmosphere to form hemihydrate ($CaSO_4½H_2O$). During rehydration of the hemihydrate, new crystals of calcium sulphate dihydrate are formed in an interlocked net. The material is low-density, has low thermal conductivity and develops enough strength so it can be used in a wide range of construction products.

'Raw gypsum is found in recovered plasterboard, selectively collected from construction or demolition sites, and so-called synthetic gypsum (calcium sulphate dehydrate as a by-product of industrial processes), usually from flue-gas desulphurisation (FGD), also called FGD gypsum. The reprocessing of calcium sulphate waste is an example of high-grade recycling, which is relatively rare in the recycling of CDW. In terms of carbon savings, the benefits of recycling plasterboard are not high. A plasterboard panel made with 25 % recycled waste plasterboard, in a low-transport scenario, saves an average of 33 kg CO_2e in associated GHG emissions per tonne compared to conventional gypsum plasterboard, i.e. around 10 % savings (WRAP, 2008b). But, in addition, when waste plasterboard is incorporated into the manufacture of recovered gypsum, it has indirect benefits on the recycling of CDW, as the segregated recovery of plasterboard removes sulphate contamination in the matrix of recycled aggregates from clean concrete wastes, increasing its recyclability and applicability (EC, 2012; Asakura, 2013).

The process for producing recycled gypsum from waste plasterboard is straightforward and very similar to any process for construction and demolition waste treatment. An example is shown in Figure 5-4. At the entrance, the waste materials are visually checked and classified by size. Metals are separated and, if required, the panels are ground to a certain size. Then, paper is separated through a grinding and sieving process, which is key for the quality of the final reprocessed gypsum. Paper is pretreated and packed for its recycling. Gypsum is sieved (or even crushed again) depending on the grades to be produced.

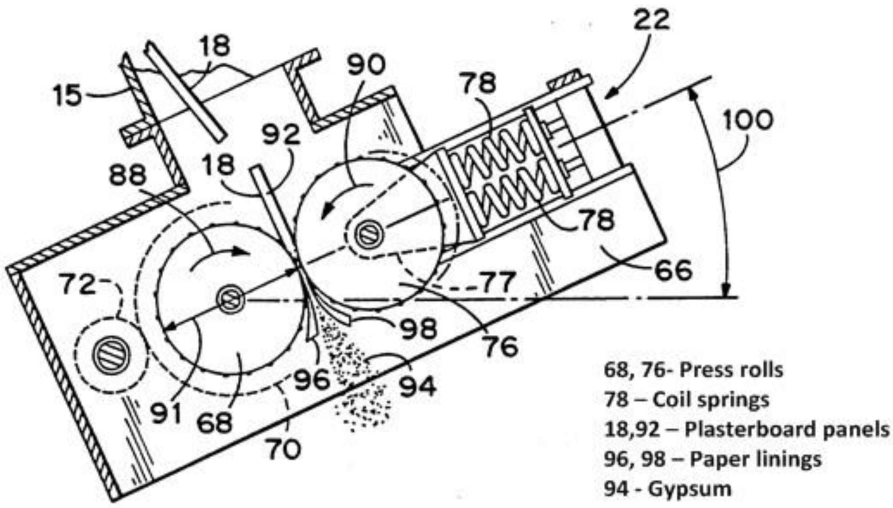


Source: Adapted from Roy Hatfield (2013)

Figure 5-4. Waste plasterboard processing

Gypsum recovered after sieving can then be used for the production of new plasterboard, usually up to 25 % of the total content of gypsum (see Achieved environmental benefits section).

In any waste plasterboard processing facility, the key step for the quality of the output material is paper separation, as it can increase the recycled material content of new plasterboard. As gypsum produces a much finer material than paper during grinding or crushing, the conventional separation is done by grinding and further sieving. The process allows a relatively high separation rate. For instance, one arrangement of a plasterboard crusher is shown in Figure 5-5 (Bauer, 1992). The press rolls rotate in different directions and have a beaded surface able to break the interior of the boards, separating the gypsum material from the large pieces of paper lining.



Source: Adapted from Bauer, 1992

Figure 5-5. Waste plasterboard crusher

Achieved environmental benefits

The recovery of gypsum from plasterboard has a lower environmental impact than the manufacture from conventional raw materials, i.e. natural gypsum and synthetic gypsum (calcium sulphate from flue-gas desulphurisation (FGD)). Additionally, the segregation of waste plasterboard at the site where it is generated reduces the contamination of the remaining CDW, improving its recyclability (see BEMP 5.4.5).

The most important study published so far (WRAP, 2008a), with the input of data from the main manufacturers in Europe, indicates that the maximum content of recycled gypsum in new plasterboard products is 25 %, as the content of fibre, from the lining of panels, has a negative effect on the product performance. For this level of recycling, the difference in the environmental performance of plasterboard production under several scenarios is relatively small, less than 10 % (see Table 5-14).

For instance, the reduction in GHG emissions from, for example, incorporating 15 % recycled materials would be only 2 % in a low-transport scenario or 1.4 % in a high-transport scenario, or 4.5 % and 3.8 % respectively for 25 % recycling. These very low reductions are due to two main factors:

- The environmental impact is mainly allocated to the thermal stages: the *calcination* process, i.e. dehydration of gypsum to produce hemihydrate ($CaSO_4½H_2O$), requires temperatures up to 200 °C and, depending on the final product, a steam atmosphere in an autoclaved process. Also, the fast *drying* required for plasterboard production consumes a significant amount of natural gas.
- A maximum recycled content of 25 % is assumed. Production of natural gypsum ready for the process (extraction, transport and pre-processing) is associated with the emission of 120 kg of CO_2e per tonne (84 kg in production), while collection, transport and pre-processing of recycled gypsum results in up to 40 kg of CO_2e per tonne. The benefits, therefore, should be extensive in a high recycling scenario. However, the presence of cellulose fibres prevents further use of recovered materials. The lower the fibre content, the higher the recyclability (see Operational data for more information).

Table 5-14. LCA results for one tonne of plasterboard

Impact category	Unit	Baseline scenario		15 % recycled content		25 % recycled content	
		LT	HT	LT	HT	LT	HT
Abiotic depletion	kg Sbe	3.1	3.1	3.0	3.1	2.93	3.0
Global warming (100-yr)	kg CO_2e	513	517	503	510	480	493
Ozone layer depletion (ODP)	kg CFC-11e	1.8E-05	1.96-05	1.8E-05	1.9E-05	1.8E-05	1.9E-05
Human toxicity	kg 1,4-DCBe	104.7	104.9	103.4	104.3	100.4	102.6
Fresh water aquatic ecotoxicity	kg 1,4-DCBe	28.0	28.0	27.6	28.0	27.6	27.6
Marine aquatic ecotoxicity	kg 1,4-DCBe	1.5E+06	1.5E+05	1. 5E+05	1.5E+05	1.4E+05	1.4E+05
Terrestrial ecotoxicity	kg 1,4-DCBe	0.45	0.45	0.44	0.45	0.43	0.44
Photochemical oxidation	kg C_2H_4	0.09	0.09	0.08	0.08	0.08	0.08
Acidification	kg SO_2e	1.4	1.4	1.3	1.4	1.3	1.3
Eutrophication	kg PO_4e	0.19	0.19	0.19	0.19	0.18	0.19

NB: LT: Low-transport scenario; HT: High-transport scenario.

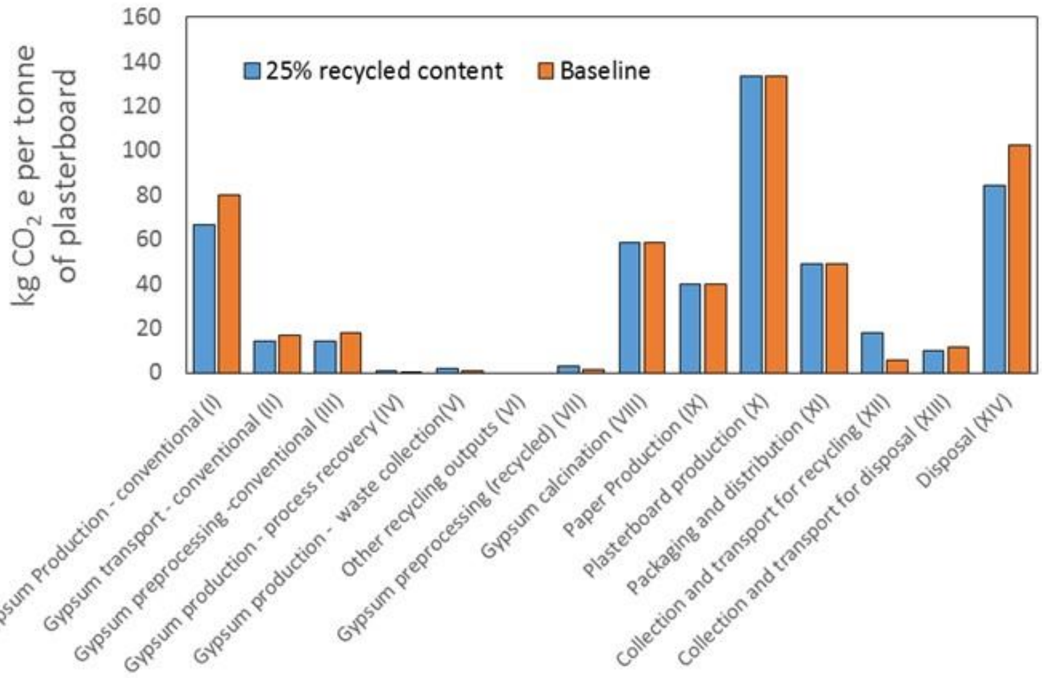
Source: adapted from WRAP, 2008a

Regarding the process contribution, Figure 5-6 shows the contribution to greenhouse gas emissions for each different stage and the life-cycle flow chart reflecting all stages assumed in the study by WRAP (2008a). As shown, the main contributors are plasterboard production (mainly drying), calcination, natural gypsum production and disposal. Disposal and production of natural materials are, of course, reduced once recycled materials are incorporated, but the benefit needs to be increased by the incorporation of more recycled material, while further reductions in the thermal processes (calcination and drying) are process-dependent and not raw-material-dependent.

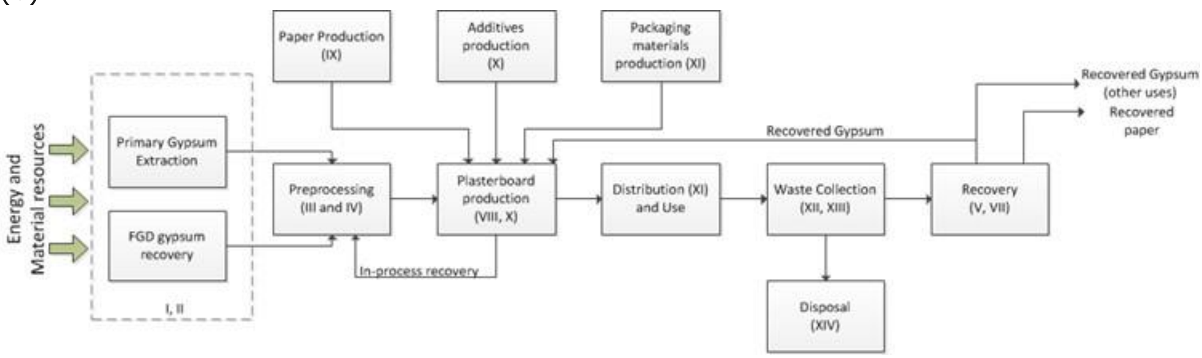
Another environmental benefit of gypsum plasterboard segregation and recycling is the removal of sulphates from the main bulk of construction and demolition waste, which mainly consists of concrete. The gypsum content in CDW is found to be around 5–10 % (Asakura, 2013), while the threshold value for the acceptability of CDW as raw material for secondary materials is around 3 %, so segregation is required. During CDW crushing to produce recycled aggregates, gypsum tends to be incorporated into the fines and semi-fines fractions, due to its lower strength (compared to concrete), creating problems when used in new concrete mixes. New approaches to separate sulphate-containing waste are being developed and successfully applied (Vegas et al., 2015).

Plasterboard waste can be problematic in landfills due to the sulphate content of gypsum. When mixed with biodegradable municipal waste in a landfill, sulphate-reducing bacteria form hydrogen sulphide in anaerobic conditions, which dissolves in the leachate in wet conditions or generates bad odours. The life-cycle assessment confirmed that a H_2S reduction of up to 17 % can be achieved in the low-transport scenario when 25 % recycled gypsum is used in the manufacture of new plasterboard.

(a)



(b)



Source: Adapted and modified from WRAP, 2008a

Figure 5-6. Greenhouse gas emissions per process stage (a) and assumed supply chain for plasterboard (b)

Appropriate environmental indicators

The indicator to assess the successful implementation of this BEMP is:

- efficiency of material recovery at the waste plasterboard processing plant (%).

The efficiency of material recovery at the waste plasterboard plant takes into account the amount of rejects generated from its operations.

Cross-media effects

No cross media effects are expected.

Operational data

Processing of waste plasterboard

The example of steps for waste plasterboard processing in Figure 5-4 presents the operations carried out at the Roy Hatfield plant in Rotherham (UK). Waste plasterboard processed there comes from a variety of sources, such as construction and demolition companies or households carrying out small works. The processing rate is 60 tonnes per hour and the treatment capacity is around 1 000 tonnes per week. New West Gypsum Recycling (NWGR) shreds the waste plasterboard and applies mechanical separation of the gypsum from the paper. The process results in less than 1 % paper contamination in the gypsum to achieve the acceptance levels for new plasterboard. The recyclable gypsum is transported back to plasterboard manufacturers, where it is combined with virgin rock or synthetic gypsum to make new plasterboard. The recycler states that it has a low fibre content in the recycled process and a gypsum use rate above 25 % in the making of the new plasterboard (Roy Hatfield, 2013).

Quality assurance

Quality assurance for gypsum produced from waste plasterboard is very important for its actual use. The manufacturers of plasterboard that use reprocessed gypsum from waste plasterboard as raw material need to ensure the quality and technical specifications of their final product.

In some cases, a set of criteria defining the quality of reprocessed gypsum has been defined as end-of-waste criteria. For instance, in the United Kingdom, a "Quality Protocol" for recycled gypsum from waste plasterboard identifies the criteria to determine when waste plasterboard is no longer considered a waste and therefore waste management controls no longer apply in England, Wales and Northern Ireland (WRAP, 2011). Although the criteria do not establish benchmarks on recycling, they give assurance to holders and processors. The Quality Protocol ensures the applicability of the reprocessed gypsum for new plasterboard, as raw material for cement as well as soil treatment for agriculture, although spreading it within 50 metres of potable groundwater should be avoided, due to the risk of pollution.

Approved specifications under the UK Quality Protocol are those gathered under the PAS 109:2013 (BSI, 2013) for the production of gypsum from waste plasterboard and the limits for metal and metalloid values shown in Table 5-15.

Table 5-15. Maximum metal and metalloid values in gypsum from waste

Parameter	Maximum contaminant values (mg/kg)
Arsenic	5.23
Cadmium	0.30
Chromium	17.9
Copper	32.8
Lead	31.9
Magnesium	2 412
Mercury	< 2
Molybdenum	7.68
Nickel	7.31
Phosphorous	87
Potassium	1 992
Selenium	7.37
Zinc	40.3
Sulphur	209 200

Source: WRAP, 2011

The PAS 109:2013 standard defines three grades of recycled plasterboard and a minimum quality specification, depending also on the final use, distinguishing between agricultural use and use as a raw material (Table 5-16). The standard also defines the minimum requirements on the Quality Management system of the reprocessor and how the acceptance criteria for waste plasterboard should be communicated. One of the most important aspects of the standard is the requirement of traceability of the reprocessed gypsum back to the batch of waste.

Table 5-16. Specifications for PAS 109:2013 reprocessed gypsum

Parameter	**Specification**					
Particle size distribution (% retained on sieve individually)	Fine grade		Coarse grade		Custom grade	
	Lower limit	Upper limit	Lower limit	Upper limit	Lower limit	Upper limit
31.5 mm	0	0	0	0	To be defined by its market. Upper limit of 31.5 mm.	
16 mm	0	0	40	80		
8 mm	0	0	20	60		
4 mm	0	0	0	40		
2 mm	0	0	0	20		
1 mm	0	10	0	10		
0.500 mm	0	20	0	5		
0.250 mm	0	40	0	2		
0.125 mm	20	60	0	2		
0.063 mm	40	80	0	2		
Residual paper / fibres						
Content	< 1 % w/w					
Size of paper pieces	Maximum 10 mm largest dimension					
Purity (% weight of $CaSO_4$ $2H_2O$)	> 85 %					
Physical contaminants	< 2 mm, upper limit 0.25 % weight (dry sample), of which 0.12 % weight is plastic					
End uses	**Agriculture**			**Plasterboard manufacture/ others**		
Chemical composition						
Soluble Chloride	< 0.1 % w/w			< 0.02 % w/w		
Magnesium oxide	n.a.			< 0.2 % w/w		
Sodium oxide	< 0.06 % w/w			< 0.06 % w/w		
Colour	White, light grey or light beige, with no coloured particles					
Smell	Odourless / neutral					

Source: (BSI, 2013) PAS 109:2013

The quality restrictions for recycled gypsum from waste plasterboard and for that coming from other industrial process, such as flue-gas desulphurisation, are very similar. Table 5-17 gives an overview of the quality parameters of recycled gypsum and flue-gas desulphurisation gypsum, as shown in EC (2012). In other countries, e.g. in Germany, no end-of-waste criteria have been agreed upon yet, although the industry has established similar criteria to those in the UK for the minimum quality requirements of recycled gypsum (BV Gips, 2013).

Table 5-17. Comparison of quality parameters of recycled and FGD (flue-gas desulphurisation) gypsum

Quality parameter	Determined as	Unit	Quality criteria	
			FGD gypsum	Recycled gypsum
Humidity	H_2O	Mass %	< 10	< 10
Calcium sulphate dihydrate	$CaSO_42H_2O$	Mass %	> 95	> 80
Magnesium salts	Water-soluble MgO	Mass %	< 0.10	< 0.02
Sodium salts	Water-soluble Na_2O	Mass %	< 0.06	< 0.02
Potassium salts	Water-soluble K_2O	Mass %		< 0.02
Chlorides	Cl	Mass %	< 0.01	< 0.01
Calcium sulphite-hemihydrate	$CaSO_4½ H_2O$	Mass %	< 0.50	< 0.50
pH	--	--	5–9	5–9
Colour		%	White	White
Odour	--	--	Neutral	Neutral
Toxic compounds	--	--	Harmless	Harmless
Grain size	--	mm	--	< 5

Source: LFU (2007) as cited in EC (2012)

Applicability

There are no technical barriers to the applicability of this BEMP. However, there are significant economic barriers: the recyclability of the waste plasterboard depends on the level of segregation at the site[70] where it is generated and poor segregation leads to cost-inefficient situations. Moreover, transport costs could be significant if the recycling plant is located far away from the source of the waste plasterboard.

The economic environment around natural gypsum is also a key driver for the implementation of the BEMP. Natural gypsum would be more favoured in countries/areas with extensive natural sources.

Economics

As mentioned in the Applicability section, economic factors are key for this BEMP. Plasterboard recycling is economically viable when the material processed is well-segregated waste plasterboard (i.e. limited amount of impurities) that is transported short distances. When investing in the construction of a new waste plasterboard processing plant, careful location planning needs to be carried out, in order to locate

[70] In some cases, segregation at the construction site may not be possible due to space constraints. In such situations, the waste plasterboard can be pretreated and segregated at different locations before being processed.

the facility in a strategic position (i.e. close to areas where significant quantities of waste plasterboard are generated) that limits the need to transport materials.

Finally, the market price of natural gypsum and the acceptability of the recycled plasterboard are crucial for the economics of this BEMP.

Driving force for implementation

The main driving force for processing waste plasterboard for the recovery of gypsum is the environmental performance of the process, which is favourable compared to the use of conventional raw materials. The improved recyclability of the remaining CDW (uncontaminated with plasterboard) is another relevant driving force for this BEMP. Moreover, limited landfill capacities, the protection of natural resources, and the expected decline of FGD gypsum quantities due to the phasing out of coal-based power plants have led to reconsideration of the recycling of gypsum from construction and demolition materials. In addition, in some countries like Germany there is currently a debate regarding stricter sulphate limit values in the recovery of secondary materials, and this could lead to significant restrictions on the use of recycled construction materials in the future if plasterboard is not very well segregated at source.

Reference organisations

Waste authorities and organisations

Waste Resources Action Programme (WRAP) has developed the EoW criteria with the industry in the UK.

Eurogypsum is the European association of gypsum product manufacturers.

Gypsum industry

KNAUF

British Gypsum

Roy Hatfield UK

Regyp recycling solutions

New West Gypsum Recycling

Reference literature

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5.4.5. Processing CDW for the production of recycled aggregates

Summary overview							
It is BEMP for waste management companies treating CDW to recover concrete from CDW as recycled concrete aggregate (RCA). This processing takes place in plants which usually consist of the following steps (for well-segregated CDW): reception, characterisation and identification of incoming CDW, (manual) preselection, screening of large materials, magnetic separation, screening for fine materials, crushing, screening and secondary crushing. The recyclability of the inert elements of CDW depends on the level of segregation at the site[71] where they are generated and poor segregation leads to the processing of CDW being cost-inefficient.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
There is no specific limitation to the applicability of this BEMP as long as the CDW is well segregated into the different fractions at the construction site.							
Specific environmental performance indicators							
- Efficiency of material recovery at the CDW processing plant (%). - Annual amount of RCA marketed (tonnes/year).							

Description

The focus of this BEMP is on the selection of the product portfolio of waste management companies treating CDW, based on the final applications. Manufacturing of recycled aggregates is based on two families of products: mixed aggregates, usually with a minimum 50 % content of concrete, and recycled concrete aggregate, with over 90 % concrete in its composition. These two types of aggregates constitute more than 80 % of the mass output of a recycling plant. Some of the techniques described in this section can be considered a common approach in some European countries, with very high recycling rates for "clean" concrete waste. However, the situation in Europe is heterogeneous regarding the implementation of recycling practices for concrete.

[71] In some cases, segregation at the construction site may not be possible due to space constraints. In such situations, the CDW can be pretreated and segregated at different locations before being processed for the production of recycled aggregates.

Therefore, concrete recovery as recycled concrete aggregate (RCA) has to be considered a BEMP and the techniques described here are of informative use for waste authorities and of practical use for waste management companies treating CDW. This section describes the range of products by application that waste management companies treating CDW may consider and that have been proven to achieve maximum recovery rates.

Prior to demolition/deconstruction, it is key to assess the construction and demolition waste stream that is going to be generated (Tecnalia et al., 2016; EC, 2016). The recyclability of the inert elements of construction and demolition waste depends on the level of segregation at the site where they are generated or, when there is insufficient space at the site, at different locations where CDW is pretreated and segregated. Poor segregation leads to cost-inefficient situations for waste recyclers, since the range of products would be strongly influenced by the segregation rate.

Processing of CDW is usually similar across Europe, although the nature of final products may vary according to the existing market (mainly local) for these products. A CDW recycling plant usually consists of:

- reception, weighing and visual inspection;
- characterisation and identification of incoming CDW;
- manual preselection and rejection to other treatments (depending on the acceptability criteria, if the original segregation is good enough, this step might not be useful);
- screening of large materials;
- magnetic separation (e.g. for reinforcement steel and metals) and screening for fine materials;
- if segregation at source is poor, manual separation of plastic, wood and other waste types may be needed;
- crushing;
- screening and secondary crushing (depending on the aggregates produced and marketing of products).

A CDW manager mainly has to deal with the inert fraction (concrete plus masonry). From well-sorted waste, waste managers are able to produce high-quality aggregate products. A normalised classification of recycled aggregate from construction waste is proposed, among many other standards in Europe, by DIN (through the standard 4226-100 for recycled aggregates). Four types are differentiated, shown in Table 5-18.

Table 5-18. Classification of aggregates according to DIN 4226-100

DIN classification	Type 1	Type 2	Type 3	Type 4
Recycled aggregates	Concrete plus crusher sand	Mixed wastes plus crusher sand	Masonry plus crusher sand	Mixed plus crusher sand
Concrete and natural aggregates	≥ 90 %	≥ 70 %	≤ 20 %	≥ 80 %
Clinker, non-porous bricks	≤ 10 %	≤ 30 %	≥ 80 %	
Sand-lime bricks			≤ 5 %	
Other mineral materials	≤ 2 %	≤ 30 %	≤ 5 %	≤ 20 %
Asphalt	≤ 1 %	≤ 1 %	≤ 1 %	
Foreign substances	≤ 0.2 %	≤ 0.5 %	≤ 0.5 %	≤ 1 %
Density (kg/m^3)	≥ 2 000	≥ 2 000	≥ 1 800	≥ 1 500

Source: Müller, 2006

A number of possibilities and routes for recycled products exist in the current construction market. The main final destination of recycled construction products is the substitution of materials as base materials in roads, as aggregates for concrete production and for filling material in earthworks. The characteristics of the final construction product should be considered when choosing the recycled aggregate and, technically, taking into consideration the natural materials' substitution rate. For example, high-quality concrete for foundations and piles may accept less recycled products than mass concrete or light concrete, which are able to accept 100 % recycled aggregates. Secondary uses for recycled materials may include sand for cement production, but this application has a limited substitution rate because of the composition of crusher sand (even from concrete crushing) (Hauer and Klein, 2007).

Table 5-19 shows applicable solutions for the two main products produced in recycling plants, i.e. concrete aggregates and mixed aggregates.

Table 5-19. Possibilities for recycled construction materials

Material	Use	Applicability	Specifications/restrictions
Concrete aggregates (e.g. minimum of 90 % concrete content)	Earthworks, filling and road sub-bases	These aggregates are usually applicable to this kind of works. There may be restrictions on the physical properties because of water absorption, sulphate content (causing expansion and fragility) and water absorption. Usually, all countries ask for the same technical properties as for natural aggregates, plus some standards on concrete and impurities.	French NF P 11-30, Spanish PG-3 technical specifications for roads and bridges. Specific requirements for recycled aggregates in terms of strength (e.g. with Los Angeles test, or with the amount of small slaps or flagstone).
	Buildings and other civil works, for structural concrete	Coarse recycled aggregates may be applied for structural concrete (mass concrete or reinforced concrete) but water demand is higher and may cause higher cement consumption for the same resistance as with natural aggregates. Compression resistance may be reduced (as a function of quality) and elasticity is lower.	Spanish recommendation of a maximum 20 % substitution of natural coarse aggregates. Additional requirements are specified for recycled aggregates in order to keep structural properties. Dutch national standards allow for a replacement of 20 % of natural primary aggregates by mixed or concrete aggregates (without additional performance tests).
	Buildings and other civil works, for non-structural concrete		Up to 100 % substitution if technical and environmental specifications are fulfilled.
	Buildings and other civil works, for mortar	Fines and small particles may be used to produce mortar.	Water demand is increased. CEDEX, 2010, recommends to use 25 % recycled mortar in order to keep properties.
	Buildings and other civil works, for cement	Fines from concrete sand crusher have similar properties to cement with natural sand.	First used in Japan. Price is less than conventional cement. Energy consumption reduction and saving of natural materials are the main benefits, but the chemistry of the mixture does not allow a substitution rate of more than 10 % (Hauer, 2007). Nevertheless, 100 % substitution is allowed if technical

Table 5-19. Possibilities for recycled construction materials

Material	Use	Applicability	Specifications/restrictions
			specifications are met.
Mixed aggregates (e.g. minimum of 50 % concrete content)	Earthworks, filling and road sub-bases	They can be applied but it is required that the gypsum content is low. Main application is as filling material. Usually, not suitable for road pavement bases.	The cost for cleaning may be high. Same specifications as for other materials. Workability may be worse, as water absorption is higher and slower than for natural aggregates.
	Buildings and other civil works, for non-structural concrete	Adequate consistence and resistance properties are achievable for *in situ* concrete for non-structural concrete. Not usable for prefabricated concrete elements.	The low density of these aggregates may be optimal for the production of light concrete. Nevertheless, durability is lower than for other aggregates.

Achieved environmental benefits

The main environmental benefit of concrete recycling is the avoidance of the impacts from the disposal of CDW and those avoided from the use of primary or natural aggregates.

In terms of life-cycle environmental performance, generalisation is not possible, and each case is different.

The analysis by Hiete (2013) of the environmental performance of concrete recycling, mainly as recycled aggregates, shows the following conclusions:

- Site characteristics are essential: location influences transport distances; composition influences recycling materials and determines the type of final application.
- During use phase, there is no fixed standard for the leachability of recycled aggregates.
- When weighing up the benefits of primary aggregate substitution, the type of application and the type and origin of the natural aggregate strongly influence the life-cycle performance.
- However, washing, which is applied when site segregation is poor, can account for more than 99 % of the total environmental impact (Korre and Durucan, 2009).
- Although there are studies confirming the better environmental performance of the recycled aggregates' supply chain, Chowdhury et al. (2010) state that the production and crushing of concrete is more energy-intensive than for primary aggregates, and the environmental impact can be compensated if the ratio of transport distances for primary aggregates versus recycled aggregates is above four.

Appropriate environmental indicators

The first indicator to assess the successful implementation of this BEMP is:

- Efficiency of material recovery at the CDW processing plant (%)

The efficiency of material recovery at the CDW plant takes into account the amount of rejects generated from its operations.

Another relevant indicator which assesses the substitution of primary aggregates thanks to RCA, is:

- Annual amount of RCA marketed (e.g. tonnes/year).

Cross-media effects

Whenever recycling products are based on concrete from CDW, there is a risk that potentially hazardous materials are contained in the original waste. Symonds (1999) showed a full list of hazardous waste found in CDW (Table 5-20). This is the case with recycled aggregates, as they come from waste, which is likely to contain some of the hazardous materials shown in Table 5-20, but also for those recycled products to be used for construction (e.g. slags, ashes). The Commission issued a mandate to CEN for a harmonisation on the assessment of dangerous substances. As a response, a new Technical Committee – CEN/TC 351 – was created: ‘Construction products: assessment of release of dangerous substances’. This Committee should provide tools and assessment methods for the quantification of dangerous substances, which may be released from construction products to the environment into the soil, groundwater, surface water and indoor air (Delgado et al., 2009). Actually, several (preliminary) technical standards and rules are at the drafting/approval stage or have been published.[72]

Table 5-20. Hazardous materials in construction and demolition waste

Product/Material	Potentially hazardous components	Hazardous properties
Concrete additives	Hydrocarbons, solvents	Flammable
Damp-proof materials	Solvents, bitumen	Flammable, toxic
Adhesives	Solvents, isocyanides	Flammable, toxic, irritant
Mastics, sealants	Solvents, bitumen	Flammable, toxic
Road surfacing	Tar-based emulsions	Toxic
Asbestos	Breathable fibre	Toxic, carcinogenic
Mineral fibres	Breathable fibre	Skin and lung irritants
Treated timber	Copper, arsenic, chrome, tar, pesticides, fungicides	Toxic, ecotoxic, flammable
Fire-resistant wasting	Halogenated compounds	Ecotoxic
Lighting	Sodium, mercury, PCBs	Ecotoxic
Air conditioning systems	CFCs	Ozone-depleting
Firefighting systems	CFCs	Ozone-depleting
Contaminated building fabric	Heavy metals, including cadmium and mercury	Toxic
Gas cylinders	Propane, butane, acetylene	Flammable
Resins/fillers, precursors	Isocyanides, anhydride	Toxic, irritant
Oils and fuels	Hydrocarbons	Ecotoxic, flammable
Plasterboard	Source of hydrogen sulphides	Flammable toxic
Road planning	Tar, asphalt, solvents	Flammable, toxic
Sub-base (ash/clinker)	Heavy metals including cadmium and mercury	Toxic
Insulation foams blown with ODS	Ozone-depleting substances	Ozone-depleting

Currently, there are not many approaches to limit the leachability of recycled aggregates. It is usually common that recycled aggregates coming from ashes, slags

[72] http://standards.cen.eu/dyn/www/f?p=204:32:0::::FSP_ORG_ID,FSP_LANG_ID:510793,25&cs=135BD767027D4B4E081006EF46B5E957C

and other wastes are regulated, while for recycled concrete some countries apply a set of different criteria. For instance, the Netherlands does not apply a waste regulation to recycled aggregates, but a common regulation is used for natural or recycled aggregates in terms of environmental criteria. In Germany, a regulation is being prepared and the leaching limit values are material-specific and refer to specific applications.

As there are no harmonised standards and limit values in Europe, a good reference point is the leachability compared to the Landfill Directive's leaching limit values. An assessment made by DHI (2011) on the leachability of some aggregates is shown in Table 5-21.

Table 5-21. Recycled aggregates' leachability: elements close to, partially exceeding or consistently exceeding the EU leaching limit values for acceptance of waste at waste landfill

Product	Close to the limit	Partially exceeding	Consistently exceeding
Recycled concrete		Ba, Cr, Pb	
Recycled brick		SO_4^-	
Recycled glass		Cu, Pb	Sb
Mixed CDW		Cd, Cl, Pb	
Recycled asphalt			
Blast furnace slag		SO_4^-	
Basic oxygen furnace slag			V
Electric arc furnace slag			
Phosphorous slag		Mo, Pb, Sb, Se	
Coal fly ash		As, Ba, Cd, Cl, Cr, Mo, Ni, Pb, V, Zn	SO_4^-
Coal bottom ash	As	Cd, Cr, Mo, Ni	
Municipal solid waste Incinerator fly ash		As, Cr, Cu, Zn	Cd, Cl, Mo, Pb, SO_4^-
Municipal solid waste Incinerator bottom ash	Cd, Se, Zn	Cr, Mo, Ni, Pb, Sb, SO_4^-	Cl, Cu
Artificial aggregates	Cd, Mo, Pb, SO_4, Zn	As, Cd, Mo, Se	
Natural aggregates	Cd, Ni, V		

Source: DHI, 2011

Another important aspect is the health and safety issue in recycling plants. At least, 20–25 % of dust in the surroundings of recycling plants has been detected to be of a diameter of less than 10 mm (Kummer et al., 2010) and, therefore, its generation and impact has to be duly controlled, e.g. through the implementation of dedusting devices in screening, crushing and handling operations. Also, the location of recycling plants close to urban areas, although good in terms of life-cycle environmental impact, has an adverse effect due to noise, vibration and emissions from the commonly used diesel engines.

Operational data

Recycling plants

CDW recycling plants can be mobile, semi-mobile or stationary. It depends on the nature of the material to be crushed, the total amount, and the purpose of the installation. For instance, stationary plants are commonly used for recycling plants, integrating several technologies to produce products of a high quality. Mobile plants can be used directly in quarries or large construction sites that produce a large quantity of construction waste (e.g. excavated soil or stone).

Common recycling processes consist of a first manual sorting and/or visual inspection step. An excavator or similar device feeds a pre-classifying sieve to separate sand and the fine fraction, which makes up one product from the facility. Then, materials are crushed to several fractions and metals are separated with a magnetic separator. Material screening and classification is then carried out and the products are stored in several piles.

Different processing technologies are compared in Table 5-22.

Table 5-22. Comparison of different crusher types in mobile, semi-mobile and stationary plants

Type	Advantages	Disadvantages	Applications
Semi-mobile and mobile plants with jaw crusher	Simple, rugged construction Low wear rate Crushes hardest rocks	Lower crushing efficiency Problems when crushing bituminous broken road paving Recycling of oversized materials practically impossible	Crushing of unproblematic building rubble where no demands are placed on product quality or capacity
Semi-mobile and mobile plants with impact crusher	Favourable crushing efficiency with all types of building rubble and broken road paving	Relatively high wear rate Can generate excessive fines	Suitable for all-round rubble crushing with a high capacity
Stationary plant with jaw and impact crushers or two impact crushers	Combines advantages of both crusher types High capacity Can crush large reinforced concrete waste pieces	Plugging problems with bituminous material High capital costs	Good for high capacities combined with high demands on product quality
Stationary plant with jaw and cone crusher	Very good product quality, sharp, cubical form Low wear rate	Susceptible to rebars and tramp metal in cone crusher High capital costs	Recommended for generation of high-quality secondary materials
Stationary plant with beater drum and impactor	Particularly good for handling large concrete lumps	Very high wear High capital costs	Ideal combination for recycling concrete waste, railway sleepers, concrete

Table 5-22. Comparison of different crusher types in mobile, semi-mobile and stationary plants

Type	Advantages	Disadvantages	Applications
			masts, etc.

Source: FAS, 2002

Construction and demolition waste recycling process: FEBA case study

An example of a construction and demolition waste recycling plant was provided by Feba, in Freiburg, Germany as shown in the Technical Report on Best Environmental Management Practice for the Building and Construction Sector (EC, 2012), where a full description is provided. According to the managers of the plant, there is a healthy demand for recycled aggregates, especially those coming from concrete. The mass balance for the years 2009, 2010 and 2011 can be observed in Table 5-23. As shown, the total input matches the total output of materials, the amount accumulated being negligible (or even negative). The main fraction is concrete, followed by excavated materials and asphalt and bituminous materials.

Table 5-23: Input-output balance of the FEBA recycling plant

Waste input	**LoW number**	**2009 (t)**	**2010 (t)**	**2011 (t)**	**2009+2010+2011**
Concrete	170101	27 400	18 000	36 500	81 800
Bricks	170102	1 800	1 800	3 500	7 100
Tiles and ceramics	170103	1 000	1 400	200	2 600
Mixed	170107	8 400	6 500	15 000	29 900
Soil and excavated materials	170504	28 500	17 000	29 100	74 600
Asphalt and bituminous (mixed)	170302	12 900	16 900	20 200	49 900
Total input		**79 900**	**61 500**	**104 600**	**246 000**

Waste output	**2009 (t)**	**2010 (t)**	**2011 (t)**	**2009+2010+2011**
Waste for disposal	50	30	40	130
Sold scrap	260	170	420	850
Total	**310**	**210**	**460**	**980**
Product output	**2009 (t)**	**2010 (t)**	**2011 (t)**	**2009+2010+2011**
Crushed brick 0/8	60	110	30	200
Crushed brick 0/16	180	360	40	590
Screening at 0/3 (sand)	6 100	3 400	9 500	19 000
Screening at 0/8	590	340	360	1 290
Screening at 0/16	4 700	3 100	2 500	10 300

FSS 0/32	9 700	10 400	8 800	28 900
FSS 0/45	48 300	61 000	44 300	153 600
STS 0/32	630	10	510	1 150
STS 0/45	2 800	12 400	13 500	28 800
Blown material 16/100	250	5 900	1 100	7 200
Special mixtures	730	2 100	1 500	4 300
Total output	**74 100**	**99 000**	**82 100**	**255 300**

Applicability

Technical and environmental criteria for recycled products

In general, the incorporation of recycled aggregates can reach up to 20 % (w/w) with no loss of mechanical properties in structural concrete. For non-structural applications, substitution rates of up to 100 % are achievable, if certain recommendations are followed (CEDEX, 2010). This indicates a high applicability of recycled aggregates, since the total production of suitable CDW for recycled aggregates is around 10 % of the total mass of concrete produced in Europe. Further restrictions to the applicability in structural concrete and non-structural concrete are shown below (Table 5-24 and Table 5-25).

Table 5-24. Proposed technical specifications to fulfil mechanical properties of structural concrete

Parameter	Value
Particles < 4 mm	< 5 %
Clay lumps content	< 0.6 % (for 20 % recycled aggregate)
Water absorption	< 7 %
Ceramics content	< 5 %
Light particles	< 1 %
Asphalt	< 1 %
Other (glass, plastic, etc.)	< 1 %

Source: CEDEX, 2010

Table 5-25. Proposed technical specifications to fulfil mechanical properties for non-structural concrete

Parameter	Value
Water absorption	< 12 %
Total S content	< 1 %
Sulphates (acid-soluble)	< 1 %
Other materials (glass, plastic, etc.)	< 1 %
LA value (Los Angeles abrasion coefficient)	< 50 %
Fines content	< 4 %
Ceramics content	< 50 %
Gypsum content	< 2 %

Source: CEDEX, 2010

The applicability is also dependent on the level of waste segregation. For instance, as described above, the gypsum content of CDW is extremely important in the applicability of recycled aggregates produced from CDW (Table 5-26).

Table 5-26. Restrictions on the gypsum and soluble salt content for recycled aggregates

Gypsum content	**Use**
< 0.2 %	Usable for any zone of embankment
0.2–2 %	Core of embankment
2–5 %	Core of embankment, with special materials in crowning point and screen walls
5–20 %	Core of embankment, with measures to avoid solution of sulphates
> 20 %	Not usable
Soluble salt	**Use**
< 0.2 %	Usable for any zone of embankment
0.2–1 %	Core of embankment
> 1 %	Not usable

Source: CEDEX, 2010

Economics

Cost of recycled products

The cost of recycled aggregates is variable and depends on the manufacturer. Nevertheless, the final price is not substantially different from the natural aggregate cost and, in some circumstances, may even be lower. The market price varies from EUR 3 to EUR 12 and depends on many local circumstances, especially transport costs (WBCSD, 2009) and quality. The high share of transport costs in the total costs is highlighted by Hiete (2013) as being a very decisive factor for CDW recycling. CDW needs to be transported from the site to the plant and the recycled aggregate from the plant to the site. For a typical recycling plant with a capacity of 100 000 tonnes per year, a utilisation factor of 80 % and with a European average of 2 tonnes of CDW per capita per year, a population of 40 000 within a radius of 10 km (a population density above 125 inhabitants per km^2) would be required for an optimal performance of the recycling system (Hiete, 2013). Of course, this is not the situation in many parts of Europe. Low population density also favours the availability of primary aggregates.

Generally, the availability of low-cost natural materials is a great disadvantage for the competitiveness of recycled aggregates. Production costs of natural aggregates are usually higher than for recycled aggregates, and logistics costs depend on the availability of quarries in the surrounding area. Good segregation of construction waste on site reduces the production cost of recycled aggregates and logistics prices are comparable to quarries in populated areas. Therefore, the cost of recycled aggregates should not be a significant barrier for the uptake of recycled aggregates in most cases.

The main factors affecting the uptake of recycled aggregates are usually:

- the proximity and quantity of natural aggregates;
- reliability of supply and quality (in theoretical terms, quality homogeneity is better for natural materials), which is largely influenced by the presence of proper recycling plants in the vicinity;
- incentives, subsidies and taxes for natural aggregates and landfills;
- standards and regulations for recycled aggregates;
- quality certification and green building systems;
- existence of illegal landfills.

Driving force for implementation

The main drivers for the application of concrete recycling are costs and the marketability of the final product, both induced through economic instruments affecting wastes or natural aggregates, or due to the scarcity of natural aggregates. Environmental credentials, although important, are of much less importance for the construction sector. Reduction of landfill volumes is also a resource efficiency driver for waste authorities.

Reference organisations

Organisations providing best practice guidance on CDW recycling and application of recycled aggregates are: WRAP (UK), BRBL Recycling (NL), GERD (ES), CEDEX (ES), RUMBA Guidelines (AT), Bundesverband der Deutschen Recycling-Baustoff-Industrie resp. Kreislaufwirtschaft Bau (DE), Rigips (Germany).

Finally, there are several reference guidance documents addressing the topic of concrete recycling developed at national level, including 'Guide de conception et de fonctionnement des installations de traitement des déchets du btp' and 'Guide technique pour l'utilisation des matériaux régionaux d'Île-de-France - Les bétons et produits de démolition recyclés' (in France) and 'Guide des déchets de chantier – service cantonal de gestion des déchets, Geneva' in Switzerland.

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6. Healthcare waste (HCW)

6.1. Introduction

The management of healthcare waste (HCW)[73] is strictly controlled, due to its hazardous characteristics, but differently regulated within EU Member States. The European List of Wastes (EC, 2014) defines the following subcategories of waste under category 18, "wastes from human or animal health care and related research" (* = hazardous waste):

- 18 01 wastes from natal care, diagnosis, treatment or prevention of disease in humans:
  - 18 01 01 sharps (except 18 01 03);
  - 18 01 02 body parts and organs including blood bags and blood preserves (except 18 01 03);
  - 18 01 03* wastes whose collection and disposal is subject to special requirements in order to prevent infection;
  - 18 01 04 wastes whose collection and disposal is not subject to special requirements in order to prevent infection (for example dressings, plaster casts, linen, disposable clothing, diapers);
  - 18 01 06* chemicals consisting of or containing hazardous substances;
  - 18 01 07 chemicals other than those mentioned in 18 01 06;
  - 18 01 08* cytotoxic and cytostatic medicines;
  - 18 01 09 medicines other than those mentioned in 18 01 08;
  - 18 01 10* amalgam waste from dental care.
- 18 02 wastes from research, diagnosis, treatment or prevention of disease involving animals:
  - 18 02 01 sharps (except 18 02 02);
  - 18 02 02* wastes whose collection and disposal is subject to special requirements in order to prevent infection;
  - 18 02 03 wastes whose collection and disposal is not subject to special requirements in order to prevent infection;
  - 18 02 05* chemicals consisting of or containing hazardous substances;
  - 18 02 06 chemicals other than those mentioned in 18 02 05;
  - 18 02 07* cytotoxic and cytostatic medicines;
  - 18 02 08 medicines other than those mentioned in 18 02 07.

The World Health Organisation (WHO) considers that about 85 % of HCW is non-hazardous waste and that only 15 % is hazardous, also referred to as healthcare risk waste (HCRW) (WHO, 2015). Other studies (Mastorakis et al., 2010) indicate a figure

[73] Healthcare waste (HCW) is the waste generated at a medical institution, hazardous and non-hazardous (including MSW-like waste), while medical waste (MW) is normally used to define waste specifically generated by the operation of health activities. There is some overlapping in both definitions. The term used in the text is HCW, as recommended by the Technical Working Group supporting the development of the document.

of 80 % non-hazardous waste while the main types of hazardous waste are described as "infectious and anatomopathological waste" (15 %) and "sharp" (3 %).

The main types of HCRW are:

- infectious waste (e.g. contaminated by bodily fluids);
- pathological waste (human tissues, organs, body parts, etc.);
- sharps (syringes, scalpels, needles etc.);
- chemical (disinfectants, broken thermometers, batteries, etc.);
- pharmaceutical (expired drugs and medicine);
- genotoxic waste including cytotoxic drugs used for cancer treatment;
- radioactive waste.

Major sources of healthcare waste are hospitals (including maternity clinics and long-term healthcare establishments), dialysis centres, laboratories, mortuaries and nursing homes for the elderly (WHO, 2014).

Although HCW is strictly defined as a result of healthcare practice, waste similar in nature can be produced in many other environments (e.g. at home or offices). In such cases, the status of the waste generated depends on the person performing the treatment. If the treatment is done by a professional from the healthcare sector (nurse, general practitioner (GP), etc.), then the waste is regarded as professional waste that has to be handled by the practitioner. If the patients treat themselves, the waste is regarded as household waste and it is the responsibility of the patient. The waste, in this last case, is then classified as MSW, falling under category 20 of the European List of Wastes, for municipal wastes:

- 20 01 separately collected fractions (except 15 01):
    - 20 01 31* cytotoxic and cytostatic medicines;
    - 20 01 32 medicines other than those mentioned in 20 01 31;
    - 20 01 99 other fractions not otherwise specified.

The classification 20 01 99 is used in the case of *offensive*[74] waste.

However, in terms of waste management, e.g. in a hospital, a simpler classification is required, since the waste handler is not only dealing with waste under category 18. For instance, in Greek regulation (EPTA, 2006), HCW is classified according to these categories for its management:

a. Non-hazardous HCW (MSW-like waste)
b. Hazardous HCW
    b1. Infectious waste
    b2. Toxic and infectious waste
    b3. Toxic waste
c. Others (radioactive, batteries, etc.)

The category under which a stream of HCW is classified will determine its treatment. Generally speaking, the following treatments are acceptable for HCW (CIWM, 2014):

[74] *Offensive* waste is a term used for non-hazardous healthcare waste that causes *offence* due to its appearance, odour or wetness.

- thermal treatment, such as high-temperature incineration, incineration and landfilling of incineration residues;
- alternative treatments, such as chemical or thermal sterilisation (autoclaving);
- others (for MSW-like waste), as recovery operations.

As a consequence of the application of strict public health regulations to the waste streams, a treatment method can be applied to each of them, as shown in the following table.

Table 6-1. Treatment method per waste category

Category	Treatment method	Disposal
Infectious clinical, 18 01 03*	Alternative treatment or hazardous waste incineration	Waste-to-energy or landfill of incineration residues
Offensive waste, 18 01 04 and 20 01 99		Waste-to-energy or landfill of incineration residues
Non-medicine contaminated sharps, 18 01 03*	Alternative treatment or hazardous waste incineration	Residual ash recovery or landfill
Medicine contaminated sharps, 18 01 03* and 18 01 09	Hazardous waste incineration	Incineration
Cytotoxic and cytostatic 18 01 03* and 18 01 08*	Hazardous waste incineration	Incineration
Medicine waste, 18 01 09	Hazardous waste incineration	Incineration
Medicine contaminated infectious clinical waste, 18 01 03* and 18 01 09	Hazardous waste incineration	Incineration
MSW-like		Reuse, recycle, energy recovery, incineration

Source: Adapted from CIWM (2014)

6.2. Technique portfolio

In terms of HCW management in healthcare institutions, the identification of best practices and frontrunners is restricted to the areas where there are no mandatory measures. Therefore, the following classification of management practices can be proposed (non-exhaustive list):

- Mandatory measures (usually regulated for hazardous wastes under the duty of care):
    - identification and labelling;
    - selective collection of hazardous waste according to its nature, final treatment, etc.;
    - individual and collective health and safety protective measures;
    - information, communication and training;
    - temporary storage: time limits, location and characteristics of containers (internal and external).
- Best environmental management practices:

- outstanding integrated HCW segregation at healthcare institutions;
- selection of alternative treatments for HCW.

While the mandatory measures are oriented to public health protection and are strongly regulated, the best environmental management practices are those oriented to minimising the environmental impact produced by HCW generation. Prevention measures are the most important but are excluded from this document, as they exclusively affect the activities of the healthcare sector and not the waste management sector.

Finally, this chapter also investigates the issue of hazardous HCW generated by households (e.g. at home, offices), specifically sharps and needles generated by home treatments. This waste stream needs to be appropriately collected and treated by a specific collection system implemented by local authorities and/or waste management companies.

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6.3. Management of HCW in healthcare institutions

6.3.1. Waste segregation

The segregation of HCW at the point of production is strongly regulated in the EU Member States and regions, under different regulation approaches, for example:

- in Italy the D.P.R. 254 15/07/2003;
- in Germany Communication No 18 of the Joint Working Group of the German federation/federal states on waste (LAGA);
- in England and Wales, but a best practice recommendation in Scotland and Northern Ireland – according to the Health Technical Memorandum (HTM) 07-01 (DH, 2013).

The World Health Organisation (WHO) regularly publishes guidelines for the safe management of healthcare waste and recommends a basic segregation scheme (Table 6-2).

Table 6-2. WHO-recommended segregation scheme

Type of waste	Colour code and marking	Container
Highly infectious waste	Yellow, with HIGHLY INFECTIOUS and biohazard symbol	Strong, leak-proof plastic bag or container capable of being autoclaved
Other infectious waste, pathological and anatomical waste	Yellow with biohazard symbol	Leak-proof plastic bag or container
Sharps	Yellow, marked SHARPS with biohazard symbol	Puncture-proof container
Chemical and pharmaceutical waste	Brown, labelled with appropriate hazard symbol	Plastic bag or rigid container
Radioactive waste	Labelled with radiation symbol	Lead box
General healthcare waste	Black	Plastic bag

Source: WHO (2014)

Beyond the basic segregation, successfully implemented in Europe, the use of a single black plastic bag for non-hazardous waste (MSW-like waste and others) prevents further recycling and material separation. The existence of health and safety regulations on the management of several hazardous waste streams reduces the resources available for non-hazardous waste management. Some healthcare organisations are able to segregate waste further in several streams:

- recyclables: paper, plastic and cans, usually generated by patients and visitors in common areas;
- food waste: generated by kitchens;

Hazardous waste other than healthcare waste, e.g. chemicals, solvents, batteries, light bulbs, is generated at higher rates by healthcare activities than by households..

However, HCW management should ensure hygiene and infection control as a top priority. All measures of prevention, reuse or recycling of waste from the healthcare sector have to meet this essential prerequisite. The environmental benefits, for example due to the substitution of primary materials, come second. A higher HCW segregation rate would eventually reduce the amount of waste incinerated at high temperature. As a consequence, less waste fuel would be supplied to the incinerator which would therefore require extra fossil fuel to achieve the required temperature. However, the energy required is largely compensated by the benefits from better recycling and the incineration at lower temperature with energy recovery (Tudor et al., 2009).

Desirably, a healthcare institution manages HCW by implementing the following measures:

- Segregating HCW at least according to the minimum recommendations of WHO, but minimising the amount of waste requiring the highest environmental impact treatment methods (landfill or high-temperature incineration).
- Segregating food waste and recyclables.
- Training all the personnel handling HCW and any other types of waste. Safe management training of HCW is mandatory in hospitals, but should also provide the required education and information on the best management option for MSW-like waste.
- Documenting all the procedures and protocols and monitoring the performance, according to EMAS, ISO 14001 or a similar standardised system.

However, segregation of HCW is dependent on the size of the healthcare institution. While small labs, clinics, dental practices, etc. generate a relatively small amount of waste with a varying proportion of hazardous waste and MSW-like waste, the total amount of waste in large hospitals is usually larger in specific terms (per patient, per bed or per doctor) than for small institutions, which is a counter-intuitive conclusion from the usual effect of scale. So, large hospitals tend to generate more HCW per patient or per bed as a consequence of the high degree of specialisation and the agglomeration of health services in hospitals (e.g. labs, in-house autoclaving and sterilisation units).

A high rate of diversion of offensive waste, which is not hazardous, is feasible due to the high costs derived from high-temperature treatments. From the waste contractor perspective, several practices have been implemented in recent years that have been very relevant to the management system such as pre-acceptance audits and offensive waste segregation. The separation rate of waste fractions has, therefore, improved, mainly motivated by the financial aspect of the management. According to Botterill (2014), waste management in hospitals is not regarded in terms of waste hierarchy but as a firefighting exercise (sic), where waste is assumed to exist and the cost of its management minimised as much as possible. Waste minimisation through prevention or reuse is still a long way off from its real potential. Also, Botterill (2014), through several interviews, identifies staff training as one of the key aspects to avoid or minimise health risks and waste contamination, while improving the waste management system performance.

Mercury-containing waste management

Mercury content in HCW is up to 50 times higher than in MSW, and emissions can be up to 60 times higher (IEC et al., 2015). It comes from thermometers, sphygmomanometers, dental amalgam, laboratory chemicals and preservatives, cleaning agents, and various electronic devices such as fluorescent lamps and computer equipment. The cost of replacing mercury-containing devices is not high; a training programme for a hospital can cost around USD 650 or less, while replacing thermometers and sphygmomanometers for example costs only USD 6 000. However, the main management of waste-containing devices or materials is segregation at source. For instance, segregation of dental amalgam is mandatory in most of the Member States in Europe (EC, 2012). Also, it is important to remark that avoidance of mercury by mercury-free purchasing policies at hospitals is the most effective way to reduce mercury in HCW (IEC et al., 2015).

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6.3.2. Healthcare waste treatment

Incineration

Incineration is the burning of waste at high temperature. In modern incinerators, a primary chamber exposes waste to lower temperatures under oxygen-starved conditions, causing pyrolysis. Then, the gases pass into a second chamber where they are burnt at a higher temperature (> 1 000 °C). Dioxins and furans in the emissions of waste incinerators have three main sources:

1. formation of PCDD/F from chlorinated hydrocarbons already in or formed in the furnace (such as chlorohydrobenzene or chlorobenzene);
2. de novo synthesis in the low temperature range (typically seen in boilers, dry electrostatic precipitators);
3. incomplete destruction of the PCDD/F supplied with the waste (EC, 2006).

The common technology for HCW incineration is rotary kilns, in contrast to the grate incinerators commonly used for MSW. Rotary kilns can achieve up to 1 450 °C, although the maximum temperature used for incineration of hazardous waste in rotary kilns is 1 200 °C (EC, 2006) in the post-combustion chamber to destroy PAHs, PCBs and PCDD/F. The rotary kiln is a horizontally rotating cylindrical vessel (from 10 to 15 metres long, up to 6 metres in diameter), where the waste is conveyed by gravity as it rotates. Normal residence times vary from 30 to 90 minutes, depending on the composition and the character of the waste. Due to the infectious character of certain fractions of HCW to be incinerated, pretreatment by shredding or milling is frequently avoided (or even banned), so the residence time required for full combustion is higher than for other wastes. The environmental impact of high-temperature incineration is relevant, as shown in Table 3.6 of the Waste Incineration BREF (EC, 2006).

A WHO review showed that small-scale HCW incinerators had “significant problems regarding the siting, operation, maintenance and management”; they are therefore only viewed as a transitional means of disposal for HCW (WHO, 2014).

Microbiological inactivation efficacy

Some of the HCW streams are required to be incinerated at high temperature due to its hazardous nature. Infectious waste, on the other hand, can be disinfected with alternative methods, not requiring high-temperature incineration, if a certain level of microbiological inactivation efficacy is attained. A consortium of regulatory agencies, called the State Territorial Association on Alternative Treatment Technologies (STAATT), developed criteria and consensus for the use of alternative treatments, establishing the levels of microbial inactivation efficacy shown in Table 6-3. They are still valid and recommended by the WHO (STAATT, 2005). These levels are accompanied by a list of indicators (i.e. concentration of microorganisms as a representative of each family) to be measured as a quantitative quality level. All alternative treatment methods should achieve STAATT level III. For instance, steam treatment in autoclaves usually requires a minimum time-temperature combination of 20 minutes and 121 °C, although it will always depend on the type of installation.

Table 6-3. Levels of microbial inactivation efficacy (STAATT, 2005)

Level I	Inactivation of vegetative bacteria, fungi, and lipophilic viruses at a 6 Log10 reduction or greater
Level II	Inactivation of vegetative bacteria, fungi, lipophilic/hydrophilic viruses, parasites, and mycobacteria at a 6 Log10 reduction or greater
Level III	Inactivation of vegetative bacteria, fungi, lipophilic/hydrophilic viruses, parasites, and mycobacteria at a 6 Log10 reduction or greater, and inactivation of *B. stearothermophilus* spores and *B. subtilis* spores at a 4 Log10 reduction or greater
Level IV	Inactivation of vegetative bacteria, fungi, lipophilic/hydrophilic viruses, parasites, and mycobacteria, and *B. stearothermophilus* spores at a 6 Log10 reduction or greater

Chemical treatment

Chemical treatment is the usual disinfection procedure for materials, floors and walls in hospitals. For HCW, the waste is mixed with a sterilisation agent, usually in wet conditions to improve the contact and the reactivity of the agent. The common chemicals used for that purpose are ozone, chlorine, formaldehyde, ethylene oxide, propylene oxide, periacetic acid (= peroxyacetic acid, $C_2H_4O_3$) and others. Usually, the sterilisation chamber also includes a shredder to reduce the size of the waste and improve the contact with the chemical agent.

Although this is the simplest treatment, it is probably the one that requires the most careful consideration. PATH (2005) detected the following issues regarding the technology:

- Not all chemical agents are effective; certain bacterial spores are resistant to chemical agents. The scale of resistance (WHO, 2014), from most to least resistant microorganisms, is bacterial spores, mycobacteria, hydrophilic viruses, lipophilic viruses, vegetative fungi, fungal spores and vegetative bacteria. A disinfectant effective against a particular group will be effective against less resistant groups.
- The process requires strict pollution control and highly specialised skills when handling certain chemicals. Sterilisation with aldehydes (e.g. formaldehyde) produces toxic gas releases and, therefore, those are not recommended for sterilisation.
- Large, bulky waste cannot be treated. This waste would require pre-shredding or simultaneous shredding, aimed to increase the reactive surface of the chemical agent.
- It produces an effluent that may be considered a hazardous waste; as treated waste is contaminated with the liquid effluent, it may be not acceptable as a MSW-like fraction.

Alkaline hydrolysis or digestion is a non-incineration method indicated to render safe and unrecognisable HCW consisting of body parts. This is done by heating the waste to a temperature between 110 °C and 127 °C in an alkaline solution (water plus sodium or potassium hydroxide) in a stirred tank for six to eight hours (WHO, 2014), removing any pathogenic microorganisms. The high pH of the final effluent requires treatment and hazardous waste management practices.

Autoclaving and steam-based treatments

Steam under pressure (autoclaving) or at atmospheric pressure (steam treatment or *wet* or *moist heat*) is used to increase the temperature of the treated waste up to a minimum of 121 °C for a certain time to achieve the desired level of sterilisation. The use of steam increases the contact with the waste and considerably improves the heat transfer, which can be improved by pre-shredding. In order to avoid excessive water condensation, the autoclave tanks can be heated, reducing the required steam temperature. The system operates at vacuum or negative pressure (for steam-based treatments) to allow steam penetration and air removal. The air released this way should be filtered through a high-efficiency particulate filter to avoid the release of pathogens. Some autoclaves release air at different pulses of pressure-vacuum repeatedly, allowing the system to gain pressure through steam addition and then applying vacuum (WHO, 2014). The released air is wet and potentially infectious; it requires further condensation and decontamination.

The operation of autoclaves requires a combination of temperature and time. The absolute minimum is 121 °C for 30 minutes, which would correspond to a pressure of 2 bar. However, time can be reduced thanks to pre-shredding and agitation of waste during the period in the autoclave. In any case, an effective sterilisation depends on many other factors: load size, stacking configuration, packing density, type of containers, physical properties of the materials, residual air and moisture content of the waste (Lemieux et al., 2006). The size of autoclaves can be from small 20 L units up to 20 m^3 and can treat from around 4 kg up to several tonnes per hour.

A drying step may be required to avoid excessive weight gain of the waste. Pre-shredding reduces the size of the waste particles and improves the sterilisation, while minimising the temperature and time parameters, but it may not be practicable under a strict control of risks of the shredder. Some devices combine shredding and sterilisation in the same chamber, but most operators shred after the sterilisation, along with compaction alternatives. Autoclaves combining sterilisation, mixing, shredding and drying are commonly known as integrated steam-based treatment systems or advanced autoclaves, and are designed for a continuous or semi-continuous operation. The investment required and the operating costs of these advanced designs are significantly higher than for conventional autoclaves.

Some aspects of the autoclaving operation are summarised below (PATH, 2005):

- The operation requires highly skilled operators.
- The input of mercury and heavy metals has to be completely avoided, to avoid water pollution. Also, volatile and semi-volatile organic compounds, chemotherapeutic waste and other hazardous waste that are reactive to water should be avoided in the feed.
- The operation generates a water effluent that needs to be treated before disposal/recycling to the process.
- The operation will generate odours, requiring an activated carbon filter. Also, it would not reduce the volume of waste. In fact, the final weight of waste will be increased due to the increase in water content if a drying step is not available.
- It requires a high amount of energy and it is not recommended for body parts or bulky wastes, as the temperature-time parameters for a full sterilisation are not easy to determine (WHO, 2014).

Dry heat

Dry heat consists of heating the element to be disinfected for a certain period of time in a closed chamber under a certain air pressure. Pressures, temperatures and times are usually higher than in steam-based systems, so its large-scale application is not competitive with other alternative treatment systems. However, it is commonly used to avoid health risks from small waste fractions at hospitals (WHO, 2014).

Radiative sterilisation (microwave)

This is a technique mainly used in the United States. It uses radiant energy (microwave or others) to heat the moisture within the waste (or water that is added to the waste). The radiation has no effect on microorganisms, but the combination of water and heat generates a steam pressure in the system for a certain period of time. A microwaving cycle may last from 30 minutes to one hour. The usual microwave unit combines the radiation with simultaneous shredding. Some of the operational aspects of the technique are summarised below (PATH, 2005):

- The capacity tends to be lower than in autoclaving processes. The use of microwaves does not allow continuous processes, so their treatment capacity is limited by loading and unloading operations.

- Some chemicals would react in the presence of microwaves and should be avoided in the feed. Mercury and other metals should also be avoided.
- It generates a water effluent that should be treated before its disposal or recycling.

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6.4. BEMPs for healthcare waste segregation

6.4.1. Encouragement of healthcare waste segregation at healthcare facilities

Summary overview

There is a significant potential to reduce the environmental impact of healthcare waste (HCW) management, in particular by targeting better prevention, segregation and treatment of non-hazardous waste, with due consideration of safety. It is BEMP for HCW management companies to:

- Organise waste audits at healthcare facilities in order to improve the knowledge of the various waste fractions and the current waste management practices.
- Help healthcare facilities with the definition of their waste management system by establishing clear guidelines for the categories of waste to be sorted.
- Organise training sessions to raise awareness among the healthcare facilities' staff and explain the rules for waste segregation (training sessions should be tailored to the different roles of staff within the healthcare facility and give special attention to addressing non-compliances identified during audits or during the handling of HCW by the HCW management company).
- Provide information material (posters, indications on containers, etc.) to help the healthcare facility's staff with instructions.
- Monitor the results and impacts of the action by defining a set of key performance indicators (including risk management and financial savings).
- Implement innovative technical solutions reducing the general environmental impact of the waste management system, e.g. on re-use of containers for the collection of HCW.

Better segregation of waste produced in healthcare facilities enables more recycling because it avoids that non-hazardous waste, including recyclables (e.g. printed paper, plastic bottles), is incorrectly put together with hazardous waste.

Waste management area

Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	HCW

Applicability

There is no specific limitation to the applicability of this BEMP by HCW management companies. However, the commitment of healthcare facilities towards an improved HCW management plays a key role for the type of measures and success of the actions implemented.

Specific environmental performance indicators

- Share of staff members of the client healthcare facility having undergone a training session about waste in the last two years (%).
- Share of correct answers given by staff members of the client healthcare

facility in post-training evaluation surveys about handling of waste in the HCW facility (%).

- Collection rates per waste fraction, per bed or per patient, according to the specific fractions collected in each healthcare facility (kg/patient/day).

Description

Healthcare waste (HCW) generated by medical institutions and healthcare facilities (including hospitals, medical laboratories and nursing homes for the elderly) includes both healthcare wastes generated by the treatment of patients (which can be hazardous or non-hazardous) and non-hazardous waste similar to municipal solid waste, such as biowaste, paper waste and residual waste.

The quantities of HCW generated range between 0.1 kg and 2 kg per bed and per day, according to various sources (PREDAS, 2009; Audit Scotland, 2001; WHO, 2015). Smaller sources of HCW are small healthcare establishments, independent practitioners, tattoo parlours, funeral homes or home treatment and mainly generate infectious waste and sharps (WHO, 2014).

Ensuring proper handling and collection and an appropriate treatment for hazardous medical waste is crucial due to the sanitary and environmental hazards it can entail. Risks include possible injuries provoked by sharp materials, poisoning through the release of pharmaceutical products and pollution through waste water, and can impact patients, health workers and staff of waste collection and treatment services. Its treatment requires special attention, i.e. preliminary disinfection prior to disposal, dedicated incinerators ensuring proper gas cleaning and limited human intervention (WHO, 2015). Sharp medical waste mixed with non-hazardous waste can provoke accidental blood exposure for waste collectors and operators on sorting lines, which can have severe consequences.

In addition to the issue of isolating hazardous waste, there is also a challenge in diverting non-hazardous waste (such as packaging or non-infectious nappies) from hazardous medical waste. This could lead to significant savings since medical waste treatment involves significant costs (Botterill, 2014).

Food waste can also be significant and is generated both during preparation in the kitchen and by patients (food wastage). In Ireland, an average of 730 g of food waste per patient bed per day was calculated (Greenhealthcare, 2009). In France, the Ministry of Agriculture assessed that food waste amounts to 260 g per person per day in healthcare facilities, with 166 g in nursing homes for the elderly and up to 363 g for short-term stay facilities. It seems there is great potential for waste prevention and separate collection. Other waste fractions are worth considering for source separation: for instance paper and packaging generated by the staff, patients and visitors.

Healthcare facilities face four main challenges when it comes to healthcare waste segregation:

- ensure the proper separation and disposal of hazardous medical waste to limit the health and environmental impacts on the staff and patients and for the waste management system;
- limit the presence of non-hazardous healthcare waste, such as non-infectious nappies or medical packaging, within hazardous flows, to limit the quantities of

waste sent to the HCRW treatment facility and thus the financial costs of the system;
- improve source separation of waste similar to MSW (food waste, paper and packaging) to increase recycled quantities and limit the quantities of residual waste sent to disposal.

The role of frontrunner HCW management companies

While the traditional role of HCW companies is to ensure the proper collection and treatment of waste generated in healthcare centres in accordance with the regulations, several HCW management companies provide additional services to healthcare facilities to optimise their waste handling system.

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- Help healthcare facilities with the definition of their waste management system by establishing clear guidelines for the categories of waste to be sorted.
- Organise training sessions to raise awareness among the healthcare facilities' staff and explain the rules for waste segregation (training sessions should be tailored to the different roles of staff within the healthcare facility and give special attention to addressing non-compliances identified during audits or during the handling of HCW by the HCW management company).
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Better segregation of waste produced in healthcare facilities enables more recycling because it avoids that non-hazardous waste, including recyclables (e.g. printed paper, plastic bottles), is incorrectly put together with hazardous waste.

The different elements listed as best practice are presented in the text below, with more details and information on actual implementations in healthcare facilities.

Waste audits

Establishing a waste audit is an important requirement for proper HCW management. HCW generation is very dependent on the type of activities carried out and treatments delivered at the healthcare facility, so relying on existing ratios and literature might not be sufficient to assess the current waste arising. Moreover, it is important to identify the current practices in terms of waste segregation by the healthcare (HC) facility staff and to identify potential non-compliance or where the most promising improvements can be achieved. Such audits are provided by several HCW management companies in cooperation with the HC facility staff.

Waste audits can be set at a regular frequency to allow a more complete assessment of the waste strategy (e.g. by performing new composition analyses and monitoring the actual evolution of the practices). While there are generally no specific obligations regarding the organisation of waste audits, national organisations can provide recommendations. For instance, the Department of Health of the UK Government gives indications on the frequencies with which they must be conducted: as a minimum, they should be conducted before any major change in the HCW procedure. For large producers (more than 5 t/year), a yearly audit is advised. For smaller producers, one audit every two years is indicated (HTM07-01, 2015).

A waste audit can be time- and resource-consuming; therefore a proper method has to be applied. It is important to organise it in collaboration with the various HC facility staff: general management that will agree on the objectives and grant access to the documents, the different managers of the audited services and the staff that will be interviewed. Before the on-site audit, it is important to plan the visits and interviews as well as to confirm the protocol with the managers. The existing documentation on waste management must be provided to the auditor for information and review.

Choosing days that are representative of the standard activity is also important. Midweek can be a relevant choice (GreenHealthcare, 2009).

Regarding the on-site audit, it must include the following elements:

- the assessment of the quantities and the types of waste produced, along with potential risks and their disposal methods;
- an overview of the collection system, including a diagram of the location of the bins, the colour coding, and the presence of a dedicated waste officer;
- a study of the current practices of the various departments, including questioning of the staff and a training overview;
- a composition analysis of the waste containers (survey on their content, including quantities) and their characteristics;
- an audit of waste storage areas (security of the rooms and containers, proper information, etc.).

Composition analysis can be performed in various ways depending on the level of detail needed: from a simple visual check allowing the identification of the presence of unwanted waste in the various bins to a complete analysis of several samples. For hazardous HCW, it is important to ensure the safety of the operator (by providing personal protective equipment and by assigning the sorting operations to staff with appropriate inoculations against potential infections). The choice of waste categories to be sorted depends on the objectives set. Templates to collect information are available from previous projects[75].

Waste audits can also lead to the optimisation of the use of containers: proper mapping of generation points, types of waste and level of segregation will provide sufficient information to calculate the need for containers and make their use more rational.

To avoid any conflict of interest and considering the fact that the outcome of the audit can affect the service provided by the HCW management company, it is strongly advised that the HCW management company adopt a very transparent approach and

[75] http://www.greenhealthcare.ie/wp-content/uploads/2013/11/Waste-survey-calculation-sheet.xlsx.

involve the healthcare facility as much as possible in the definition and implementation of the procedure.

Regarding traceability and indicators, HCW management companies play a relevant role by providing easy access to legal traceability documents, invoices and collected quantities. Several HCW companies propose a traceability system using barcodes which allows the monitoring of the waste from collection until its disposal. The barcode can include information on the type of waste, the ward that generated it, the time and location of its collection and of its arrival at the treatment site, etc.

Training the staff

Training is a very important element for improving HCW segregation. Several HCW management companies deliver training services for the HC facility staff tailored to the different needs and tasks of their daily activities:

- The **directors** must be trained in the issues and challenges of HCW management, which is both a legal obligation, a safety measure and a significant budget item.
- The **management staff** in charge of the various units or of environmental and quality management must be involved in the implementation and the monitoring of the waste management system. Making them aware of the safety issues and of the costs can be relevant to ensure their involvement.
- The **management staff of other services** (administration, logistics, etc.) which will be responsible for grouping and storing waste before its collection and treatment.
- The **staff performing medical treatment** (nurses, doctors, etc.) carry out the segregation of waste at the source; therefore their involvement and awareness are essential.
- **Other staff members** that are involved in waste production and waste handling (gathering and storing of waste, preparation and distribution of meals, etc.) also receive training.

The different training programmes can focus on the following topics:

- **A clear classification of HCW** according to its hazardousness and to the treatment options available. The classification has to be easy for the medical staff to understand and to apply, detailing the following fractions: s**harps** (needles, syringes, scalpels, etc.), **HCW with a risk of infection** (materials tainted by bodily fluids and pathological waste, **other hazardous HCW with specific treatment** (expired/unused pharmaceutical waste, radioactive waste, X-ray photographic material, cytotoxic drugs, etc.) and **non-hazardous waste**.
- **Adapted collection equipment** for each category of waste, ensuring the safety of the staff handling it (e.g. puncture-resistant containers for sharp waste), along with how to identify it (colour, EWC codes, etc.). The specific protocols regarding its handling (closing after use, sealing when full) can also be presented.
- **Specific recommendations to comply with national regulations**: adapted packaging for transportation of waste, waste operators with legal agreements to collect waste and the use of treatment units with legal agreements to receive and handle HCW, documents ensuring traceability and proving the proper treatment of hazardous waste, etc.

The World Health Organisation provides guidance for the organisation of training sessions which can be interesting to take into account (WHO, 2014):

- identify the employees to be trained in order of priority and define the objectives of the training sessions for the main target audience;
- define the form of the training according to the possibilities and constraints (time available for the target group and work schedule, etc.);
- for each programmed session, prepare the various elements such as main topics, main outcomes, teaching methods, associated documents, preparation required of the participant, etc.
- foresee pre- and post- evaluation of the target audience;
- promote and communicate information on the training session.

It is also important to remind healthcare facilities that training has to be a continuous process. Training must be provided to new staff members and reminders have to be provided so that the staff do not forget about the sorting guidelines. HCW companies can develop a service ensuring frequent training sessions, as well as regulatory surveillance.

Communication material

Several HCW companies assist HC facilities with waste segregation by making communication material available. Communication material can serve different purposes:

- **inform** about the waste segregation procedures and remind the staff about the different categories and recommendations;
- **raise awareness** of the necessity of HCW segregation and on the potential risks;
- **present the results** of a new strategy or alert the staff that the quality of sorting is decreasing.

Traditional communication materials for HCW are short guidelines, leaflets and posters that are distributed to the staff and put up in the rooms where wastes are sorted. It is important to make the communication material dedicated to the staff simple, visually attractive and very concrete.

Posters must present in a very clear way the main fractions to be sorted (identified by pictures or pictograms representing the main types of waste), the adapted waste containers (with pictures of the bins/bags to be used) and the treatment destination. Grouping them by categories (e.g. hazardous HCW / other hazardous waste / non-hazardous waste) can improve the readability. They must be put up on walls close to the containers and where treatments are performed. The company SRCL makes available a list of posters and proposes to tailor them to their client's needs (http://www.srcl.com/resources/posters/#). The posters can be adapted to specific healthcare activities and generally display very simple messages so that they can be displayed next to specific bins. The various templates include:

- for specific streams, the accepted and non-accepted waste as well as the appropriate container;
- poster summarising the different fractions sorted and the appropriate containers;
- practical recommendations for the handling and storage of waste and containers (the obligation to close HCRW bins, the method to optimise the stacking of sharps containers, etc.).

Posters can also be used to display information and feedback regarding the waste segregation performance. For these posters, the information must be very clear and use indicators that are the staff are familiar with and that can be compared to well-known benchmarking elements. Such indicators can be the overall sorting rate, the share of HCW in the total waste arising, the number of accidents linked to bad HCW handling, etc.

Source: Paris Region, 2012

Figure 6-1. Template of a poster presenting the result of the HCW strategy – *"18 % of HCW are sorted..."*

Consenur also proposes adaptable communication materials: sheets, posters, triptychs, and leaflets. They mainly address waste segregation solutions with very practical information:

- cards displaying the **various categories of waste**;
- posters explaining **how to fill and close** containers;
- signs to indicate **waste storage**.

These various materials are chosen according to the needs of the customer and designed in collaboration with the waste officer.

Source: Consenur

Figure 6-2. Information card explaining the closing and sealing of sharps containers

Implementing waste strategies

To assist healthcare facilities with the implementation and monitoring of their waste strategy, HCW management companies can provide integrated solutions incorporating several services. For instance, Stericycle proposes an "Integrated Waste Stream Solution" based around an online platform the healthcare facility can use to monitor and manage its various waste streams and set specific targets either for compliance or for sustainable strategies. The HCW management company provides all the relevant information regarding quantities for the various streams managed in the healthcare facility and divided among the various services, allowing the efficiency of the system to be improved. These data are based on the various tracking systems used during collection of HCW and when entering the treatment facility (consisting of a barcode system ensuring traceability).

Reusable containers

One of the main precautions when managing HCW is to limit interaction between workers and waste as much as possible, which generally leads to the disposal of HCW containers with the waste it contains. However, several HCW management companies offer a sharp waste service with reusable containers. With this system, containers are handled with an automated system, preventing contact with workers. The sharps are put on a conveyer belt then emptied in the incineration kiln, then cleaned and autoclaved. A final check is done to ensure they have been properly disinfected. If not reusable, the container can then be sent for recycling (see also specific case study in Operational data).

Such a system can also be used for non-sharp HCRW through a combination of a plastic bag with a plastic container. Bags are closed when full and containers are secured before treatment. At the treatment facility, the container is emptied, cleaned and disinfected then sent for reuse.

Achieved environmental benefits

Improving source segregation for HCW entails several positive environmental outcomes:

- Reduce reliance on high temperature treatment by limiting the presence of non-hazardous waste in hazardous streams;
- Reduce reliance on disposal by increasing the fractions sent to recycling, also allowing to limit the environmental impact linked with disposal (GHG and air pollutant emissions…);
- Environmental benefits linked with the increasing of recycling.

Diverting non-hazardous waste from HCW allows reducing the quantities sent to specific treatment such as high temperature incineration. HCW incineration generally requires higher temperature and longer incineration times than MSW incineration, which leads to a higher energy consumption (EC, 2006).

Most environmental savings can be achieved thanks to the diversion of recyclable fractions from residual waste, for instance food waste and paper. Proper treatment of organic waste (e.g. anaerobic digestion) and material recycling from recyclables allows achieving greater environmental savings than sending waste to incineration and landfil.

The use of re-usable containers for sharps instead of one-way containers also provides an interesting environmental benefit. Two hospitals located in the U.S. reported to reduce their carbon emissions by 60 to 70 tonnes of CO_2 emissions in one year due to the use of re-usable containers (ICT, 2011).

The implementation of the bag+container system as an alternative to the single-use containers allows reducing the quantities of waste produced and sent to disposal to about 15-20%. According to a case study presented by Consenur, a hospital implementing the Programme can achieve to use 98% of its containers with a bag+container system.

Appropriate environmental indicators

The monitoring of the BEMP can be achieved at two levels: monitoring of the activities and direct results (e.g. monitoring of the waste handling practices during waste auditing, or monitoring of the level of knowledge of the staff before and after training) and monitoring of the impact of the BEMP on the waste data.

To monitor the activities and the direct impact of these various activities, several indicators can be considered:

- If the HCW management company conducts a waste audit, several elements can be monitored to assess the initial situation and then to measure the progress:
    - Regular waste audits are carried out on the HCW generated (y/n).
    - Collection rates per waste fraction, per bed or per patient, according to the specific fractions collected in each HCW facility (kg/patient/day).
    - Share of hazardous waste in the total waste generation of the HCW facility (%). However, this indicator can be difficult to determine, because the hazardousness of the waste may be difficult to detect (e.g. contamination with blood or body fluids).
    - The proportion of staff members having undergone a training session about waste in the last two years (%).
    - The number and list of non-compliances and their evolution.
- If the HCW management company organises training sessions:
    - the proportion of staff members of the HC facility to undergo the organised training sessions (%);
    - Share of correct answers given by staff members of the client HCW institutions in post-evaluation surveys about handling of waste in the HCW facility (%).
    - the success rate for training sessions based on an assessment of the staff waste management practices, as mentioned in the following paragraph;
    - the progress of knowledge of the various employees, assessed with pre- and post-evaluation surveys.

The evaluation of training programmes can be achieved by monitoring the staff behaviour through direct observation and surveys, but this type of evaluation is not necessarily performed by the reference organisations identified. Most of the time, monitoring the quantities of waste and non-compliances after training sessions allows their effectiveness to be assessed.

To assess the results of training on waste segregation and the right implementation of waste handling, the following indicators can be monitored:

- **Collection rates** for the various fractions (hazardous, non-hazardous medical waste, food waste, paper waste, residual waste, etc.). To ensure the data are comparable, ratios per bed or per patient can be used.
- **Recycling of "similar waste"**: recycled quantities for waste assimilated to MSW, e.g. evolution of the quantities of food waste sorted and sent for recycling. Relevant indicators can be: total sorted quantities (in t/month or in t/year), sorted

quantities per patient or per occupied bed (allowing the monitoring of the quantities while taking into account the evolution of HC activities), % of residues in the sorted fractions (these data can be retrieved from the waste treatment units/recyclers).

- **Diversion rate of residual waste from final disposal.**
- To assess the **presence of non-hazardous waste in hazardous streams**, the most common indicator is the share of hazardous waste in the total amount of waste. If a preliminary waste audit has been performed, the average proportion of hazardous waste compared to the total waste generation is known. Comparing these reference data with the monitored percentage allows the tracking of important deviations which can be linked to the presence of non-hazardous waste in hazardous streams. This indicator is often used by HC facilities; it is however important to take into account possible seasonal variations (e.g. periods when administrative archives are disposed of). Another method is to conduct random checks of hazardous containers for the presence of unwanted waste and to monitor this evaluation. Conducting more in-depth composition analyses provides a more accurate assessment but they are time- and resource-consuming.
- The identification and reporting of **non-compliances** by the HCW management company can also be regarded as a relevant indicator.

It can be interesting to consider sharing these indicators with the HC facilities to demonstrate the quality of the service and identify any decrease in the quality of the waste segregation by the healthcare staff.

Several indicators can help monitor the sanitary impact of the BEMP. These indicators can also be used to assess the presence of hazardous waste (and sharps) in other waste streams:

- Number of accidents linked with contact with hazardous medical waste (injuries with sharps, etc.).
- Number of accidental blood exposure incidents linked with the handling of waste.
- Number of times radioactivity is detected in treatment units receiving non-hazardous medical waste. Treatment units for non-hazardous waste such as incinerators and landfill sites are generally equipped with detectors and register the number of incidents and level of radioactivity. Procedures generally imply that the waste generating the radioactivity is identified. Therefore, this indicator can be assessed with the support of treatment facilities.

It is also possible to define qualitative indicators based on the number of non-hazardous bags containing infectious waste. To ease the implementation of this indicator, the HCW management company can provide transparent plastic bags (instead of the traditional black ones), allowing quick identification of non-compliance.

The number of accidents can be monitored both in the HC facilities and during the management of waste (collection, sorting and treatment). It is generally obligatory to monitor and report these elements, according to national legislation.

Both Consenur and Stericycle also propose an **environmental impact assessment tool** to assess the environmental impact linked with the implementation of their solutions:

- Stericycle's tool allows the assessment of the potential CO_2 savings based on the number of beds in the hospitals[76];
- Consenur's tool provides very simple indicators on CO_2 and the saved quantities of virgin plastics[77].

Cross-media effects

As stated before, better segregation of waste produced in healthcare facilities enables more recycling.

The implementation of the practices presented above (organisation of training sessions, waste audits, etc.) are likely to have limited cross-media effects as they do not involve resource-consuming or polluting processes.

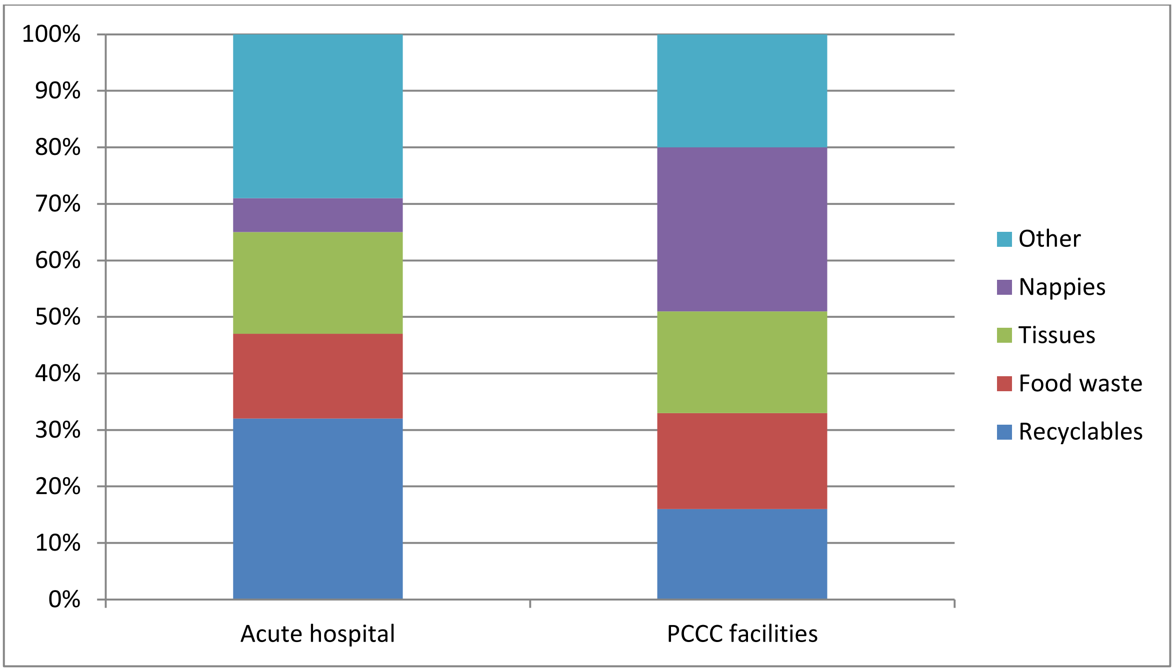
Full implementation of this BEMP will generate cross-media effects linked with the creation of new collection routes and new treatments, yet the effects are likely to be limited compared to the avoided impacts. The implementation of the BEMP will lead mostly to environmental benefits.

Operational data

As said before, it is difficult to provide relevant benchmarking elements regarding the arising of HCW and the composition of the main waste fractions. HCRW amounts to about 10–20 % of the total waste arising depending on the activities. Regarding recyclable fractions, the Green Healthcare Programme analysed the composition of non-hazardous waste in acute hospitals and in primary community continuing care facilities based on a survey and identified the following quantities.

[76] Available here: http://www.stericycle.com/compliance/tools-and-resources/carbon-footprint-estimator/

[77] Available here: http://www.srclconsenur.es/calculo-medioambiental



Source: GreenHealthcare, 2009

Figure 6-3. Average composition of non-hazardous waste in Irish acute hospitals and PCCC facilities

These data show that the potential for recycling ranges between 30 % and 50 % for non-hazardous waste (taking into account only recyclable fractions and food waste).

To assess the potential savings achievable through staff training, audits and communication activities, several case studies documented with quantitative data are presented.

Case 1: Reduction of non-hazardous waste in hazardous streams through staff training

This action was implemented by the Cochin Hospital in Paris, France. In 2005, a significant issue was identified regarding HCW segregation: healthcare risk waste amounted to 36 % of the total waste arising, when national figures tended to show that only 20 % of HCW belonged to the infectious category. To reduce the presence of non-hazardous HCW in the infectious fraction and the associated costs, several training sessions were contracted by the hospital. It consisted of general training in plenary sessions and smaller training sessions for groups composed of 20–100 staff members. In total, around 600 people were trained, and a part on waste was added to the welcome training for new employees.

Within one year, the proportion of infectious waste decreased from 36 % to 22 %, generating savings of about EUR 200 000. In 2009, a study on HCW by the Health Ministry and five other HC facilities led to further training sessions specifically designed for the various wards; this allowed the share of infectious HCW to be brought down to 19 %.

To ensure proper monitoring, monthly figures are proposed to quickly identify non-compliance. Adapted responses have been defined depending on these results:

- at 20 % or below, the performance is considered good and no actions are foreseen;

- at 21 %, closer monitoring of the collected quantities is set in order to gather more detailed data;
- at 22 %, investigations into the sources of the increase and the reason behind it are launched;
- at 23 % or more, a new set of training sessions is programmed to improve the performance.

Monthly monitoring is also done on costs and savings resulting from the good or poor performances compared to the 20 % objective. For instance in 2010, the annual average was calculated at 19 %, implying savings of about EUR 19 400.

Case 2: Programme for the Sustainable Management of HCW by Consenur, Spain (Sanmartin, 2016)

The Spanish company Consenur developed a Programme for the Sustainable Management of HCW after acknowledging that most of the improvements in HCW management have to be achieved by improving the management inside healthcare facilities. This programme encompasses various services provided to HC facilities:

Waste audits: a general method for waste audits is established, based on interviews with the staff, site visits to the different places where waste is generated on the facilities and analysis of waste data. These audits are conducted both to assess the room for improvement (non-compliance and environmental improvements) and evaluate specific actions such as training sessions. Audits are conducted by auditors in coordination with the waste officer. This collaboration and an evidence-based approach ensure the objectivity of the results.

Consenur’s audits are organised around different steps:

- **A preliminary meeting** where the auditor and the staff agree on the scope of the audit, the criteria for the assessment and the timeline.
- **A planning phase**: a schedule for carrying out visits and interviews is set and presented to the HC facility management to ensure an agreement on the objective, confirm the protocols and confidentiality of results, and grant access to documentation.
- **Preparation of the documentation**: with the use of a checklist, the audit team reviews and analyses the existing documentation (protocols, specific information and data related to the various services, etc.).
- **On-site audit**: a meeting is organised with the managers of the different services concerned by the audit. Data and information are collected through interviews with the staff managing waste or interacting with the waste management system as well as through sampling in the different containers. This allows the existing fractions to be mapped, the compliance of containers to be verified, behaviour to be compared with the established protocols and the level of awareness of the staff to be assessed.
- **Drawing conclusions**: in collaboration with the waste officer, the main observations and conclusions are drawn (non-compliance, mass-flow analysis), as well as the identification of corrective actions.
- **Final report and closing meeting**: a final meeting with the different managers is set to validate the audit and the results and discuss corrective actions. A final report is delivered presenting the audit and the main conclusions regarding strengths and weaknesses.

The close collaboration with the HC staff and management ensures proper transparency which limits the risk of conflicts of interest.

Training plans: after agreeing with the healthcare facility about the objectives of the training and its content, a training programme is presented and implemented. In general, training focuses on both the practical aspects of waste segregation (identification of waste, handling procedures, etc.) and awareness-raising on the environmental and sanitary impact of an improper HCW management system, especially by highlighting the possible risks to the waste chain in the case of improper segregation. Training also tackles the issues of high costs linked with poor segregation or the protocols in case of injuries with waste. The different types of staff members can be targeted by the various training possibilities.

The training plan is drawn up with a technical working group and validated by the management. The technical working group regularly meets to discuss the progress of the plan. Trainings are developed from the viewpoints of the healthcare staff, taking into account their initial level of awareness. Trainings are composed of face-to-face meetings, online training courses and practical workshops to put into practice the knowledge gained. The online platform also allows trainees to access relevant communication material, general information, monitoring of their progress and a forum/chat to ask specific questions.

At the end of the training programme, the trained staff undergo several tests with immediate correction. An evaluation form is also filled in to assess the relevancy of the training method. Constant evaluation by trainers is performed over the course of the training, allowing the progress to be monitored. When the evaluation is complete, certifications are provided. A general evaluation of the training programme is also performed to identify potential improvements for the programme.

Optimising the use of containers: this action consists of different steps:

- identification of the generation points, the types of waste and the level of segregation;
- calculating the need for containers according to the first step, and rationalisation of the number and the location of containers;
- concrete implementation with definition of collection frequencies.

These actions allow the facility to optimise the number of containers in use, rationalise their location and ensure better monitoring of their use.

Consenur has also developed two new practices to limit the impact of HCW containers:

- **Recyclable containers**: a new treatment process avoids the crushing of the containers with the waste and sending it in landfills. Instead, the containers are emptied and sent to material recycling.
- **Bag plus plastic container system**: these containers are redesigned to improve their durability and are reused. Bags are closed when full and containers secured prior to their transfer to the treatment unit, where mechanised unloading is performed. The HCW inside the bag is treated while the container is sent for disinfection and washing. Both internal and external tests are performed. An external, independent auditor carries out the external tests.

It is important to note that this system cannot be used for sharp objects.

Case 3: Waste audits and training provided by SRCL, UK

SRCL provides guidance to healthcare companies on conducting pre-acceptance audits that are imposed by the UK Environmental Agency to inform treatment units about the nature and composition of the waste they will receive. Among others, it provides guidelines and a checklist for healthcare facilities as well as technical assistance to implement the audit (UK Environmental Agency, 2010).

In particular, these audits must include, among others, the following elements:

- the assessment of the quantities and the types of waste produced, along with potential risks and their disposal methods;
- an overview of the collection system, including a diagram of the location of the bins, the colour coding, and the presence of a dedicated waste officer;
- a study of the current practices of the various departments, including questioning of the staff and a training overview
- a composition analysis of the waste containers (survey on their content, including quantities) and their characteristics;
- an audit of waste storage areas (security of the rooms and the containers, proper information, etc.).

Regarding the composition of the various containers, SRCL performs visual composition audits by looking directly at the content of the bin to determine the general composition and spot non-compliances and irregularities.

The audit focuses exclusively on non-compliance issues in coordination with the healthcare facilities, thus ensuring commercial considerations are excluded from the audit. Involving external, independent contractors can also be an option.

SRCL also provides several training options for healthcare companies: an online training centre with streamlined training as well as on-site training with the specific presentations. The online training system includes an attendance tracking system to ensure the proper training of the staff, with the possibility to register members of staff to specific training sessions (either focusing on one particular type of waste or general procedure). More specific training activities can be organised following on-site evaluations of waste management and linked with site-level reporting, allowing the optimisation of the general organisation of training.

Training courses proposed by SRCL include the following elements:

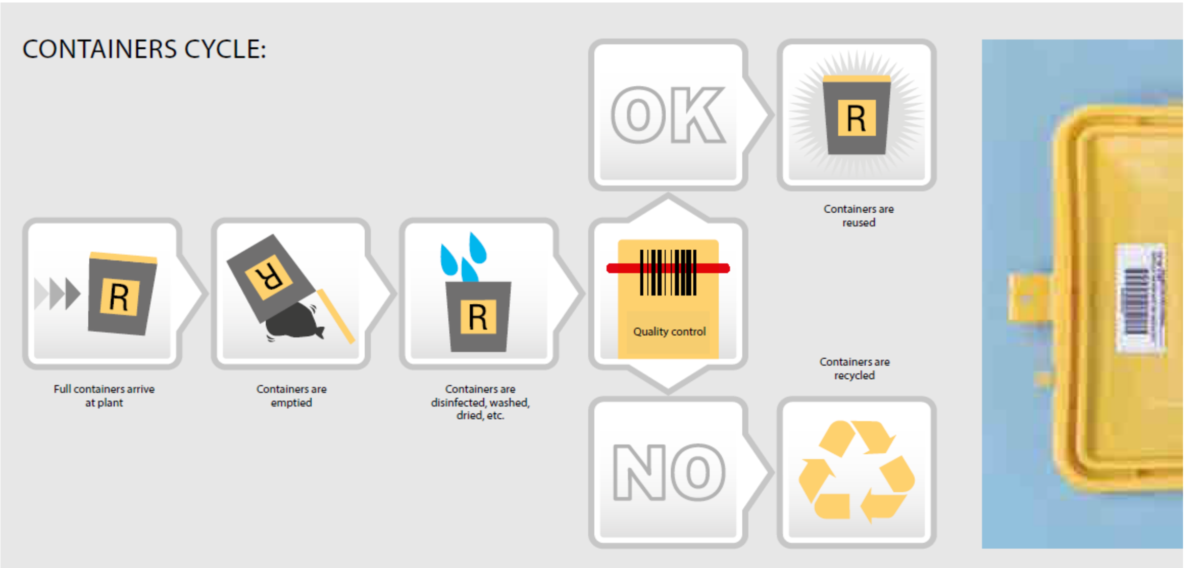
- **presentation of the regulation**: general environmental regulations, legislation associated with hazardous waste and waste management;
- the **main obligations** of healthcare facilities: ensure proper collection, storage, transfer and treatment for all the generated waste, authorised collectors and treatment units, etc.
- **possible risks** of non-compliances or improper management: fines, safety, higher costs linked with specific disposal options;
- the **various types of waste** generated by category: clinical waste (i.e. healthcare risk waste (HCRW)), offensive waste;
- the **main waste fractions to be segregated** along with the appropriate containers and the colour code;
- **recommendations** and an explanation regarding the labelling and tracking of bins (using barcodes).

SRCL helps with the monitoring of information by providing an online client platform, as mentioned above.

Case 4: Reusable HCRW containers – Mengozzi's and Stericycle's systems

The company Mengozzi, specialised in HCRW treatment, has developed a system of reusable containers since the 1980s in order to reduce the amount of waste sent for hazardous waste disposal (thus ensuring more treatment capacities for other hazardous waste), reduce the costs associated with HCRW management and reduce the associated CO_2 emissions.

Mengozzi's system consists of reusable containers that can be reused up to 12 times before being recycled. When entering the treatment process, the containers are put on a conveyer belt, automatically emptied in the incineration plant and disinfected, washed (first wash with a soda-based solution, second wash with chlorine and bromine solution then rinse with warm water) and dried. Then they undergo a quality control using ATP-bioluminescence meters to detect any biological residues and their barcode is scanned to monitor the number of uses. If the barcode is scanned for the twelfth time, the container is sent for recycling and reprocessed into a new one, with the introduction of 7 % virgin materials. Otherwise, the container is ready to be reused and sent to a customer.



Source: Mengozzi brochure

Figure 6-4. Scheme explaining Mengozzi's container cycle

Stericycle (Williams, 2016) also provides such a service for sharp containers. The system is very similar. Disinfection is achieved through a high-temperature/high-pressure wash and a high-temperature forced air drying process before undergoing a quality control. Stericycle states that one container can be used up to 600 times, thus replacing 600 single-use containers. The system was implemented by SRCL in four main HC sites in 2015: Salford Royal Hospital, Papworth Hospital, St Mary's Hospital on the Isle of Wight and the Heart of England NHS Foundation Trust (SRCL, 2016).

Case 5: Food waste segregation by implementing a new collection system and staff training (*Source*: WRAP, 2014)

In 2011, the service provider Sodexo initiated a programme with three healthcare clients: Central Manchester Hospital, Queen's Hospital in Romford and Queen Mary's Hospital in Roehampton (UK). The programme consisted of a review of the waste management system by applying the waste hierarchy principle to the various waste streams. Following this survey, several actions were conducted including the introduction of food waste segregation, in parallel with the separation of dry recyclables.

The introduction of food waste segregation was done by setting a specific collection system: food waste is collected by domestic staff in 35-litre caddies with biodegradable liners located in the kitchen, and then the liners are put in 240-litre bins located in local bin stores adjacent to each ward. These containers are collected three times a week and sent to an anaerobic digestion plant, and then cleaned by the waste collectors. The implementation of the new sorting scheme was promoted via communication and training activities mainly targeting the staff members in charge of waste collection. Training sessions were organised at the beginning of the programme and an on-going programme was also launched, e.g. to address contamination issues. The communication materials consisted mainly of bin stickers explaining which waste is included and which waste is not. Sorting performances are communicated to the healthcare facility, along with the energy production achieved by the anaerobic digestion of food waste.

For the Central Manchester Hospital, about 1.9 kg/bed/week of waste has been diverted from the residual fraction. About 12.5 tonnes of food waste could be recovered each month. The introduction of this scheme also had a positive effect on the segregation of other recyclable fractions: the overall recycling rate went from 25 % to 95 %, allowing the diversion of more than 1 000 t/year of waste from landfills and generating about EUR 20 000/year of savings for the food waste alone.

Other examples of such practices could be identified:

- The network of regional HCW management companies Réseau GC provides waste audits consisting of an on-site assessment performed in collaboration with the healthcare facility's waste officer and aiming at the identification of potential risks linked with HCRW management, the assessment of the containers in use and the potential savings that can be achieved with more effective waste management. It also delivers staff training directed at healthcare facilities, focusing on the following aspects: general definition and related risks of HCW, HCW segregation and sorting material, legal and technical aspects related to the segregation and storage of HCRW, appropriate treatment as well as traceability and reporting.
- The company Rhenus Logistics organises **three-day seminars** targeted at waste experts from healthcare facilities (such as waste officers appointed by healthcare facilities to manage the general waste management system). These seminars mix lectures, workshops, site visits and practical demonstrations focusing on waste regulation, waste types, case studies and good practices and procedures to comply with the national requirements.

Applicability

Considering the expertise of HCW management companies when it comes to regulation, legal requirements and the collection of data linked to tracking and reporting obligations which they perform within the framework of their traditional activities, it is likely that most of them can adapt these practices and propose adapted services to their customers. From the general information gathered in different national guidelines (WHO, 2015; GreenHealthCare, 2009; HTM07-01, 2015; C2DS, 2015), it seems that most healthcare facilities are concerned by the need to comply with the regulation and to reduce costs thanks to a more effective waste management system. The degree of improvement will depend on several parameters:

- current waste arising and practices: setting and promoting new separate collection schemes is technically and economically feasible if sufficient quantities are generated;
- the economics and financial balance;
- the involvement of the staff;
- if considering new recycling routes, the availability of local treatment facilities: an example could be found where a hospital (Ambroise Paré Clinic located in Beuvry, France) has to send pharmaceutical glass waste to a dedicated treatment plant in Denmark, leading to relevant transport costs (C2DS, 2015).

If no waste protocol is defined in a healthcare facility, it is then likely that the staff lack knowledge on legal and technical aspects of HCW management. HCW management companies can provide interesting input related to regulatory surveillance, specific training for both managers and staff members performing treatment and generating waste, and technical competences to establish waste audits on current practices, compliance with the regulation and potential improvements.

In general and as presented in the BEMP description, it is recommended to set waste audits and staff training on a regular basis, making it relevant for HCW companies to consider providing such services. As mentioned above, due to the potential risk of conflicts of interest and considering the fact that the outcome of the audit can affect the service provided by the HCW management company, it is strongly advised that the HCW management company adopt a very transparent approach and involve the healthcare facility staff members, such as the waste officer, while proposing such services.

Economics

The services described here can either be integrated in the tenders issued by healthcare facilities for the handling of their waste or be delivered as a occasional service, in which case a specific fee will be paid.

For waste audits, a range between EUR 5 000 and EUR 20 000 could be identified for one-off audits. It largely depends on the size of the HC facility and the level of detail. For a yearly waste audit service, the common costs range between EUR 1 000 and EUR 3 500 per year, depending on the size of the facility. A detailed composition analysis can reach about EUR 10 000. A cost of EUR 4 000–5,000 for the training of a HCW officer was reported by a HC facility. However, these costs depend to a large extent on the scope of the services provided.

Driving force for implementation

The main driving force for the implementation of the BEMP for a HCW management company is the demand from its clients. Indeed, HCW management is subject to strict legislation and the management of hazardous waste can represent a significant budget item. For instance, Council Directive 2010/32/EU imposes several measures to prevent injuries linked with sharps management: implementation of specific waste procedures, use of technically safe and clearly marked containers, and training. Awareness-raising and training are explicitly mentioned in clauses 7 and 8 of the Directive. HCW regulations evolve, so healthcare centres might prefer to rely on the expertise provided by HCW management companies to ensure their compliance with the legislation as well as the safety of their staff.

Therefore, HCW management companies usually provide these services to satisfy the demand of their customers. These services are provided as part of the contract or for a specific fee.

Specific regulations can also be a driving force for the implementation of this BEMP. For instance, the UK Environmental Agency has made it mandatory for HC facilities to establish a pre-acceptance audit to inform the treatment companies about the nature and composition of the waste they will receive. Therefore, HCW companies are encouraged to cooperate with healthcare facilities for the creation of these audits to comply with the regulation.

Most healthcare facilities aim at optimising their budget. Several elements make it important for them to focus on improving HCW segregation:

- The cost of non-compliance can be high depending on national legislation, with possible fines.
- The significant cost for hazardous medical waste handling. Depending on local situations, management costs for healthcare hazardous waste are reported to be between 5 and 10 times higher than for non-hazardous waste. HCRW management costs between EUR 500 and EUR 1 000 per tonne while residual waste management costs generally range between EUR 100 and EUR 200 per tonne.
- The potential costs linked with food wastage.
- The potential positive economic balance of recycling for certain fractions (food waste, paper, etc.), depending on current costs for disposal (and associated taxes), the cost of separate collection and the market value for dry recyclables (e.g. for paper waste). This depends on the local framework, especially taxes on disposal which can make recycling less expensive than incineration or landfilling.

The cost of a waste audit and staff training can be significant for a healthcare facility. In addition to the cost of the service, the healthcare facility must also include staff costs linked with the staff members attending training courses. However, most case studies indicate significant savings due to diversion of non-hazardous waste. Based on the collection of good practices implemented in several French healthcare facilities, the following savings were identified (ARMEN project, 2012):

- an average 10 % saving on all HCW management costs is achievable by carrying out an external audit on HCW management;
- an average 25 % saving on all HCW management costs is achievable through the optimisation of HCRW management by diverting non-hazardous waste, by carrying out audits and training and implementing new waste sorting protocols.

The financial balance of diverting the recyclable waste from residual waste is generally positive; however, the local cost for disposal and taxes on landfilling and incineration might make disposal less expensive than organic recovery (C2DS, 2015).

The use of reusable containers as presented above can also represent a source of savings. Their use can entail cost savings of 15–20 % compared to single-use containers.

Besides hazardous waste, national legislation can make selective collection of several types of waste mandatory for non-household waste producers. For instance, the Irish

Waste Management (Food Waste) Regulations 2009 make it mandatory for commercial producers to separate their food waste. The Article L 541-21-1 French Environmental Law makes it mandatory for entities producing more than 10 tonnes per year of biowaste to organise its selective collection. These obligations can also exist for other recyclable fractions.
Improving HCW management can also have direct benefits for the HCW management companies, by limiting the possibilities of accidents for their collection and treatment staff, such as injuries due to sharp waste. In the event of many non-compliances, a HCW company can strongly suggest to its customer to implement an audit and provide training and communication material to limit the risks and comply with the regulation.

Reference organisations

Several HCW management companies performing services relevant to this BEMP have been identified:

- Stericycle (UK): a HCW management company that also provides services in training, communication support and on-site evaluation regarding HCW management.
- SRCL (UK): a national contract for healthcare risk waste management is signed with this company to collect healthcare risk waste from HSE facilities. SRCL offers training on risk waste segregation to hospitals.
- Mengozzi Rifiuti Sanitari (IT): an Italian waste company handling HCW that also provides HCW training and proposes reusable HCRW containers.
- Consenur (ES): a HCW management company in Catalonia providing training and support for HC facilities. It has developed a specific programme aiming at making the management of HCW more sustainable by addressing the various steps of HCW management.
- Rhenus eonova GmbH (DE): a company specialised in HC logistics that provides HCW management services. It proposes staff training focusing on compliance and good practices as well as audits for healthcare facilities (Rhenus, 2016).
- Réseau GC (FR): a network of regional HCW management companies providing audits and training for better segregation of HCW in various HC facilities.
- Other organisations and projects were consulted to cross-check the relevancy of the good practices identified:

- Paris region (FR): within the framework of its Regional Plan of Infectious Medical Waste, the Paris region has conducted studies with hospitals in order to improve waste management, and especially the handling of hazardous waste. The study has led to several concrete implementations, especially waste audits and staff training which could contribute to the case studies. (Elements available here: http://espaceprojets.iledefrance.fr/jahia/Jahia/planification_dechets/site/projets/pid/6022.)
- GreenHealthCare Programme (IE): this programme is an initiative by the Irish EPA under the National Waste Prevention Programme aiming at preventing waste and reducing costs linked to waste management. HC facilities have benefited from waste audits, exchange of good practices and benchmarking. The programme is an interesting source regarding waste arising and methodological elements that can be applied at local level. It displays several

relevant case studies concerning HCW management. (http://www.greenhealthcare.ie/resources/)

- EU-HCWM project: an EU project aiming at providing a common approach for the development of National Occupational Standards and Vocational Educational Training Programmes for Healthcare Waste Management across EU Member States. http://www.hcwm.eu/knowledge-base
- Health Care Without Harm Europe: a non-profit association of hospitals and HC actors working, among other topics, on waste management issues. They take part in a project called "UNDP GEF Project on Global Healthcare Waste" that produces a significant amount of training material focusing on sustainable waste management in HC facilities.

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6.4.2. Healthcare waste collection for households

Summary overview							
This BEMP focuses on collection systems implemented by local authorities and/or waste management companies to collect hazardous HCW generated by households, specifically sharps and needles generated from treatments performed at home. It is BEMP to adopt a separate HCW collection scheme for households that ensures safe and environmentally friendly HCW collection and management by: - assessing the quantities of HCW arising; - providing appropriate boxes for collection; - selecting collection methods and frequency of collection according to local conditions; - involving stakeholders, typically: pharmacies and other healthcare actors (such as medical doctors and nurses), patients performing home treatment and the medical industry; - putting in place controls and corrective actions for the HCW collection system.							
Waste management area							
Cross-cutting	**MSW - strategy**	**MSW - prevention**	**MSW - collection**	**MSW - EPR**	**MSW - treatment**	**CDW**	**HCW**
Applicability							
The BEMP is applicable to all local authorities and/or waste management companies.							
Specific environmental performance indicators							
- Number of collection points for HCW generated by households per 10 000 residents, by type (civic amenity sites, pharmacies, street containers) - Number of individual boxes for HCW generated by households distributed via collection points or on request. - Quantity of HCW generated by households collected (kg/capita/year). - Share of HCW (e.g. sharps) in mixed household waste (%).							

Description

Healthcare waste generated by households

Healthcare waste can be generated in a private household/office: pharmaceutical waste, e.g. unused or expired medicines and sharps generated by self-administered treatment (e.g. insulin for diabetes). As mentioned in Section 6.1, the status of the waste generated at home depends on the person performing the treatment. Waste generated from home treatments performed directly by patients is considered household waste and is the responsibility of the patient.

Household healthcare waste (HCW) can be non-hazardous (e.g. diapers), in which case it is generally accepted as residual waste in the MSW collection system as long as it is properly bagged and sealed. However, hazardous waste (also called healthcare risk waste or HCRW) should not be mixed with residual waste and for this waste type a specific collection scheme has to be implemented. Expired medicines are managed through dedicated systems across Europe, generally involving take-back schemes in pharmacies (*Medsdisposal, 2015)* to avoid improper disposal (in the waste treatment system or in landfills). However, the practices regarding the management of household sharps are less widespread and more heterogeneous across Europe (*Medsdisposal, 2015).* Therefore, this BEMP will focus on collection systems implemented by local authorities and/or waste management companies to collect sharps and needles generated by home treatments, i.e. treatments performed by residents.

Local HCRW management: sharps and needles

While HCRW is the responsibility of the patient (if he/she is performing the treatment), local authorities must ensure the safety of the waste operators collecting municipal waste and must provide home-treatment patients with guidance and information (DEFRA, 2013). With this in mind, they can themselves promote or implement a dedicated collection scheme for HCRW such as sharps and needles, and any potentially infectious waste.

Sharps and needles present a significant risk for patients, waste operators and any person in contact with the sharps due to their potentially infectious character and the fact that they can provoke injuries favouring infections (the most common being Hepatitis B and C and HIV) (Bristol-Myer Squibb Company, 2016). Potential injuries can occur during collection (especially if residual waste is collected in bags) and on manual sorting lines in dry recyclables sorting centres. Therefore, sharp waste requires a collection system that prevents contact between waste and the operators, as well as any other person. They are also produced in small quantities, making traditional collection systems (e.g. door-to-door collection) difficult to implement. It is therefore important to separate and isolate them, and to ensure they undergo proper treatment.

Scope of HCRW generated by households

The scope of action of the HCRW collection system has to include: waste generated by self-treating patients, waste that can injure waste operators and possibly a list of pathologies caused by sharps. Sharps waste commonly encompasses any medical waste that includes devices used to puncture or lacerate the skin. Sharps included in local collection schemes generally include needles, syringes and lancets as the main categories. The Bristol-Myers Squibb Company provides the following list of waste encompassed in the "sharps" category: needles, syringes, lancets, auto injectors, infusion sets and connection needles (Bristol-Myer Squibb Company, 2016). The French EPR scheme has detailed a list of 18 pathologies and 10 types of waste that are included in the system: lancets, separated needles, needles for pen needles, micro drip, transfer sets, catheter, all-in-one catheter, pen needles, syringes and Imiject syringes (DASTRI, 2016).

When it comes to pathologies and as was mentioned above, it is possible to identify about 20 pathologies that can potentially be caused by household sharps and needles (ORS Rhône-Alpes, 2005). Patients can be divided into two categories: long-term patients (whose treatment can last all their life) and short-term patients (who undergo a treatment over a limited period of time, generally under six months). The various pathologies generate waste in different ways, either linked to the injection of medicine, to the monitoring of a pathology (e.g. blood sugar level for diabetes), or to a specific home-treatment system (e.g. haemodialysis at home). Waste generation depends on several factors: the occurrence rate of the pathology, the share of patients that are diagnosed with the pathology, the frequency of the treatment, the existence of alternative treatment without injection, etc. The type of syringe used also has an impact on the quantities generated: for some, only the needle will be disposed of, while others such as pen needles will generate bigger quantities.

Diabetics represent the most important population of patients performing home treatment and the most important amount of HCRW (DASTRI, 2016). In France, it is estimated that diabetics represent about 50 % of the home-treatment patients generating sharps, and that they produce about 75 % of the total household HCRW (GIRUS, 2009). Focusing first on this population is a relevant way of targeting a significant part of the total HRCW produced at local level.

Overview of existing systems in various EU Member States

The following table summarises information on both pharmaceutical and sharp waste presented on the *Medsdisposal* website that gathers information for most EU countries.

Table 6-4. Existing collection systems for household sharps in various EU Member States

Country	Sharps
Ireland	Can be returned to local pharmacies but no legal obligation to accept them. Mobile collection systems set up by certain local authorities.
UK	Needle clipper or sharp bins can be obtained on prescription and returned to the doctor or collected by the local authority. The organisation depends on the local council which can charge for the service.
France	Take-back system, mostly in pharmacies, using sharps bins – EPR scheme (PRO: DASTRI).
Belgium	Used needles and sharps should be collected in a syringe container (for sale at community pharmacies) and should be disposed of in the municipal container park as biohazardous waste.
Netherlands	Most municipalities have local agreements on the collection of sharps and syringes by the pharmacies.
Luxembourg	Used needles and cannulas are collected in special security boxes. Narcotics should be returned to the Health Ministry's competent service.
Austria	Community pharmacies and communal recycling centres collect sharps (needles and syringes), which should be returned separately.
Denmark	Needles and sharps can also be returned to pharmacies (in approved sharps containers, which can be obtained at the pharmacy).
Sweden	Sharps (needles and syringes) are managed separately according to schemes set up by the municipalities, sometimes in collaboration with the pharmacies.
Finland	Both pharmacies and local collection points collect sharps (needles and syringes), which should be returned in a separate container.
Poland	Sharp medical waste (needles, syringes) is collected at outpatient clinics.
Hungary	Some pharmacies accept needles and syringes, but the waste regulation does not cover this (only medicines).
Slovenia	Sharps (used needles) used by individuals are classified and collected as waste metal in waste collection centres; it is recommended that they are delivered to the waste collection centre in closed plastic containers. Pharmacies are not allowed to collect sharps, medical technical accessories, chemicals, radiopharmaceutical products or blood and plasma products.

Croatia	Sharps are collected only by health centres and medical practices; in exceptional cases when consumers deposit sharps in unsorted municipal waste (which is in principle prohibited), they are recommended to deliver them in hard plastic containers.

Source: Medsdisposal campaign, 2015

The table shows that specific collection systems have been implemented in many EU countries. In general, the systems seem to be decided on by the local authorities managing municipal waste or depend on the willingness of local pharmacies to do so. In some cases, a national framework is set (e.g. the EPR system designed by the PRO DATRI in France which is then implemented by local authorities in a homogeneous way) or guidelines at national level (with no legal obligation) are available. Data are unavailable for several Member States, which means that either no information is available or no system is available for households.

Most examples presented above are based on the same principle, with some exceptions:

- **Pre-collection equipment:** providing home patients with specific pre-collection equipment: puncture-resistant boxes and possibly needle clippers. These boxes are either sold (e.g. in Belgium) or provided for free to patients (e.g. in France).
- **Bring system:** the patient has to take their boxes when full or when the treatment is over either to local pharmacies, container parks or to mobile collection systems.
- **Proper disposal:** waste is then managed by a collector and sent to a proper treatment unit for healthcare risk waste.

Designing the HCRW collection service

Setting a collection service for HCRW is essential to limit the risk of contamination for any person that might enter into contact with the waste. The most important element of the system is to isolate the waste to avoid any contact between used sharps, needles or syringes and the patients, the waste collectors, the waste treatment staff (e.g. in sorting centres) and any other person. Therefore, collection systems for HCRW have to rely on the following elements:

- **individual, puncture-resistant boxes**, bins or any container available to home-treatment patients to deposit their waste in after the treatment;
- **secured bring points** that prevent contact between people and the boxes or direct collections by authorised collectors;
- a **well-organised collection scheme** ensuring the proper transport of HCRW from bring points to dedicated treatment units (disinfection or incineration).

For all these different steps, several technical and organisational possibilities are available to local authorities, as detailed below.

Assessing the HCRW arising

The assessment of the HCRW arising from home-treatment patients is an important issue in order to assess the need for individual collection boxes, the number of collection points and the frequency of collection.

Such an assessment was performed before the implementation of the EPR scheme in France (Girus, 2009). The assessment relied on:

- an assessment of the number of patients performing home treatment, by pathology;
- an assessment of the number of sharps, needles and syringes produced by each patient and the frequency;

To assess the number of patients performing home treatment, the study relies on literature reviews providing prevalence rates for the various pathologies that can then be applied to local populations. These prevalence rates are based on various sources: feedback from Ministry of Health and Social Security data, as well as from surveys targeted to pharmacies. As an example, data on diabetes can be retrieved from various federations and presented by the International Diabetes Federation (IDF, 2014). Data provided by the IDF show raw prevalence rates ranging from 2 % to 12 % in the EU countries covered. While prevalence rates might not be available at the local level, it is possible to apply national ratios to the local population to obtain an estimate of the number of patients. It is important to note that prevalence rates can evolve over time, reflecting the fact that the diagnosis of diabetes improves as well as other factors, such as the aging of the population (IDF, 2014)

The waste to be collected and treated includes both the HCRW itself (i.e. the used needles, lancets or syringes) and the individual boxes where patients deposit the waste. To assess the total HCRW arising, it is possible to assess the average number of sharps and needles used by a patient for each type of pathology, depending on the treatment available. This assessment can be achieved by gathering information from pharmacies, general practitioners and local patient associations.

Delivery of boxes

Ensuring that home-treatment patients have access to individual boxes for depositing used sharps and needles is crucial for the safety of the general waste management route. Specifications for these boxes can be given in national legislation (e.g. the colour to be used). If not, the World Health Organisation provides recommendations for the containers to handle HCRW such as sharps: a yellow puncture-proof container labelled "sharps" (WHO, 2014).

Available volumes for individual boxes range from 0.5 litres to 4 litres depending on the HCRW production and the type of pathology. The most common type is a 1- or 2-litre plastic box. Several elements are also important to consider:

- a lid allowing the box to be closed after each use;
- a sealing system that prevents the opening of the box when full;
- a system (possibly integrated in the lid) that allows the needle to be detached from the main part of the injection device.

Several methods are used by local authorities across Europe to make the boxes available and deliver them to home-treatment patients, the most common ones being to make them available at local pharmacies for free (Utrecht, NL), using a prescription from a GP (NHS system, UK; DASTRI system, FR), or sold (Brussels Region, BE). In Utrecht, the box is given for free by the pharmacy when a patient buys sharps or needles, and a new one is given when a full one is brought back (Afvalverwijdering Utrecht, 2016).

In some local authorities' territories in the UK, where boxes are collected directly at home (e.g. in the London Borough of Newham), new boxes are provided when the filled ones are collected on request.

Collection methods

Several collection methods are available for HCRW from households:

- collection from the patient's home;
- collection at bring points;
- collection in residual waste.

Some local authorities provide the possibility to request a HCRW collection for sharps by filling in an online form or by calling a phone number. Some examples can be found in the UK: the Bristol Borough Council provides such a service for free to its diabetic citizens, boxes being provided in pharmacies with a prescription. For other sharps (excluding diabetes-related sharps), charges apply. Most public authorities in the UK provide this type of service for free (NHS, 2016). For some of them, forms must be filled in and signed by a GP.

Figure 6-5. Self-service container in a civic amenity site in Cambrai (France)

Bring systems using collection points are a very common system across Europe. Two main types of collection points are in use: local pharmacies and civic amenity sites (Medsdisposal, 2015). Other bring points can be used: healthcare centres, medical laboratories, and mobile collection points similar to mobile hazardous collection trucks. Collection can be achieved either by manually putting them in storage containers or in self-service containers. In any case, the anonymity of the waste producer must be ensured by setting a dedicated protocol.

Self-service containers are generally used in civic amenity sites or in any other location where citizens have to segregate their waste themselves. They have to be secured to prevent their opening by anyone but home-treated patients. Opening can be operated by codes, badges delivered to home-treated patients or by barcodes printed on the sharps bins. Containers must be easily accessible and information on their location must be made available to home-treated patients, either by providing a map online or by indicating the details in sorting guidelines. The picture presented above displays an automatic container available on the civic amenity site in Cambrai, France. Its opening is achieved by scanning the individual box.

Some public authorities also ask inhabitants to dispose of the sharps box in residual waste, as is the case in Bayern (Bayerisches Landesamt für Umwelt, 2014). This option is made available for all territories in Bayern exept two which send residual waste to mechanical biological treatment and not incineration. However, this type of collection has not been identified in other EU Member States; instead, a dedicated collection system is provided.

Organising collection rounds

The collection must be entrusted to an authorised collector complying with the national or regional regulations. The local authority is responsible for ensuring that the collector complies with the regulation regarding hazardous waste collection and transport.

If a bring system is used, the collection can either be at a regular frequency or on request. The HCRW has to be sent to an authorised treatment plant, either an incineration unit for HCRW or a disinfection unit followed by traditional incineration or landfilling, depending on the treatment units available.

Reporting is essential to monitor the effectiveness of the system. Data on collected quantities as well as documents regarding the traceability of the waste have to be provided by the company operating the HCRW collection, in order to assess the effectiveness of the system and the evolution of collected quantities.

In the Rivierenland region in the Netherlands, 30 pharmacies have reached an agreement to organise a monthly collection scheme for the household sharps they collect. The collection frequency depends on the number of collection points per patients as well as the storage capacities of these collection points. (*Frans, 2016*)

Stakeholder involvement

Actors involved in the collection of HCRW from citizens are in general the following:

- **Pharmacies** and **other healthcare actors** (such as the GP who prescribes the treatment and healthcare centres) that act as suppliers of sharps bins and collection points and that constitute key interlocutors for home-treatment patients. In several EU Member States, pharmacies are key collection points, and have two missions: distribution of empty boxes and bring points for full boxes.
- **Patients performing home treatment:** there are a wide variety of pathologies that can be treated at home, making the patients a heterogeneous group. As mentioned before, the largest proportion of this group is diabetic, so this is the first group to be targeted when implementing such a system. Diabetes associations are common in all EU Member States. A list, along with data on diabetes, is available on the International Diabetes Federation website (http://www.idf.org/membership/EUR).
- **Federation of pharmacies or of medical industries** might facilitate the organisation of the service (possibly through a PRO) or contribute to its organisation with public authorities. They can also provide current data on the HCRW put on the market which can provide an indication of the amount of HCRW to be collected.

When pharmacies are considered collection points, there are various ways to ensure their involvement: communication through the local/national federation, direct contact with them to present the system. This direct approach was successfully implemented by DASTRI staff for a specific territory with low performances to increase the number of collection points, and can be easily replicated at local level (DASTRI, 2016). The issue of cost coverage is also important when it comes to the involvement of pharmacies, even more so when pharmacies have no legal obligation to collect household HCRW.

Communication activities targeting these various stakeholders must ensure a clear message and detailed information on how the system is organised. The dissemination of information must focus on the following:

- **Patients**: they need practical information on the various elements: where and how boxes can be obtained, how to use the box properly and how it is collected (collection on request, location and opening hours of collection points, etc.). GPs are mentioned as one key communication channel in many local authorities, since they provide the prescription for the self-injection devices and in many cases for the individual boxes. When boxes are distributed in pharmacies, it is useful to provide them with additional communication material (brochures, guides, etc.) for patients.

- **External collection points**, such as local pharmacies, must be provided with clear information on the organisation of the system: how and to whom to deliver boxes, how to collect them, information about the collection dates and protocol (weighing, traceability documents, etc.).
- **Information in case of injuries**: this information might be useful to deliver to patients and collection points. DASTRI makes a poster available detailing the first aid to be carried out following an injury and the organisations to contact for a diagnosis (DASTRI, 2014).

To promote the selective collection to patients, it seems important to highlight the potential risks of injuries and contamination for the waste management workers to patients, as is mentioned below.

Control and corrective actions

One of the goals of the implementation of such a system is to limit the presence of HCRW in household waste, thus limiting the risk of infection due to injuries of the waste operators, mainly waste collectors and workers on sorting lines for co-mingled recyclables. Allocating a specific organisation to monitor the occurrence of this presence, as well as targeted corrective actions when the traceability of the waste can be established, is an effective way of addressing this issue.

Achieved environmental benefits

Providing a separate collection for household HCW mainly addresses issues related to safety. However, inappropriate management of home HCRW can have adverse effects on the environment:

- Environment: the improper disposal of healthcare risk waste (in the waste water system, in landfills, etc.) can lead to soil and water pollution.
- HCRW in sorting centres: the presence of HCRW in sorting centres can have negative consequences for the sorting performance: if spotted at an early stage, the whole waste lorry can be redirected to disposal to avoid any problem on sorting lines. If spotted on manual sorting lines or provoking an injury, it can entail the stop of the sorting centre and limit its sorting capacity.

Appropriate environmental indicators

The implementation of a household HCW collection scheme can be monitored thanks to various indicators.

Indicators for monitoring the resources in use:

- Number of collection points for HCW generated by households per 10 000 residents, by type (civic amenity sites, pharmacies, street containers).
- Number of individual boxes for HCW generated by households distributed via collection points or on request.

To monitor the results of the system, the following indicator can be monitored:

- Quantity of HCW generated by households collected (kg/capita/year).

To monitor the effectiveness of the system, it can also be important to monitor the following element:

- Share of HCW (e.g. sharps) in mixed household waste (%). Composition analyses can be performed; however, due to the small quantities, a large number of samples are required to obtain a statistically relevant assessment, so it might not be a practical solution.

Surveys of the various stakeholders might also provide relevant information on how to improve. For instance, DASTRI conducts such a survey on patients, pharmacies and GPs on a regular basis to monitor the habits of the various stakeholders, e.g. the number of GPs aware of the system, the number of GPs giving indications on HCRW handling to patients, or the specific behaviour of patients when away from home (DASTRI, 2016).

Cross-media effects

Considering the small amount of this waste arising, the impact of the separate collection of household HCW is limited. Collection and treatment of HCW from self-administered treatment does not present specific environmental impacts compared to traditional waste collection and incineration. The specific collection routes are likely to slightly increase the environmental impact but, since very low quantities are concerned, the impact is limited.

Operational data

Assessment of the HCW arising

An assessment of the HCW arising was conducted in the preliminary study carried out for the implementation of the French EPR scheme (GIRUS, 2009). The number of boxes will depend on their volume and the volume of the various types of HCRW. In the preparatory study, the following figures were presented:

- for short-term treatment (below 6 months): one 1-litre box;
- for long-term treatment (above 6 months): four 1-litre boxes per year.

To convert the production of conditioned HCRW into waste quantities, it is possible to use ratios concerning densities. The preparatory study provides an average of 0.3 kg/litre for conditioned HCRW: 0.18 kg/l for the sharps/needles and 0.12 kg/l for boxes. With these data, annual HCRW production is assessed to be 1.2 kg/year for long-term patients and 0.3 kg/year for short-term patients, including the individual boxes (GIRUS, 2009). The total amount of sharps arising in France is assessed to be 360 tonnes (1 135 tonnes when adding the containers that will be incinerated with the waste), which can be translated to 0.26 kg/patient/year without containers and about 0.80 kg/patient/year of waste treated.

Delivery of boxes

Source: **DASTRI**

Figure 6-6 shows the lid of a one-litre box used in the DASTRI system. The top part of the box allows it to be closed after use and there are systems that help with the detachment of needles inside the lid. The strip located on the lid can be used to seal the box when full or when the treatment is over, to prevent opening. Boxes generally have a level marked inside them, and they must not be filled above this mark. Boxes have to be stored in a secure place, preventing children from accessing them.

Source: DASTRI

Figure 6-6. HCRW box provided for household by DASTRI

In France, about 1.79 million individual boxes were distributed in 2015 (about 1 280 for every 1 000 patients) by the 22 500 local pharmacies. It means there is about one delivery point for boxes for every 60 patients.

Collection methods

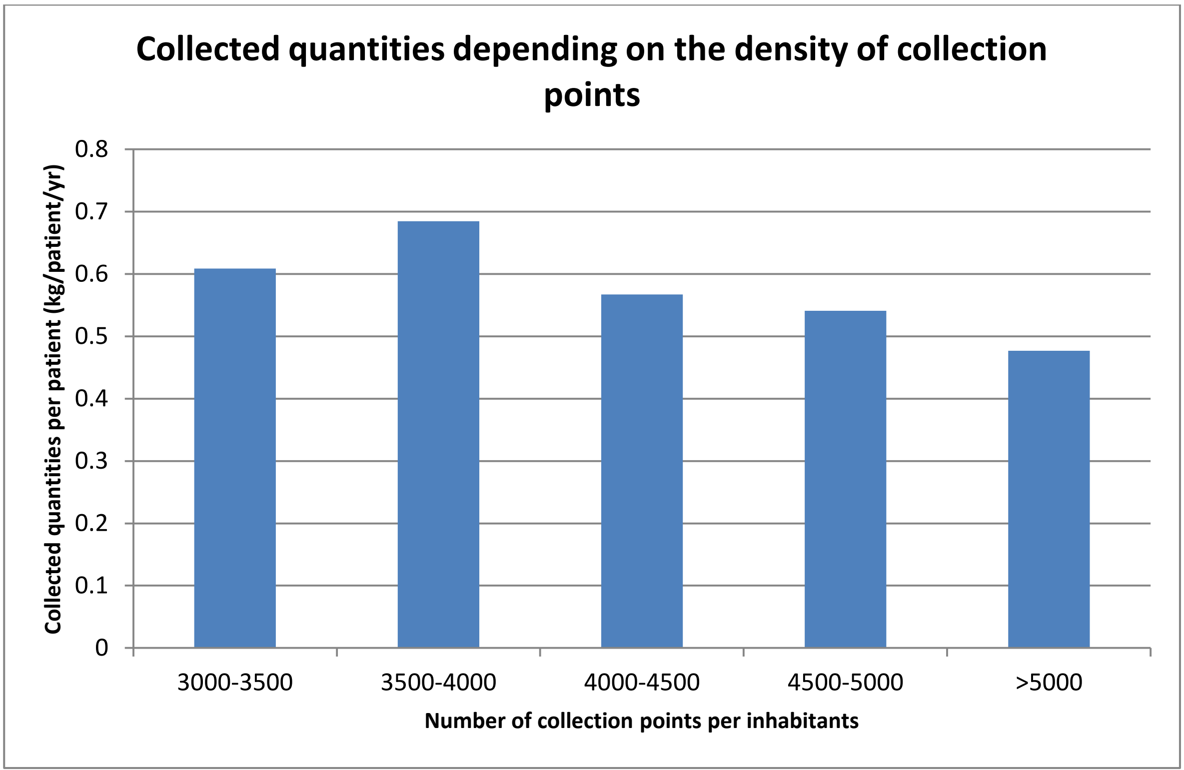
For collection in pharmacies or any other place where individual boxes are stored manually, simpler and smaller containers can be used. Within the DASTRI system, 50-litre cardboard and 50-litre plastic containers are made available to local pharmacies: the cardboard boxes are used for secured individual boxes while the puncture-resistant plastic boxes are used for any HCRW that is not properly packed (this format was used during the first months of the implementation). The containers also have a sealing system that has to be used when full. They have to be secured in a place not accessible to the public and comply with the requirements mentioned above.

The French PRO DASTRI had specific targets regarding the network of collection points stated in its bill of specifications: at least one collection point every 15 km and at least one point for 50 000 inhabitants (DASTRI, 2016). In other places (such as the Netherlands), all pharmacies selling sharps are expected to provide and accept boxes (Dante Pharmacy, 2016).

Regarding collection points, about 14 633 were reported in 2015, of which 13 400 are pharmacies and the rest mainly civic amenity sites. This amounts to slightly less than one collection point for every 100 patients. About 47 750 collections were organised in these collection points, collecting 772 tonnes of waste, which represents about 68 % of the assessed HCW arising. About 70 % of the collection points collected less than 15 kg per year while 28 % collected between 15 kg and 45 kg per year. However, there are important discrepancies among territories: while some regions achieved very important capture rates (some being above 100 % of the estimated HCW arising), some are underachieving. The main reason behind the differences is the density of collection points.

The DASTRI system foresees once every three months collection from pharmacies, and full containers are replaced with new ones. Pharmacies have the possibility to postpone the collection date if the collected quantities are too low (DASTRI, 2016).

The following figure shows the average collected quantities per patient depending on the density of collection points, based on regional figures.



Source: based on DASTRI, 2016

Figure 6-7. Collected quantities in French regions according to the density of collection points

The figure shows that there is a correlation between these two parameters. However, it is important to note that the density of collection points might not perfectly reflect their accessibility for the patients.

Control and corrective actions

The SYCTOM (waste agency for the Parisian metropolitan area), a public authority in charge of waste treatment for the central area of the Paris region, has developed a specific strategy to address the issue of HCRW found on sorting lines of the packaging waste sorting centres it is in charge of. This is a problem of increasing importance that can have an important impact on the safety of the workers as well as the sorting capacity of the sorting centres, with about 1 300 stops of the sorting lines and 30 injuries in 2013.

Since 2008, the private companies operating the sorting centres report incidents to the SYCTOM when HCRW is identified on the sorting line. The protocol includes the stop of the sorting line and the identification of any document with a postal address found around the identified waste (which will provide an indication of the collection route concerned). A monthly report summarising this information is sent to the SYCTOM for corrective actions.

These reports have led to several corrective actions. The municipalities concerned by irregularities are contacted and specific work done to identify potential sources of errors (SYCTOM, 2012).

Moreover, the SYCTOM has issued a communication kit for municipalities composed of short and long articles explaining the issue as well as pictograms to illustrate communication material. The articles highlight the potential hazardousness of HCRW for waste workers and provide information on the specific collection schemes (which can be adapted by the municipality if different from the traditional one). Municipalities that are members of the SYCTOM can download the kit by simply filling in an online form (SYCTOM, 2014).

Applicability

The BEMP is applicable to all local authorities and/or waste management companies since household HCRW originating from self-administered treatment is generated in any territory, although at various levels. The need of setting up a separate collection scheme allowing these fractions to be excluded from non-hazardous management systems is not limited to specific parts of Europe. While HCRW can be regarded as the responsibility of the patient, local authorities have to ensure the safety of the workers performing waste collection, sorting and treatment operations.

As for sharps, various systems are in use. The fact that householders have to put their waste in a specific box makes it important to involve either doctors that prescribe the medicine or pharmacies that sell it, to provide the boxes and/or to collect them when full. It is likely that a clear legal framework will contribute to the organisation and help clarify the financing of their collection.

However, the examples presented here can in general be applied at local level without a national framework; it appears that most existing systems across Europe were either implemented at local level by public authorities without a national framework or entrusted to local pharmacies.

Economics

The costs of the implementation of this BEMP include:

- cost for the equipment: sharps bins, containers;
- cost for the collection;
- cost for treatment in an authorised unit.

Many elements of the costs related to sharp waste are presented in the preliminary study carried out for the implementation of the French EPR scheme (GIRUS, 2009). The average costs reported by this study for the various elements of the collection system are presented in the table below (without VAT). They are based on various suppliers and studies.

Table 6-5. Costs for household HCRW collection (GIRUS, 2009)

Items	Average cost (EUR, without VAT)
Individual boxes	
0.5-litre box	0.80/unit
1-litre box	0.85/unit
2-litre box	1.05/unit
Collection points	
Plastic 50-l container	4.20/unit
Cardboard 50-l container	1.10/unit
Delivery fees	10/year per collection point
Automated collection bank:	
Annual running costs	30/year
Investment costs	2 600
Annual renting costs	3 000
Collection	

Collection costs	15 per 50-l container
Treatment	
Incineration	350/t
Disinfection	450/t

All these costs have to be considered as average and might vary depending on the potential mutualisation of equipment or the suppliers, or any other local specificity (taxes, local market, etc.).

In its annual report, DASTRI presents the overall costs of the system which include the organisation’s costs as well as the costs related to the running of the system and the communication activities. Of its annual budget of EUR 8.43 million, about 66 % is dedicated to the running of the system (supply of boxes and containers, collection and treatment of sharp waste, etc.) and 11 % to communication activities. Considering that about 1.4 million people perform home treatment, these costs represent about EUR 4/patient for the running costs of the system and EUR 0.66/patient for communication activities.

A crucial element is the financing of the system. In the case of collection by request, as organised by various local authorities in the UK (mentioned above), the costs are borne by the public authorities, and in rare cases the patient is asked to contribute. When collection is organised in pharmacies, there are various possibilities. An interesting case study is the Netherlands, where collection of sharps is traditionally performed by the pharmacies, yet the detailed organisation is decided at local level with public authorities (https://dante.medsenapotheek.nl/Default.asp?&HTTPSHASH). There are different organisations when it comes to the financing of the system:

- In the province of Utrecht, the local government covers all the costs for collection and treatment.
- In about half of the local authorities, public authorities pay for the disposal of the waste yet pharmacies have to arrange the transport of the waste to the treatment units.
- In the remaining 45 % of the local authorities, the pharmacies pay for the transport and disposal. This can amount to EUR 300/month for the pharmacy (*Frans, 2016*).

However, since there is no legal obligation for pharmacies to manage municipal HCRW, an increasing number of pharmacies refuse to accept waste (Brondijk, 2016). Another solution is being tested in the Rivierenland region, where 30 pharmacies have reached an agreement with the waste management company Avri for a mutualised monthly collection scheme, allowing the price to be cut to EUR 0.41 per individual container (Frans, 2016). This represents a collection cost of EUR 20 per 50-l container, which is close to the figure presented above.

Benefits are difficult to assess since the main objective is to divert hazardous waste from the non-hazardous waste flows. The benefit for the local authorities is related to the decrease in the disruption of sorting lines in sorting centres.

Driving force for implementation

The implementation of such a system has two main drivers: environmental protection and concern for the safety of waste operators. Pharmaceutical waste and infectious waste generated at home are produced in small quantities but they can have a very serious impact, e.g. infection of waste workers with HIV. Several factors can favour the implementation of the BEMP.

For local authorities, there are several driving forces:

- safety for the waste management system: mixing infectious waste with residual waste or other waste fractions (e.g. dry packaging) leads to risks of exposure for waste operators, e.g. waste collectors (especially in the case of collection in plastic bags, operators working on manual sorting lines in sorting centres, etc.);
- disruption of the waste management service: the presence of HCRW in non-hazardous waste fractions can provoke significant disruption, especially in sorting centres;
- the impact of unmanaged HCRW is also more important when residual waste is treated by MBT or direct landfilling, increasing the risk of contamination.

Therefore, European local authorities are regarded as key actors for the promotion and implementation of dedicated systems for household HCRW, considering that it affects the safety of the waste collection and treatment staff.

In general, strong regulation and a clear national framework making separate collection mandatory are relevant:

- a strong regulation making separate collection mandatory for this specific fraction;
- a strong commitment of the healthcare sector (pharmacies, GPs, medical companies putting products on the market) to design and fund the system and help with the dissemination;
- involvement of associations of patients performing self-administration (e.g. diabetes associations).

The importance of a national framework can be illustrated by the current discussions occurring in the Netherlands between the Federation of Pharmacies (KNMP), the Association of Dutch Municipalities (VNG), the Ministry of Health and the Ministry of Environment to share the responsibilities for household HCRW management (KNMP, 2016)

As seen with the French system, the implementation of an EPR scheme might be considered an interesting driving force that proposes a concrete framework for the operational and financial organisation of the collection with the contribution of companies putting sharps and needles on the market. This also allows the promotion of synergies for the technical implementation and the homogenisation of the collection scheme, by defining common containers for patients and by assigning one collection company per region.

Another important element is the communication with medical professionals performing medical treatment generating medical waste at home (nurses, GPs, etc.), who might use the service provided by the local authority for free. The capture rates of over 100 % achieved in several French regions tend to show that non-household waste is indeed collected with municipal waste. The city of Copenhagen offers a municipal service for private producers that might help reduce this situation by applying specific prices: about EUR 29 to subscribe to the service, then about EUR 26 per collection, to which the cost for treatment has to be added (City of Copenhagen, 2016).

Reference organisations

Reference organisations for the collection of sharp waste are as follows:

- DASTRI (France): PRO for the French EPR scheme on household sharp waste. It coordinates the financing and collection of household sharps and monitors the performance. Their position allows them to have a good overview of the various instruments in use for the collection of HCRW: collection modes, communication material. They have also considerable data on the quantities arising and collected.
- Afval Verwijdering Utrecht: local authorities in charge of waste management on behalf of the Utrecht municipalities. In its territory, household HCRW collection is done by local pharmacies while the costs are borne by the local authority.
- KNMP (Netherlands): the Dutch Association of Pharmacies is currently working on the issue of municipal HCW collection (not limited to sharps).
- SYCTOM (France): this public authority organising waste treatment for the municipalities in the Paris region has developed actions to reduce the presence of sharp waste on the manual sorting lines in the sorting centres treating its waste.
- AESGP: the Association of the European Self-Medication Industry: it created the Medsdisposal website to draw attention to HCRW collection from households, along with other European federations.
- Health Service Executive (Ireland): organisation providing all of Ireland's public health services. Sharps disposal containers are generally available at local health centres. HSE provides information material for household sharps disposal.

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6.5. BEMPs for the treatment of healthcare waste

6.5.1. Alternative treatments for healthcare waste

Summary overview
High-temperature incineration is the most common treatment method for healthcare waste (HCW) because of safety concerns; however, it has significant environmental impacts such as high energy use, natural resources depletion and emissions. There are alternative treatments that can also guarantee safety levels for waste streams of concern (e.g. infectious waste, anatomical waste, sharps and pharmaceutical waste) and they can achieve a better environmental performance than high-temperature incineration, e.g. thanks to reduced energy use or better resource efficiency (increasing the rate of recycling from HCW). When using alternative treatments for HCW, it is BEMP to meet the following criteria: - Autoclaving: • optimal segregation at source; • homogeneous particle size at the inlet; • steam-based sterilisation with simultaneous/post-shredding; • drying step after treatment; • output separated per material stream when possible and sent for recycling; • waste-to-energy applied to the output when incineration is admissible. - Microwaving: • optimal segregation at source; • water addition at the inlet; • drying step after treatment; • output separated per material stream when possible and sent for recycling; • waste-to-energy applied to the output when incineration is admissible. - Chemical treatments: • optimal segregation at source; • output not considered hazardous waste or treated for optimum recovery; • sterilisation agent is recyclable within the process; • output separated per material stream when possible and sent for recycling; • waste-to-energy applied to the output when incineration is admissible.

Waste management area							
Cross-cutting	MSW - strategy	MSW - prevention	MSW - collection	MSW - EPR	MSW - treatment	CDW	**HCW**

Applicability
High-temperature incineration is still the most common treatment for HCW. Four main factors affect the applicability of alternative treatments: source segregation, proving the safety of alternative treatments in treating certain fractions of segregated waste, the optimum operating capacity for incineration and the national legal framework for HCW treatment.

Specific environmental performance indicators
- Share of HCW managed by the HCW management company processed by alternative treatments (%). - Amount of HCW processed by alternative treatments (kg HCW per hour, day or cycle). - Water consumption per kg of waste processed by alternative treatments (litres/kg)

Description

The Stockholm Convention on Persistent Organic Pollutants in 2004 and the World Health Organisation Policy Paper on Safe Health-Care Waste Management made many countries around the world prioritise, with varying levels of success, the implementation of technologies that prevent the release and formation of dioxins and furans, scaling up the so-called alternative treatments for healthcare waste consisting of non-incineration technologies (HCWH, 2007). As seen in Section 6.3.2, there are different treatments suitable for HCW; however, the lack of proper segregation considerably increases the fraction of HCW that needs to be treated.

An important part of the environmental impact of incineration can be avoided with the use of alternative treatment methods that remove, for example, the infectious character and therefore allow the HCW to be treated as MSW-like waste streams. However, the use of alternative HCW treatments, described in the previous section, should comply with certain requirements in order to be considered suitable. For instance, in the UK, all treatment activities have to render safe all treated HCW (DH, 2013; Tudor et al., 2009) under the following criteria:

- For infectious waste: the alternative treatment should have demonstrated the ability to reduce the number of infectious organisms in order to reduce the risk of infection. The minimum level required is a Level III STAATT inactivation (see Operational data for more information), which is a common reference level all over Europe.
- For anatomical waste: it should be destroyed in such a way that it is no longer generally recognisable.
- For other HCW: it destroys the shape and form of syringes, needles and other sharps, so they become *unusable* and *unrecognisable*.
- For pharmaceutical waste: it destroys the component chemicals to a non-hazardous, non-polluting form.

Therefore, alternative techniques may constitute best environmental management practice if these criteria are met and are able to show a better environmental performance than high-temperature incineration, e.g. by avoiding the emission of certain pollutants, having a better life-cycle environmental performance and/or increasing the rate of recycling from HCW. In Germany, a similar approach is used, although the provisions vary slightly. The Robert Koch Institute (RKI) – a leading institution of the government for the safeguarding of public health in Germany – indicates the processes that are considered acceptable and under which conditions, e.g. shredding is not allowed unless disinfection occurs at the same time (LAGA, 2009). A more systematic approach for when to consider alternative treatments

BEMPs is shown in Table 6-6. Admission criteria (Table 6-6) are all the requirements for a waste stream to be treated under each treatment. Minimum environmental criteria are those to be considered by the waste treatment service when comparing the performance of alternative treatments to high-temperature incineration. Best practice criteria are those oriented not only to best operational results, but also how waste is supplied (i.e. its segregation at source) and how the residue after the treatment is managed (e.g. waste-to-energy).

Table 6-6. Admission criteria, minimum environmental criteria and best practice criteria for alternative treatments for HCW

	Autoclaving	**Microwave**	**Chemical treatment**
Admission criteria	Treatment to render waste safe.. Non-bulky wastes; or bulky wastes suitable for shredding operations, if applicable. Not applicable to mercury- or heavy-metal-containing wastes. Not applicable to medicine-contaminated, cytotoxic and cytostatic waste, infectious or not.		
Minimum environmental criteria	Segregation at source meeting minimum standards. Exhaust air decontamination unit. Waste water treatment. Output safely disposed of and incinerated if PVC content is negligible; otherwise, to be deposited in safety landfill.		Segregation at source meeting minimum standards. Post-treatment of liquid waste. Waste water treatment. Output safely disposed of as hazardous waste if applicable
Best practice criteria	Optimal segregation at source. Homogeneous particle size at the inlet. Steam-based sterilisation with simultaneous/post-shredding. Drying step after treatment. Output separated per material stream when possible and sent for recycling. Incineration with energy recovery of the suitable non-recyclable outputs.	Optimal segregation at source. Water addition at the inlet. Drying step after treatment. Output separated per material stream when possible and sent for recycling. Incineration with energy recovery of the suitable non-recyclable outputs.	Optimal segregation at source. Output not considered hazardous waste or treated for optimum recovery. Sterilisation agent is recyclable within the process. Output separated per material stream when possible and sent for recycling. Incineration with energy recovery of the suitable non-recyclable outputs.

Achieved environmental benefits

Townend and Cheeseman (2005) reported some of the environmental impacts from alternative treatment of HCW in comparison with incineration practices (Table 6-7).

Table 6-7. Some environmental characteristics of HCW incineration and alternative treatments

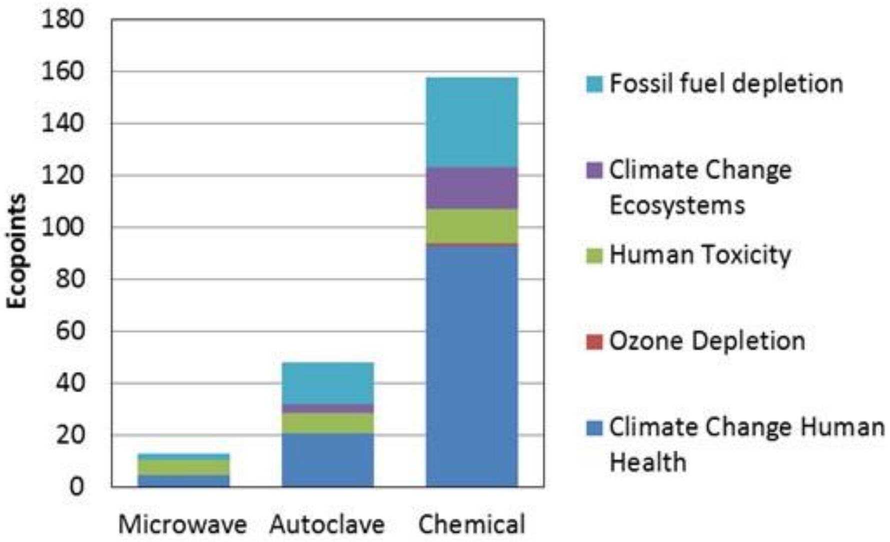
Characteristic	**Autoclave and steam-based**	**Microwave radiation**	**Chemical disinfection**	**Incineration**
Waste volume and weight	Do not reduce weight, but increase in the case of the addition of water/chemical/additives. Volume can only be reduced with shredding operations.			Reduces volume and weight by more than 90 %.

Impacts on the environment	Toxic volatile organic compounds and odours (requires abatement system). Generates waste water.	Toxic volatile organic compounds and odours. Generates waste water.	May generate toxic volatile organic compounds and odours. Generates liquid hazardous waste and/or waste water.	High volume of air emissions that require an appropriate pollution abatement system. High risks of dioxin and mercury emissions.

Source: Townend and Cheeseman (2005)

Source: **Soares et al., 2013**

Figure 6-8 shows the aggregated value associated to the environmental impact of different alternative treatment techniques per tonne of HCW for installations achieving the same level of disinfection, treating 250 kg/h of waste. The aggregated value, in Ecopoints, uses the ReCiPe method. According to this, the technique with the lowest environmental impact associated is microwaving, followed by autoclaving. The environmental impact of both techniques is associated with the production and use of energy (so greenhouse gas emissions and fossil fuel depletion are the main impact categories considered), the consumption for microwaving being much less than for pressurised steam, as anticipated by the authors of the study (Soares et al., 2013). In the case of chemical disinfection, the assumption of alkaline hydrolysis with lime makes the main life-cycle environmental hotspot the production of the chemical agent.



Source: Soares et al., 2013

Figure 6-8. Environmental impacts of different alternative HCW treatment techniques

Zhao et al. (2009) published a LCA comparison of incineration and alternative treatments for HCW (Figure 6-9). Results were favourable to incineration for GHG emissions, calculated using CML1999, and other energy-related categories when waste heat is used and electricity is cogenerated. Autoclaving has a higher eutrophication impact (not shown) due to the production of a leachate from the sanitary landfill. Currently, a best practice approach would include waste sorting, to recover recyclable materials for compatible waste streams, and waste incineration of SRF in a larger municipal solid waste incineration (MSWI) plant that operates at lower temperature but with an efficient waste heat and electricity cogeneration. If, for instance, the results from Figure 6-9 are adjusted for the energy balance of MSWI, the final GHG emissions for autoclaving would be reduced by at least 40 kg of CO_2 and potentially much more from the balance of recyclables. Costs are affected considerably, with reductions of up to 60 %.

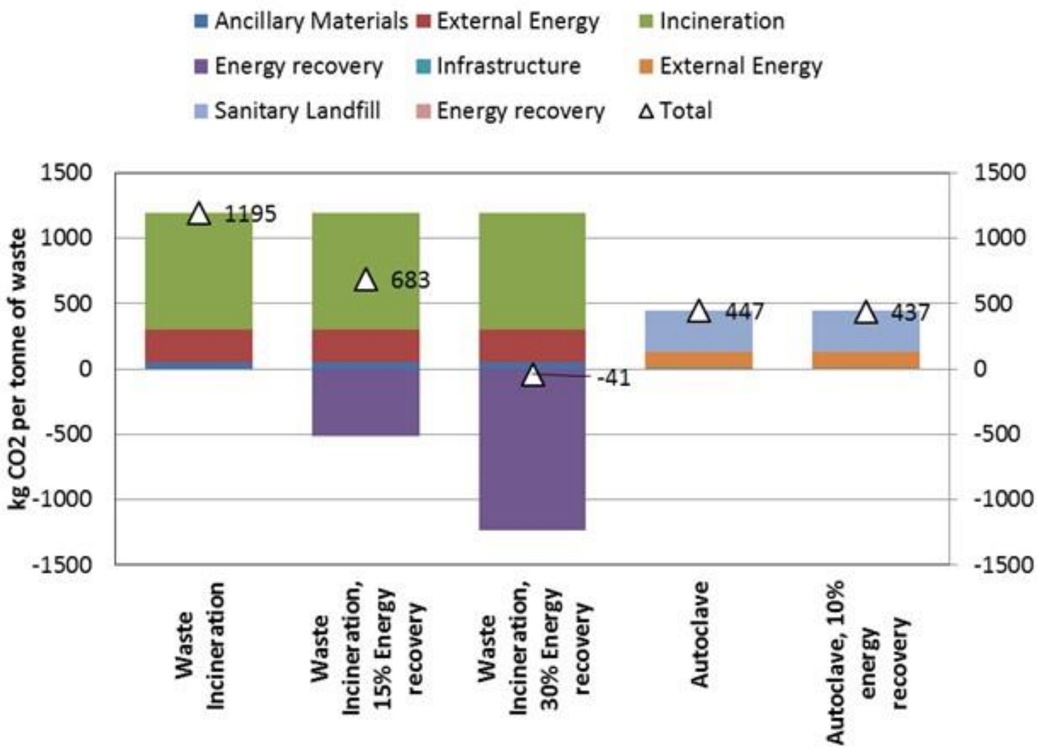


Figure 6-9. GHG emissions from several alternatives of HCW high-temperature incineration and autoclaving

From the discussion above, it can be deduced that the application of alternative treatments cannot be considered a BEMP per se if they are not accompanied by some so-called enabling techniques: waste has to be segregated and diverted from landfill, mercury-containing waste is segregated according to regulations and certain downstream processes take place (shredding, sorting, recycling, incineration, etc.).

Appropriate environmental indicators

Several indicators can be useful to monitor HCW flows:

- Share of HCW managed by the HCW management company processed by alternative treatments (%).

For each of the alternative treatments, there are a number of useful technical parameters that help to understand the performance of the technique:

- Amount of HCW processed by alternative treatments (kg HCW per hour, day or cycle);
- Water consumption per kg of waste processed by alternative treatments (l/kg).

Cross-media effects

As shown in the Achieved environmental benefits section, the trade-offs of environmental impacts can be complex. While some techniques are oriented to avoid the burning of recyclable materials, they produce waste water effluents that, in many cases, need to be treated or, as in the case of chemical disinfection, are considered hazardous waste.

In addition to the need for safe treatment of the waste, it is necessary to note that separate waste collection in healthcare is far from optimal (e.g. chemicals and pharmaceuticals are still not well segregated), and the waste should not be treated in autoclaves or microwave systems and then incinerated in MSW incinerators. This is the case of HCW with a high PVC content, which may require a safer treatment than incineration (Simon, 2015).

Operational data

The decision-making for waste treatment and management falls to the waste contractor and the service provider, and to the waste producer, as the legal liability of rendering safe a waste stream lies with the HC organisation, or the HC organisation may prefer the safest route for proper treatment (i.e. high-temperature incineration) and avoid the option of alternative treatments. Nevertheless, at the same time, application of alternative treatments is usually cheaper for the waste producer and more profitable for the contractor. The service provider, then, may use its influence for the best achievable performance by the following measures:

1. Sourcing better segregated waste. Segregation at origin is key for the whole treatment to achieve an optimal performance, with or without the application of alternative treatments. An optimal performance can be achieved through the availability of resources for segregation (e.g. differentiated containers for the categories shown in Table 6-2), pre-acceptance audits and through awareness campaigns and regular and up-to-date training. Some case studies are described in Table 6-8, which were applied to hospitals, usually with the collaboration of a waste contractor or consultant.

Table 6-8. Achieved environmental benefits reported for several case studies

Case Study	Description	Quantifiable benefit	Reference
University College London (UK)	Implementation of a segregation scheme for HCW from research and teaching activities (an important amount of the clinical waste is non-hazardous offensive waste, which can be sorted and not incinerated) Implementation of MSW-like waste recycling scheme Reduction of waste collection journeys by half Stakeholder engagement programme	18 % diversion from high-temperature incineration Maximum savings of 1 kWh per month from new incineration routes for offensive waste 28 tonnes of CO_2 saved per month in waste transport	Monk (2011) Stratton (2011)
Opole Hospital (PL)	Improved staff training to avoid inefficiency	Infectious waste reduction of 50 %; 14.7 tonnes of waste sent for recycling (approx. 29.4 kg/bed yr)	HCWH (2007)
Freiburg Hospital (DE)	Phase-out and reuse programmes for paper towels, dishes, baby bottles, shoe protectors	Reduction of 577 tonnes of waste per year	HCWH (2007)
Gloucestershire Hospitals NHS Foundation Trust, UK	Implementation of separation of offensive waste stream in hospitals to divert other waste from high-temperature incineration Implementation of a top-down training programme, centralised by the trust	Not quantified yet	DH (2013)

In some countries, *pre-acceptance audits* are required of waste contractors for a HCW treatment permit, in order to check for compliance with the minimum standards of segregation, composition and amount of waste streams and, in the case of the HC organisation, to monitor compliance with minimisation and prevention policies, and to show compliance with regulations. These audits, if not mandatory, can be considered best practice. In general, four types of audits can be performed by the contractor: observation of practices, observation of waste facilities, staff questionnaire, and detailed examination of waste. The benefits of such a practice will produce recommendations for improvement, easily identifying free or low-cost opportunities, and can recover costs through the implementation of improved practices at source (DH, 2013).

Training programmes and awareness campaigns are extremely important for better sourced waste. In general, staff handling HCW should receive appropriate instruction and training on all relevant aspects of health and safety, and it is the responsibility of the HC institution to provide it. Waste contractors may be involved in some of the following training issues:

- waste management arrangements such as appropriate classification and segregation of the waste;
- the standard operation procedures (SOPs) for its safe storage, carriage, treatment and disposal, including spillages, leakages, etc.

Delivery of training depends on the target group; while training in general waste policy is required for every staff member, the waste contractor is responsible for the technical instructions relevant to each of the target groups and developing or delivering draft SOPs to hospitals and other HC institutions.

Information posters, signs and other communication material are also supplied by the waste contractor. As an example of best practice, Source: http://www.bio-bin.org/assets/img/YELLOW-POSTER.png

Figure 6-10 shows the poster supplied by Econix Ltd for their disposable waste bins in the UK, which is freely downloadable from the internet.

Source: **http://www.bio-bin.org/assets/img/YELLOW-POSTER.png**

Figure 6-10. Example of information poster supplied by waste contractor

2. Better understanding of logistics issues for HCW. Four approaches can be identified in the use of alternative treatments and incinerators by contractors and/or health organisations (HCWH, 2007):

- Centralised treatment. This approach takes advantage of the economy of scale by the use of large-scale treatment, fed by the waste from several locations. Although costs are relatively lower, it requires a large infrastructure and collection system in specialised vehicles, increasing the risk derived from infectious waste handling.
- Decentralised treatment. Every single waste generator has an on-site treatment unit. This avoids any risk derived from waste transport but its cost is higher, as the marginal cost is increased in small units. In addition, the required training on the use of the unit is extended to a large fraction of the hospital staff. However, it may present advantages in terms of cost and feasibility for rural areas.
- Mobile treatment systems. The treatment unit is mounted on trucks and travels to each generation site. The total cost of treatment is the highest of the four options.
- Treatment within clusters. A major hospital has a scaled-up facility for the treatment of waste generated on site plus that generated in the area or district.

3. Selecting vendors and technologies. Some examples of vendors per technology are shown in Table 6-9.

Table 6-9. Examples of vendors, per technology

Technology	Vendor
Autoclave	Tuttnauer
Shredding, Steam, Mixing, Drying	Ecodas
Steam, Mixing, Shredding, Drying	Hydroclave Systems Corp
Shredding, Steam, Mixing, Drying, Chemical	Steriflash
Vacuum, Steam, Drying, Shredding	Sterival, Starifant Vetriebs GmbH
Shredding, Steam, Drying, Chemical	STI Chem-Clav, Waste Reduction Europe Ltd
Shredding, Steam, Mixing, Compaction	STS, Erdwich Zerkleinerungssysteme GmbH
Vacuum steam, Drying, Shredding	System Drauschkle, GOK Consulting AG
Vacuum, Vacuum steam, Drying	WEBECO GmbH
Steam-fragmenting, Drying	ZDA-M3, Maschinenvertrieb fuer Umwelttechnik GmbH
Microwave treatment	Ecosteryl, AMB; Medister, Meteka; Sanitec, Sintion
Fragmenting-Steam-Chemical	Newster, Multiservice Frist SRL
Alkaline hydrolysis	WR2, Waste Reduction Europe Ltd

Source: HCWH, 2007

Of course, the alternative treatments market is relatively innovative and moves fast to achieve tailored solutions. As an example, microwave sterilisation, which is usually a discontinuous or semi-continuous operation, can however be redesigned to offer a continuous process that includes pre-shredding, a continuous screw-driven feed with no water addition, and storage. The final product is shown in Figure 6-11. The redesigned system achieves costs reductions of 45 % for service providers in some Belgian hospitals (AMB, 2015) and is frequently applied within large hospital facilities to reduce transportation costs, although they are mainly operated by service providers.

Figure 6-11. Shredded and unrecognisable microwaved healthcare waste

<u>4. Understanding and reporting on the applied technology.</u> As stated in the Appropriate environmental indicators section, technical information on the technology is key to understanding its economic and environmental potential. Table 6-10 below shows the performance of alternative treatments.

Table 6-10. Technical parameters and indicators of alternative treatments

Parameter/Indicator	Autoclave	Microwave	Chemical disinfection
Temperature-time	Depends on the waste. Min. 121 °C, 30 min. May be less if pre-shredded and agitated	Depends on the waste and water content. Min. 121 °C, 30 min	> 100 °C Several hours
Batch/Continuous/Semi-continuous operation	Batch, semi-continuous, advanced treatments can operate continuously	Batch, semi-continuous	Batch
Throughput (kg waste per hour, day or cycle)	Max. 1.5 tonne per hour	Max. 0.4 tonne per hour	n.a.
Water treatment and recovery (y/n)	Yes	No	No
Suitable for bulky materials (y/n)	Requires pre-shredding	Requires pre-shredding	No
Pre/Post-shredding	Pre-shredding not recommended by WHO	Pre-shredding not recommended by WHO	Pre-shredding not recommended by WHO
Production of hazardous waste/effluent	No (if downstream drying)	No (if downstream drying)	Yes
Level of disinfection	III or higher	III or higher	III or higher
Volume variation	Reduction after shredding and drying	No, only after shredding	No, only after shredding
Weight variation	Increase > 5 %	Increase > 1 % if water is added	n.a.

Source: WHO (2014), Townend and Cheeseman (2005)

Applicability

Although alternative treatments should be encouraged and maximum diversion from incineration should be achieved, high-temperature incineration will always be necessary for the treatment of a significant fraction of HCW (Tudor et al., 2009). It is therefore required for contractors to maintain a certain throughput of their incinerators, which are usually much smaller than MSW incinerators and quite scattered around Europe. The need for waste bulking would restrict the amounts that can be actually diversified to alternative treatments, especially in a sector, healthcare, where waste amounts cannot be accurately predicted. Tudor et al. (2009) identified three main factors affecting the applicability of alternative treatments: source segregation, proving the efficacy of alternative treatments for certain fractions of segregated waste and the optimum operating capacity for incineration.

Of course, the applicability by waste managers, as service providers, is also limited by the decision-making processes of waste producers, which may avoid alternative treatments due to health and safety risks.

Economics

One of the main drivers for the implementation of alternative treatments is cost, as high-temperature incineration is reported to be very expensive due to the use of support fuels and pollution abatement, while alternative treatments have reported up to 60 % savings in optimal scenarios. In 1990, the US already reported a cost two to five times higher for incineration than for alternative treatments (USCOTA, 1990). In actualised terms, using the price index for industrial commodities, those costs would correspond to a maximum of USD 1.90 per kg of waste incinerated, and USD 0.40 per kg of waste sent to alternative treatment (post-treatment not included). More recently, Tudor et al. (2009) reported a cost of GBP 500–800 per tonne for incineration, which corresponds to a maximum of USD 1.30 per kg of waste. In this regard, there was a shift from local, small incinerators installed in hospitals to centralised and/or treatment clusters in order to have a) installations at higher scale working at less marginal costs, and b) incinerators with appropriate exhaust treatment systems. In 2013, calculations by Soares et al. reflected a cost of USD 0.12 per kg for microwave treatments, USD 1.10 per kg for autoclaves, and USD 1.53 per kg for alkaline hydrolysis (these figures include full waste treatment). While the use of alternative treatment reduces associated costs in the whole treatment chain, higher savings are more likely to be achieved by the service providers. However, the scale factor is considered to be extremely important in the cost-benefit analysis of alternative treatments for HCW.

Driving force for implementation

Risk minimisation is the primary objective of any HCW management strategy. Therefore, the diversion from incineration to alternative treatment should consider health risks and safety as the primary priority. Under certain circumstances, alternative treatments are also shown to be driven by a better environmental and economic performance.

Reference organisations

World Health Organisation (United Nations public health arm, who.int)

Directorate-General for Health and Food Safety, European Commission, http://ec.europa.eu/health/index_en.htm

Health Care Without Harm, HCWH, noharm.org. A comprehensive list of technology vendors can be found in the publication from HCWH (2007).

US Environmental Protection Agency

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7. Conclusions

The conclusions from this report are summarised in the following two tables. The first (Table 7-1) presents the common environmental performance indicators identified for municipal solid waste, together with the corresponding benchmarks of excellence. The second (Table 7-2) lists all the best environmental management practices presented in the previous chapters, the indicators, the BEMP-specific benchmarks of excellence and some aspects of their applicability.

Table 7-1. Common environmental performance indicators for municipal solid waste presented in Chapter 2 of this document with the related benchmarks of excellence

Common environmental performance indicator	Brief explanation	Benchmarks of excellence
Indicators for the overall municipal solid waste management system		
1) MSW generation (Section 2.4.1)	The indicator describes the amount of total MSW generated within the territory administered by a local waste authority per year, in relation to the resident population. Waste monitoring is key in order to regularly record waste quantities for each different waste stream collected separately by all the different collection systems available in the territory (e.g. door-to-door, civic amenity sites, street bins). This indicator is useful for assessing overall waste generation trends as well as the results of any effort to promote waste prevention.	The annual generation of MSW in the territory administered or managed (collected by all different waste collection systems available in the area) is: - lower than 75 % of the national average of municipal waste generation[78], using the national definition of municipal waste of their own country; **or** - lower than 360 kg/capita, if calculated only for the following waste fractions[79]: (i) organic/biowaste (e.g. green cuttings, food, kitchen waste), (ii) co-mingled packaging, (iii) paper and cardboard, (iv) glass, (v) plastics, (vi) metals, (vii) bulky, (viii) WEEE and (ix) mixed waste.

[78] As reported by national authorities or by the statistical office of the European Union (Eurostat).
[79] The following fractions have been selected because they are commonly monitored in the EU by local waste authorities and waste management companies and they are generally the most relevant fractions (by weight) in MSW.

Common environmental performance indicator	Brief explanation	Benchmarks of excellence
2) Amount of mixed MSW collected (Section 2.4.1)	The indicator describes the amount of mixed MSW collected per capita per year. Its calculation takes into account the waste collected as non-source separated mixed waste. Mixed MSW contains all waste fractions for which no separate container or other collection system is available. In systems where most of the waste is segregated at source and collected separately, this is often referred to as "residual waste". The calculation of the indicator amount of mixed MSW collected can be integrated by adding the amount of separately collected fractions that cannot be recycled (i.e. rejects from sorting/recycling plants), provided that the local waste authority (or the waste management company) is aware of these quantities. The amounts of rejects from sorting/recycling can be based on actual data (from sorting/recycling plants) or reliable estimations based on the amount of misthrows found in the separately collected fractions. Similarly, in the event that mixed waste is pretreated (e.g. in an MBT plant) and the local waste authority (or the waste management company) is aware of the dry recyclables that are sorted out from mixed waste and sent for recycling, the quantity of dry recyclables can be subtracted from the amount of mixed MSW collected.	N/A

Common environmental performance indicator	Brief explanation	Benchmarks of excellence
3) MSW sent to energy recovery and/or disposal (Section 2.4.1)	The indicator measures the annual amount of MSW that is treated by either incineration with energy recovery and/or disposal operations, such as landfilling or incineration without energy recovery. If this information is not available as such (e.g. in the case of waste authorities or waste management companies not managing the whole process), it can be calculated as follows. The fate of the mixed waste collected is taken into account: if mixed waste is directly sent to energy recovery and/or disposal, the quantity can be directly used for the calculation. In the event that the mixed waste is pretreated (e.g. in an MBT plant), the local waste authority (or waste management company) includes in the calculation of the indicator the actual quantities of waste that, after the pretreatment, are sent to energy recovery and/or disposal. Similarly, it is important that the local waste authority (or the waste management company) also takes into account in the calculation of the indicator the amount of rejects from the sorting/recycling of the separately collected fractions that are not recycled but sent to energy recovery and/or disposal. The amounts of rejects from sorting/recycling can be based on actual data (from sorting/recycling plants) or estimations based on the amount of misthrows found in the separately collected fractions. In the event that the local waste authority (or waste management company) cannot fully calculate the indicator, considering all its factors, it can report only the amount of mixed waste sent to energy recovery and/or disposal, acknowledging that the indicator is partially calculated. In such cases, it is important to clearly state the elements that are not included in the calculation (e.g. rejects from separately collected fractions sent to energy recovery and disposal). Moreover, appropriate measures to obtain reliable data for the full calculation of the indicator can be put in place to improve the usefulness of this indicator.	The annual amount of collected mixed MSW sent to energy recovery and/or disposal is: - lower than 15 %[80] of the national average of municipal waste generation[81]; **<u>or</u>** - lower than 70 kg/capita.

[80] Please note that the formulation 'the annual amount of collected mixed MSW sent to energy recovery and/or disposal is lower than 15%...' does not necessarily mean that 85% of municipal waste is separately collected for reuse and recycling. For municipalities that generate less municipal waste than the national average in their country, 15% of the national average would correspond to a higher share (e.g. 20-30%) of their own municipal waste generation.
[81] As reported by National Authorities or by the statistical office of the European Union (Eurostat)

Common environmental performance indicator	Brief explanation	Benchmarks of excellence
4) MSW sent to disposal (Section 2.4.1)	The indicator measures the annual amount of MSW that is sent to disposal, such as landfill or incineration, without energy recovery (all disposal operations are defined in Annex I to the WFD, see Annex 8.1). If this information is not available as such (e.g. in the case of waste authorities or waste management companies not managing the whole process), it can be calculated as follows. Firstly, the fate of the MSW collected as mixed waste is taken into account for the calculation: if mixed waste is sent directly to incineration without energy recovery the quantity can be directly used for the calculation. If mixed waste instead undergoes pretreatment (e.g. in an MBT plant), the quantities actually sent to disposal after treatment are needed. Finally, for the calculation of this indicator, it is important to include also the amount of rejects from sorting/recycling of separately collected fractions that are sent to disposal, if known by the local waste authority/waste management company.	The annual amount of MSW sent to disposal is: - lower than 2 % of the national average of municipal waste generation[82]; **or** - lower than 10 kg/capita.
Waste-stream-specific indicators		

[82] As reported by National Authorities or by the statistical office of the European Union (Eurostat)

Common environmental performance indicator	Brief explanation	Benchmarks of excellence
5) Capture rate of a specific waste stream (Section 2.4.2)	The capture rate is the percentage of the estimated generation of a specific waste fraction that is collected separately. It provides insights into the efficiency (i.e. how efficient in intercepting the recyclables) of a separate collection system. The precondition for the calculation of this indicator is that a composition analysis of the mixed waste has been performed. In addition, the amounts collected by each collection system for each material can be compared to the total amount of the same material generated within the territory administered by a local authority. The capture rate can be calculated for the separately collected fractions, e.g.: -plastic; -metal; -paper and cardboard; -glass; -co-mingled packaging; -biowaste.	- The capture rate for waste glass separately collected as a single fraction (i.e. not in a co-mingled collection system) is higher than 90 %. - The capture rate for waste paper and cardboard separately collected as a single fraction (i.e. not in a co-mingled collection system) is higher than 85 %. - The capture rate for waste metals separately collected as a single fraction (i.e. not in a co-mingled collection system) is higher than 75 %. - The capture rate for co-mingled waste packaging is higher than 65 %.

<table>
<tr><th>Common environmental performance indicator</th><th>Brief explanation</th><th>Benchmarks of excellence</th></tr>
<tr><td>6) Impurity rate of a specific waste stream (Section 2.4.2)</td><td>The impurity rate of a specific waste stream refers to the amount of non-target materials in the separately collected waste stream. This indicator is closely linked to the previous indicator (capture rate) as it monitors the effectiveness (i.e. how effective in selecting the recyclables at home the residents are) of a separate collection. It provides information about the amount of misthrows and materials contained in the separately collected recyclables that cannot be recycled.

The impurity rate can be calculated for the separately collected fractions, e.g.:
-plastic;
-metal;
-paper and cardboard;
-glass;
-co-mingled packaging;
-biowaste.

Two indicators may be calculated for biowaste if kitchen waste and garden waste are collected separately:
a) impurity rate in separately collected kitchen waste;
b) impurity rate in separately collected garden waste.</td><td>N/A</td></tr>
<tr><td>7) Biowaste in mixed waste (Section 2.4.2)</td><td>The indicator describes the annual amount of biowaste included in mixed waste, which is identified by a composition analysis.</td><td>The annual amount of biowaste in mixed waste is lower than 10 kg/capita.</td></tr>
<tr><th colspan="3">Additional waste stream specific indicators</th></tr>
<tr><td>8) Collection scheme for glass bottles (Section 2.5)</td><td>The indicator on the presence of a deposit refund scheme (DRS) for glass bottles (y/n) is needed to complement the capture rate and the impurity rate for glass waste, because of the very significant influence of such a deposit refund scheme on the results obtained with the capture and impurity rate indicators.</td><td>N/A</td></tr>
</table>

Common environmental performance indicator	Brief explanation	Benchmarks of excellence
9) Amount of used and waste textiles collected separately	The indicator reflects the annual amount of used and waste textiles collected separately through the collection scheme established by the local waste authority. This includes both used textiles sent for reuse and waste textiles sent to either preparation for reuse or recycling.	N/A
10) Textiles in mixed waste	The share of textiles found in mixed waste can be used to monitor the correct source separation by households of waste textiles and the efficiency of the used and waste textlies collection system. This metric allows the assessment of the quantity of textiles that are not correctly source separated and are thus disposed of in the mixed waste.	N/A
11) Capture rate for textiles	The capture rate is the share of the estimated generation of a specific waste fraction that is collected separately. It provides insights into the efficiency of a separate collection system. The precondition for the calculation of the capture rate for textiles is that a composition analysis of the mixed waste has been performed. In addition, all the amounts of waste textiles collected by each collection system (public and private) are needed in order to calculate the indicator.	N/A

Table 7-2. BEMPs presented in this document with related environmental performance indicators and benchmarks of excellence

BEMPs	Applicability	Benchmarks of excellence	Environmental performance indicators
All waste streams - Cross-cutting			
1) Integrated waste management strategy (Section 3.3.1)	This BEMP is primarily targeted to waste authorities with control, or at least significant influence over, waste management strategy at the local or regional level – primarily local authorities. The waste authority may need to outsource aspects of strategic planning where particular specialist expertise, such as analytical data skills, are required.	• An integrated waste management strategy that includes long-term (i.e. 10–20 years) and short-term (i.e. 1–5 years) overall targets for the improvement of the performance of the waste management system is in place and regularly reviewed (at least every 3 years).	• Overall targets for the improvement of the waste management system (e.g. based on the indicators defined in this report) are in place (y/n). • Specific targets for waste prevention and reuse are in place (y/n).
2) Life-cycle assessment of waste management options (Section 3.3.2)	Any waste management organisation may apply life-cycle thinking and review LCA studies. Buying bespoke LCA services and/or paying for staff training in LCA may only be economically viable for larger organisations.	• The waste management strategy is designed and implemented on the basis of systematic application of life-cycle thinking and, when needed, ad-hoc life-cycle assessment studies.	• Systematic application of life-cycle thinking, and, where necessary, undertaking of life-cycle assessments, throughout waste management strategy design and implementation (y/n).

3) Economic instruments (Section 3.3.3)	The regulatory framework and its enforcement are the main barriers for the application of economic instruments at local level. In addition, the existence of environmental awareness, good management skills and innovation-driven behaviour at the local government level are prerequisites for the implementation of local economic instruments.	• Economic instruments set at local level in the form of taxes and tax modulation, product levies, waste pricing, extended producer responsibility schemes and deposit refund schemes are systematically implemented as a means to achieve the objectives set in the local waste management strategy. • For local authorities, a deposit refund scheme for glasses, cups, dishes and cutlery is in place for all festivals and large public events organised in the territory of the local authority.	• Use of economic instruments at local level to stimulate good behaviour (y/n). • Share of residents/businesses using a voluntary economic instrument (%).
4) Link to other relevant reference documents for best practices (Section 3.3.4)	This BEMP is targeted to local waste authorities and waste management companies planning and carrying out operations in the areas of waste treatment, material recycling, energy recovery and waste disposal.	N/A	• Relevant state-of-the-art techniques described in the reference documents listed in this BEMP are implemented (y/n).
Municipal solid waste (MSW)			

1) Cost benchmarking (Section 4.3.1)	Cost benchmarking can be applied within an area (at local or national level) where waste management conditions are comparable and where there is a uniform legal framework. However, in some cases, strong deviations occur due to specific conditions.	N/A	• Regular participation in a detailed cost benchmarking study (y/n). • Total MSW management cost per resident per year (EUR/capita/year).
2) Advanced waste monitoring (Section 4.3.2)	Detailed waste monitoring is applicable to all local authorities and waste management companies managing municipal solid waste.	• Composition analysis of mixed waste is carried out at least four times a year (during different seasons) every three years or after any substantial change of the waste management system.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • use of web-based tools for tracking and reporting waste data (y/n); • frequency of composition analysis of mixed waste (one composition analysis every # months or years).
3) Pay-as-you-throw (Section 4.3.3)	While the approach is broadly applicable, existing infrastructure must be adapted (e.g. collection). Door-to-door collection is usually necessary to fully implement PAYT principles.	• A pay-as-you-throw system is in place, according to which at least 40 % of the cost is charged to the users depending on the quantity (kg or m^3) of mixed waste collected, the size of the waste collection bins and/or the number of collection rounds. • The PAYT system also includes the waste conferred to civic amenity sites.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this • A pay-as-you-throw system is in place (y/n); • inclusion of waste conferred to civic amenity sites in the PAYT system (y/n); • share of users with zero waste generation (%).

4) Performance-based waste management contracting (Section 4.3.4)	The existence of an effective waste management performance monitoring system is a prerequisite to performance-based waste monitoring system (building on internal management practices to expand to contract management).	N/A	• Share of the contract value depending on the achievement of the environmental objectives / environmental performance levels (%). • Customer satisfaction (% of residents satisfied with household waste collection and specifically with the collection of the separately collected fractions).
5) Awareness-raising (Section 4.3.5)	Awareness-raising can be implemented at some level in any context.	• Awareness campaigns are systematically implemented for different types of target groups (e.g. pupils, general public, users of civic amenity sites) and the annual budget devoted to awareness-raising activities is at least EUR 5 per resident.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • budget spent on awareness-raising per resident per year (EUR/capita/year); • share of total MSW management budget spent on awareness-raising (%); • share of population in the waste management catchment area having received awareness-raising messages over a given time period (e.g. % of population per month).

6) Establishment of a network of waste advisers (Section 4.3.6)	This BEMP can be implemented at any level. However, their scope of action is more focused on the local level since they address operational issues (waste prevention and recycling guidelines).	• A network of waste advisers is in place with at least one waste adviser per 20 000 residents.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • Share of population in the waste management catchment area advised by waste advisers over a given time period (e.g. % of population per month); • Number of waste advisers per 100 000 residents.
7) Home and community composting (Section 4.3.7)	In cases when home and community composting is the most appropriate waste management option for biowaste, there are no major restrictions to implementing this BEMP.	• All residents have access to either separate collection of biowaste or home and community composting of biowaste.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • share of population doing home composting or to which community composting is available (% of total population in the waste management catchment area); • share of population implementing home/community composting correctly, on the basis of an annual visit and analysis of the compost produced (% of the population doing home composting or to which community composting is available); • system in place for regular follow-up with residents doing home composting (y/n); • Share of home composters visited annually (% of the households doing home composting).

8) Local waste prevention programmes (Section 4.4.1)	Waste prevention measures need to be carefully selected based on local circumstances and well implemented (e.g. some may need support by financial incentives) but there are suitable measures for any context.	• Waste prevention has strategic relevance in the waste management strategy, which includes a local waste prevention programme underpinning long-term (i.e. 10–20 years) and short-term (i.e. 1–5 years) waste prevention targets and including provisions for regular monitoring.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • establishment of a local waste prevention plan, including long-term and short-term targets and provisions for regular monitoring (y/n); • budget dedicated to waste prevention programmes per resident per year (EUR/capita/year); • share of total MSW management budget devoted to waste prevention (%); • number of stakeholders involved in prevention programmes.
9) Schemes fostering the reuse of products and the preparation for reuse of waste (Section 4.4.2)	This BEMP applies to all waste management organisations that handle any type of reusable products and waste, in particular garments, furniture and electrical and electronic equipment.	• In civic amenity sites, product/material exchange areas aimed at fostering reuse are available.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • Number of reuse centres/community repair points per 100 000 residents; • Number or quantity (i.e. weight or volume) of end-of-life products collected for reuse and waste items sent for preparation for reuse; • Annual number of customers of the reuse centres/community repair points; • Availability of products/materials exchange areas aimed at fostering reuse in civic amenity sites (y/n).

10) Waste collection strategy (Section 4.5.1)	The prevailing socio-economic status and recycling consciousness within the area from which waste is collected needs to be considered in the definition of the waste collection strategy. More costly strategies, such as door-to-door collection, may prove more cost-effective once fully running, but require initial investment.	• Door-to-door waste collection of at least four waste fractions is implemented in the whole territory administered.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • Participation rate, i.e. the share of the population using the waste collection system (%); • Share of the local area covered with a specific waste collection system (%); • Customer satisfaction (% of residents satisfied with household waste collection and specifically with the collection of the separately collected fractions); • Collection of bulky waste on demand (y/n).
11) Inter-municipal cooperation (IMC) among small municipalities (Section 4.5.2)	There are no specific barriers for the application of inter-municipal cooperation in waste management. However, benefits from the economy of scale are only evident for small and medium municipalities.	N/A	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicator to assess the successful implementation of this BEMP is: • implementation of inter-municipal cooperation with other municipalities (y/n).

12) Civic amenity sites (Section 4.5.3)	The concept of collection centres is broadly applicable. The ultimate recyclability of the waste streams collected also depends on the availability of downstream markets.	• For municipalities with at least 1 000 residents, there is at least one civic amenity site in their territory. • At the civic amenity sites, at least 20 different waste fractions are collected. • In civic amenity sites, product/material exchange areas aimed at fostering reuse are available.	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • number of civic amenity sites per 100 000 residents; • number of different fractions collected at the civic amenity sites; • availability of product/material exchange areas aimed at fostering reuse in civic amenity sites (y/n); • easy accessibility of civic amenity sites, e.g. without a car (y/n).
13) Logistics optimisation for waste collection (Section 4.5.4)	All organisations involved in waste collection can implement some degree of logistics optimisation (e.g. planning the location of waste bins). However, the actions are limited in some cases by existing organisational structures	N/A	In addition to the common environmental performance indicators presented in Chapter 2, the most appropriate indicators to assess the successful implementation of this BEMP are: • fuel consumption per tonne of waste collected (litres/t); • GHG emissions per tonne of waste and km travelled (kg CO_2e/tkm);
14) Low-emission vehicles (Section 4.5.5)	This BEMP is broadly applicable.	• All new refuse collection vehicles purchased or leased by the waste management organisation are Euro 6 and are fuelled by either compressed natural gas or biogas, or are hybrid-electric.	• Average fuel consumption of the waste collection vehicles (litres/100 km). • Share of vehicles that are Euro 6 in the total waste collection vehicle fleet (%). • Share of waste collection vehicles that are hybrid, electric, natural-gas- or biogas-powered (%).

15) Best use of incentives by producer responsibility organisations (PROs) (Section 4.6.1)	The actual leverage that a PRO has on the EPR depends on the national setup and legal allocation of roles and responsibilities. For the application of some incentives, proper allocation of finances is needed.	N/A	• Recycling rate (% of waste that is actually recycled or sent for recycling out of the total waste covered by the EPR scheme). • Preparation for reuse rate (% of waste that is delivered as input to a centre for preparation for reuse out of the total waste covered by the EPR scheme). • (applicable at the local level for a specific local area where the EPR scheme is in place) Share of EPR-covered products found in residual waste based on composition analysis (% of the total quantity of mixed waste). • (applicable for a specific national, regional or local area where an EPR scheme for packaging waste is in place) Share of EPR-covered packaging that is targeted by the selective separate collection system (% of the total quantity of EPR-covered packaging put on the market).
16) Sorting of co-mingled light packaging waste to maximise recycling yields for high-quality output (Section 4.7.1)	There are no barriers to building and operating a packaging waste sorting plant. However, careful planning is required as part of an integrated waste management concept. An important factor that needs to be considered is related to the optimal plant capacity.	• Material recovery facilities sorting co-mingled light packaging waste have a plant sorting rate of at least 88 %.	• Plant sorting rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of co-mingled packaging waste processed. • Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of co-mingled packaging waste processed. • GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (Scope 1 and 2) of the plant divided by the quantity of co-mingled packaging waste processed.

17) Processing of mixed plastic packaging waste to maximise recycling yields for high-quality output (Section 4.7.2)	Good waste collection systems and the good quality of the collected materials need to be assured in order for the recycled output to be suitable for the market. As with the previous BEMP, there are no general barriers to building and operating such a plant. However, careful planning and determination of the optimal plant capacity are important.	• Plastic recovery facilities processing mixed plastics packaging waste have a plant processing rate of at least 60 %.	• Plant processing rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of mixed plastic packaging waste processed. • Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of mixed plastic packaging waste processed. • GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (scope 1 and 2) of the plant divided by the quantity of mixed plastic packaging waste processed. • Water use (m^3/t), calculated as the annual total water used on site divided by the quantity of mixed plastic packaging waste processed.
18) Treatment of mattresses for improved recycling of materials (Section 4.7.3)	There are no main technical barriers to the applicability of this BEMP. The simplicity of the treatment process does not require significant investments, even for the most automated processes.	• Facilities treating waste mattresses have a plant sorting rate of at least 91 %.	• Plant sorting rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of waste mattresses processed. • Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of waste mattresses processed. • GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (scope 1 and 2) of the plant divided by the quantity of waste mattresses processed.

19) Treatment of absorbent hygiene products for improved recycling of materials (Section 4.7.4)	This BEMP is broadly applicable as no particular geographical or technical barriers exist. However, some specific conditions can influence the technical and economic viability of this treatment solution.	• Facilities treating absorbent hygiene products waste have a plant sorting rate of at least 90%.	• Plant sorting rate (weight %), calculated as the annual quantity of materials sent for recycling divided by the annual quantity of AHP waste processed. • Energy efficiency (kJ/t), calculated as the annual total energy consumption of the plant divided by the quantity of AHP waste processed. • GHG emissions (t CO_2e/t), calculated as the annual total CO_2 equivalent emissions (scope 1 and 2) of the plant divided by the quantity of AHP waste processed. • Water use (m^3/t), calculated as the annual total water used on site divided by the quantity of AHP waste processed.
Construction and demolition waste (CDW)			
1) Integrated construction and demolition waste plans (Section 5.4.1)	The formulation and implementation of local waste management plans for CDW is a commonly used instrument by regions and large municipalities.	• An integrated CDW management plan is implemented with a target CDW recycling rate in 2020 of at least 80 % and provisions for monitoring and enforcement mechanisms.	• Share of total collected CDW that is correctly segregated and managed towards reuse, recycling or recovery (%). • Provision for pre-demolition audits aimed at reuse (y/n).
2) Avoidance of PCBs contamination of CDW (Section 5.4.2)	This BEMP is broadly applicable for waste authorities responsible for CDW.	N/A	• Inclusion of provisions for the mapping and separate removal and collection of PCB-containing materials in the CDW plan (y/n).

3) Local schemes for proper management of waste asbestos removed by residents (Section 5.4.3)	This BEMP is applicable only to certain cement-bonded asbestos (such as asbestos cement roofs, wall and ceiling cladding; asbestos down pipes and gutters, etc.) in good condition	• There is at least one collection point per 100 000 residents or free home collection for waste asbestos removed by residents.	• Number of collection points for asbestos waste per 100 000 residents. • Total amount of asbestos collected through the scheme, expressed in weight (tonnes) or surface area (m^2). • Number of sealable bags for collection/disposal of asbestos used by residents.
4) Processing waste plasterboard to foster recycling (Section 5.4.4)	There are no technical barriers to the applicability of this BEMP. However, there are significant economic barriers (i.e. well segregation of waste plasterboard and transport costs).	N/A	• Efficiency of material recovery at the waste plasterboard processing plant (%).
5) Processing CDW for the production of recycled aggregates (Section 5.4.5)	There is no specific limitation to the applicability of this BEMP as long as the CDW is well segregated into the different fractions at the construction site.	N/A	• Efficiency of material recovery at the CDW processing plant (%). • Annual amount of RCA marketed (tonnes/year).
Healthcare waste (HCW)			

1) Encouragement of healthcare waste segregation at healthcare facilities (Section 6.4.1)	There is no specific limitation to the applicability of this BEMP by HCW management companies.	N/A	• Share of staff members of the client healthcare facility having undergone a training session about waste in the last two years (%). • Share of correct answers given by staff members of the client healthcare facility in post-training evaluation surveys about handling of waste in the HCW facility (%). • Collection rates per waste fraction, per bed or per patient, according to the specific fractions collected in each healthcare facility (kg/patient/day).
2) Healthcare waste collection for households (Section 6.4.2)	The BEMP is applicable to all local authorities and/or waste management companies.	N/A	• Number of collection points for HCW generated by households per 10 000 residents, by type (civic amenity sites, pharmacies, street containers). • Number of individual boxes for HCW generated by households distributed via collection points or on request. • Quantity of HCW generated by households collected (kg/capita/year). • Share of HCW (e.g. sharps) in mixed household waste (%).

3) Alternative treatments for healthcare waste (Section 6.5.1)	Four main factors affect the applicability of alternative treatments: source segregation, proving the safety of alternative treatments in treating certain fractions of segregated waste, the optimum operating capacity for incineration and the national legal framework for HCW treatment.	N/A	• Share of HCW managed by the HCW management company processed by alternative treatments (%). • Amount of HCW processed by alternative treatments (kg HCW per hour, day or cycle). • Water consumption per kg of waste processed by alternative treatments (litres/kg).

8. Annexes

8.1. Annex 1: Treatment and recovery operations according to the WFD

DISPOSAL OPERATIONS

D 1 Deposit into or on to land (e.g. landfill, etc.)

D 2 Land treatment (e.g. biodegradation of liquid or sludgy discards in soils, etc.)

D 3 Deep injection (e.g. injection of pumpable discards into wells, salt domes or naturally occurring repositories, etc.)

D 4 Surface impoundment (e.g. placement of liquid or sludgy discards into pits, ponds or lagoons, etc.)

D 5 Specially engineered landfill (e.g. placement into lined discrete cells which are capped and isolated from one another and the environment, etc.)

D 6 Release into a water body except seas/oceans

D 7 Release to seas/oceans including sea-bed insertion

D 8 Biological treatment not specified elsewhere in this Annex which results in final compounds or mixtures which are discarded by means of any of the operations numbered D 1 to D 12

D 9 Physico-chemical treatment not specified elsewhere in this Annex which results in final compounds or mixtures which are discarded by means of any of the operations numbered D 1 to D 12 (e.g. evaporation, drying, calcination, etc.)

D 10 Incineration on land

D 11 Incineration at sea ([1])

D 12 Permanent storage (e.g. emplacement of containers in a mine, etc.)

D 13 Blending or mixing prior to submission to any of the operations numbered D 1 to D 12 ([2])

D 14 Repackaging prior to submission to any of the operations numbered D 1 to D 13

D 15 Storage pending any of the operations numbered D 1 to D 14 (excluding temporary storage, pending collection, on the site where the waste is produced) ([3])

([1]) This operation is prohibited by EU legislation and international conventions.

([2]) If there is no other appropriate D code, this can include preliminary operations prior to disposal including preprocessing such as, inter alia, sorting, crushing, compacting, pelletising, drying, shredding, conditioning or separating prior to submission to any of the operations numbered D1 to D12.

([3]) Temporary storage means preliminary storage according to point (10) of Article 3.

RECOVERY OPERATIONS

R 1	Use principally as a fuel or other means to generate energy ([1])
R 2	Solvent reclamation/regeneration
R 3	Recycling/reclamation of organic substances which are not used as solvents (including composting and other biological transformation processes) ([2])
R 4	Recycling/reclamation of metals and metal compounds
R 5	Recycling/reclamation of other inorganic materials ([3])
R 6	Regeneration of acids or bases
R 7	Recovery of components used for pollution abatement
R 8	Recovery of components from catalysts
R 9	Oil re-refining or other reuses of oil
R 10	Land treatment resulting in benefit to agriculture or ecological improvement
R 11	Use of waste obtained from any of the operations numbered R 1 to R 10
R 12	Exchange of waste for submission to any of the operations numbered R 1 to R 11 ([4])
R 13	Storage of waste pending any of the operations numbered R 1 to R 12 (excluding temporary storage, pending collection, on the site where the waste is produced) ([5])

([1]) This includes incineration facilities dedicated to the processing of municipal solid waste only where their energy efficiency is equal to or above:

— 0.60 for installations in operation and permitted in accordance with applicable Community legislation before 1 January 2009,

— 0.65 for installations permitted after 31 December 2008,

using the following formula: Energy efficiency = (Ep - (Ef + Ei))/(0.97 × (Ew + Ef)), in which:

Ep means annual energy produced as heat or electricity. It is calculated with energy in the form of electricity being multiplied by 2.6 and heat produced for commercial use multiplied by 1.1 (GJ/year);

Ef means annual energy input to the system from fuels contributing to the production of steam (GJ/year);

Ew means annual energy contained in the treated waste calculated using the net calorific value of the waste (GJ/year);

Ei means annual energy imported excluding Ew and Ef (GJ/year);

0.97 is a factor accounting for energy losses due to bottom ash and radiation.

This formula shall be applied in accordance with the reference document on Best Available Techniques for waste incineration.

([2]) This includes gasification and pyrolysis using the components as chemicals.

([3]) This includes soil cleaning resulting in recovery of the soil and recycling of inorganic construction materials.

([4]) If there is no other appropriate R code, this can include preliminary operations prior to recovery including preprocessing such as, inter alia, dismantling, sorting, crushing, compacting, pelletising, drying, shredding, conditioning, repackaging, separating, blending or mixing prior to submission to any of the operations numbered R1 to R11.

([5]) Temporary storage means preliminary storage according to point (10) of Article 3.

8.2. Annex 2: Waste composition analysis from Portugal

Waste composition analysis methodology in Portugal

Portugal approved in 2009 the technical standards for the Municipal Waste composition analysis methodology (*Portaria n.º 851/2009, de 7 de Agosto*).

The entities responsible for municipal waste management (urban waste management systems) shall ensure the characterization of municipal waste produced in its geographical area of intervention. Municipal waste produced annually must be characterized in terms of the categories and subcategories indicated in the PT ordinance

The entities must report to the Portuguese Environment Agency [APA - Portuguese Environment Agency (APA)] the data by 31 March of the year following that to which the data relate.

The waste sector comprises the collection, treatment and disposal of municipal waste. The waste services are classified as bulk services or retail services according to the performed activities. Bulk service is related to the wholesale activity of urban waste management, whereas the retail services focus only on the interaction with the end-user.

Waste composition analysis methodology

Main categories	subcategories
Fines <20 mm.	
Bio-waste (Bio-waste includes biodegradable garden waste, food and kitchen waste from households, restaurants, catering and retail units and similar waste from food processing plants)	Food and kitchen waste Garden waste Other putrescible waste
Paper and cardboard	Waste paper and waste cardboard packaging Newspapers and magazines. Other paper /cardboard waste (non packaging)
Plastic	Packaging waste in PE film. Waste from rigid PET packaging. Waste from rigid packaging in HDPE. EPS rigid packaging waste. Other waste plastic packaging. Other plastic waste (non packaging)
Glass	Waste glass packaging. Other glass waste (non packaging)
Composites	Waste paperboard packaging for liquid food (ex. Tetra Pak or similar) Other composite packaging waste. Small appliances (small WEEEs)

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	Other composite wastes (non packaging)
Textiles	Waste textile packaging. Other textile waste (non packaging)
Helth care textiles (diapers, sanitary towels,...)	
Metals	Ferrous waste packaging. Non-ferrous waste packaging. Other ferrous waste. Other metal waste.
Wood	Waste wood packaging. Other wood waste.
Hazardous waste	Chemicals. Fluorescent tubes and low consumption lamps. Batteries and accumulators. Other hazardous waste.
Other waste	Other packaging waste. Other non-packaging waste.
Green waste (from selective collection)	
Bulky waste	

According the PT Ordinance, the urban waste streams produced are characterized by sampling, and an annual Waste composition analysis campaign must to be carried out, consisting of two sampling periods, one in autumn-winter (wet season) and one in spring-summer (dry season).

In order to guarantee the representativeness of the data, atypical or exceptional periods, such as festive seasons or holidays, should be avoided, with repercussions on the characteristics of municipal waste produced.

Minimum number of samples for characterization of urban waste produced	
Waste stream	Number of samples / campaign
Mixed waste (un-sorted municipal waste)	21
Bio-waste selective collection	10
Paper and cardboard selective colection	5
Co-mingled light packaging waste (plastic, metals and composite packaging (Tetra Pak or similar for food and liquids)	10
Glass selective collection	2
Other selective waste sterams	5

The number of samples should be distributed proportionally to the estimated annual production of waste, according to the various waste streams and recovery/final deposition. Sampling should also integrate the contribution of different sectors, such as different types of settlement (urban, rural, sparsely populated areas) or producers.

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Amount of material per sample	
Waste stream	Sample weight (kg), wet weight
Mixed waste (un-sorted waste)	350
Other streams (selective collection)	250

Sampling must be random. The material for the sampling should be leaked, followed by the quartet method involving the following steps:

- Mixing of waste with shovel, carrying out several turnings;
- Scattering of the waste to form a coarse disc with a height of up to about 50 cm;
- Divide this disc into four equal parts, rejecting two opposing quarters;
- Mixture of remaining quarters;

Repeat the sequence of the previous steps until the desired weight for the sample is reached (350 kg or 250 kg).

The Figure below shows the procedure (in PT):

In order to guarantee the confidence of results, waste composition analysis of the samples in the established categories and subcategories shall take place up to twenty-four hours after their preparation (as mentioned previously), with the registration of the results on a specific record sheet, as well as any anomalous or relevant situations.

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The average physical composition of the different streams shall be expressed in terms of the mean values obtained for the percentage by weight of each category and subcategory on the basis of the wet weight (without moisture correction). The following statistical parameters should also be determined for each category: Minimum; Maximum; Median; Standard deviation; Coefficient of variation; Confidence interval of the mean, with 95% probability; Percent error, with 95% probability.

According the PT ordinance, the relative accuracy (percentage error) of the mean values obtained for the bio-waste, paper and cardboard, plastic, glass, metals and fines categories shall be less than 20%. In the absence of such a situation, the sampling conditions for the waste composition analysis to be carried out in the following year shall be reviewed and the minimum number of samples to be considered shall be statistically determined in order to achieve that minimum precision.

LIPOR example

In 2016, the waste composition analysis campaigns (sample preparation, sorting and weighing to determine the percentage physical composition, determination of the specific weight) of the various waste streams were carried out by a team consisting of 4 full-time workers and a technicien (half time).

In the Annual Campaign (2016), a total of 60 samples of the Mixed Waste stream, 20 samples of the co-mingled light packaging waste stream, 20 samples of the selective collection stream of Paper/Cardboard, 12 samples of the selective collection stream of Glass, 4 samples of the Bulky waste selective collection, 44 samples of the Bio-Waste selective collection stream, were prepared and analysed.

For the mixed waste stream, the amount of MSW produced by each municipality was considered in order to obtain a representative randomness. For the other streams, a simple random sampling procedure was used, collected from the total mass of waste.

The results regarding the municipal waste composition can be seen at http://portal.lipor.pt:7777/pls/apex/f?p=2021:24:0

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In terms of costs, we can refer, as an average value, about €8,500 for carrying out 40 analyses.

The Portuguese Environment Agency publishes in its website the portuguese data regarding the waste composition.

https://www.apambiente.pt/index.php?ref=16&subref=84&sub2ref=933&sub3ref=936

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List of abbreviations

AD	Anaerobic digestion
BEMP	Best environmental management practice
BREF	Best Available Techniques Reference Document
CDW	Construction and demolition waste
CED	Cumulative energy demand
CNG	Compressed natural gas
CO_2e	Carbon dioxide equivalent (measure for global warming potential)
CVRS	Computerised vehicle routing and scheduling
1,4-DCBe	1,4-Dichlorobenzene equivalent (measure for human toxicity potential)
DMC	Domestic material consumption
EMAS	Eco-Management and Audit Scheme
EoW	End-of-waste
EPR	Extended product / producer responsibility
EUR	Euro (€)
FGD	Flue-gas desulphurisation
FRDP	Fossil resource depletion potential
GBP	Pound sterling (£)
GWP	Global warming potential
HCW	Healthcare waste
HGV	Heavy goods vehicles
IED	Industrial Emissions Directive (Directive 2010/75/EU of the European Parliament and of the Council)
IMC	Inter-municipal cooperation
LCA	Life-cycle assessment
LCI	Life-cycle inventory

LCIA	Life-cycle impact assessment
MBT	Mechanical and biological treatment
MJ_e	Megajoule equivalent (measure for fossil resource depletion potential)
MRF	Material recovery facility
MSW	Municipal solid waste
MSWI	Municipal solid waste incineration (plant)
MW	Medical waste
NMVOC	Non-methane volatile organic compounds
NO_x	Nitrogen oxides
OHW	Organic household waste
PAH	Polycyclic aromatic hydrocarbons
PAYT	Pay-as-you-throw
PCB	Polychlorinated biphenyls
PCDD/F	Polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs)
PDF	Potentially disappeared fraction (measure for land use)
PM	Particulate matter
PO_4e	Phosphate equivalent (measure for eutrophication potential)
PRO	Producer responsibility organisation
RCA	Recycled concrete aggregates
RDF	Refuse-derived fuels
Sb_e	Antimony equivalent (measure for abiotic resource depletion potential)
SO_2e	Sulphur dioxide equivalent (measure for acidification potential)
SOPs	standard operation procedures (SOPs)
SO_x	Sulphur oxides
SRD	Sectoral Reference Document
SRF	Solid recovered fuels

SWMP	Site waste management plan
TWG	Technical Working Group supporting the development of the EMAS SRD
USD	US Dollar ($)
VA	Voluntary agreements
VOC	Volatile organic compounds
WEEE	Waste from Electrical and Electronic Equipment
WFD	Waste Framework Directive (Directive 2008/98/EC of the European Parliament and of the Council)
WtE	Waste-to-energy

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