Background Report on Best Environmental Management Practice in the Waste Management Sector

Preparatory findings to support the development of an EMAS Sectoral Reference Document

Report for the European Commission's Joint Research Centre

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Index

List of Abbreviations

AD	Anaerobic Digestion
ARDP	Abiotic Resource Depletion Potential
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 (£)
GVA	Gross Value Added
GWP	Global Warming Potential
HCW	Healthcare Waste
HGV	Heavy Goods Vehicles
HWCC	Household Waste Collection Centre
IED	Industrial Emissions Directive
IMC	Inter-municipal Cooperation

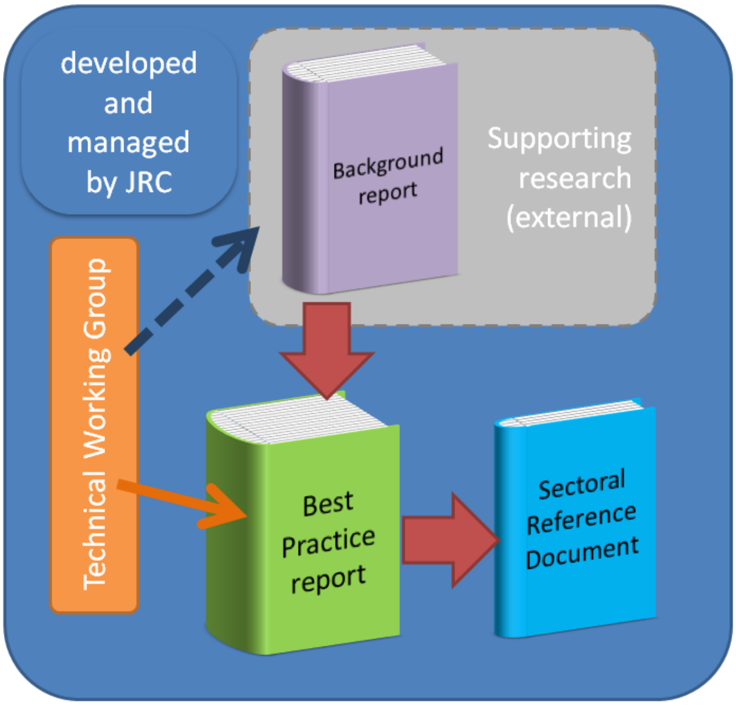
IWMP	Integrated Waste Management Plan
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MBT	Mechanical and Biological Treatment
MJ_e	Megajoule Equivalent (Measure for Fossile Resource Depletion Potential)
MRF	Materials Recovery Facility
MSW	Municipal Solid Waste
MSWI	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	Sulfur Dioxide Equivalent (Measure for Acidification Potential)
SO_x	Sulfur Oxides

SRD	Sectoral Reference Document
SRF	Solid Recovered Fuels
SWMP	Site Waste Management Plan
TWG	Technical Working Group
USD	U.S. Dollar ($)
VA	Voluntary Agreements
VOC	Volatile Organic Compounds
WEEE	Waste from Electrical and Electronic Equipment
WFD	Waste Framework Directive
WMO	Waste Management Organisation
WtE	Waste-to-Energy

PREFACE

This draft background report provides an overview of techniques that may be considered **Best Environmental Management Practices** (BEMPs) in the waste management sector. The document was developed by *BZL Kommunikation und Projektsteuerung GmbH* (Germany) and *E3 Environmental Consultants Ltd.* (UK) under a contract with the European Commission's Joint Research Centre (JRC) on the basis of desk research, interviews with experts and site visits. This background report is intended to provide a preliminary basis for further discussions between the JRC and technical experts via the forum of a Technical Working Group (TWG). **The contents of this report therefore represent early findings that will be further developed through discussions with the TWG**, according to a structured process 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[1].

The final findings will be presented in a best practice report produced by the JRC and used for the development of an EMAS Sectoral Reference Document (SRD), as illustrated below.



Source: JRC

Figure I: The present background report in the overall development of the Sectoral Reference Document (SRD)

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

[1] http://susproc.jrc.ec.europa.eu/activities/emas/documents/DevelopmentSRD.pdf

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 waste management sector.

Nevertheless, it is important to note that the guidance on BEMP is not only for EMAS participants, but rather is intended to be a useful reference document for any relevant organisation[2] 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 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 organisations (or by at least one organisation, but is replicable/applicable by 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, yet at the same time 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 I.

[2] The word "organisation", in the context of the EMAS regulation and throughout this report, refers to any "company, corporation, firm, enterprise, authority or institution, located inside or outside the Community, or part or combination thereof, whether incorporated or not, public or private, which has its own functions and administration" (Regulation (EC) 1221/2009, Art. 2(21)).

Table I: 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.
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 performance data from specific implementations 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.

- **Environmental Performance Indicators** represent the metrics that are employed by organisations in the sector to monitor either the implementation of the BEMPs described or, when possible, directly their environmental performance.
- **Benchmarks of Excellence** represent the highest environmental standards that have been achieved by organisations 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 is possible to achieve (under stated conditions) that organisations can use to set priorities for action in the framework of continuous improvement of environmental performance.

Conclusions on sector-specific Environmental Performance Indicators and Benchmarks of Excellence are drawn by the experts of the TWG at the end of their interaction with the JRC. Therefore the proposals for indicators (and, eventually, for benchmarks) contained in this background report are to be considered no more than *preliminary proposals* from the authors of this background report.

Role and purpose of this document

The present background report provides a basis to be used by the JRC and the Technical Working Group for the elaboration of the "JRC Scientific and Policy Report on Best Environmental Management Practice in the Waste Management Sector", or simply "Best Practice Report", containing the technical basis for the Sectoral Reference Document (SRD).

Organisations from the waste management sector interested in implementing best practice in the improvement of environmental performance are recommended to refer instead to the final Best Practice Report that will be available on-line[3] as soon as it is finalised and published.

3 See: http://susproc.jrc.ec.europa.eu/activities/emas/waste_mgmt.html

EXECUTIVE SUMMARY

Target group

The proposed best environmental management practices (BEMPs) described in this report are intended to support the efforts of all waste management organisations (WMO) to improve their environmental performance, i.e. waste authorities (local public administrations in charge of waste management) and waste management companies / waste contractors.

Scope

The document covers best environmental management practices which can be implemented by waste authorities and waste managers/contractors on the level of municipalities, cities, counties or regions, i.e. to strategies and integrated management plans, prevention, re-use, extended product responsibility (EPR), collection and treatment of municipal solid waste (MSW), construction and demolition waste (CDW) and healthcare waste (MW). All practices already covered in other SRDs or European Commission technical reports for other initiatives or legislations are excluded from this document. The main aspects of the sector that have been excluded are product policy, end-of-waste criteria based on the Waste Framework Directive, and waste treatment technologies covered under the Industrial Emissions Directive. (IED).

Information sources

Information has been sourced from available public sources including comprehensive reports and scientific literature. Also, information has been collected directly from waste authorities, waste managers, consultancy firms, non-governmental organisations, and technology providers. A number of site visits proved to be very useful for obtaining technical and performance data and information on economic aspects.

Structure of the document

The document is divided in the following chapters:

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

In this chapter, the macroeconomic situation of the sector in 2014 is described, along with its main environmental challenges and environmental aspects, direct or indirect, that the organisations from the sector are managing. Statistical information on EMAS implementation is also provided.

Chapter 2. Cross-cutting issues

Cross-cutting issues are those concerning municipal solid waste, construction and demolition waste, healthcare waste, and even other types of wastes that waste authorities and/or waste contractors have to manage. Overall strategies and the minimisation of the environmental impact of operations through assessment tools, as life cycle assessment, are analysed.

Chapter 3. Municipal Solid Waste (MSW)

This chapter sequentially addresses a range of best practice techniques to manage MSW. The content of this chapter refers to overall strategies for MSW, prevention, collection, re-use, treatment, and extended product responsibilities, along with the so-called enabling techniques. The latter are techniques oriented to the implementation of other best practices, but are not best practices *per se*. The structure of this part of the document is shown in Figure A.1.

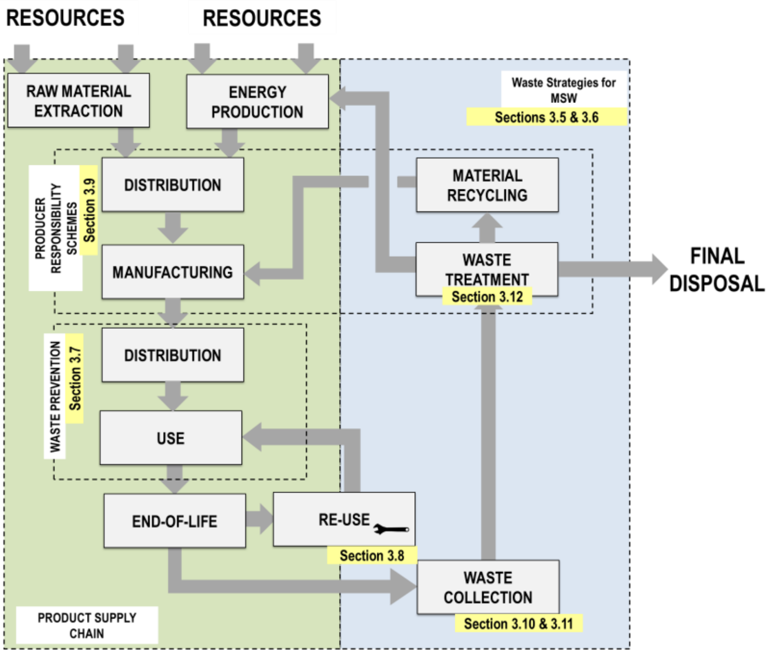


Figure A.1. The *waste* supply chain and the structure of the chapter on MSW

Chapter 4. Construction and Demolition Waste (CDW)

This chapter focuses on the involvement of waste authorities and waste management companies directly or indirectly responsible for the main environmental aspects of CDW management. However, several aspects of CDW logistics, on-site management and treatment operations will be already covered in the EMAS SRD for the Building and Construction Sector[4]. Therefore, this chapter is oriented to fill the gaps and extend the scope of the treatment options described in that document.

Chapter 5. Healthcare Waste (HCW)

In the case of HCW, prevention measures are the most important but excluded from this document, as they are exclusively associated with activities of the health care sector and not with the waste management sector. Integrated segregation and collection and alternative treatments that can be implemented by waste management companies dealing with HCW are the main focus of this chapter.

[4] The SRD for the building and construction sector will be based on the related best practice report 'Best Environmental Management Practice for the Building and Construction sector' available at: http://susproc.jrc.ec.europa.eu/activities/emas/documents/ConstructionSector.pdf

Chapter 6. Applicability to small organisations

This chapter summarises the main aspects and relevant best environmental management practices for small organisations (both SMEs – small and medium enterprises – and small waste authorities – i.e. local authorities in charge of the management of waste from small populations).

Conclusions

This background report proposes a series of best environmental management practices, for each of the relevant covered aspects, and environmental performance indicators that can be used to report their performance. Where available, ranges of environmental performance are also given in the detailed description of each technique.

Based on these, further research by the European Commission, and all the information provided by the experts involved in the process, the Technical Working Group will conclude on the final list of BEMPs and environmental performance indicators as well as on a series of Benchmarks of Excellence to be included in the final best practice report and, ultimately, in the SRD.

RESUME

Cible

Les meilleures pratiques de management environnemental (BEMPs) décrites dans ce rapport visent à soutenir les efforts de toutes les organisations de gestion des déchets (WMO), à savoir les autorités en charge des déchets (administrations publiques locales en charge de la gestion des déchets), les compagnies de gestion des déchets et les entrepreneurs dans le domaine des déchets, en vue d'améliorer leurs performances environnementales.

Objectif

Ce document couvre les meilleures pratiques de management environnemental (BEMPs) qui peuvent être implémentées par les autorités en charge des déchets et les gestionnaires/entrepreneurs dans le domaine des déchets au niveau des municipalités, villes, départements ou régions, à savoir les stratégies et les plans de gestion intégrés, la prévention, la réutilisation, la responsabilité élargie du produit (EPR) ainsi que la collecte et le traitement des déchets ménagers (MSW), des déchets de construction et de démolition (CDW) et des déchets médicaux (HCW). Toutes les pratiques déjà couvertes par d'autres documents de référence sectoriels (SRDs) ou rapports techniques de la Commission Européenne sur d'autres initiatives ou législations sont exclues de ce document. Les principaux aspects du secteur qui ont été exclus sont la politique de produit, les critères de fin de la qualité de déchet basés sur la directive-cadre sur les déchets (WFD) et les technologies de traitement des déchets couvertes par la directive sur les émissions industrielles (IED).

Sources d'information

Certaines informations ont été obtenues à partir de sources publiques disponibles, parmi lesquelles des rapports complets et de la littérature scientifique. D'autres informations ont été obtenues directement de la part d'autorités en charge des déchets, de gestionnaires dans le domaine des déchets, de sociétés de conseil, d'organisations non gouvernementales et de fournisseurs de technologies. Plusieurs visites de sites se sont révélées être très utile afin d'obtenir des données techniques et des données de performance ainsi que des informations sur les aspects économiques.

Structure du document

Le document est composé des chapitres suivants :

Chapitre 1. Informations générales sur le secteur de la gestion des déchets, son intérêt environnemental et l'implémentation du système communautaire de management environnemental et d'audit (EMAS) dans le secteur

Dans ce chapitre, la situation macroéconomique du secteur en 2014 est décrite ainsi que les principaux défis et aspects environnementaux, directs ou indirects, gérés par les organisations du secteur. Des informations statistiques sur la mise en œuvre du système communautaire de management environnemental et d'audit (EMAS) sont également fournies.

Chapitre 2. Problèmes transversaux

Les problèmes transversaux correspondent aux problèmes liés aux déchets ménagers (MSW), aux déchets de construction et de démolition (CDW), aux déchets médicaux (HCW) et aux autres types de déchets qui doivent être gérés par les autorités en charge des déchets et/ou par les entrepreneurs dans le domaine des déchets. Des stratégies globales ainsi que la minimisation de l'impact environnemental des différentes opérations par l'utilisation d'outils d'évaluation, comme l'analyse du cycle de vie (LCA), sont analysées.

Chapitre 3. Les déchets ménagers (MSW)

Ce chapitre présente successivement plusieurs méthodes de meilleure pratique de gestion des déchets ménagers. Ce chapitre contient des stratégies globales pour les déchets ménagers : leur prévention, collecte, réutilisation et traitement ainsi que les responsabilités élargies des produits et les techniques dites « de créativité ». Ces dernières sont des techniques orientées vers l'implémentation d'autres meilleures pratiques mais ne sont pas des meilleures pratiques en soi. La structure de cette partie du document est présentée en figure A.1.

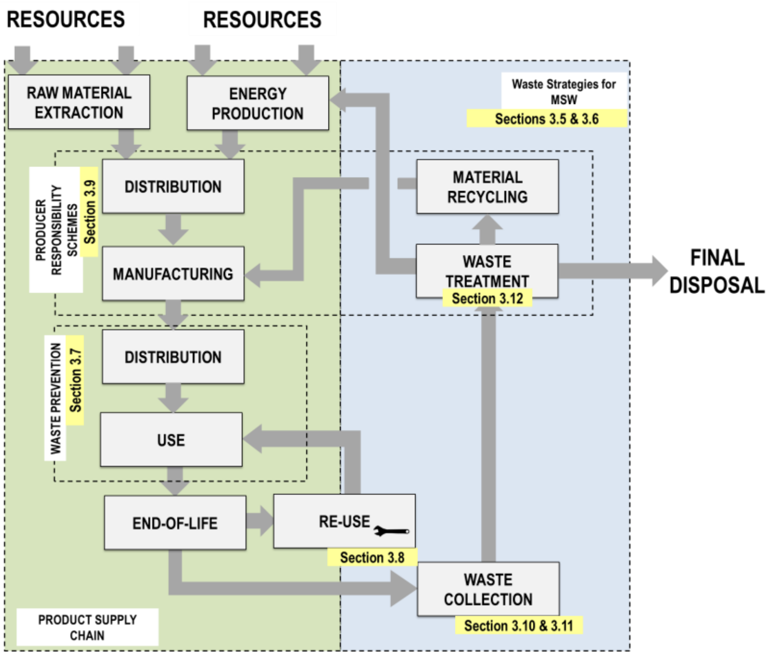


Figure A.1. Chaine logistique des déchets et structure du chapitre sur les déchets ménagers (MSW)

Chapitre 4. Les déchets de construction et de démolition (CDW)

Ce chapitre se concentre sur l'implication des autorités en charge des déchets et des compagnies de gestion des déchets qui sont, directement ou indirectement, responsables des principaux aspects environnementaux de la gestion des déchets de construction et de démolition. Cependant, plusieurs aspects de la logistique, de la gestion sur place et des opérations de traitement des déchets de construction et de démolition sont déjà couverts par le document de référence sectoriel (SRD) du système communautaire de management environnemental et d'audit (EMAS) pour le secteur du bâtiment et de la construction[5]. Ce chapitre est donc axé sur le remplissage des blancs et sur l'extension du champ des options de traitement décrites dans ledit document.

Chapitre 5. Les déchets médicaux (HCW)

Dans le cas des déchets médicaux, les mesures de prévention sont les plus importantes mais elles sont cependant exclues de ce document car exclusivement associées aux activités du secteur de la santé et non à celles du secteur de la gestion des déchets. Ce chapitre se focalise principalement sur la ségrégation et la collecte intégrées ainsi que sur les traitements alternatifs qui peuvent être implémentés par les compagnies de gestion des déchets s'occupant de déchets médicaux.

Chapitre 6. Applicabilité dans le cas de petites organisations

Ce chapitre résume les aspects principaux des meilleures pratiques de management environnemental pertinentes pour de petites organisations (petites et moyennes entreprises – SMEs – et petites autorités en charge des déchets, par exemple des autorités locales en charge de la gestion des déchets pour de petites populations).

Conclusions

Ce rapport de fond propose une série de meilleures pratiques de management environnemental pour chacun des aspects pertinents qui ont été couverts ainsi que des indicateurs de performance environnementale qui peuvent être utilisés pour rendre compte de leur performance. Quand elles sont disponibles, des plages de performance environnementale sont également fournies dans la description détaillée de chaque technique.

A partir de ce rapport, des recherches supplémentaires menées par la Commission Européenne et des informations fournies par les experts impliqués dans la procédure, le groupe de travail technique (TWG) pourra établir la liste finale des meilleures pratiques de management environnemental (BEMPs) et des indicateurs de performance environnementale ainsi qu'une série de repères d'excellence à inclure dans le rapport final sur les meilleures pratiques de management environnemental (BEMPs) et, finalement, dans le document de référence sectoriel (SRD).

[5] Le document de référence sectoriel pour le secteur du bâtiment et de la construction est basé sur le rapport de meilleure pratique associé "Meilleures pratiques de management environnemental pour le secteur du bâtiment et de la construction" disponible ici:
http://susproc.jrc.ec.europa.eu/activities/emas/documents/ConstructionSector.pdf

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 economic system 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.

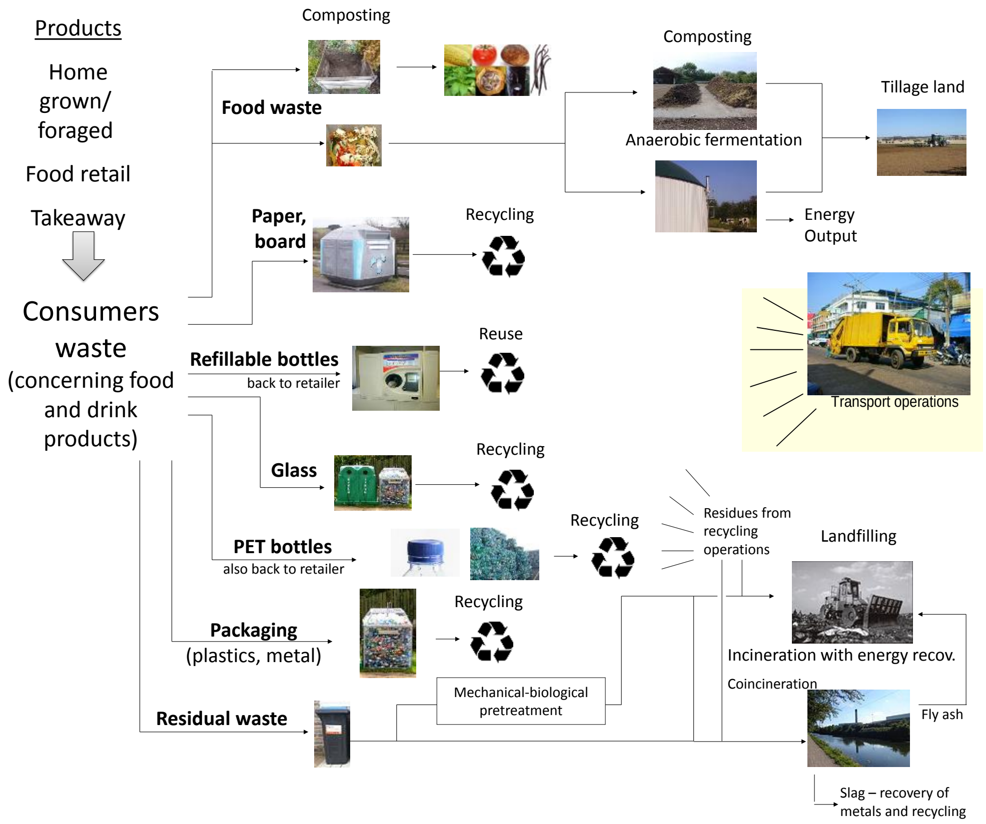


Figure 1.1. Re-use, recovery, recycling and disposal of consumer waste including the associated transport activities

On average, each EU citizen consumes 16 tonnes[6] of materials annually, of which six 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 the largest share, 34 %, from the construction sector (Figure 1.2). In total, 4 % of the waste generated is estimated as hazardous.

[6] 1 tonne is a non-SI metric unit of mass equal to 1,000 kilograms and is thus equivalent to one megagram (Mg).

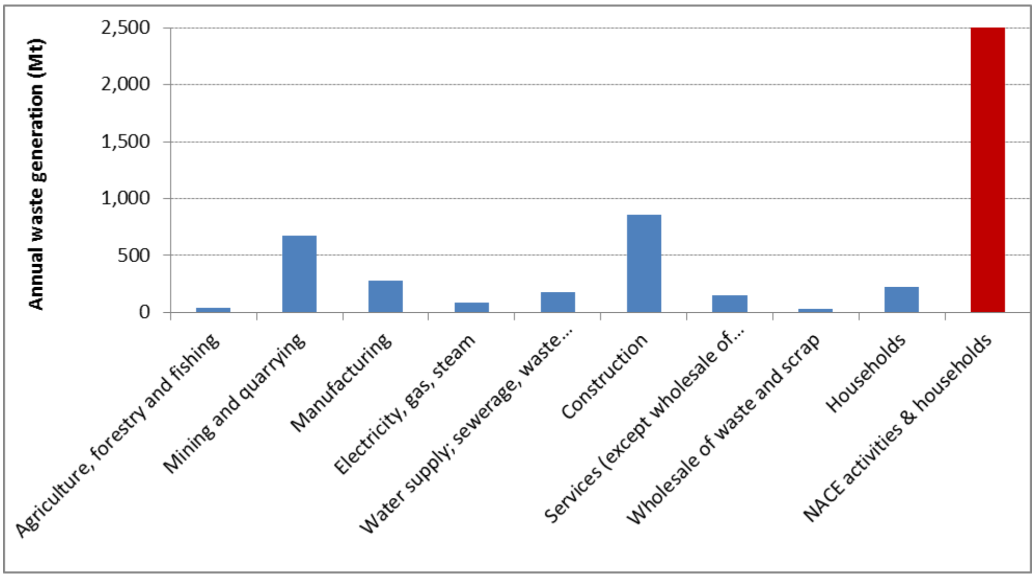


Figure 1.2. Waste generated by NACE sectors across the EU-28 in 2010 in Mt (= Million tonnes) (Data from Eurostat, 2014, (env_wasgen)

Germany, France and the UK together account for more than 39 % of the total amount of waste generated in Europe (Eurostat, 2014) (see Figure 1.3).

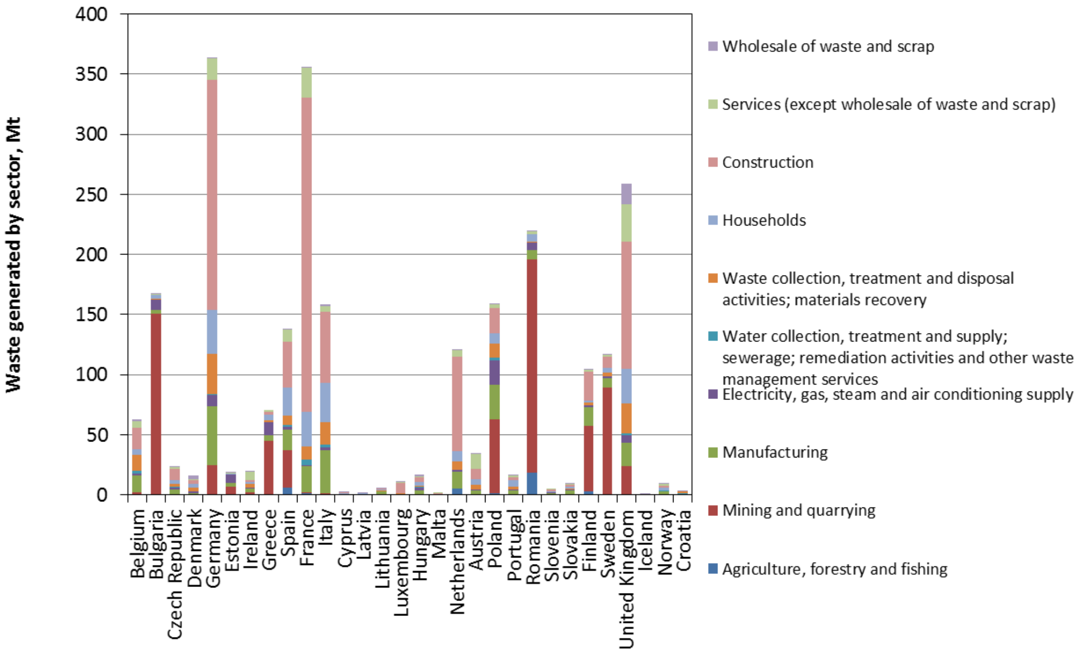


Figure 1.3. Waste generated by NACE sectors in European countries in 2010 in Mt (Data from Eurostat, 2014, env_wasgen)

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

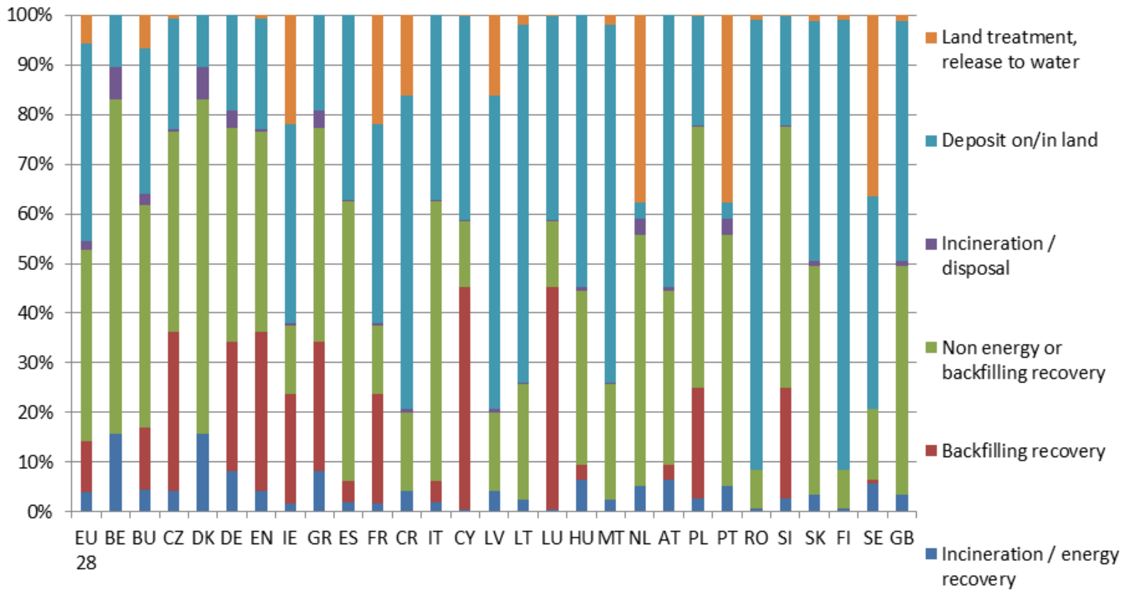
Waste management systems in the EU Member States differ significantly, varying from zero to 90 % disposal of untreated waste on landfills (Figure 1.4).

The latest statistics from Eurostat show that 45.4 % of **total waste** is disposed in Europe, while 49 % was sent to any recovery operation. The remaining 5.6 % corresponds to incineration.

Although representing only around 10 % of total waste generated in the EU-28 by mass (Figure 1.5), **municipal solid waste (MSW)**, i.e. household waste and similar commercial, industrial and institutional waste (EC, 2014), is one of the most polluting categories of waste, and the category with the highest potential for environmental improvement through better management. It is a highly political issue due to its composition, distribution and its inevitable link to consumption patterns.

This waste fraction is generated by households and commercial enterprises, and 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 methodologies used to produce waste statistics across Member States. The statistical value is mainly affected by how household-type waste from commerce, industry and institutions is considered.

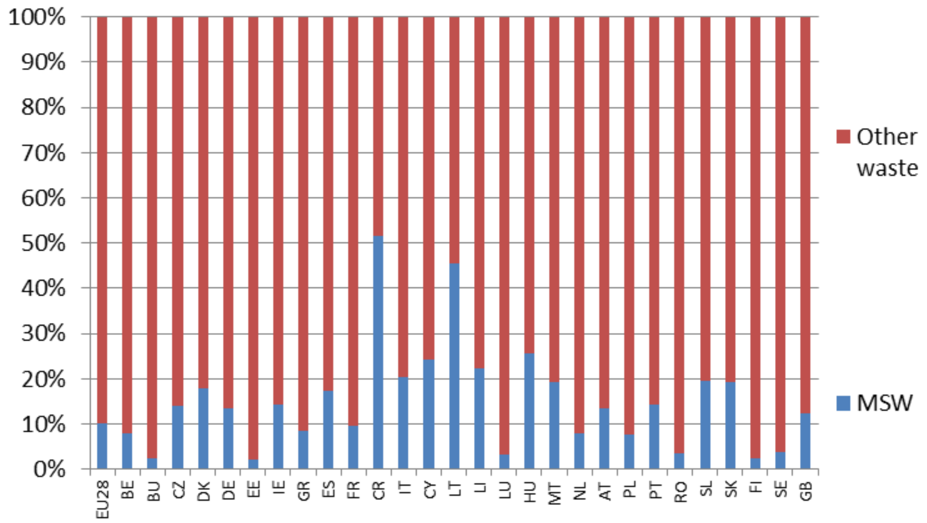
On average, each EU citizen generated 492 kg MSW in 2012, down from 522 kg in 2007 (Eurostat, 2014). On average, only a limited share (40 %) of the municipal waste generated is recycled, with the rest being landfilled (37 %) or incinerated (23 %).



Source: Eurostat (2014)

Figure 1.4. Percentages of total wastes undergoing different treatment or disposal options across the EU-28 in 2010[7]

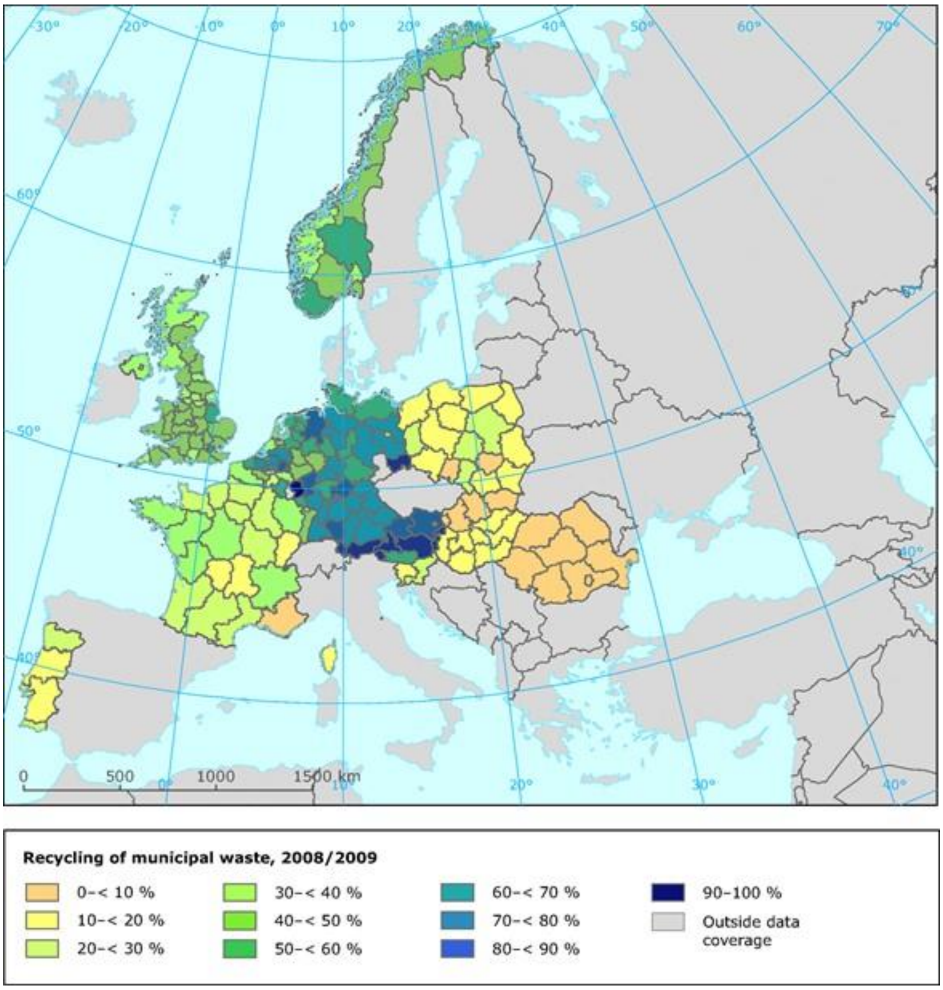
[7] As seen in the original publication, some of the country abbreviations are not standard.



Source: Eurostat (2014)

Figure 1.5. The percentage of total waste that is categorised as Municipal Solid Waste (MSW) across Member States of the EU-28[7]

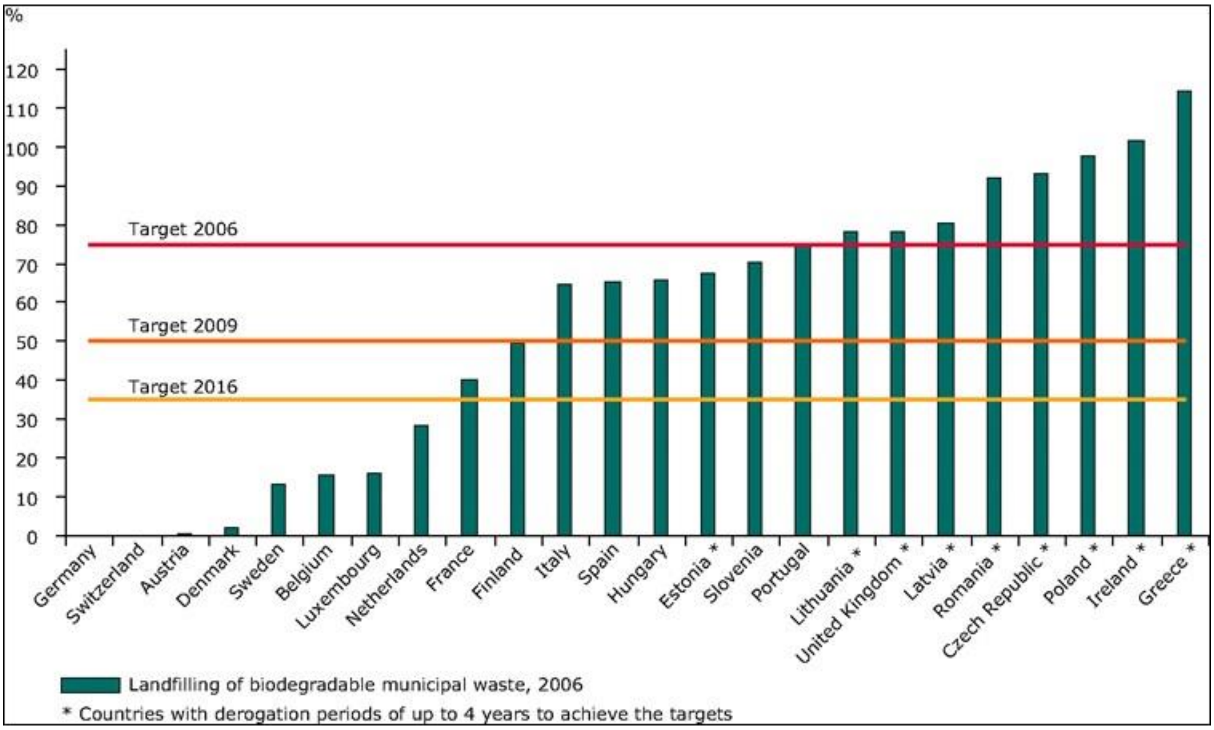
The European Environment Agency reported (EEA, 2013a) that whilst eleven Member States have already met, or are on track to meet, the Waste Framework Directive's target for 50 % of MSW to be recycled by 2020, the majority of Member States will have to make unprecedented progress in increasing recycling rates (the ones are presented in the figure below for the time period 2008-2009) in order to meet this target (Figure 1.6).



Source: EEA (2013a)

Figure 1.6. 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 1.7). Whilst meeting these targets is ultimately the responsibility of national and local government, also private companies, including small and medium enterprises are heavily involved in delivering waste management and recycling services.



Source: EEA (2012)

Figure 1.7. 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 so-called "waste hierarchy" (Figure 1.8).

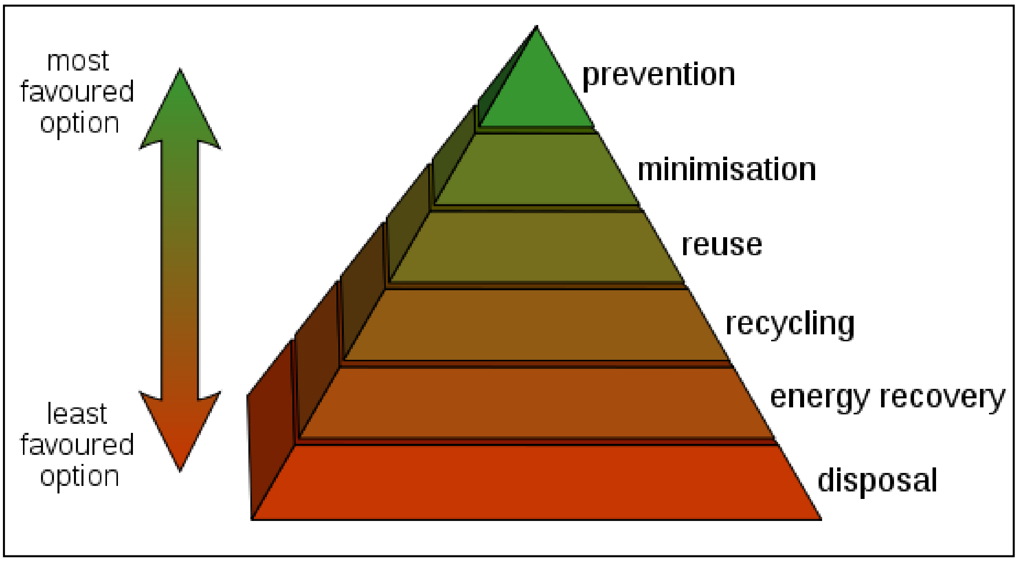


Figure 1.8. Waste hierarchy according to the Waste Framework Directive 2008/98/EC (Source: wikipedia (https://en.wikipedia.org/wiki/Waste_hierarchy)

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 1.9.

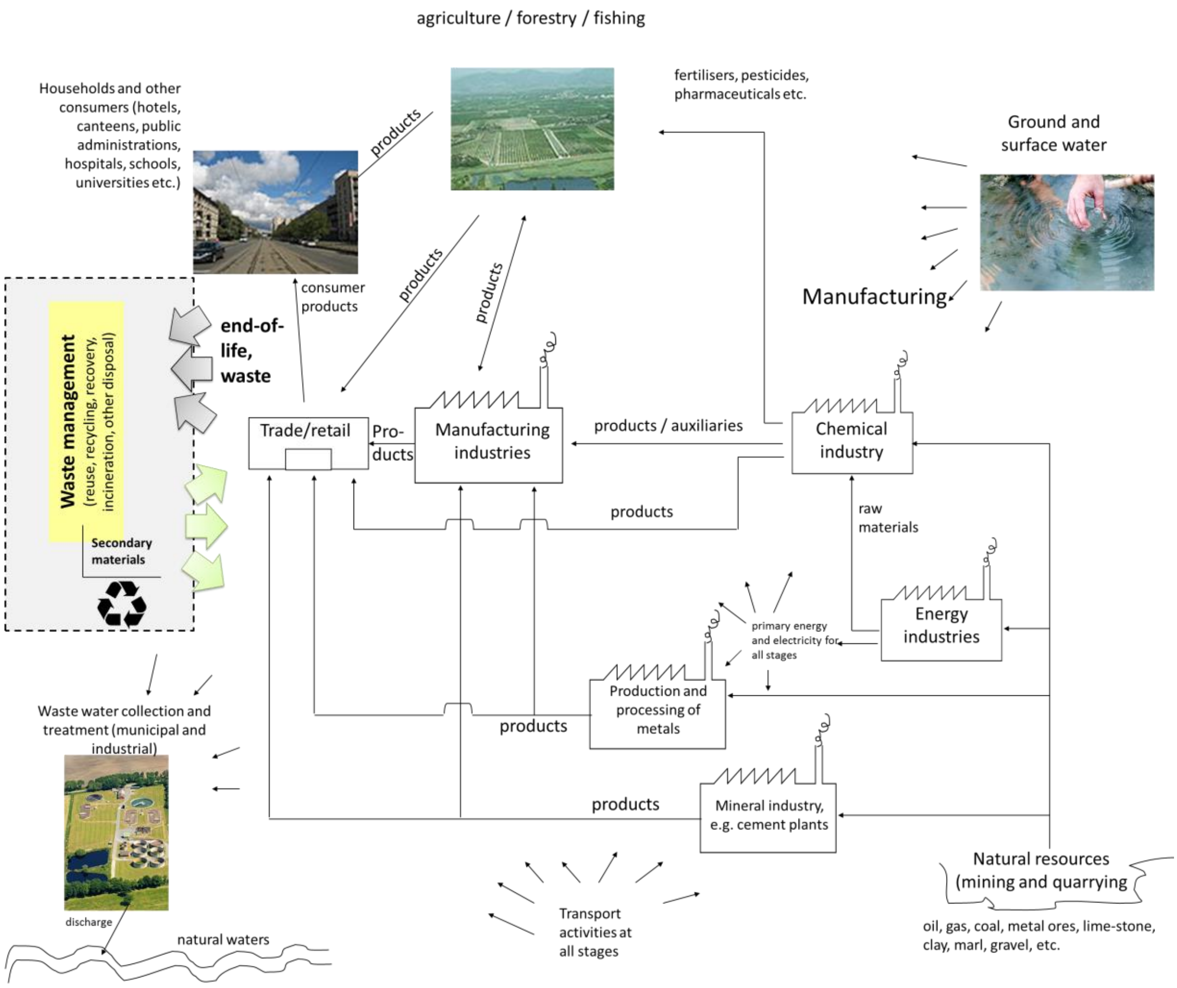
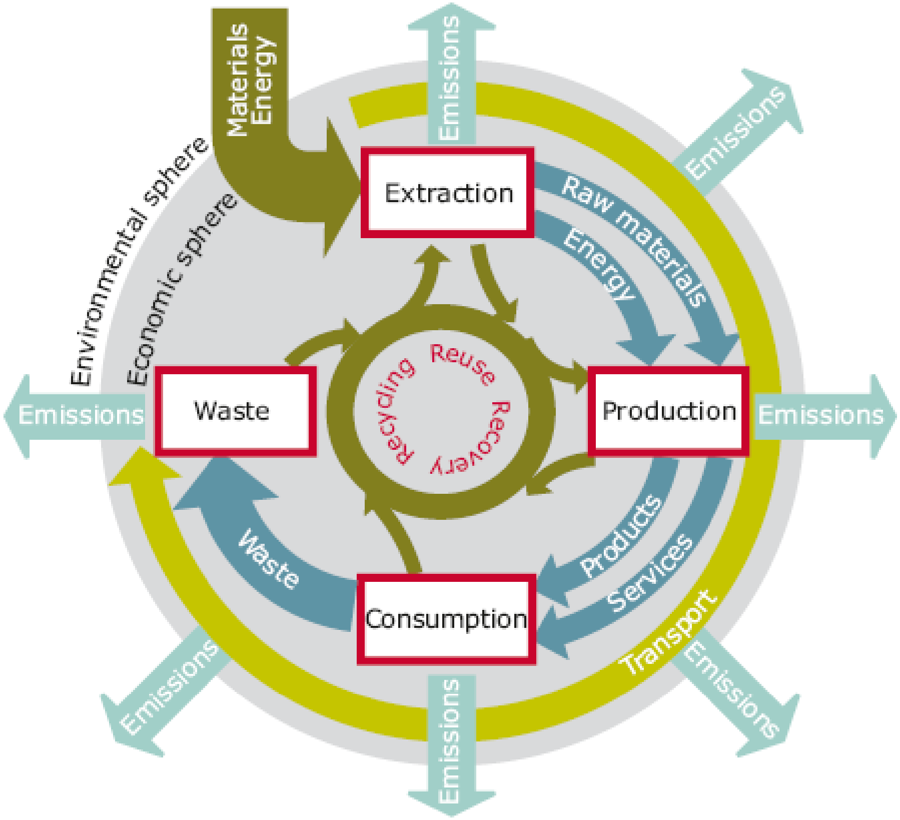


Figure 1.9. Basic scheme for the mass streams of current economic system

The European Commission has a long-term objective to foster a sustainable circular economy in which materials are extensively re-used and recycled through feedback loops that both support and directly generate economic activity (Figure 1.10). This objective is integral to achieving long-term economic stability, prosperity and a high quality of life for European citizens.



Source: EEA (2010)

Figure 1.10. A conceptual representation of raw materials and energy flows, services and transport, in the European economy. In a circular economy, inputs of virgin finite resources are minimised.

Efficient waste management, in particular waste prevention, re-use and recycling, is a critical component of a resource-efficient economy that is targeted by EU law. Key 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 Ecodesign 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)

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 waste authorities (public administration in charge of managing wastes from their citizens, policies and regulations) are considered within the boundaries of the sector, because the consequences of the decisions made at public administration level are key to determine 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 ten employees, specialised usually in collection and materials recovery. Indeed, from a total of 44,424 companies in NACE division 38 (according to Eurostat), 77 % are micro and 99.7 % are SMEs (less than 250 employees). Besides the number of companies, it is important to note also 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 would often be waste departments in municipalities or other local authorities. However, many of the SMEs reported below are public companies.

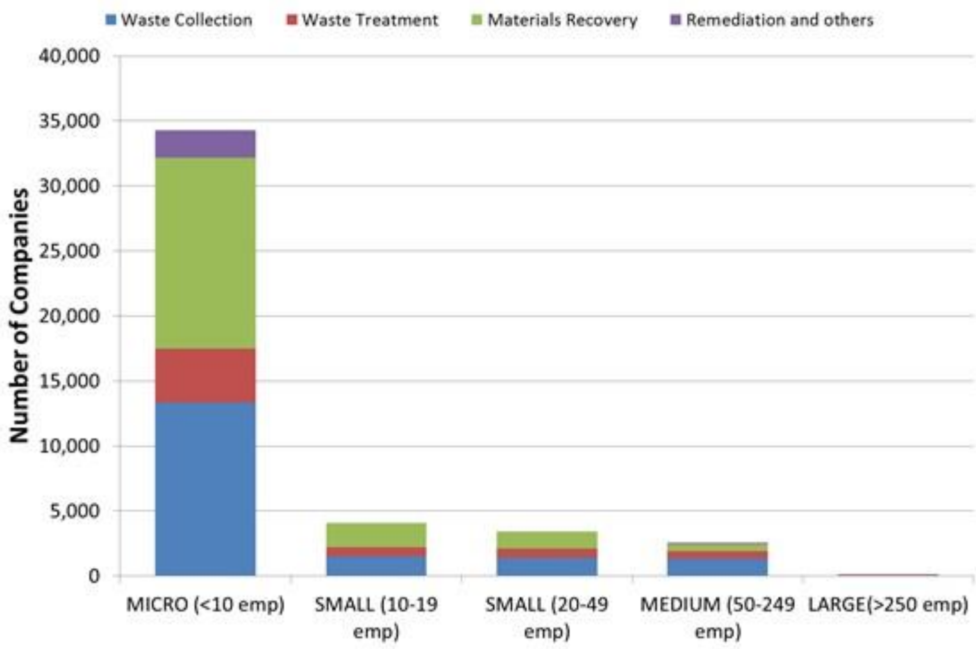


Figure 1.11. Number of companies in Europe (EU-28) per waste subsector and size (Data from Eurostat, sbs_na_ind_r2)

The structure per country is extremely heterogeneous regarding the size and the number of companies (Figure 1.12), 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 1.13). The turnover of the whole waste collection subsector (including all types of wastes) sums EUR 50,000 million, the waste treatment around EUR 35,000 million, and the 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 1.14. 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 materials recovery subsector, however, is dominated by smaller companies.

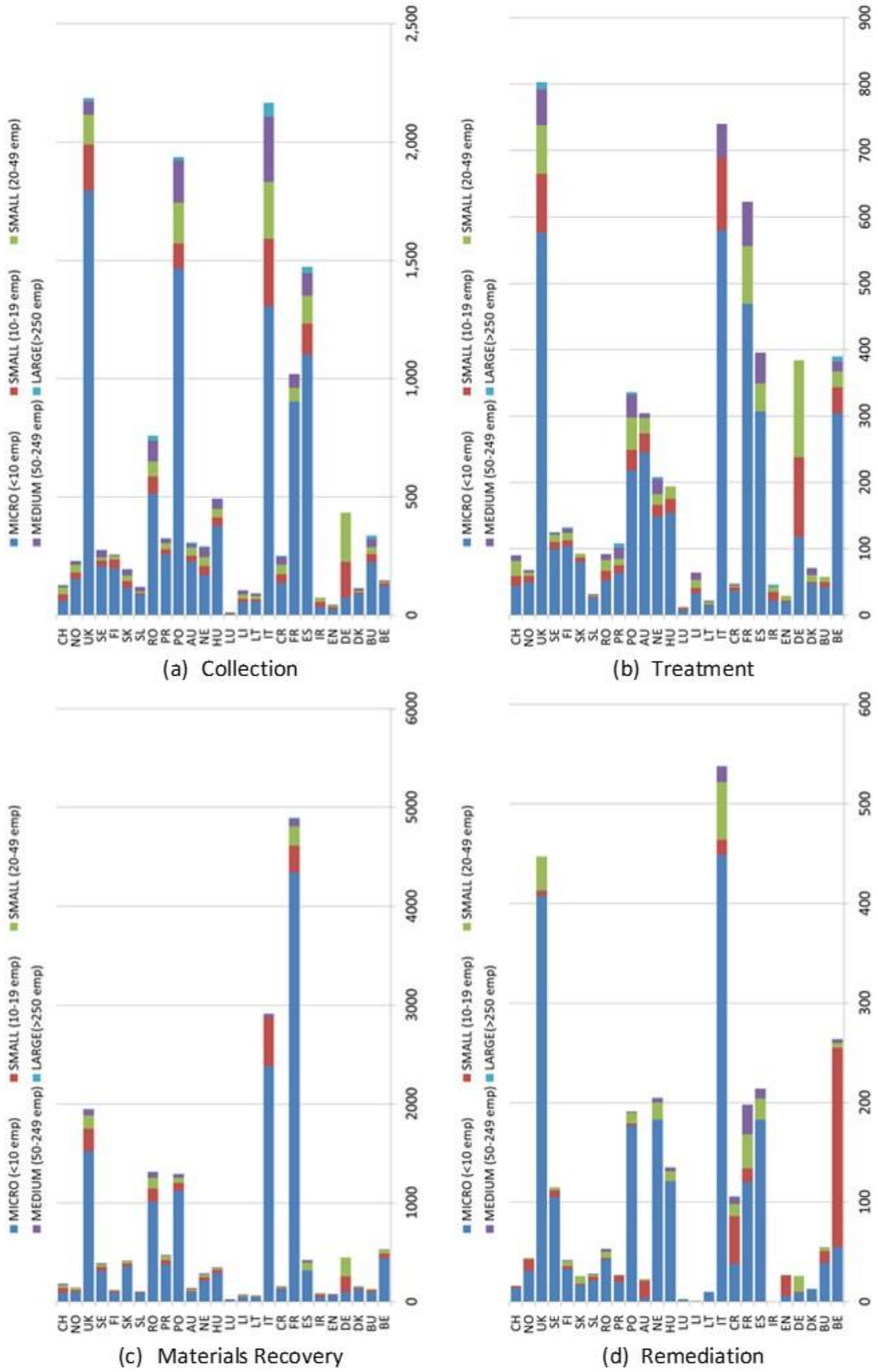


Figure 1.12. Number of companies per country and size for a) waste collection, b) waste treatment, c) materials recovery and d) remediation (Data from Eurostat, sbs_na_ind_r2)

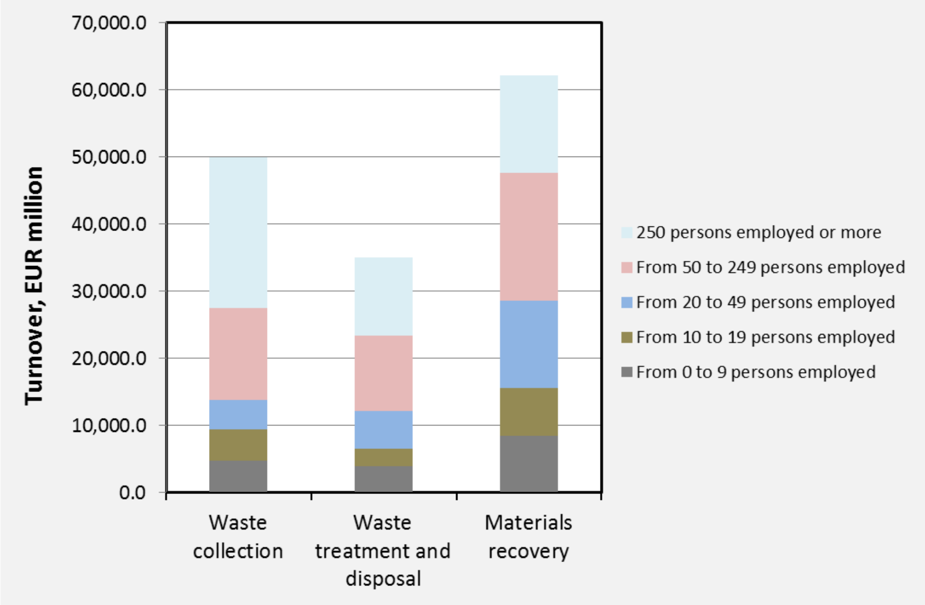


Figure 1.13. Turnover per waste subsector and size of company (remediation excluded) (data from Eurostat, sbs_na_ind_r2, 2013)

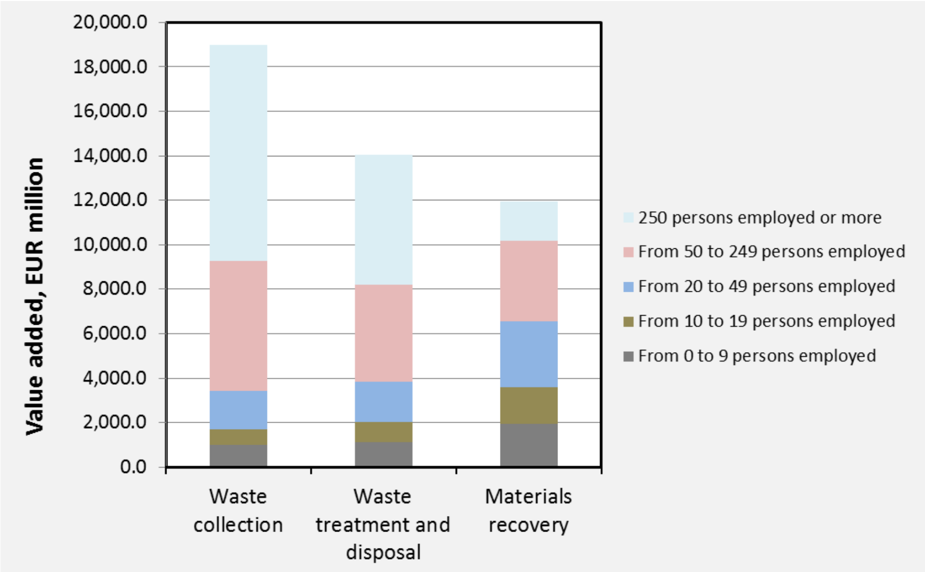


Figure 1.14. Value added per waste subsector and size of company (remediation excluded) (Data from Eurostat, sbs_na_ind_r2, 2013)

The number of persons employed per subsector and size of company is shown in Figure 1.15. In total, 900,000 people are accounted as employed by the sector, but this number could be 20 to 30 % higher due to different statistical approaches (Hall and Nguyen, 2012).

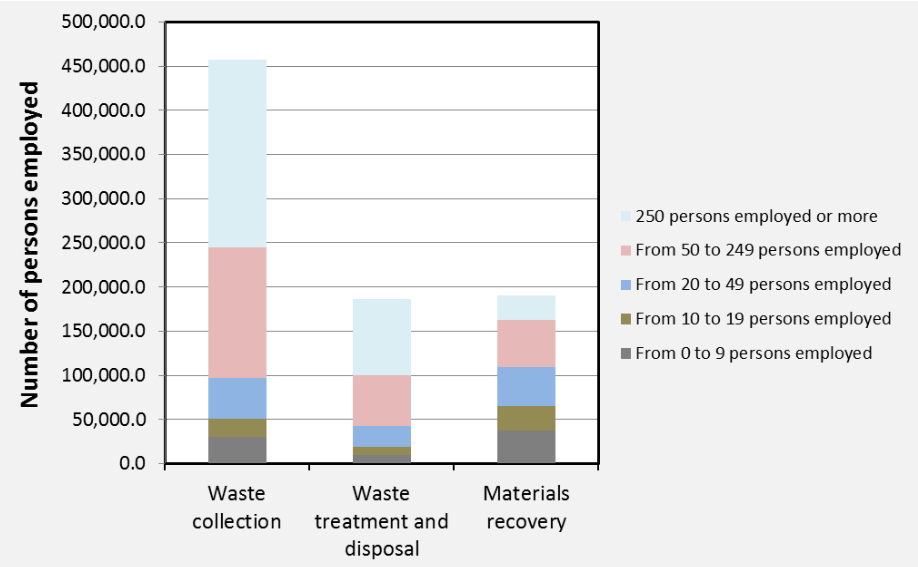


Figure 1.15. Persons employed by the waste sector in Europe (Data from Eurostat, sbs_na_ind_r2, 2013)

There is an evident high labour intensity in the waste collection subsector, while waste treatment or materials recovery have a similar 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, value added per person employed, varies with the labour intensity and the size of the company (Figure 1.16).

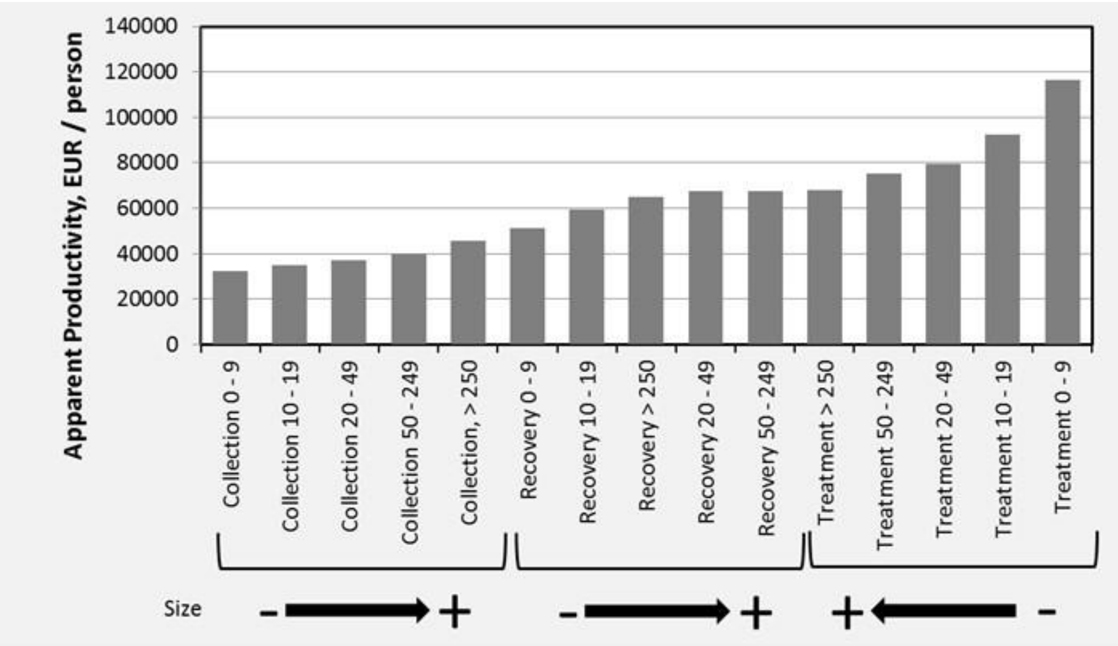


Figure 1.16. Apparent productivity of the waste sector in Europe (Data from Eurostat, sbs_na_ind_r2, 2013)

Treatment has higher productivity, probably due to the existence of larger facilities, with lower unitary cost of treatment and higher throughput per employee. Results show also 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 a lower apparent productivity, as its labour intensity is higher and its performance is quite limited by transport capacities and fuel costs. Large companies perform better, but a lower productivity compared to other sectors is observed. Materials recovery productivity is not varying 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 practice are rather limited and their investment capacities are probably low for those with lower productivity. A higher number of employees require 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 quite 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 the practices mainly driven by the 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 from the 16 biggest private organisations in waste management sums 40 % of the total revenue of the sector, mainly 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 for 2006 (Table 1.2). Although the data is 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 2006: percentage 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 would be addressed in the SRDs for their respective sectors.

1.2.2. Waste management activities

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 IED (Industrial Emission Directive)[8].
- 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[9] (developed under the **Waste Framework Directive**) which specify when certain waste ceases to be considered waste and obtains a 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, 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.).
- **Waste re-use** (e.g. schemes promoting repairing and reselling of end-of-life electronic equipment and furniture).

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[8] 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 taken 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.

[9] 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 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 to recycle plastics).

For other phases (i.e. other waste treatment and disposal facilities, recycling and recovery operations) reference will be made to the relevant reference documents, legislation, or criteria. The figure below illustrates the waste management phases in relation to the project: in green the ones covered, in yellow the one partially covered and in red the one not addressed.

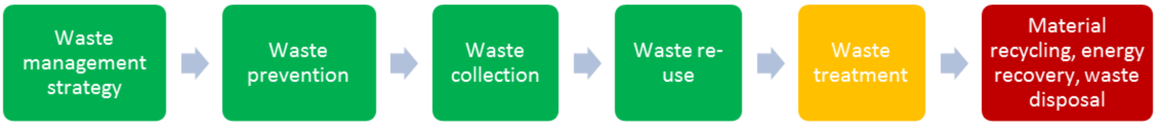


Figure 1.17. Waste management activities covered in the scope of this document

Detailed description of the waste management activities covered

In general, the activities of organisations belonging to NACE class 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; these include construction and demolition waste, debris and the operation of transfer facilities;
- Collection of recyclable materials;
- Collection of refuse in litter-bins in public places.

The collection of hazardous wastes (class 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 out of the scope of the activities to be covered. Collection of bio-hazardous and healthcare waste, used batteries, used oil from small garages, etc. are within the scope of activities to be considered in the background document and may be included in the final SRD.

Treatment and disposal of non-hazardous waste (class 38.21) is not covered completely in the document: operation of landfills is excluded as well as 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 (class 38.22). These activities may thus be only 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 group 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[10]. Under the scope of this background document, materials recovery activities are considered if they are (i) performed by a waste manager, public or private, and (ii) are excluded from the IED BREF waste-related best available techniques. Waste processing by companies not belonging to the waste management sector would only be considered if required as part of integrated management strategies.

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

NACE class 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 on the environmental performance of waste trading activities and its impact on the environmental performance of the waste (or end-of-waste) material supply chain (e.g. transportation and movement of traded waste reduces considerably the carbon reduction achievable by its use in manufacturing processes from an LCA perspective). However, the impact of trade activities on the performance of waste management is excluded from the purpose of these activities.

This background document covers the activities under class 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" is included (Eurostat, 2008). Indeed, many strategic decisions, 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 better economic and environmental performance than private schemes (Bel and Mur, 2009). In recent years, the waste management sector is also subject to a *remunicipalisation* 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 to the citizens, but, in some case studies, it has also been caused by 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

[10] 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).

production of secondary materials. As the service remains profitable, 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:

- Municipal solid waste (MSW): household waste and assimilated, e.g. those services producing waste of similar composition. 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 not specifically addressed in other European best practice reference documents.

Industrial waste and commercial waste non assimilated to household waste are not targeted in this document as they are better addressed in the specific document(s) addressed to the specific sector where the waste is generated (e.g. end-of-use vehicles are addressed in the document on car manufacturing[11]).

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 high environmental impact (MSW), or with high volumes (CDW), or with 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 of further specific consideration if regulated by regional or national legislation, or are out of the scope if they fall under the IED scope.

[11] 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 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

Table 1.3. Categories of waste to be considered under the European list of wastes (EC, 2014)

Chapter	Subchapter	Category
	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 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

Table 1.3. Categories of waste to be considered under the European list of wastes (EC, 2014)

Chapter	Subchapter	Category
		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
	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

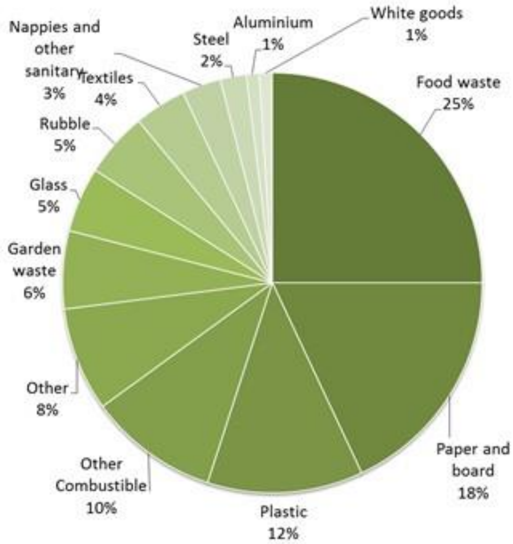
(*) Hazardous components from electrical and electronic equipment may include accumulators and batteries mentioned in 16 06 and marked as hazardous, mercury switches, glass from cathode ray tubes and other activated glass, etc.

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 similar waste composition as households (e.g. offices, administration services, schools, etc.).

However, the European Environment Agency (EEA) in 2013 found that European countries have very different approaches in 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.

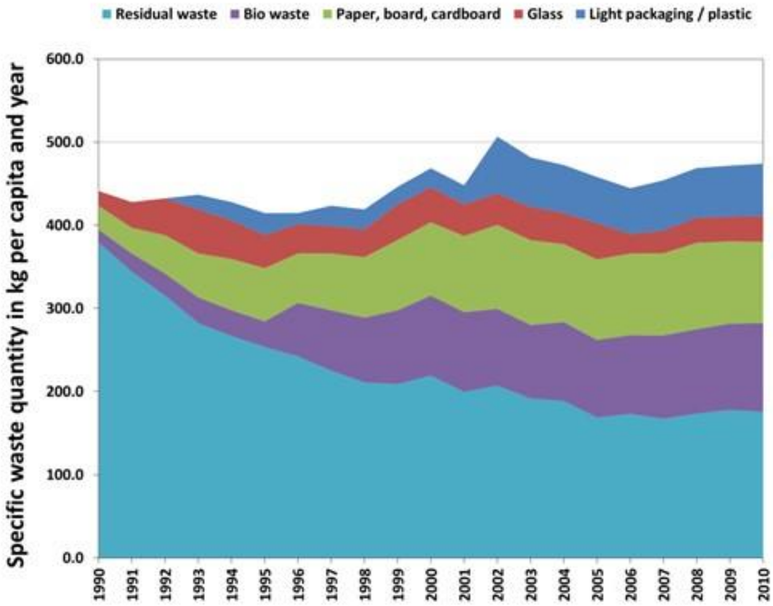
As this document focuses on environmental management practice, the most appropriate definition is according to “nature” or “compositional” characteristics of the waste. The typical qualitative composition of municipal waste (Figure 1.18) is used to classify materials and practices described in this document.



Data source: Zero Waste Europe, 2015

Figure 1.18. Sample composition of municipal solid waste in Europe

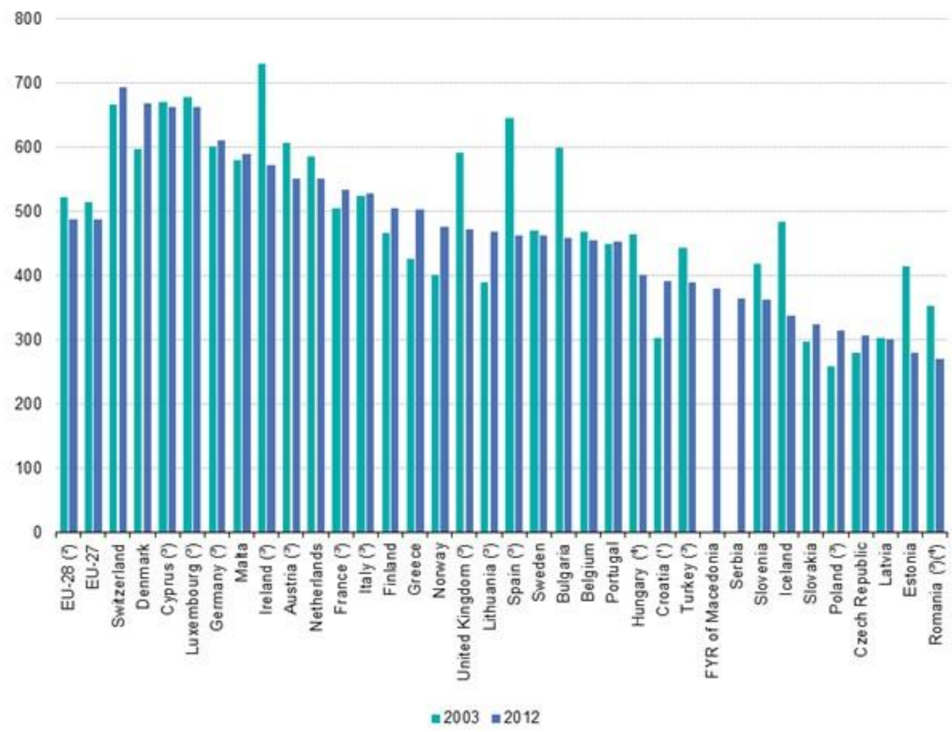
During the last 25 years, a huge change has happened in the way municipal waste is managed. Many countries (see, for instance, data for Germany in Figure 1.19) have reduced the production of unsorted residual waste, thanks to the separate collection of recyclable fractions, such as paper, glass and plastics. Also, organic waste collection schemes were 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.



Data source: Eurostat, 2014

Figure 1.19. Development of the quantities of certain waste fractions in Germany from 1990 – 2010

Figure 1.20 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 1.20. Municipal waste generated by country in 2003 and 2012 in kg per capita and year, 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, UK or Italy). Under 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 1.21 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 has increased until the year 2007. The decrease of 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.

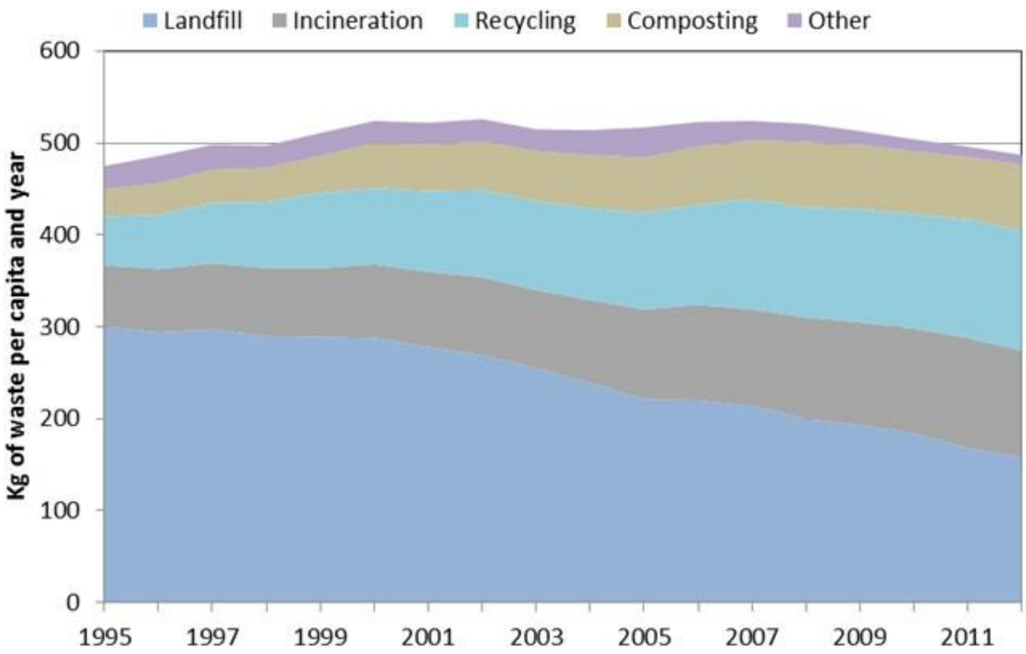


Figure 1.21. Municipal waste by type of treatment, EU-27 in kg per capita[12] (Data from Eurostat, 2014)

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, it is banned to landfill any combustible waste, and Belgium, Austria and Germany banned the landfilling of any untreated waste. As a consequence, these countries are not landfilling any municipal waste (see Table 1.4).

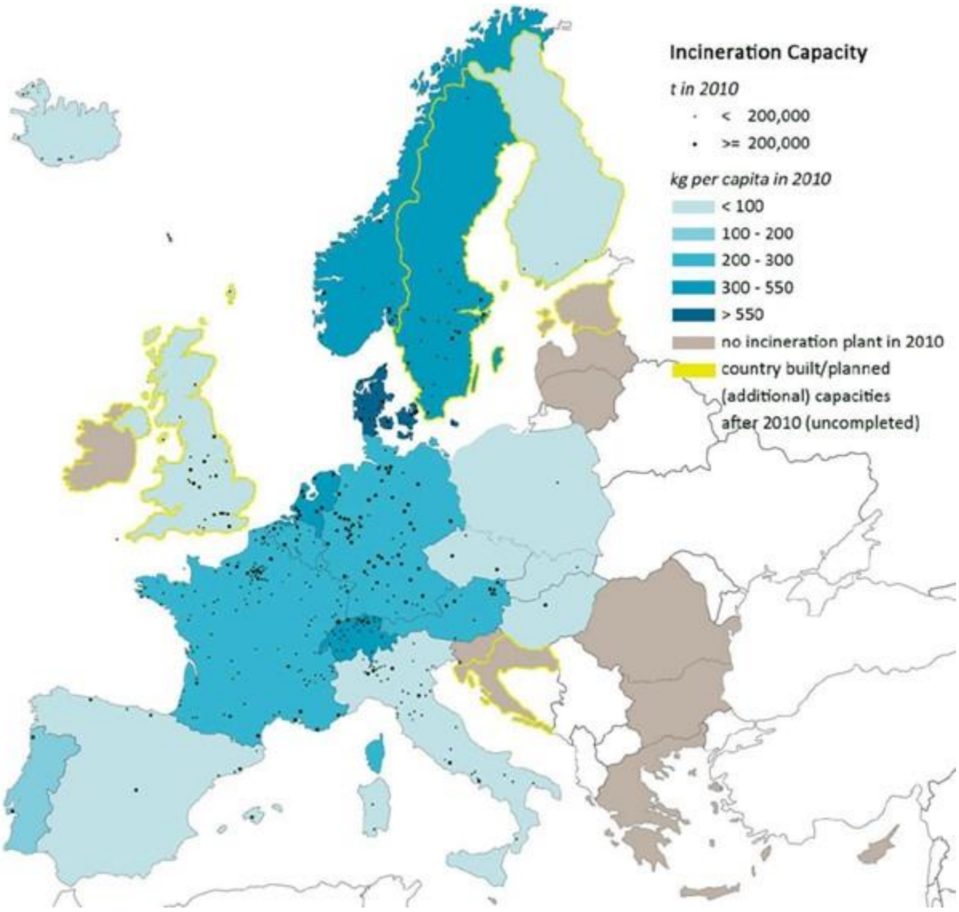
[12] As Croatia is Member State of the European Union from 1 July 2013 it is therefore not included here.

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 high content of biodegradables Since 2006 in Flanders for combustible household waste and industrial / commercial waste (exceptions possible until 2015) Since 2007 throughout entire 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 an early political aim and investment schemes, while the others have similar evolution but apparently delayed. The geographical disparities in Europe are quite evident and reflect the level of economic development, the level of investment in environmental policies, as well as the different historical approaches in waste management. Wilts and von Gries (2014) published recently an ETC/SCP[13] Working Paper where the capacities for municipal waste management in Europe were analysed. Most of European countries have an incineration capacity of less than a quarter of their municipal solid waste generation, but in some specific regions there is 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 1.22.

[13] European Topic Centre on Sustainable Consumption and Production



Source: Wilts and von Gries (2014)

Figure 1.22. Incineration capacity in Europe and incinerators

Figure 1.23 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 still some improvement can be done in other treatments. In blue, countries with very high landfill rates and still lack of incineration, recycling or composting, are represented. 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)[14]. With Figure 1.23, it is shown how the treatment strategy differs across Europe. This chart also distils where the best practices are most likely to be found. Countries like Germany, Denmark, Netherlands, Sweden, etc. have applied a zero landfill policy very successfully during the last ten to twenty 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.

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

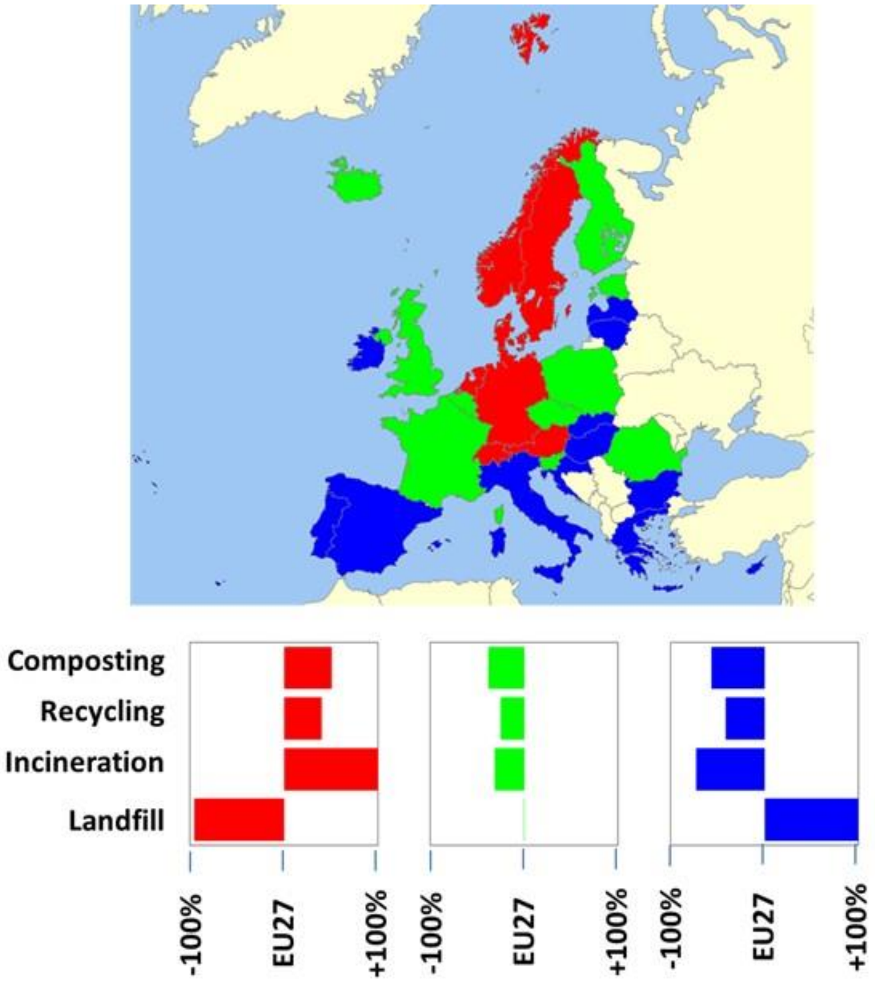


Figure 1.23. Geographical distribution of waste treatment practices, compared to EU-27 average. Colour classification was done based on the total waste landfilled per capita. (Data from Eurostat, table: env_wasgen, 2013)

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 on waste treatment compared to the European average are those with an on average higher municipal waste generation. That can be seen in Figure 1.24, 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 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 much higher landfilling rate (390 kg/yr 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.

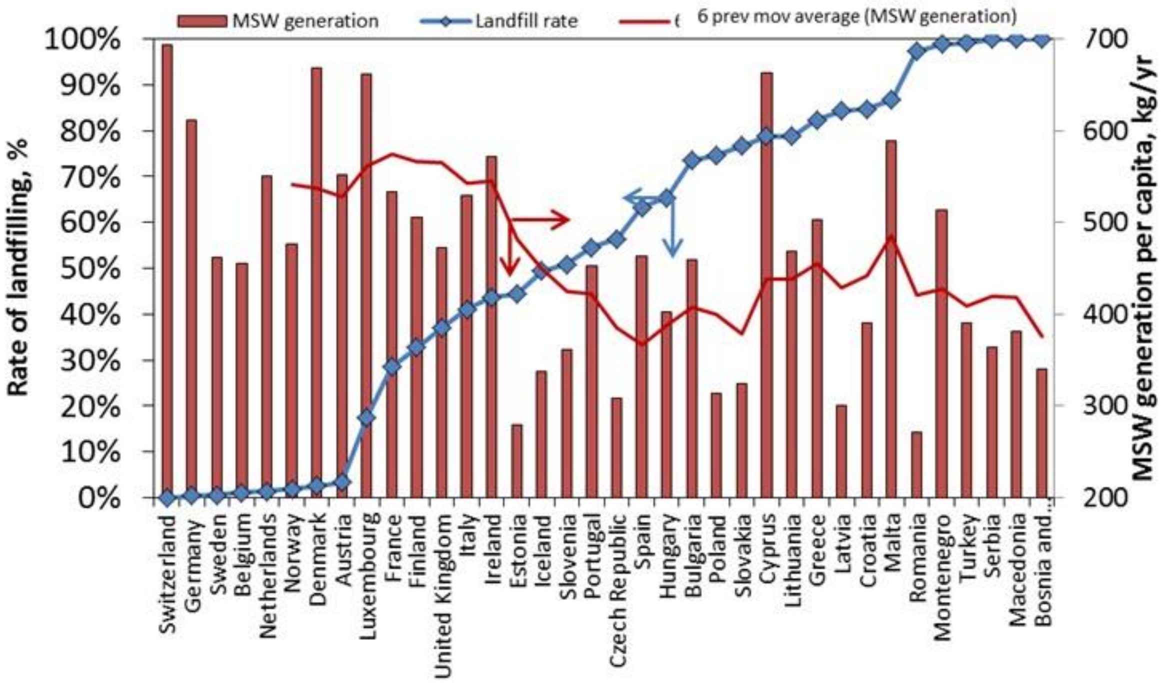


Figure 1.24. Rate of landfilling and MSW generation in 2012 for European Countries. The red line plots the average of the six previous values of MSW generation (moving average) (Data from Eurostat, 2013)

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 up (see Table 1.5). In general, except for some exemptions, the recovery and recycling targets have been achieved. The packaging waste separately collected is 159 kg per capita per year and has been kept constant in the last ten years. 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 re-use 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 re-use 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 re-use. This may be the main reason for disparities on 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 re-use 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 are accounted for the year 2012 in Europe according to Eurostat, which is 34 % of the total waste generated. However, the great part of this waste is inert excavated soil, with 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 is around 40 to 85 % of the total waste generated on site (Rimoldi, 2010). "Clean" concrete waste is barely 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 a main drawback 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 on 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 gases emissions is highly dependent on the transport. These factors contribute to a very different scenario for CDW if compared to other wastes, and require different driving forces (i.e. regulation, taxation, etc.) for best practice implementation.

A cultural misunderstanding of the application of recycled aggregates in concrete is that these aggregates have 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 impact on performance.

Europe consumes around 3 billion tonnes of aggregates (European Concrete Platform, 2007). If the whole amount of CDW is transformed to recycled aggregates, only a 2 % substitution would be achieved (or 17 % if excavated materials are required). In the UK, 25 % of the aggregates market came from secondary sources or recycled materials in 2007 (The Concrete Centre, 2009). Therefore, there are virtually no technical barriers for the maximum possible recycling of CDW. Aggregates from masonry and ceramic wastes, even mixed with concrete, are less applicable, but its volume is certainly smaller and many applications have succeeded. Several showcases around Europe showed more than 95 % CDW recycling (European Commission, 2012), simplified the market barriers to (i) availability, (ii) economics and (iii) acceptability. The profit margin on recycled aggregates also depends on the localisation of the source, which has to be closer than other quarries, and the taxes schemes on 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 is 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		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 of concrete (more than 50 % in most of the cases) and masonry. Gypsum-based materials, timber and metal are also of relevance in the final mass of wastes. Only the volume of concrete wastes is equivalent to the amount of MSW. The mineral fraction of construction waste constitutes category 12.1 of the European Regulation on waste management statistics. In the year 2012, Member States reported the treatment of this fraction as shown in Figure 1.25.

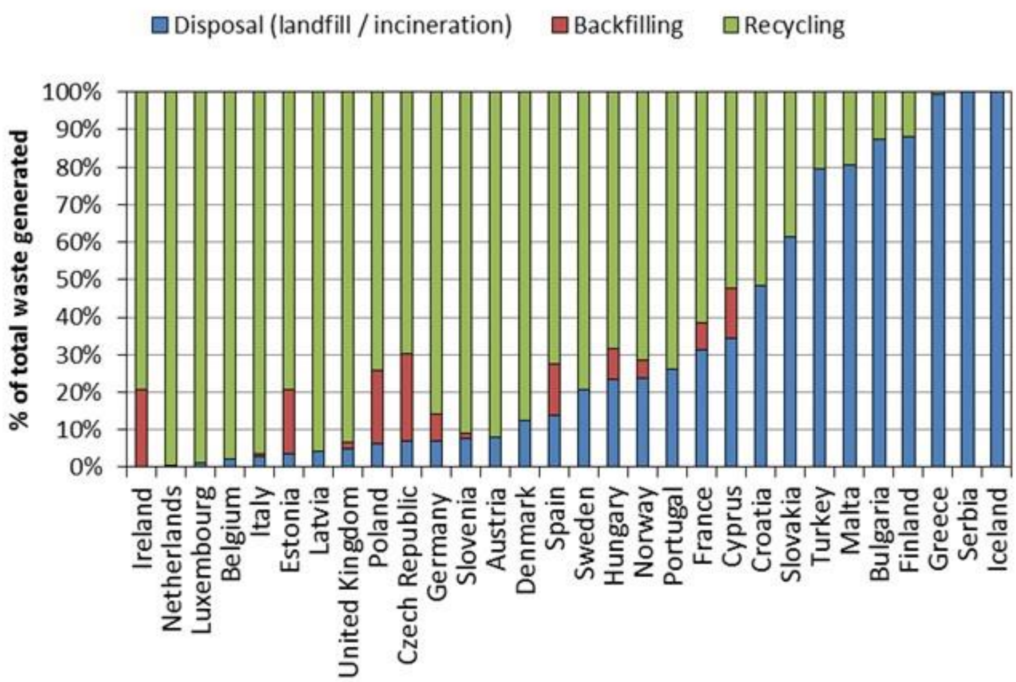


Figure 1.25. Construction and Demolition Waste Mineral fraction treatment in 2012 (Data from Eurostat, env_wasgen, 2013)

As observed, many countries 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 of the text 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 really 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 immunization 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:

- Infection 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 cannot be determined with accuracy due to the different approach in segregation.

Hazardous waste has to be disposed 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 under stoichiometric amount of oxygen for combustion. The process is energetically self-sustained.
- Alternative treatments (usually referred as non-combustion treatments) reduce or eliminate the hazardous component of the waste. Examples of these are:
  - 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, etc.

The suitability of each treatment to each HCW stream is shown in Table 1.8.

Table 1.8. Treatment type per healthcare waste stream

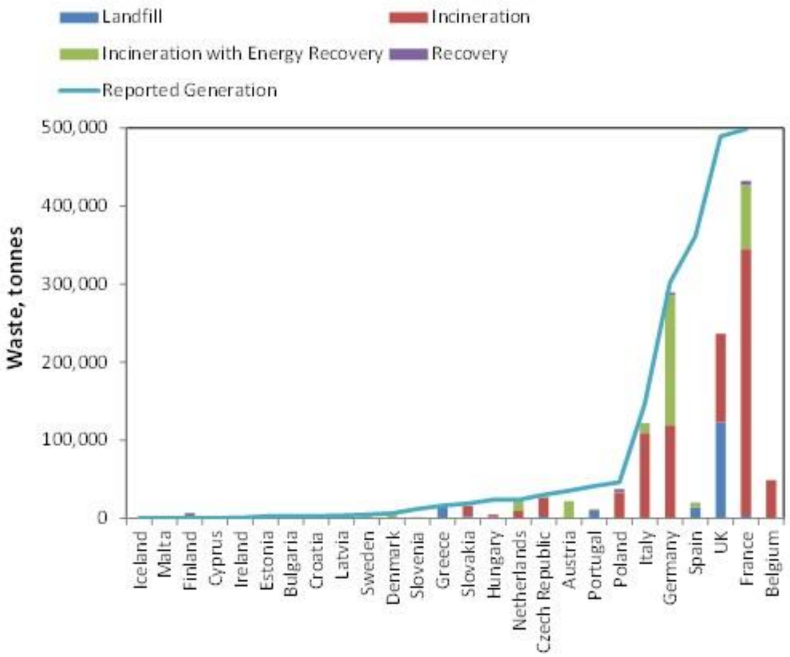
Waste	Code	Treatment
Clinical (chemicals)	18-01-03	High temperature
Clinical (swabs, soiled dressings, gloves, etc.)	18-01-03	Alternative
Sharps	18-01-01	High temperature
Anatomical	18-01-02	High temperature
Offensive (e.g. diapers)	18-01-04	Alternative
Cytotoxic and cytostatic	18-01-08	High temperature (>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 fraction recovered. Current practices in the UK indicate that most of the recyclable waste is not well sorted and fed to the high temperature incinerators as a support fuel to improve the efficiency.

Eurostat in 2014 reported the data shown in Figure 1.26 for the year 2012. The level of reporting of Member States for HCW seems heterogeneous and the quantities per capita are not comparable. Generation of waste and treated waste do not match. In total, for the countries reported in Figure 1.26, about 2.7 million tonnes of waste were generated, while 1.4 million tons were reported as treated. Probably, the difference relies on the different accountability of the MSW-like waste.

a)



b)

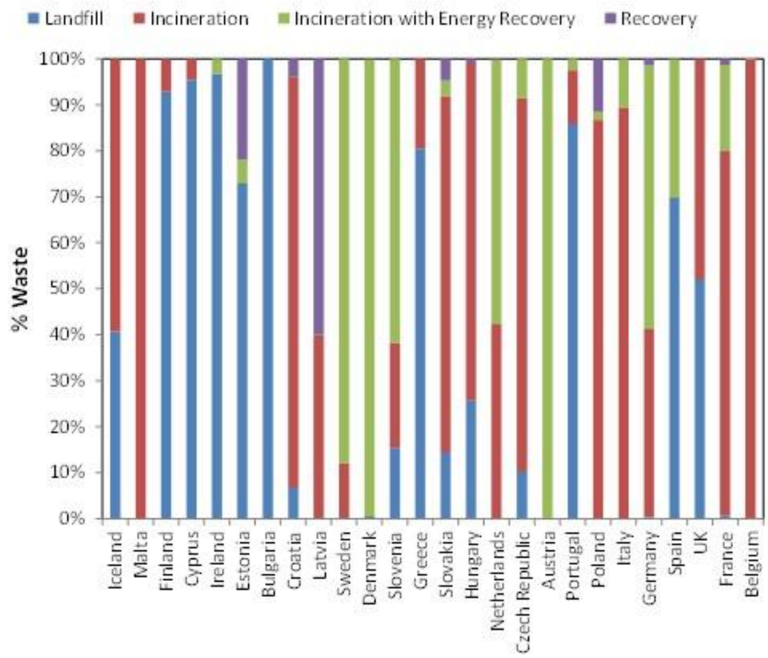


Figure 1.26. Healthcare waste generation and treatment in Europe (a) in tonnes and (b) as a percentage of the total. (Data from Eurostat, env_wasgen, 2013)

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 bed/day	6.40 kg/occupied	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 gases emissions, eutrophication and eco-toxicity in waters from leachate in the case of landfilling, or resource depletion, and acidification and eco-toxicity 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 re-use 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 disposal options such as landfill and incineration do not represent best practice, it is important to quantify the impacts associated with them in order to quantify the environmental benefits realised through best practice implementation. 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 "risk" associated with environmental aspects.

Table 1.10. Main activities in the waste management sector, and associated environmental aspects, pressures and credits

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 re-usable 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/ material re-use	- Collection and transport operations - Heating and lighting of distribution centres - Disposal of non-re-used fraction	- Climate change (GHG emissions) - Air pollution - Fossil resource depletion - Traffic	- Avoided abiotic resource use - Avoided fossil energy use - Avoided waste disposal	- Employee safety risks (heavy machinery) - Operational efficiency risks of changes - Cost of infrastructure & machinery repair and upgrade

Service or activity	Main environmental aspects	Main environmental impacts	Main environmental credits	Main environmental risks
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 affects 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 and eco-toxicity 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 recycling	- Machinery operations - Energy consumption - Infrastructure construction and maintenance - Equipment production - Transport of materials - Raw material substitution	- Climate change (GHG emissions) - Air pollution - Fossil resource depletion - Traffic - Finite resource depletion	- Avoided abiotic resource use - Avoided fossil energy use - Avoided waste disposal	- Employee safety risks (heavy machinery) - Operational efficiency risks of changes - Cost of infrastructure & machinery repair and upgrade

Service or activity	Main environmental aspects	Main environmental impacts	Main environmental credits	Main environmental risks
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 eco-toxicity) - 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 and eco-toxicity	- 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 and water	- Land occupation - Climate change (GHG emissions) - Water pollution (leachates) - Eco-toxicity	-	- Major clean-up costs borne by municipality - 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 1.27 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/yr, 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 1.28. 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.

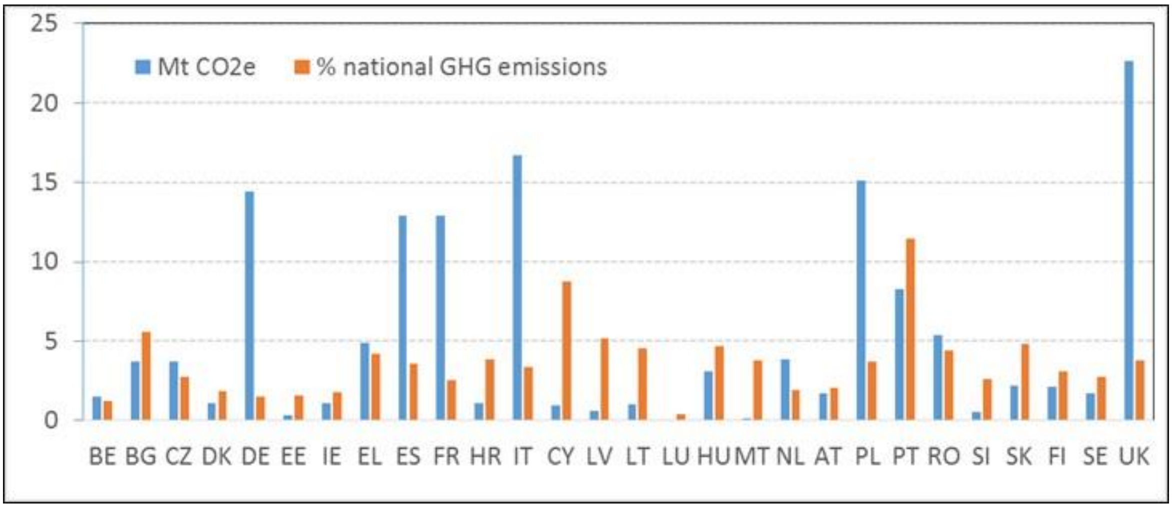
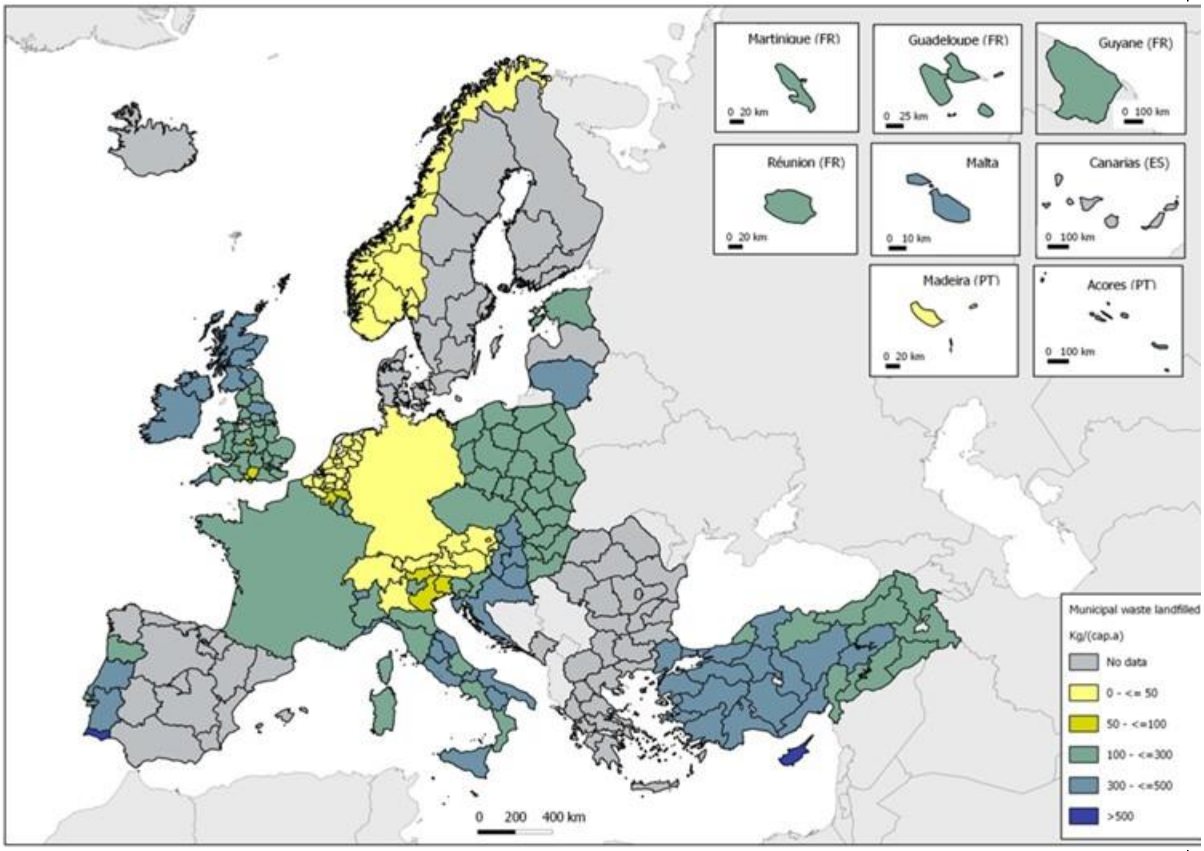


Figure 1.27. GHG emissions arising from waste management across the EU-28 in 2011 (left bars, blue), and the share of national emissions they represent (right bars, orange) (Based on data from Eurostat, 2014)

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 e.g. waste collection and transport emissions, electricity consumption for waste handling and processing, 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 1.28. Quantity of municipal solid waste per capita land-filled 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 compounds (NMVOC) in 2011. The waste sector accounts only for a trivial share of NO_x and SO_x emissions (Eurostat, 2014).

Figure 1.29 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).

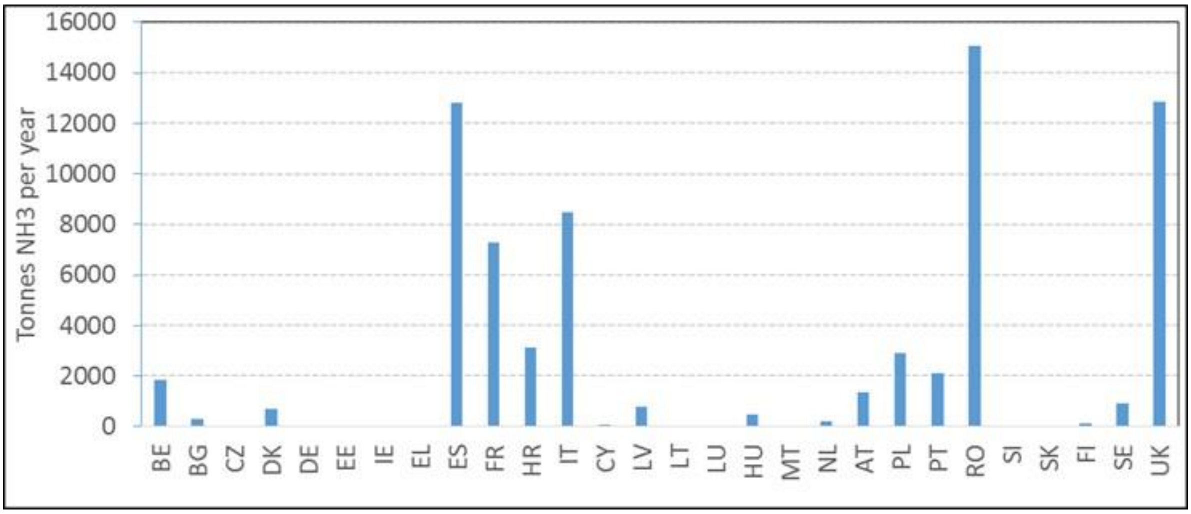


Figure 1.29. Ammonia emissions arising from waste management across the EU-28 in 2013. (Based on data from Eurostat, 2014)

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 1.30) 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.

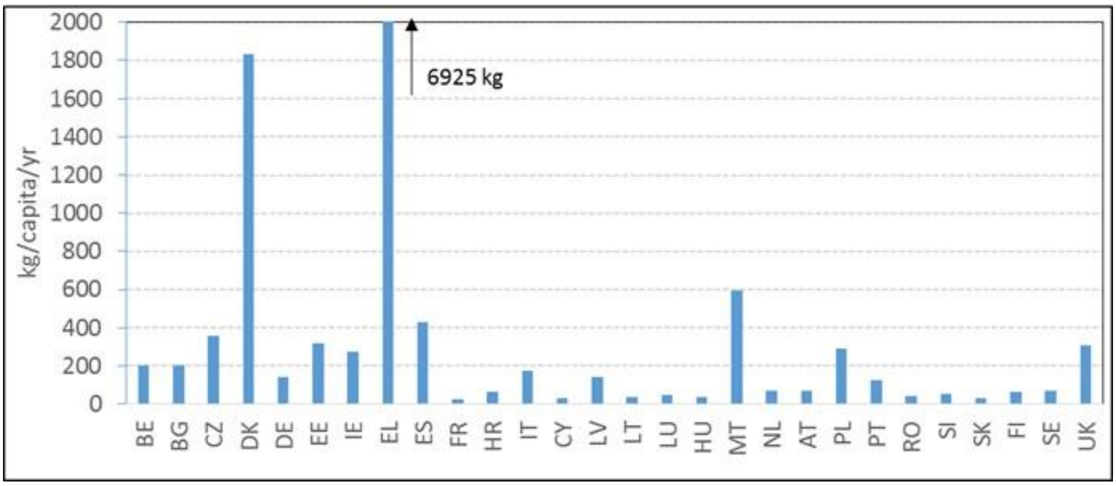


Figure 1.30. Hazardous waste generation across EU Member States in 2012 (Based on data from Eurostat, 2014)

Some important emissions with respect to eco-toxicity 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 licenced) 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, PCBs 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 %) of samples in 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 1.31). 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 plasticizer 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

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Figure 1.31. 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 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 microplastic (< 4.75 mm) and meso- and macroplastic (> 4.75 mm), and proposed various mechanisms of microplastic loss from the sea surface that include ingestion into the food 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 by-passes regulatory controls on waste handling and emissions. This can be a particular problem for e.g. waste oils and white goods, 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 total environmental impacts attributable to EU consumption (Tukker et al., 2006). Many of these may arise outside of 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 un-used 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 1.32 displays 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 particularly high DMC per capita, all above 25 tonnes per capita per year. Re-use and recycling of materials can significantly reduce DMC.

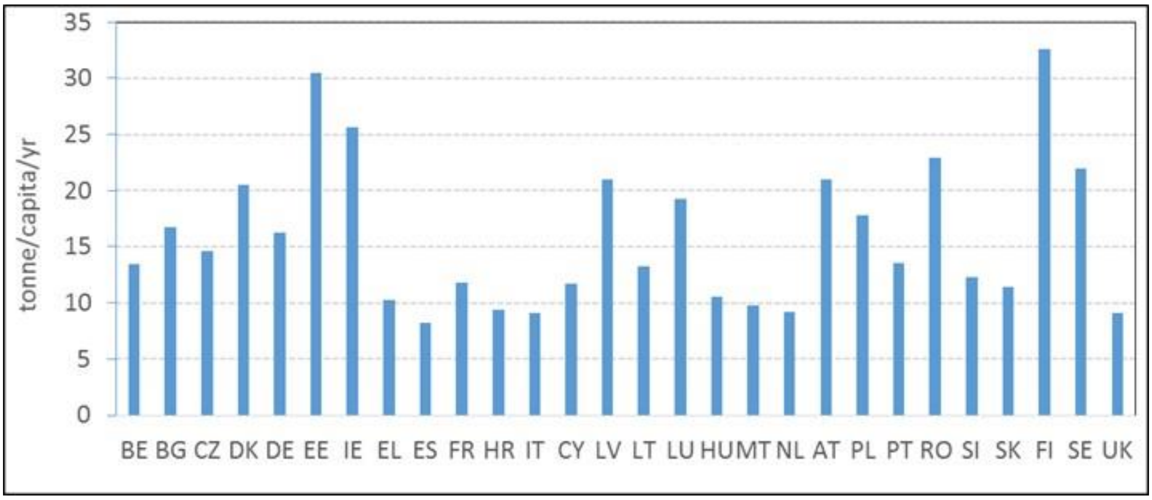


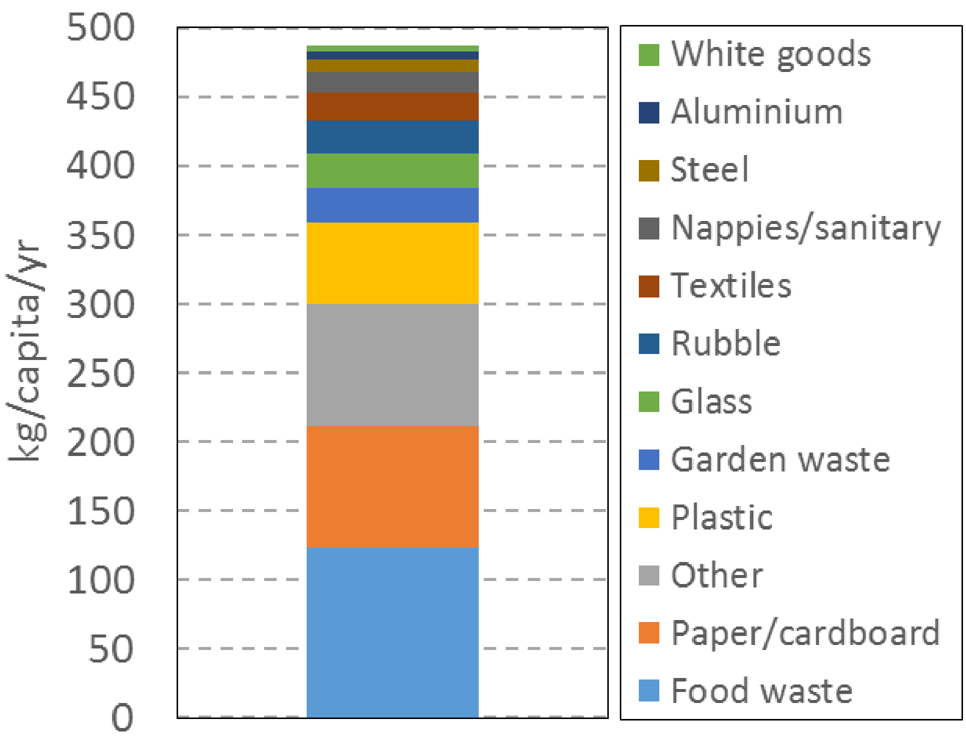
Figure 1.32. Domestic Material Consumption (DMC) per capita across the EU-28 in 2012. (Based on data from Eurostat, 2014)

Table 1.12 below summarises some of the major environmental burdens, expressed as environmental impact potentials used in LCA, arising from the extraction and primary processing of a selection of major raw materials. These burdens can be avoided through waste prevention, including re-use 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 1.33 presents the quantities of different materials sent for disposal or re-use by an average EU citizen over the course of one year. On average, each EU citizen generates over 490 kg 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.

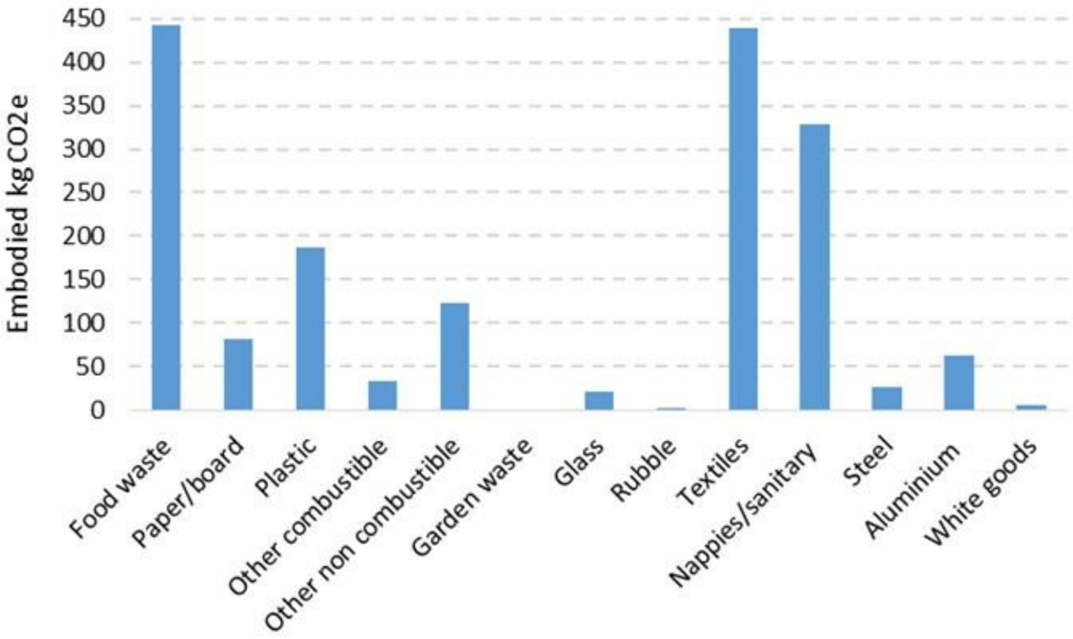


Source: Derived from data in Eurostat (2014).

Figure 1.33. Typical composition of MSW in the EU, expressed as mass of different fractions generated per person 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/yr, 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 1.33). 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, re-use and recycling in order to effectively reduce the environmental burden of waste (management).

Although insufficient data are available to undertake 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 and 2 billion tonnes of waste (FAO, 2011). The 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 1.34. 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 EUR 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 yr). 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 yr).

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 throughout this document. Below two short examples are listed.

One example of a largely avoidable waste stream, and 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), transport of bottled water incurs environmental impacts via energy consumption, GHG emissions, air emissions 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 under legal requirements, in other countries eateries are not required to provide tap water on request. In France, it is 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 spices, the carafe of water belongs to the meal and the guest cannot be charged for this separately (Die Zeit, 2013).

Waste prevention is particularly important for the voluminous CDW waste 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 as "inert", as it is characterised by the 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^2a
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 quick wins, and the large impact of landfill operations. In the Netherlands, the recycling rate of CDW is around 95 %. However, this fraction can only substitute 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 design 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 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 will be 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 point of use/disposal to re-use or treatment locations. From a life cycle perspective, transport of waste may give rise to significant GHG and NO_x emissions, and result in significant fossil resource depletion and traffic. 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 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

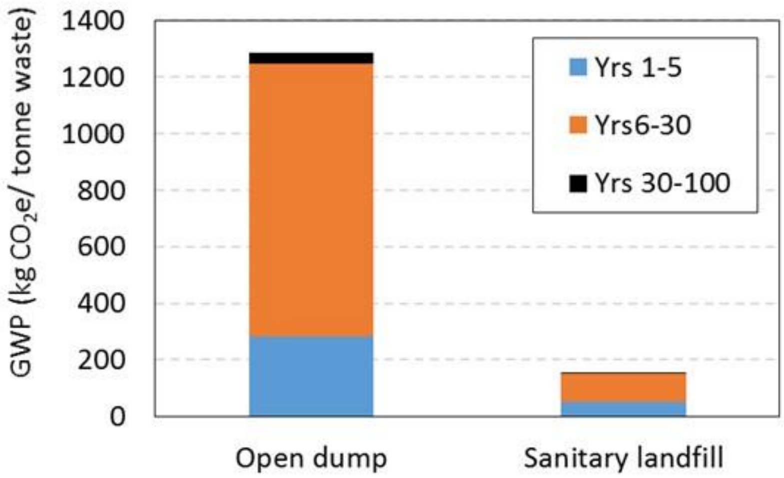
Landfill and incineration are long established as the most common treatment options for unsorted MSW or residual waste, and are associated with various environmental impacts that can be minimised through good design (specified in the Waste Treatments BREF: JRC, 2006), but more importantly through measures to minimise waste sent to landfill or incineration, as will be 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 landfill varies considerably depending on the landfill design and management and the type of material going into it. 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, which have the lowest environmental impacts, are those which 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 1.35). 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

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 (VOC and NO_x) – Odour nuisance
Landfill gas capture and energy recovery	– Avoided fossil fuel combustion burdens – Acidification – Photochemical ozone formation
Leachate generation	– Eutrophication – Eco-toxicity – Waste water treatment plant burdens



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

Figure 1.35. 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 and eco-toxicity 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 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 (volatile organic compounds 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 – Eco-toxicity – 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 ^{o}C and selective catalytic reduction, and accounts for a trivial 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 inputted MSW, and may be used in construction, for road construction, etc. (Defra, 2013). Pollution control residues including fly ash, reagents and wastewater can represent 2-6 % of the weight of inputted waste, and can contribute towards toxicity effects depending on their management. Metals representing 2-5 % by weight of inputted materials may be recovered from bottom ash and re-smelted.

In terms of GWP, incineration with energy recovery can perform comparatively well 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 14-27 %, 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 in e.g. cement plants (Galvez-Martos and Schoenberger, 2014). In these cases, emissions of mercury, NO_x and dioxins/furans, among others, are uncontrolled or less tightly controlled, respectively, leading to greater toxicity effects.

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 examples Figure 1.36. In reality, a wider range of counterfactual fates may apply to waste that is collected for centralised composting or anaerobic digestion, and in some case 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.

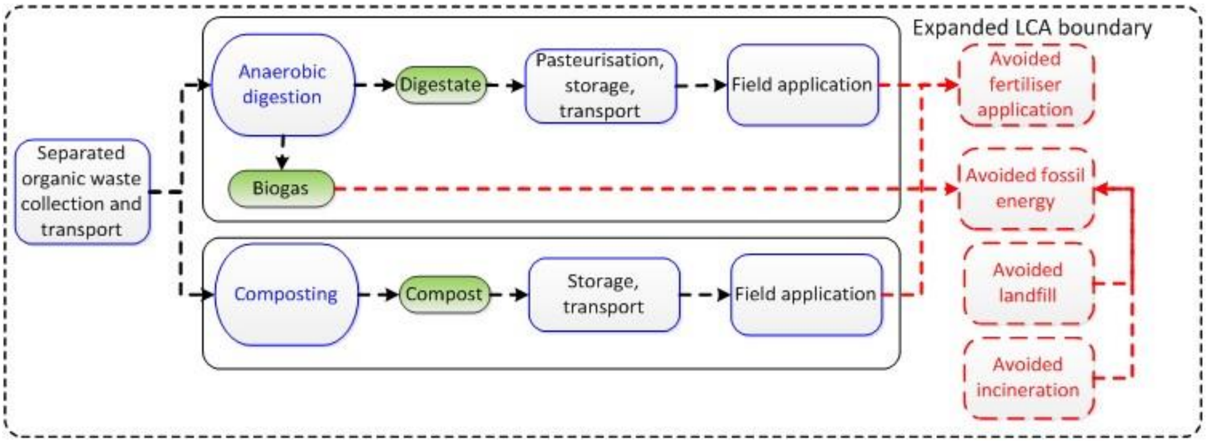


Figure 1.36. Major stages and processes affecting the life cycle balance of organic waste going to anaerobic digestion or composting, in a simplified scenario that assumes counterfactual landfill or incineration is avoided

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).

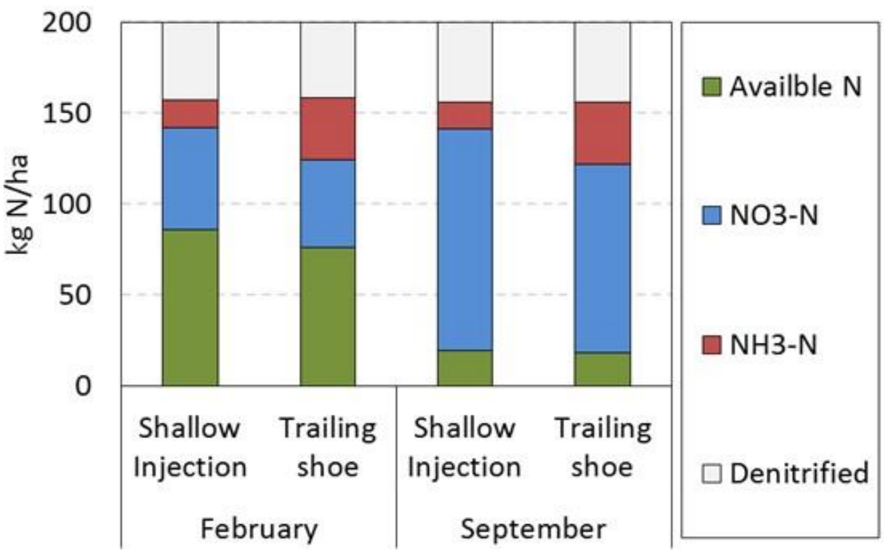
Table 1.16. Main environmental impacts arising from organic waste recycling

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 (volatile organic compounds and NO_x) – Human toxicity (particulates and polycyclic aromatic hydrocarbons) – Avoided fossil fuel combustion burdens
Extracted non-organic materials and combustion ash	– Landfill burdens

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 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.

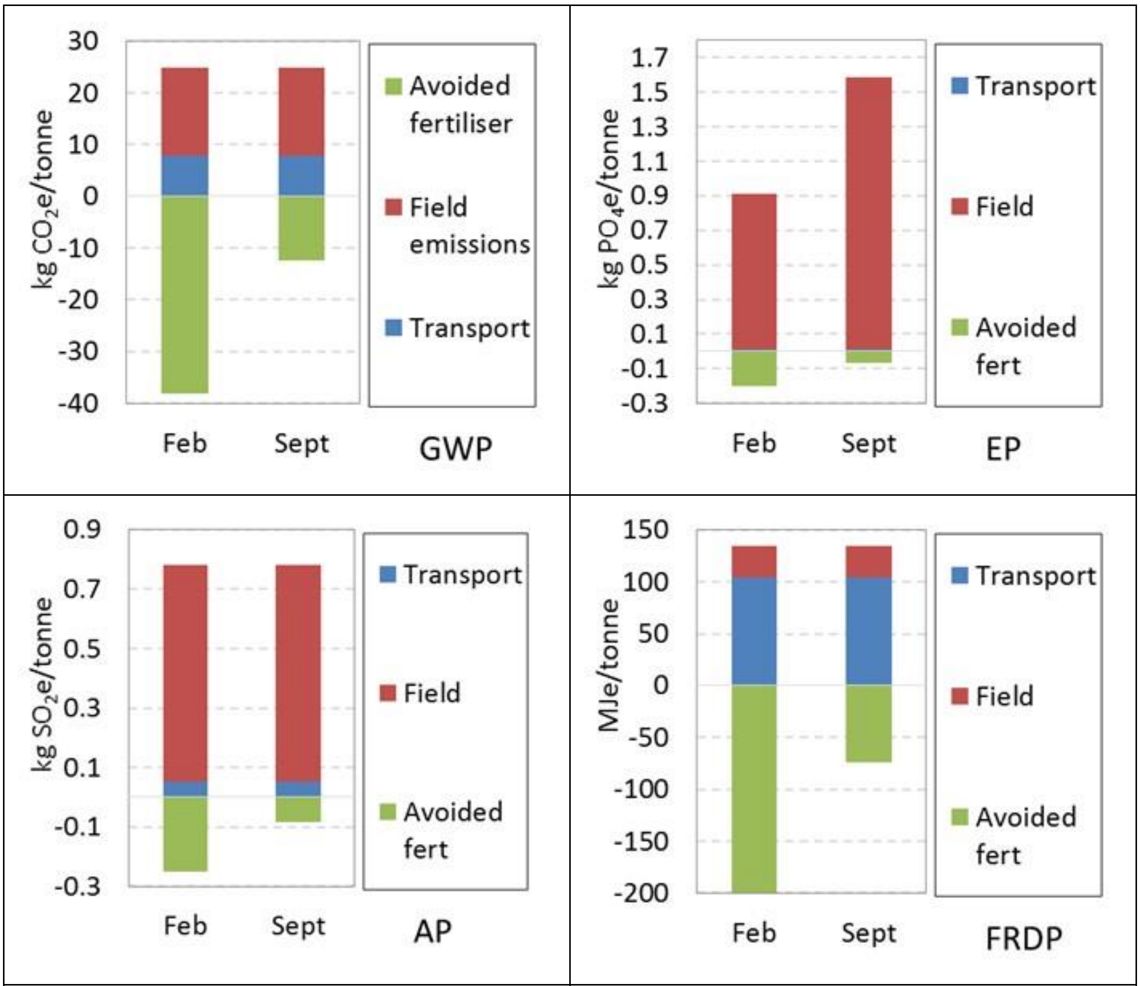
Digestate application to land as a bio-fertiliser is a hotspot for eutrophication and acidification impacts in the AD life cycle, and can sometime results in these impacts exceeding those for otherwise less efficient organic waste treatment options. Figure 1.37 shows the fate of nitrogen (N) applied 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.

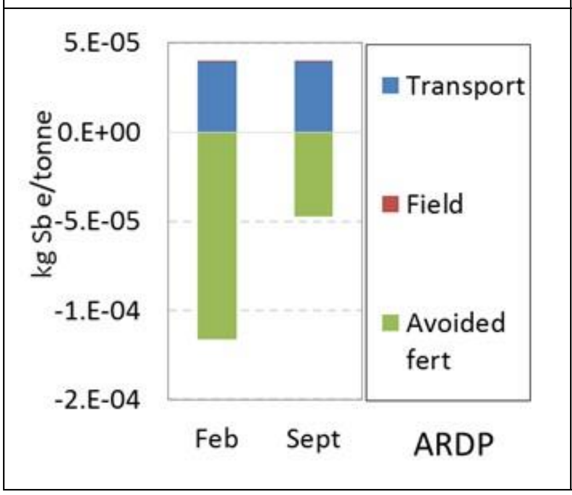


Source: Data from MANNER NPK (Nicholson et al., 2013)

Figure 1.37. 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 1.38. 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 net GWP and fossil resource depletion impacts.





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

Figure 1.38. Environmental balance for one tonne 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 for sanitised landfilling (typical UK landfill with 70 % CH_4 capture), composting and anaerobic digestion of organic waste. These impacts reflect avoided marginal grid (natural gas combined cycle turbine) electricity generation for landfill and anaerobic digestion, and avoided fertiliser manufacture and application for composting and anaerobic digestion. Overall, anaerobic digestion exhibits the best environmental performance, though 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 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, in some MBTs in addition 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 and eco-toxicity (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 fates

	Re-use	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 same product
Source: Data from Defra (2014)

Table 1.19 summarises GHG emissions across alternative fates (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 over 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 – Eco-toxicity (discharges to water)
Material recovery	– Avoided resource depletion (credit) – Avoided raw material processing (credit)
Rejected materials	– Waste disposal impacts

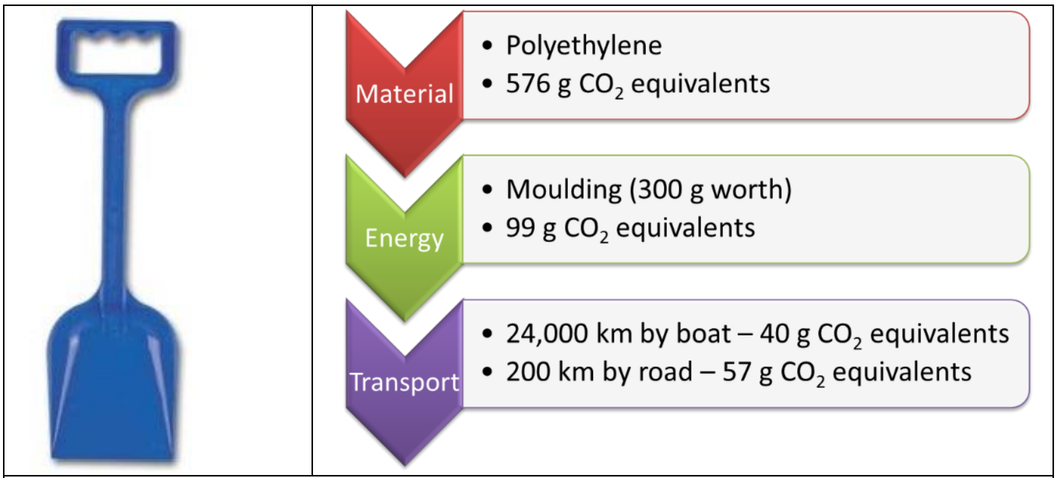
Recycling is usually associated with lower environmental impacts than virgin production for most materials, especially metals with high embodied energy (Table 1.21). For example, recycled aluminium gives rise to energy and air pollution impacts 75-90 % lower than virgin aluminium, and avoids most of 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	kg CO_2e/t	390	1,080	990	1,200	1,080	3,300

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

The effect of recycling compared with landfill or incineration is illustrated with the following example of a plastic spade carbon footprint, below.



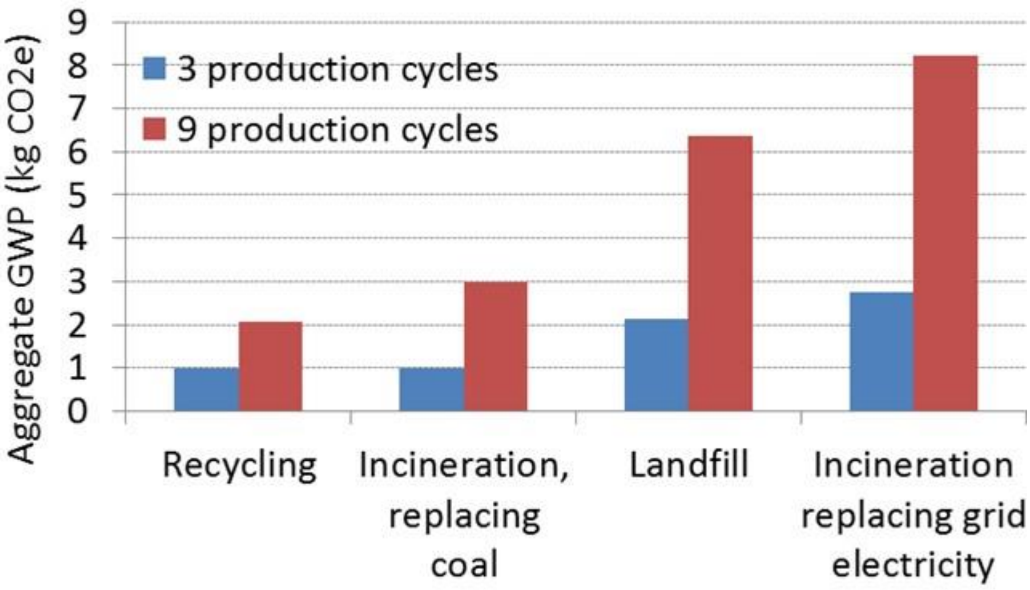
Source: Based on own elaboration and Ecoinvent (2014).

Figure 1.39. Greenhouse gas emissions from the manufacture and transport of a polyethylene spade manufactured in China

Landfill, incineration or recycling of the polyethylene plastic in the spade give rise to GHG emissions of 0.03, 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 1.40 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 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 by some margin the lowest carbon footprint of all the options considered.



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

Figure 1.40. 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 1.40 is the poor performance of incineration with electricity generation, with a higher GWP impacts 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 lower net carbon emission to the atmosphere. However, landfill also exerts a wide range of other environmental impacts that must be considered alongside these GWP results. The key message is that, in order to achieve 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 1.40). Although as the number of recycling loops increase, the comparative efficiency of recycling continues to improve beyond all other options.

1.4.7. Product re-use

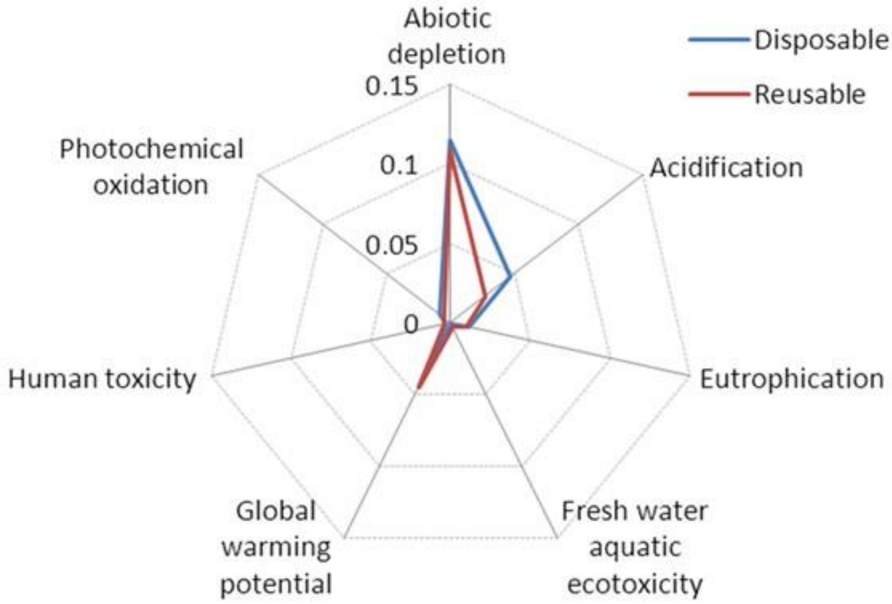
Waste management organisations can play an important role in encouraging and facilitating product re-use, diverting potential waste away from their own operations. Such diversion, if managed appropriately and associated with effective preparation for re-use, can play an important role in waste prevention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

In general, the environmental balance of product re-use 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 re-use

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 – Eco-toxicity (discharges to water)
Avoided production	– Avoided resource depletion (credit) – Avoided raw material processing (credit) – Avoided manufacturing and transport burdens (credit)

In some cases, re-use 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 re-usable nappies. The UK Environment Agency compiled a report in 2008 looking at the environmental balance of disposable and re-usable nappies, considering average UK landfill/incineration mix for disposable nappies and average UK wash temperatures, loads, share of tumble-dried washing, etc., for re-usable nappies. The results indicated only a marginal advantage for re-usable nappies owing to high energy demand for washing and drying (Figure 1.41), 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 re-usable 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 1.41. Environmental profile of disposable and re-usable 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)[15]. This value represents less than 1 % of the total sector (around 45,000 organisations in NACE division 38 and 39)[16]. These companies are mainly classified as SMEs, although many of them may belong to bigger companies (see Figure 1.42a). The proportions of waste management activities are equally represented in the EMAS register (see Figure 1.42b), i.e. collection, treatment and recovery, with a very low proportion of remediation companies.

[15] 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.

[16] 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)

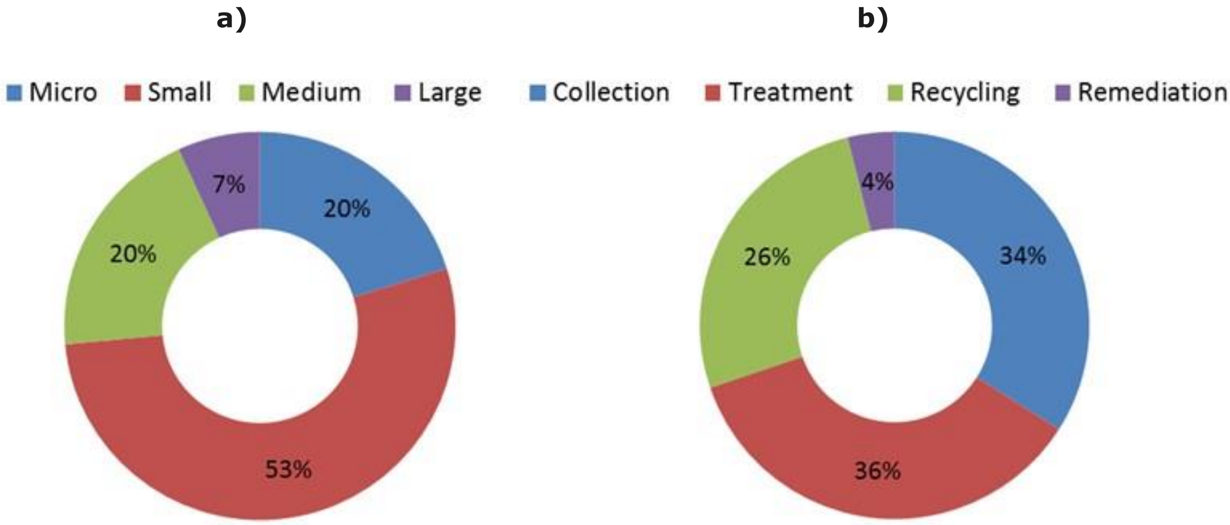


Figure 1.42. Percentage of EMAS registered companies in Europe per site (a) and per registered activity

Table 1.23 presents the number of the EMAS registered sites and companies in the different European countries; from the same table it is shown 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

Likewise, 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 has registered its waste collection activities for non-hazardous and hazardous waste and also any recovery activity that they may undertake. Therefore, Table 1.24 shows the number of registrations covering each activity per European country, but the total sum of these values would be much higher than the real number of registrations.

Table 1.24. Number of EMAS registrations covering waste main 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 in the last few years a large increase 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 in order to set higher standards of environmental performance. Within this understanding, the background document for the EMAS SRD on Best Environmental Management Practice in the Waste Management Sector does not only address organisations implementing EMAS or ISO 14001, but the activities of all European waste sector companies and waste authorities wishing to improve their environmental performance.

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2. Cross-cutting issues

2.1. Scope

Looking at the current economic system (see Figure 1.9), 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 re-use, recycle and recover the waste materials. Following the overall scope of this report, the cross-cutting issues are those concerning all of 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 3, 4 and 5).

2.2. Techniques Portfolio

The focus is laid on the development of a waste strategy. This strategy is, based on a profound analysis of the waste situation for a given municipality, city, county or region which should include the knowledge of the quality and quantity of as many as possible waste streams. The waste strategy could also be called a waste management plan which includes waste management targets in terms of rates for waste prevention, re-use, 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 more resource-efficiency and a circular economy. The efficient collection of the different fraction is also part of it. In the following chapters, for the mentioned three waste groups, 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 to certain waste streams and it may happen that it is not obvious which of those is the most environmentally friendly or most sustainable. Then, it is adequate to use life cycle considerations in order to identify the best option or to justify the selected one (see section 2.4).

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 (section 2.5).

2.3. Best Environmental Management Practices for Integrated Waste Management Strategies

Description

Waste management deals with a considerable number of different waste streams and a multitude of processes, including MSW (Figure 2.2) but also various hazardous wastes, construction and demolition waste (chapter 4) and health care waste (chapter 5).

Integrated waste management strategies should be guided by the well-documented waste hierarchy (Figure 2.1), prioritising prevention, minimisation and re-use as the most sustainable options, followed by recycling, with energy recovery and disposal as the least sustainable options. In some cases, more detailed evaluation of options through life cycle assessment (LCA) may be required to identify options with the best environmental profile (see BEMP on LCA of waste management options).

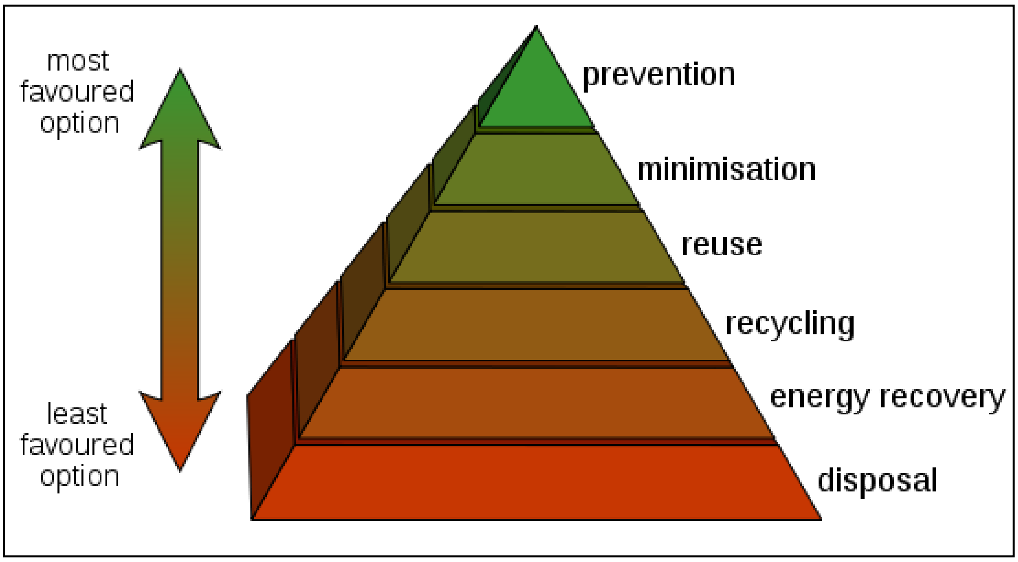


Figure 2.1. Waste hierarchy according to the Waste Framework Directive 2008/98/EC (Source: wikipedia (https://en.wikipedia.org/wiki/Waste_hierarchy)

For the development of an integrated waste management strategy, the quality and quantity of each major waste (mass) stream needs to be known, and alternative management options compared. Therefore, whilst subsequent BEMPs address specific aspects of waste management for the major waste streams, this BEMP focuses on the prerequisite data monitoring and approach necessary to develop a coherent and overarching waste management strategy at the municipality level. Thus, this BEMP is primarily targeted at waste authorities with control, or at least significant influence over, waste management strategy at the local or regional level – primarily local authorities.

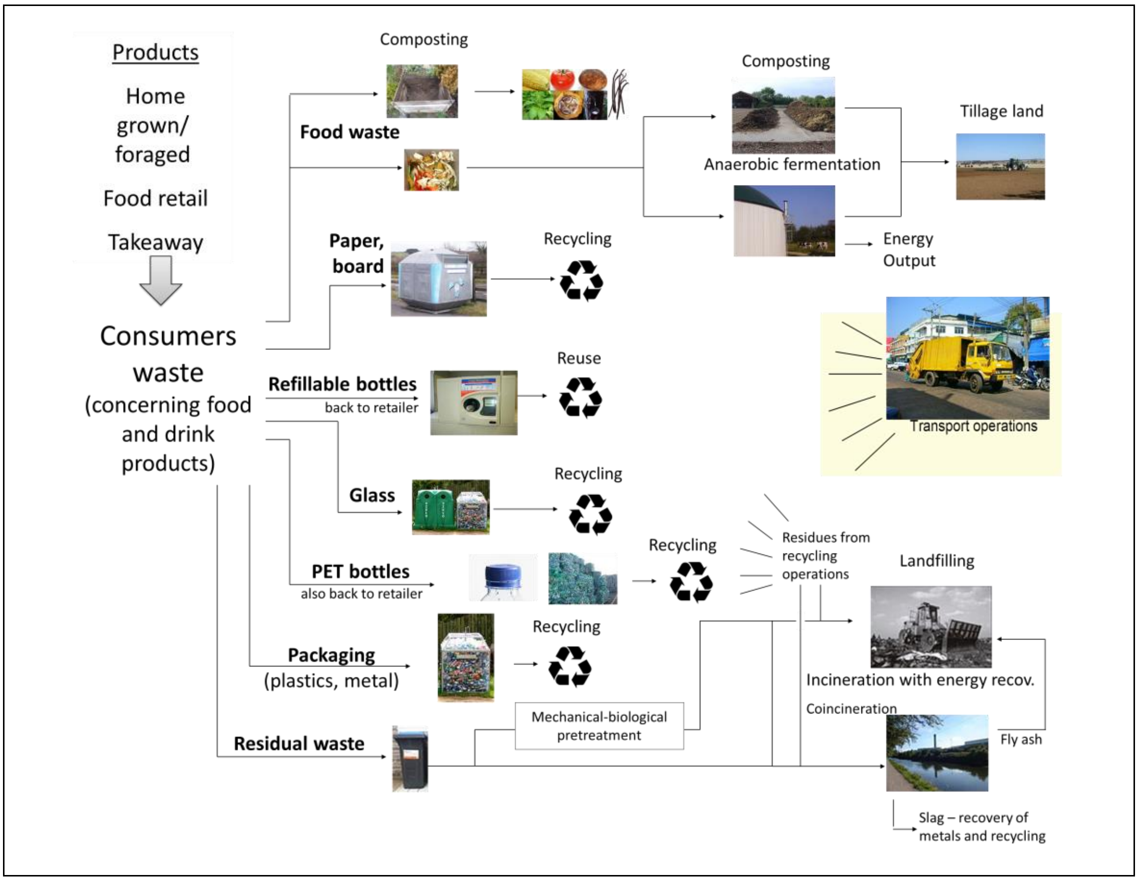


Figure 2.2. Major waste stream flows within a municipality

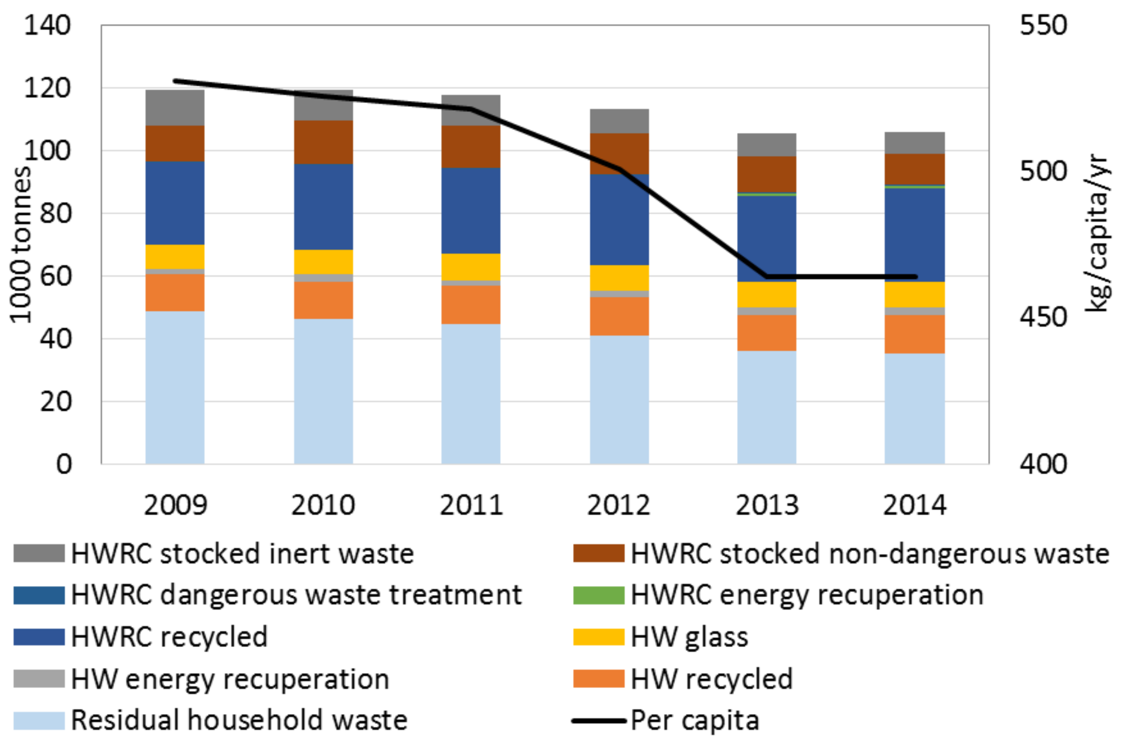
Data collation

Table 1.3 in the introductory chapter of this document specifies the list of waste categories to be considered according to the European list of wastes. For each waste stream, the total quantity generated within the waste catchment must be known, and also expressed per capita. The proportions of each waste stream going to alternative fates, including re-use, recycling, anaerobic digestion, landfilling and incineration should be recorded. Table 2.1 provides an example of relevant waste streams to be managed by a municipality, city, county or region. It is just an example and further waste streams may be added.

Table 2.1. . Example for the documentation of different waste streams which are managed by a municipality, city, county or region

		[tonnes/yr]	[kg/cap x yr]	Fate/ treatment
Recycables	Waste glass			
	Waste paper			
	Scrap metal			
	Waste tyres			
	Waste plastic and packaging composites			
	Textiles			
	Shoes			
	Green cuttings from cizitens			
	Green cuttings from public parks/gardens			
	Leaves from public parks, gardens and streets			
	Bio waste			
	Waste wood			
	Waste mineral oil			
	Waste edible fat			
	Windows/ flat glass			
	Aluminium and other non-ferrous metals			
	Waste cable			
	Polystyrene			
	Waste polyurethane foam cans			
	Waste toner cartridges			
	Waste electronic and electrical equipment (WEEE)			
	Other recyclables such as cork, CDs, PV panels etc.			
Residual, bulky, hazardous and commercial waste	Residual waste			
	Bulky waste (without wood)			
	Hazardous waste			
	Street sweepings (not recycable)			
	Commercial (household-type) waste			
Total municipal waste				
Construction and demolition waste (CDW)	Excavation earth			
	Demolition waste			
	Construction waste			
	Road construction waste			
Total CDW				
Total waste				

For each of these streams, such as plastic waste, paper/cardboard, glass, bio-waste and green cuttings (high quantities) and hazardous waste (pesticides, waste paints, waste solvents, waste mineral oil, etc.), the method of quantification has to be defined, not only for residual waste but also for the different streams of recyclables. Management of every waste stream can then be reported on and developed. Figure 2.3 provides an example of waste stream accounting according to management over time. More disaggregated breakdowns of waste streams and fates should be possible. Only once these data are collated meaningful benchmarking of performance can be undertaken, as required to compare performance with best practice described in subsequent BEMPs.



Source: SYBERT (2015)

Figure 2.3. An example of quantification of different waste streams over time, divided into Household Waste (RHW) and waste collected at Household Waste Recovery Centres (HWRC)

Developing integrated strategies

Technical and economic instruments as well as psychological aspects of citizens' behaviour, such as raising awareness, should be taken into consideration. Long-term planning is required as the implementation can only be achieved step by step, i.e. waste stream by waste stream. So, prioritisation is needed and the start should aim at the most relevant waste streams whereby the relevance comprises quantity and hazard.

So, a waste strategy should not consist of a one-sided approach but of an appropriate mix of different approaches, including the technical, economic and psychological aspects but also producer responsibility (European Commission, 2003; OECD, 2007; Milankov, 2013). This also includes the effective marketing of recycled waste streams.

On the local or regional level, the possibilities to implement waste prevention measures are limited. Nevertheless, this option has to be considered for the different waste streams and concrete measures can be set up such as mobile dish washers, subsidies for the use of reusable nappies, installation of platforms to exchange goods. The BEMPs on waste prevention provide more details (section 3.7). The BEMPs on horizontal approaches (section 3.5), on re-use (section 3.8.1) as well as on producer responsibility can be seen to be complementary to waste prevention and should also be considered for setting up the general framework of an integrated waste strategy.

The defined general strategy directly leads on to the approach for collecting the different waste streams. For instance, concerning bio-waste, technical issues concern the manner of collection (sections 3.9 and 3.11), e.g. weighing the bio-waste, and the technique to treat it, such as anaerobic digestion. As an economic incentive for bio-waste separation, no additional fee is charged, and citizens are informed by campaigns and from time to time e.g. by means of flyers explaining why the separate collection and anaerobic treatment is the best approach for the environment (raising awareness).

When looking for the best way to collect waste fractions, there are different options. One option is to go for the collection of single fractions such as paper/cardboard, glass in different colours, metal tins, and plastic foils or to choose the collection of co-mingled packaging consisting of paper, plastic, composite packaging and metals with subsequent sorting. The costs for collecting co-mingled packaging is certainly lower but the quality of the paper fractions can be lower compared to the separate collection of it. The quality of the separately collected paper fraction (and the revenues for it) can be increased on the other hand, if the paper is source-separated (see Table 2.2). The quality of the paper/cardboard fraction may also depend on the awareness of the citizens and the used sorting technology. So, there may not be one single best approach to collection. New developments have to be considered, such as the Dutch 'Conversed Collection'[17] and the aforementioned combination of different instruments/approaches may lead to best results and efficiency.

[17] The Dutch Conversed Collection system encourages separation of reusable waste fractions at home by offering: (i) more services and facilities for the separation and collection of recyclable waste fractions; (ii) fewer services and amenities for the collection of residual waste (Nijmegen, 2014).

Table 2.2. Dealer prices (ex-factory) for recovered paper in the UK, in GBP/t, July 2015 (EUWID, 2015)

Grade and title	July 2015	June 2015	July 2014
1.02 Mixed papers and boards (sorted)	55 – 70	50 – 70	50 – 60
1.05 Ordinary corrugated board	76 – 88	80 – 90	65 – 75
1.08/1.09 Mixed newspapers and magazines	60 – 75	55 – 75	80 – 93
2.01 Newspapers	75 – 90	70 – 90	80 – 95
2.05 Ordinary sorted office paper*	132 – 137	130 – 135	–
2.13 Multigrade	125 – 132	125 – 132	115 – 130
3.14 White newsprint	175 – 200	170 – 200	170 – 200
3.18.01 White woodfree uncoated shavings	270 – 295	265 – 295	250 – 290
* As of May 2015, the paper grade 2.05 was included in the price comparison for UK, the recovered paper 2.06 and 3.05 are omitted.			

Based on the proper analysis of the existing waste stream quantities and qualities, the waste strategy defines:

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

If required, life cycle assessment may be carried out in order to identify the most effective pathways to meet environmental performance objectives (see Section 2.4).

In order to provide the required transparency to citizens, an annual waste management report should be published providing an overview of the operation of the existing facilities and of the quantities of all collected, processed and recycled waste streams.

On the level of a city, a county or a region, the different municipalities should be part of a common strategy and should be supported, especially with respect to the installation of collection centres.

The County of Aschaffenburg/Germany may serve as an excellent example for the development of an integrated waste management strategy and its systematic implementation. Thereby, the strategy is continuously under optimisation. Table 2.3 shows important milestones.

Table 2.3.Important milestones of 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 bio-waste, later also for bulky waste, close co-operation 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
Re-assessment of the collection and disposal of green cuttings (Morlok, 2013)	2013
Waste sorting analysis of residual waste, bio-waste, paper, light packaging, glass and metal packaging in order to identify additional optimisation potentials (Hoeß and Ammon, 2014)	2014
Latest annual waste management report for 2013 (Aschaffenburg, 2014)	2014

Achieved Environmental Benefit

The implementation of an integrated waste management strategy will certainly be associated with environmental benefits, specifically with the considerable reduction of residual waste and the significant increase in the percentage of recycled waste.

Appropriate environmental indicator

The direct environmental benefit of an implemented integrated waste management strategy is strongly related to the mass of residual waste disposal avoided. Thus, the following key indicator is relevant:

- kg residual waste per capita and year

The most practical definition of "residual waste" from the perspective of WMOs 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 WMO is involved in subsequent sorting (e.g. in sorting plants following co-mingled collection, or in mechanical and biological treatment plants).

The percentage of recycling for the most important waste streams, such as paper/cardboard, glass, plastic waste, bio-waste and green cuttings, also provides a useful indicator of waste authority performance. Residual waste and recycling rates should be based on data for material exiting (not entering) sorting and recycling plants, to account for contamination of recycling waste streams.

Life cycle assessment (LCA) is an important tool to inform waste management strategy and to track progress. LCA indicators specified in BEMP 2.4 are therefore also highly relevant.

Integrated strategies ultimately need to incorporate all indicators relevant to processes undertaken by the waste authority, as detailed for particular processes in subsequent sections of this document.

Cross-media effects

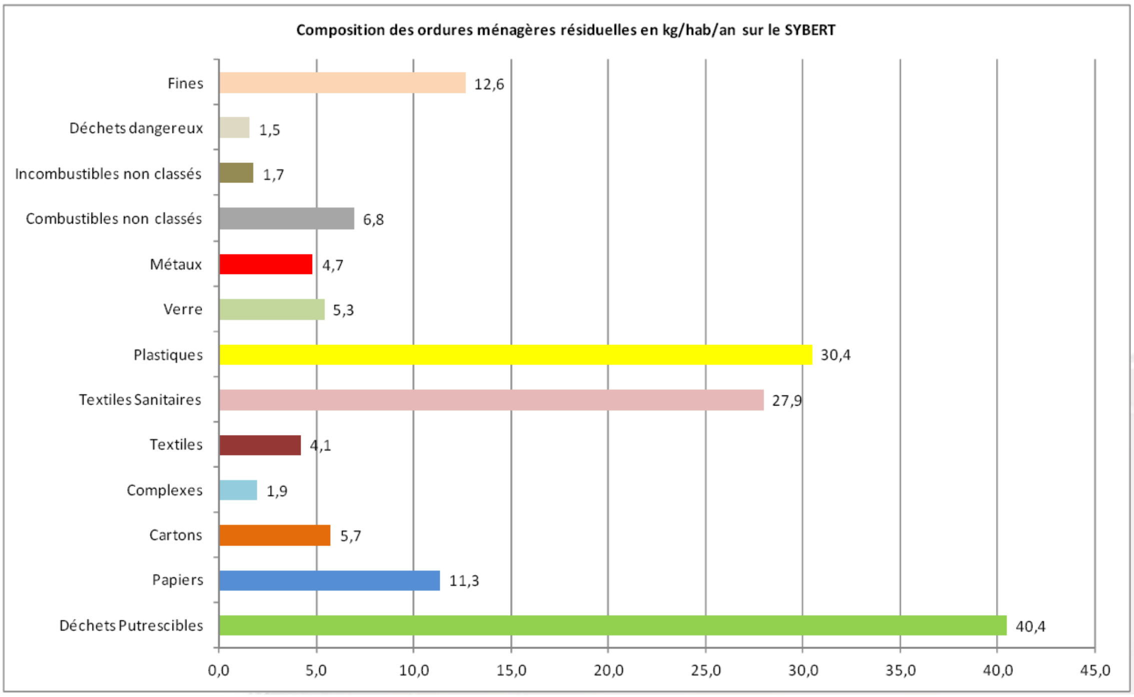
As indicated, life cycle thinking should be part of the waste management strategy in order to minimise cross-media effects, i.e. to minimise the energy consumption for collecting the different waste streams and to identify the most environmentally friendly and sustainable way for waste recycling and residual waste treatment.

Operational data

Characterising residual waste

Characterisation of residual waste is an important step towards understanding the improvement potential for waste management in a particular municipality. Obtaining a representative sample is essential, and it may be useful to undertake residual waste characterisation across sub-areas (e.g. rural and urban) and seasons to obtain a more detailed understanding of driving forces and mitigation options. An example is presented below.

 1. Residual waste is collected and arrives for characterisation.	 2. Residual waste is mixed to obtain a representative average sample
 3. The sample is manually sorted into separate fractions, loaded into containers for weighing.	 4. Fine fractions are sieved through and also weighed.



5. Results are compiled and analysed.

Source: SYBERT (2015).

Evaluation of waste management options

One key element of this BEMP is the systematic evaluation of waste management options in terms of their environmental performance. The most reliable approach to evaluate options is to consider them for each waste stream separately in the first instance, benchmarking them against identified best practice methods and performance levels as described throughout this document. An example of best practice evaluation throughout the life cycle of a particular waste stream is given for kitchen bio-waste in Table 2.4, below. This example highlights the importance of considering the multiple stages and processes applicable to individual waste streams, and how waste management organisations may directly control or indirectly influence these processes through a multitude of best practice measures as described throughout this document and elsewhere. Each measure will be associated with particular key performance indicators that can be used by waste authorities to benchmark the efficiency of their operations. Finally, a decision can be made on the best overall strategy based on the life cycle environmental performance (Life cycle assessment of waste management options BEMP, section 2.4). This may involve evaluation of trade-offs, e.g. higher waste collection emissions associated with separate collection considerably outweighed by reduced treatment emissions after accounting for grid electricity and fertiliser replacement from anaerobic digestion.

Table 2.4. Best practice measures and associated and key performance indicators for four main stages of kitchen bio-waste management

Stage	WMO best practice measures	Key performance indicators	References
Avoidance	Citizen education and awareness raising to reduce food waste	Total kitchen waste generated (kg per capita per yr)	Awareness raising BEMP (3.5.4)
Collection	Separate kerbside collection	Percentage of kitchen waste collected separately (% annual mass)	Waste collection strategy BEMP (3.9.5)
	Logistics optimisation (route planning) to minimise transport distance	Fuel consumption per tonne collected (L/tonne) Cumulative Energy Demand per tonne collected (MJ/tonne) GHG emissions per tonne collected (kg CO_2e/tonne)	Logistics optimisation BEMP (3.9.7)
Treatment	Anaerobic digestion of wet organic waste fractions	Mass of bio-waste diverted from landfill through anaerobic digestion (kg/household/yr) Percentage of bio-waste present in collected residual waste (% annual mass)	Waste treatments BREF (JRC, 2006)
	(Decentralised composting, where separate collection is not possible)	Mass of bio-waste diverted from landfill or incineration through decentralised composting (kg/household/yr) Percentage of bio-waste present in collected residual waste (% annual mass)	Waste treatments BREF (JRC, 2006), Decentralised composting BEMP (3.11.2)

Table 2.4. Best practice measures and associated and key performance indicators for four main stages of kitchen bio-waste management

Stage	WMO best practice measures	Key performance indicators	References
Post-treatment	Use of biomethane to power collection vehicles	Vehicle rated CO_2 emissions (g CO_2e/km) Engine PM, NO_x, VOC emissions (g/kWh) Percentage vehicles that are EURO VI compliant Percentage vehicles that are hybrid-electric or natural gas/biomethane powered	Low emission vehicle BEMP (3.9.8)
	Efficient use of digestate on agricultural land to recycle nutrients and increase soil organic matter	The maximum fertiliser nutrients applied do not exceed those required to achieve the agronomic optimum crop yield, after fully accounting for crop-available nutrients supplied by: (i) organic amendments, (ii) soil nutrient supply, (iii) crop residues Digestate is applied to land via injection, or trailing shoe where injection not possible	Best environmental management practice for the crop and animal production sector (JRC, 2015).
Overall	Life cycle assessment of environmental performance over the material flow chain	Environmental burdens per capita arising from management of kitchen waste (kg CO_2e per capita per yr, kg PO_4e per capita per yr/ MJe per capita per yr, etc.)	Life cycle assessment of waste management options BEMP (2.4).

In many cases, it has to be expected that the waste management strategy cannot go for the optimal solutions owing to organisational, financial and operational constraints. Instead, organisations should then ensure they go for the second best option, with a long-term objective to implement the optimal solution e.g. to go first for decentralised composting of bio-waste and later for the collection of bio-waste with anaerobic fermentation.

Organic waste case study

The following paragraphs outline the rationale for a hierarchy of options for organic waste management.

Results of life cycle assessment presented in the next BEMP (section 2.4) demonstrate that anaerobic digestion is the best treatment option for wet organic waste, such as food waste, with respect to overall environmental performance – unless that waste can be used to feed animals. Biogas provides renewable energy, whilst digestate returns readily available nutrients and organic carbon to soils – replacing fossil energy and synthetic fertilisers whilst enhancing soil quality (see section 2.4). Anaerobic digestion may be performed following separate collection of the wet organic waste fraction, or following separation from residual waste in Mechanical and Biological Treatment (MBT) plants.

However, in situations where neither separate collection nor MBT is possible, composting is the preferred option because the compost produced is an excellent soil conditioner – replacing a small amount of synthetic fertilisers, adding a significant amount of organic carbon to the soil, and improving soil structure. Composting is also the best option for certain "green waste" fractions, such as garden cuttings, that do not break down very easily in the digestion process but that contain significant nutrients.

Centralised composting requires collection of organic waste from households and businesses, and can facilitate the return of nutrients and carbon back to agricultural land. This is particularly important in areas of intensive arable agriculture, where soil organic matter is being depleted through insufficient organic inputs. EC (2012) reports that almost 75 % of analysed soils in Southern Europe had low (3.4 %) or very low (1.7 %) soil organic matter content, putting the latter category at risk of future desertification. Phosphorus recycling is also very important owing to limited proven reserves of this element on the one hand (Cordell et al., 2009) and to the risk of an overload of phosphorus (and nitrogen) biogeochemical cycles on the other (Rockström et al., 2009; Steffen et al., 2015; Stockholm Resilience Centre, 2015). Compost improves soil structure, water holding capacity and overall fertility, and reduces erosion risk (Andersen et al., 2012). Centralised composting is likely to result in more efficient nutrient cycling on agricultural land than decentralised (home) composting, but decentralised composting can avoid the environmental and economic costs of waste collection, and may be regarded as waste "prevention" according to official waste statistics (though is not a prevention measure in reality).

Finally, combustion is the preferred option for woody organic material because this material does not break down easily via digestion or composting, but has a relatively high lower heating value and can therefore achieve significant fossil energy replacement (Avfall Sverige, 2010, Aschaffenburg Local Authority, 2015).

Based on the above factors, the following prioritisation of waste management options can be made for three main types of organic waste fraction (as undertaken by Aschaffenburg Local Authority).

Table 2.5. Example of waste management prioritisation

	Wet organic waste (e.g. food waste)	Green cuttings	Woody waste
Animal feed	1 (if applicable)	NA	NA
Anaerobic digestion	2	NA	NA
Composting/mulching	3	1	2
Combustion with energy recovery	4	2	1

Applicability

The development of a waste management strategy is possible for all municipalities, cities, counties or regions which are in charge of waste management at a strategic level.

An effective integrated waste management strategy requires that the WMO full engages all staff with its development and delivery, to ensure high levels of motivation and performance, and to encourage continuous improvement and appropriate corrective actions. It may be necessary to outsource aspects of strategic planning where particular specialist expertise, such as data analytical skills, are required.

Economics

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

There is no information available concerning the costs for the elaboration 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 elaboration and further development of waste management strategies may be required by authorities but will form the basis of a modern and sustainable waste infrastructure.

Reference organisations

The County of Aschaffenburg/Germany is an excellent example, also with respect to the annually published waste management report (Aschaffenburg, 2014). The counties of Rems-Murr (Germany) and Breisgau-Hochschwarzwald (Germany) and of Besançon (France) as well as the Cities of Vienna (City of Vienna, 2012) and Munich (Schmidt, 2013) are good references, too.

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2.4. Life cycle assessment of waste management options

Description

Why undertake 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, practises 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 e.g. recycling and extraction of virgin raw materials.

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 e.g. 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 via 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 2.3), 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 for external dissemination of results, but good practice even when results are only used internally).

Case study example

Throughout this BEMP, reference will be made to a case study 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).

Achieved Environmental Benefit

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 indicator

Management indicators

The following indicators and possible management benchmarks are proposed for this technique:

- Systematic application of life cycle thinking, and where necessary undertaking of life cycle assessment, throughout waste management strategy design and implementation.
- Management strategies for all waste streams are supported by documented life cycle environmental performance data.

Environmental burden indicators

Recommended environmental burden indicators for use in LCA studies are described under operational data (below) – for example kg CO_2e to represent the contribution of a system towards global warming potential (climate change). Environmental indicators integral to LCA may be complemented with economic and social indicators if undertaking wider social LCA (UNEP, 2009).

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

Operational data

Scope and boundary definition

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 foot-printing (e.g. BSI, 2011) or organisation carbon foot-printing (WRI, 2004, 2011a) are useful for straight forward attributional LCA of waste management systems, which is likely to require accounting for processes managed by other organisations (i.e. processes occurring outside of a waste management organisation's operational boundary) (WRI, 2011b).

Figure 2.4 provides an example of the boundaries and main processes considered for attributional LCA of organic waste treatment by anaerobic digestion (AD). Two main products are generated by AD systems: biogas and digestate bio-fertiliser. Attributional LCA may be used to benchmark the environmental efficiency of AD against other forms of bioenergy and bio-fertiliser production, but this requires allocation of environmental burdens arising from the AD system across the products/services delivered (or "functional unit" in LCA nomenclature). Allocation may be undertaken based on the relative mass, embodied energy or financial value of the products/services. However, this approach is not so useful for comparing AD with other options whose primary "service" is waste management.

A better approach to compare waste management options that may generate multiple products is to expand LCA boundaries in order to consider processes avoided by the product outputs (e.g. Castellani et al., 2015). In the AD example (Figure 2.2), this would include grid electricity generation and fertiliser manufacture/application replaced by bio-electricity and bio-fertiliser (digestate) application, respectively. In the case of recycling, this would involve accounting for the quantities of raw material extraction, transport and processing that are avoided when materials are recycled. Then, results may be expressed as environmental burden *changes* expected from a particular change of strategy, or the introduction of a new system – as appropriate to inform waste management strategy from a wider public good perspective. It is important to note that changes may also lead to the *indirect* substitution or implementation of processes *via market signals*, which may be captured by consequential LCA based on economic modelling. Consequential LCA should be based on predicted marginal effects, rather than average effects: e.g. what kind of marginal electricity generation is replaced by new bio-electricity fed into the grid from biogas generation?

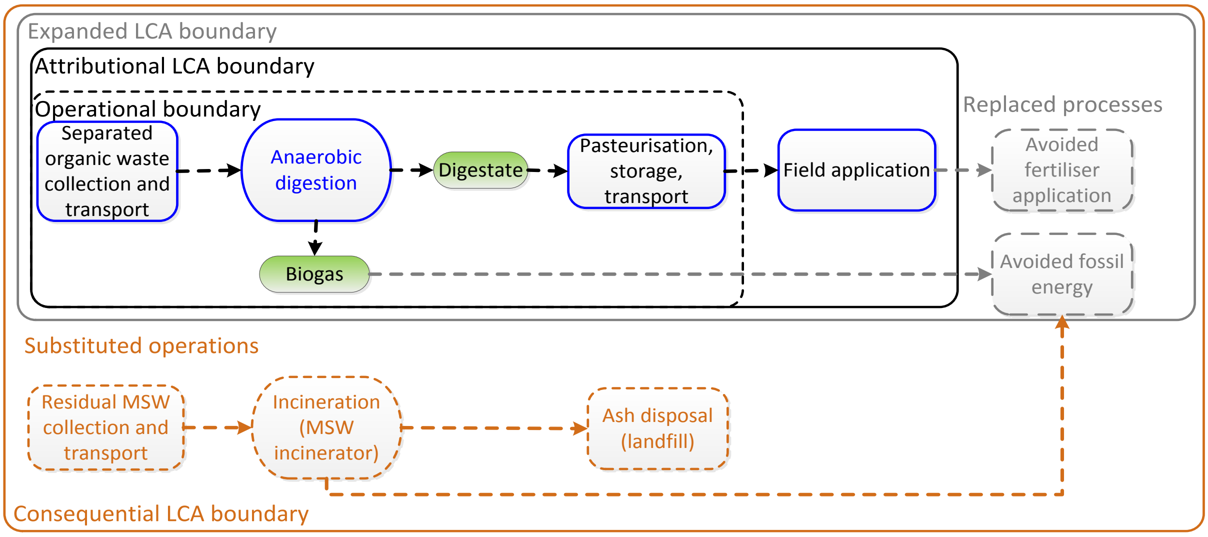

Figure 2.4. Boundaries and processes considered within different LCA approaches to evaluate the environmental balance of anaerobic digestion

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 compositing and animal feeding – in the latter case avoided cultivation of wheat as an animal feed).

Inventory compilation

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 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-kms 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 yr) soil organic carbon sequestration credit (a CO_2e “credit”) equivalent to 13 % and 14 % of 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 e.g. low, medium, high uncertainty, 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 onsite 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, excluding, for example, diesel combustion, nutrient losses and fertiliser replacement incurred by digestate application.

Life cycle impact assessment (LCIA)

Life cycle impact assessment (LCIA) involves the characterisation of inputs and emissions according to their environmental damage potentials, 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., 2009).

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, ionizing radiation, ecotoxicity, photochemical ozone formation, land use, and resource depletion (materials, energy, water).

Table 2.6 summarises LCIA methods recommended for the International Reference Life Cycle Data System (JRC, 2011).

Table 2.6. 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 human (CTUh)	Units of items (cases)
Non-cancer human health effects, CTUh	Comparative Toxic Unit for human (CTUh)	Units of items (cases)
Respiratory inorganics, PM2.5eq	Mass PM2.5-equivalents	Units of mass (kg)
Ionizing 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 freshwater, P equivalents	Mass P-equivalents	Units of mass (kg)
Eutrophication marine, N equivalents	Mass N-equivalents	Units of mass (kg)
Ecotoxicity freshwater, 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, freshwater 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-equivalents (contributions to annual per capita loadings, divided by 1,000).

In Figure 2.5, 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 additional environmental burdens, 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 2.5).

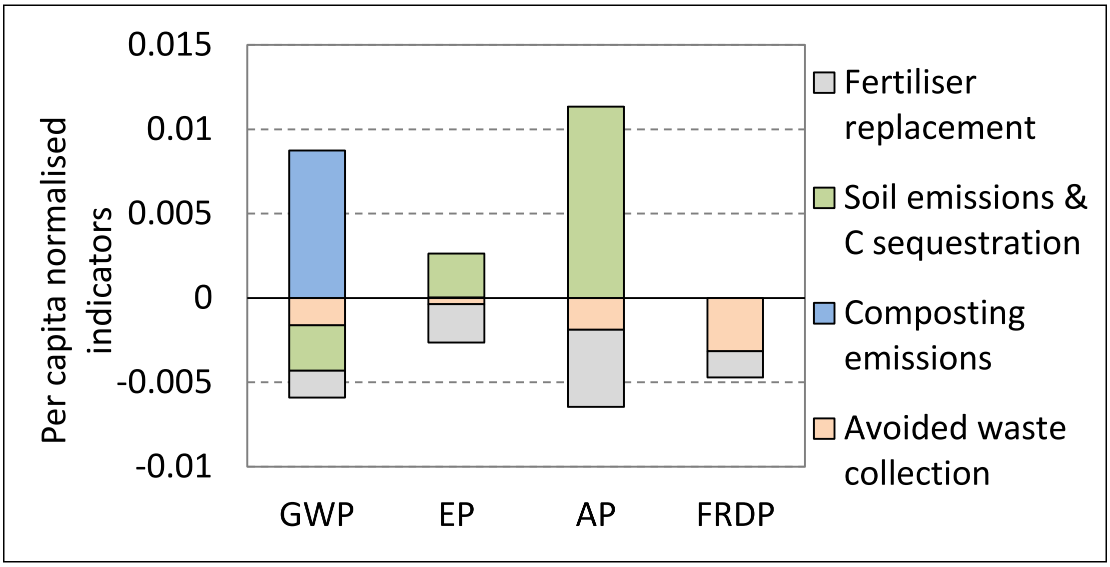

Figure 2.5. Results for global warming potential (GWP), eutrophication potential (EP), acidification potential (AP) and fossil resource depletion potential (FRDP) for decentralised composting of household organic waste (see section 3.11.2)

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 AD case study are displayed in the next section, expressed using the same four environmental indicators used in Figure 2.5.

Interpretation and reporting

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 e.g. 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.

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 (LCI and 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 analyses 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 analyses (e.g. Clavreul et al., 2013). Uncertainties for individual interventions of processes 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.
- Study report is transparent and consistent.

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

Table 2.7 and Figure 2.6, below, summarise the environmental changes that arise, expressed as credits (negative values) and burdens (positive values) across avoided and incurred processes (Figure 2.6), and expressed as *net* environmental burden change (Table 2.7), 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. Where e.g. food factory waste can be used as animal feed, 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.

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 therefore be interpreted cautiously and always in relation to the precise scenarios considered.

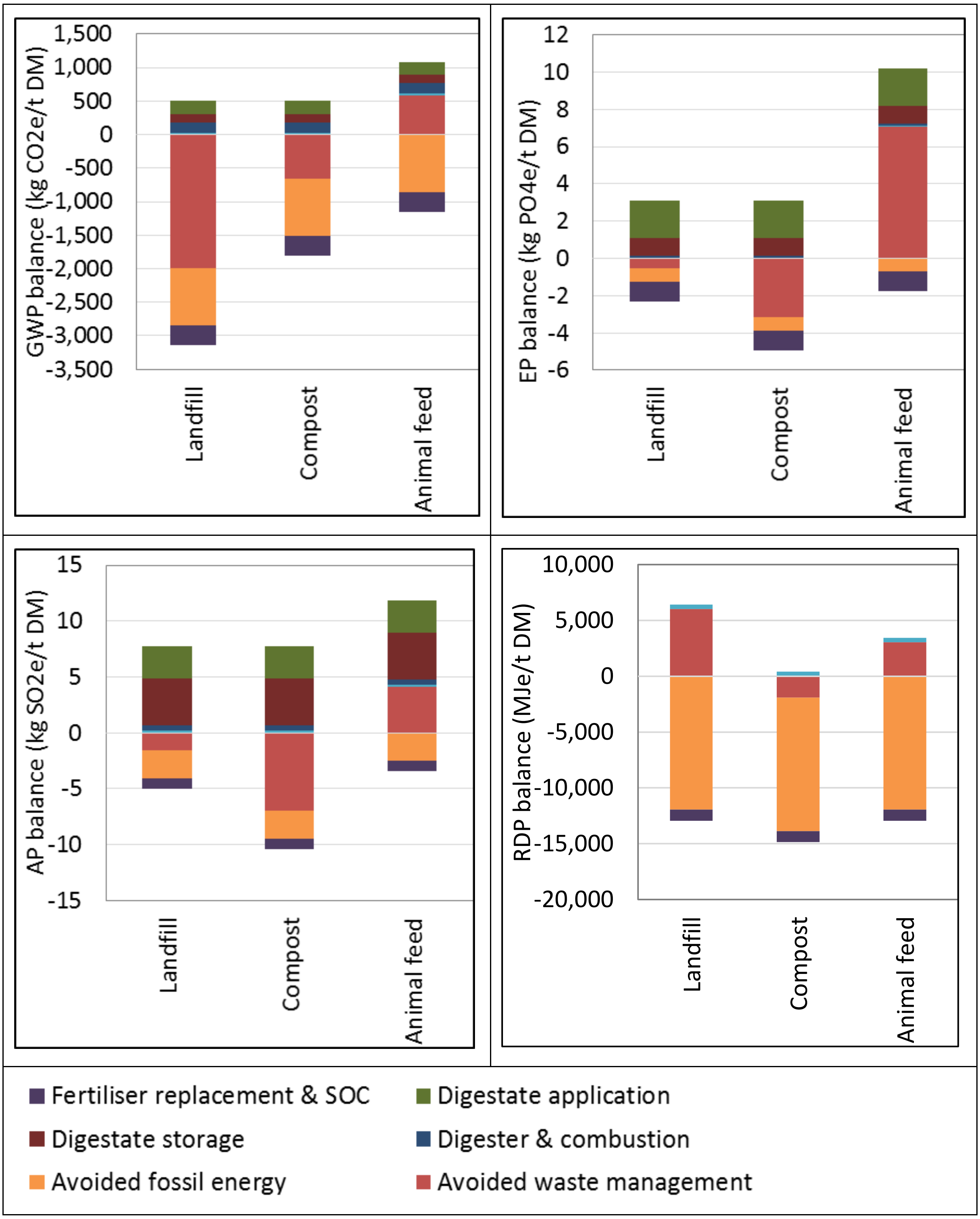

Figure 2.6. Net environmental burden changes, expressed per tonne of dry matter organic waste processed, when anaerobic digestion replaces landfilling, in-vessel composting or use of hygienic organic waste for animal feed

Table 2.7. 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

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). EASEWASTE is 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 on the waste hierarchy, and of management changes that affect whole-waste-chain performance, are often required.

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.

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 EURO, but could avoid costs associated with licencing 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 and 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 2.3).

The Technical University of Denmark (DTU) is a leading authority on LCA accounting for waste systems, and 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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2.5. Economic instruments

Description

Aim

This BEMP gathers useful information and practical examples on economic instruments that can be applied by waste management organisations and authorities, with a main focus on the local scope of its implementation. Although most of the described measures 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 'economics instruments' refers to regional or national policies or regulations. Herein, the term 'local economic instrument' is used as a reference to the 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 its 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 out of 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, up to 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
- Etc.

In general, economic instruments aim at

- reducing the amount of waste generated,
- reducing the proportion of hazardous waste,
- improving product design,
- encouraging recovery, re-use 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, this tool is 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 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 when based on weight, while volume based systems performance is not considered outstanding. A specific BEMP description for MSW can be found in section 3.5.3.

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. It consists of payments or rewards given to the users to encourage people to recycle more, typically with vouchers to individuals, vouchers to communities or paid to individuals (Holmes et al., 2014). Most of the examples that are applied in Europe are pilot schemes or partial-coverage schemes, which were implemented after the success of the pilot trial. From these, some selected case studies are described in this document. It is important to remark:

- Legal regulation at local level is a key factor for its implementation. While recycling incentive schemes are usually acceptable, PAYT has certain legal connotations that make its implementation difficult under certain regulatory environments. This is the particular 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 its 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 its 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), the 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 its use is considered a best environmental management practice (Hogg et al., 2010; Schoenberger et al., 2013). However, its 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:

- Portable batteries are charged a deposit by the local government of Osthamar, Sweden (OECD, 2014), achieving a capture rate close to 100 %.
- Police regulation, e.g. City of Schwäbisch Gmünd, Germany (2005): mandatory deposit of at minimum EUR 2.00 for drinking vessels during the city festival.
- Waste management statutes, e.g. City of Nürnberg, 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 Nürnberg, 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, Germany (2012): participation conditions/regulation for christmas market: prohibition of one way tableware, mandatory use of reuseable glogg cups, mandatory deposit of at minimum EUR 1.00, or City of Graz, Austria: charging of EUR 1.00 per beverage packaging in football stadiums to avoid littering.

Construction and Demolition Waste and Healthcare waste

As this BEMP refers to cross-cutting issues, it is worth mentioning the different approach to several economic instruments for different types of wastes. CDW management contracts include a fee per unit of collected volume, which vary for different fractions, being the more expensive for the mixed waste fraction (up to EUR 100 per tonne) compared to metals or clean concrete (from EUR 5 to 25 per tonne). A very similar approach is observed on how HCW is managed: the waste contractor usually charges the waste treatment cost per bin or container the waste is collected and stored in. 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 are charging a deposit to 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 re-paid to the contractor when “waste management certificates” are submitted to the authority. This deposit system managed by municipalities has potential to become a BEMP, but its current performance is far from such consideration due to the following reasons:

- It is oriented to avoid 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 becomes, then, 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, as extended product responsibility, HCW, e.g. for waste medicines, product levies, CDW, e.g. for natural aggregates, adaptation of VAT, e.g. for recycled aggregates.

Achieved Environmental Benefit

Municipal Solid Waste

The performance of several case studies on the application of local economic instruments in municipalities is shown in Table 2.8.

Table 2.8. Examples of reward schemes and PAYT performance[18]

Municipality or County	Instrument	Results	Additional comments	Refe-rence
Bracknell Forest, UK	Recycling incentive scheme	Enhanced public perception and widely acceptability of recycling Increase of a total 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 Llobrgat, ES	Pay-as-you-throw, unit based	Increase of separately collected materials from 33 % to 89 %, reduction of residual 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 residual 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 Region, IE	Pay-as-you-throw, weight system	Reduction of residual waste from 79 % to 65 %, and increased 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 residual waste disposal costs, reduction of residual costs down to around 50 kg per capita per year	Urban and rural, all waste streams	Section 3.5.3
Rotterdam, Barendrecht and Krimpen aan den Ijssel, NL	Recycling incentive system	Increased collection of 24 % (total waste), reduction of residual waste of 37 %.	Called 'Cash for Trash', rewards are direct cash paid back to citizens	OECD, 2015
Bradford, Aire Valley Recycling, UK	Recycling incentive scheme	Increase of 36.5 kg 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

[18] The most practical definition of "residual 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 WMO is involved in subsequent sorting (e.g. in sorting plants following co-mingled collection, or in mechanical and biological treatment plants).

Table 2.8. Examples of reward schemes and PAYT performance[18]

Municipality or County	Instrument	Results	Additional comments	Refe-rence
Gloucestershire, UK	Recycling incentive scheme	No increase or decrease of recyclables per participant per year	Urban and rural, all recyclables	Defra, 2013
Norfolk County, UK	*Reuse* and recycling incentive scheme	Increase of 99 kg of re-usables 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	Re-use incentive scheme	Increase of 67 kg recyclables and re-usables 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 re-used are also transported by the waste manager). The results were compared to a hypothetical 100 % recycling scenario for 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 emissions levels (Table 2.9). It can be seen that the performance of reverse-logistics is significantly better.

Table 2.9. Greenhouse gases 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
Trade-marked 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 indicator

The most important environmental performance indicators to monitor the performance of instruments are those directly linked to the potential benefit. For all those systems described in this BEMP, the following can be considered:

Local deposit refund schemes. The participation of the municipality in the implementation or facilitation of a local deposit refund scheme, a Y/N indicator would suffice to inform about the performance of the management practice. For instance

- The waste authority participates, regulates or manages deposit refund schemes of e.g. waste beverage containers at local level (Y/N)

The capture rate would also be an appropriate indicator, although difficult to measure if there is no direct control over the scheme:

- $Capture\ rate\ (\%) = 100\ \frac{Number\ of\ beverage\ containers\ returned}{Number\ of\ beverage\ containers\ sold\ with\ a\ deposit}$

Pay-as-you-throw and local deposit schemes. Both techniques aim to an increase of selectively collected waste and a reduction of residual waste. So, the proposed indicators are:

- Percentage of MSW generated that is selectively collected (% weight)
- Percentage of MSW generated that is recycled (% weight exiting material recovery facilities in separated fractions)

Therefore, not only the amount of recyclables is considered but also their fate. However, it is acknowledge the technical and managerial difficulties of quantifying the fate of county levels at plants that operate with regional, national and sometimes internationally sourced waste (e.g. paper mills). See section 3.5.3.

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 seems to be much lower than the savings that economic instruments could bring. Waste authorities relatively isolated in the application of e.g. PAYT in their geographical area may have a *waste tourism* effect, i.e. disposing 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 accounted in the implementation of a system that would allow the use of local economic instruments on waste separation at households:

- Produce a cost estimation that allows the waste authority to identify the priority areas of action and design how the new system would integrate in the existing structure.
- Based on the results of the cost audit, set quantifiable objectives (section 2.3), set costs benchmarks (section 3.5.1), and establish a reliable waste accounting system (section 3.5.2).
- Create a deposit fee-response model that allows further optimisation.
- Enforce the implementation by avoiding the so-called waste crime.

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

Cost Estimation

Although they are environmentally sound, local economic instruments are designed as a cost saving measure. They directly affect to 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, 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 then usually belongs to the knowhow of its practitioner, which may be confidential in cases where a private consultant audits municipalities (as the BEMP example case of section 3.5.1 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 in 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 casual 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 way 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 U.S. Environmental Protection Agency (EPA) for waste services. Some of the principles used by FCA are detailed in the following:

- 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 in these management areas: Collection, Disposal (Landfilling and Waste-to-Energy) and Recovery (Consumer products and packaging, and composting)
- Cost per area are:
    - Up-front 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 % 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 damages, 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 contracted out services 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 on the management, supervision, human resources, etc., of the service should be given a fair share from the local government expenses.
- The allocation method of shared costs between areas can be:
  - Per budget (only for administration services): the allocation of a share cost is calculated as the proportion of the total municipal budget. This would allocate an administration cost to the whole waste management service respect to other services.
  - Personnel share method: Similar to the budget method, but taking into account the number of people working at 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
  - Taxes revenues: income from taxes not directly linked to waste management
  - Transfer revenues, as subsidies or other funding received.

Regarding the above, a cost per ton 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 developed for, 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 behaviours model. Just 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) have performed a study on waste prevention and separation at markets and street festivals. To their experience a deposit has different effects, depending on the amount of the deposit. If the beer costs EUR 3.00 per cup and the deposit is EUR 0.10, the majority of the cup does not come back, but goes into the garbage. The caterer makes a considerable profit by the non-reimbursed deposit and has no expenses with 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 5.00 – returning of cups is attractive for the caterer. As the cups come back, 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 by the organizer or the local authority, the caterers generally tend to simple cups and low deposits.

Case study on the implementation of a recycling incentive scheme

Bracknell Forest City Council, in the south of England, manages the waste from a total population of 118,000 citizens through a contract with SITA. 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 the 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 the citizens through the so-called Council Tax). It is considered that a potential saving of GBP 300,000 could be achieved only from avoidable landfill tax in 3 years. The key objectives were to increase the number of households participating in the kerbside recycling service from 75 % to 82 % in 2 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 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 at 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:
    - 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 in 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 households). The amount of residual waste was reduced by 1,000 tonnes, representing a saving of GBP 90,000 (from 1st 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, as the possibility of targeted awareness campaigns through the e-mail of system users, gained insights of 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 discovering, deterring, rehabilitating and punishing. 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 compliant citizens should be engaged in the community to keep them fulfilling their obligations. Enabling and educating citizens should also be considered as appropriate measures to reduce the extension of enforcement. In general, enforcement is out of 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). In any case, it is acknowledged that enforcement and, especially, the 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 e.g. 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 legislations. 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.

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 quite relevant, reducing collection costs from 6.5 to 9.5 %. However, a reduction in the income from recycling was detected, compensating the reduction of 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 quite evident (Retorna, 2013).

Other case studies in PAYT schemes

Box 2.1. Torelles de Llobregat (OECD, 2006)

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

Implementation of the system

- Bio waste (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 sacks, supplied by municipality, same collection as bio waste. Large branches excluded.
- Garden waste as large branches, no charge, bring scheme.

Results

- Reduction of residual waste by 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 accounted), calculated from the avoidance of treatments. Increase in private transport and illegal disposal not accounted in the balance.

Box 2.2. Landkreis Schweinfurt (OECD, 2006)

Landkreis Schweinfurt (OECD, 2006)

Implementation of the system

Tariffs:

- 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 for 240 l 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 bio waste.

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 accounted in the balance.

Box 2.3. Limerick, Clare and Kerry regions (R4R, 2014)

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 lift
- A fixed fee, e.g. as annual service charge, is also paid by the user
- Recyclables, bio waste and glass are usually free of charge
- The charge per kg is EUR 0.12 – EUR 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 background report for best environmental management practices in the 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 2 or 3 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 on 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 from 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 re-sizing 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, as UK or Greece, do not allow (or do not *ease*) 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 innovative-driven behaviour at the local government, with some good accounting practices, are pre-requisites 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 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 part 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. Norfolk county and Bristol students association case studies refer to re-use shops that also produce recyclable materials.

Table 2.10. 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 in a full time basis. In kind contributions evaluate 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 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)
- Deposit Refund Schemes at events:
    - Directly applied: Graz (AT)
    - Locally regulated: Schwäbisch Gmünd, Nürnberg, Reinheim (DE)

B2B approaches:

- BEMP: London Construction Consolidation Centre (UK)

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3. Municipal Solid Waste (MSW)

3.1. Introduction

This chapter contains 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 %). MSW is one of the most polluting categories of waste, and the category with the highest potential for environmental improvement through better management. 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.

3.2. Environmental burden

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 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/yr 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.

3.3. Best practice portfolio

This chapter will sequentially address a range of best practice techniques to manage MSW, starting with overarching waste strategy formulation in sub-chapter 3.5, and culminating in waste treatment in sub-chapter 3.11 (see table below).

Table 3.1. Best practice portfolio for municipal solid waste management

Sub-chapter	BEMP
3.5. Waste Strategies for MSW	Cost benchmarking
	Waste monitoring
	Pay-As-You-Throw
	Awareness raising
3.6 Enabling techniques on waste strategies	Performance-based waste management contracting
3.7. Waste Prevention	Local waste prevention programmes
3.8. Product Re-use	Product re-use schemes
3.9. Waste Collection	Waste Collection Strategy
	Infrastructure to recycle or to recover waste streams and to dispose of hazardous compounds
	Logistics optimisation for waste collection
	Low emission vehicles
3.10. Enabling techniques on waste collection	Best practice in the application of inter-municipal cooperation (IMC) for waste management in small municipalities
3.11. Waste Treatments	Sorting of co-mingled packaging waste
	Decentralised composting

Sub-chapter 3.5 will provide waste management organisations (WMOs) with an overview of best practice measures and indicators related to the development of waste management strategies that systematically and comprehensively deliver best environmental outcomes, are record performance by: (i) monitoring and benchmarking key aspects of performance, (ii) using pay-as-you-throw to change behaviour and increase recycling, (iii) raising awareness to drive waste prevention and increase recycling rates, (iv) interaction with deposit schemes to increase rates of re-use.

Sub-chapter 3.6 describes techniques that are not in themselves “best” practice, but that can play an important role supporting and enabling best practice.

Sub-chapter 3.7 describes best practice techniques for WMOs to drive waste prevention through local waste prevention programmes.

Sub-chapter 3.8 comprises one BEMP on measures that WMOs can take to facilitate product re-use.

Sub-chapter 3.9 covers waste collection. The first and second BEMP describe elements of best practice in the design of collection strategies that minimise environmental impact whilst delivering the level of service required to maximise recycling rates within the framework of overarching waste management strategy targets and infrastructure to support take-back obligations. Logistics optimisation and low emission vehicle BEMPs focus on operational efficiency and alternative fuelling of refuse collection vehicles, respectively.

The interaction between municipalities is covered in an enabling technique in sub-chapter 3.10.

Sub-chapter 3.11 addresses waste treatment options that are not described in other best practice documentations, in particular IED BREFs. Thus, just two waste treatment options are covered: (i) recyclable waste sorting plants, with an emphasis on separation efficiency to maximise recycling rates, (ii) decentralised composting, undertaken by householders and community groups with support and guidance from WMOs.

3.4. Reference literature

EEA (2013). Managing municipal solid waste — a review of achievements in 32 European countries. EEA, Copenhagen.

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3.5. Best Environmental Management Practice on Strategies for Municipal Solid Waste

3.5.1. Cost benchmarking

Description

Waste management is heavily 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 local public administration of 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 sale of recyclables. All relevant waste fractions that are part of municipal solid waste (paper/cardboard, glass, plastics, bio waste, green cuttings, scrap metal, non-ferrous metals, residual waste from households etc.) must be taken into account in the study.

More in detail, in the evaluation of total costs, the following costs are usually considered:

- costs for collecting the different waste fractions (e.g. residual waste, bio waste, 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 a 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 bio waste 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 identifications of improvements 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 costs of waste prevention measures with the cost savings from the decreased amount of waste to be managed.

Figure 3.1 shows an example for the evaluation of the main cost categories for 33 counties and 11 cities in Germany (ia GmbH, 2015).

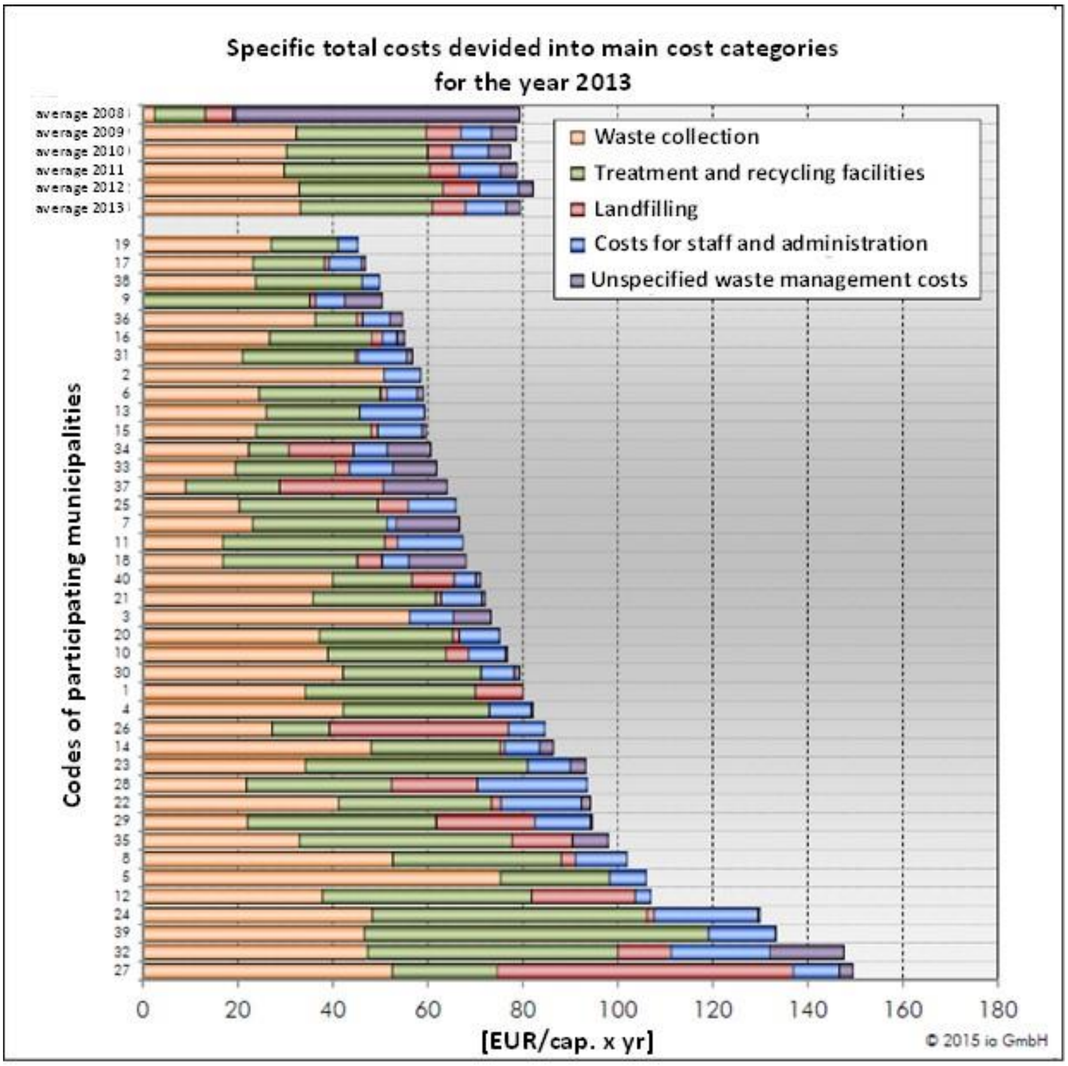


Figure 3.1.Specific waste management costs for the main cost categories for 2013 of 33 counties and 11 cities in Germany providing waste management services to 6.3 million citizens in total, based on ia GmbH (2015)

The corresponding annual waste quantities per capita are illustrated in Figure 3.2.

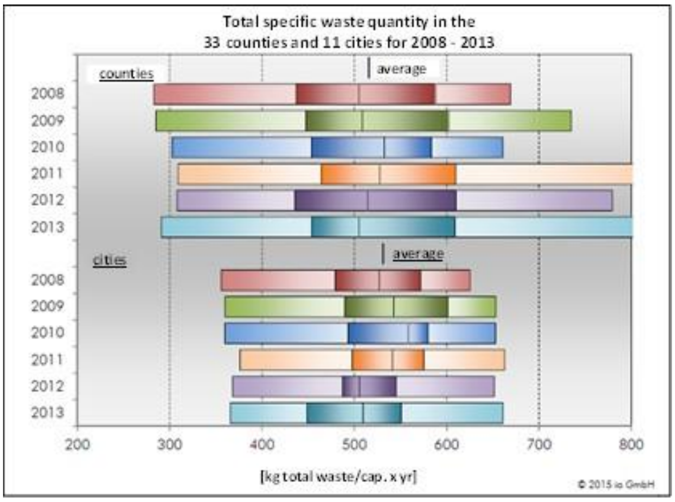


Figure 3.2.Total specific waste quantities of the participating 33 counties and 11 cities in Germany from 2008 – 2013, based on ia GmbH (2015)[19]

Achieved Environmental Benefit

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 3.3).

Appropriate environmental indicators

The regular participation in a detailed cost benchmarking as described above (yes/no criterion) is an appropriate environmental indicator. The uncovered cost [EUR/cap x yr] could also be a good indicator to be used when carrying out a cost benchmarking study.

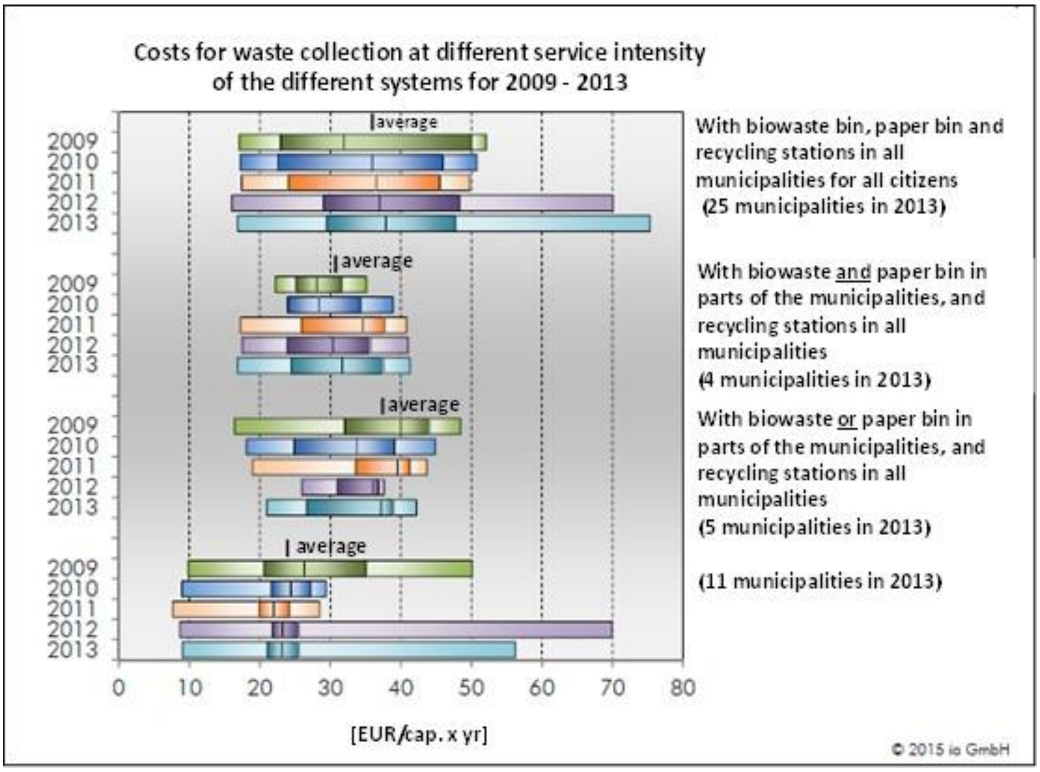


Figure 3.3.Costs for waste collection at different service intensity of the different systems for 2009 – 2013, based on ia GmbH (2015), see explanations in the footnote[19]

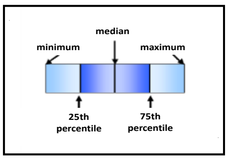
Cross-media effects

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 which is prepared to provide data in the required form and to share them with others in an anonymous form can participate in the cost benchmarking exercise. The more organisations take part the more reliable is the assessment of them.

A specific case of cost benchmarking has been carried out by a network of municipalities and local authorities in Germany, called ForumZ, which promotes the inter-municipal co-operation in the field of waste management (www.forumz.de).

19 

The values are presented as median, minimum, maximum and 25th/75th percentiles as indicated in the figure above.

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 of costs can be indicated as illustrated in Figure 3.4.

The working group while developing the questionnaire decided that based on the annual data collection and responses from the participating municipalities, the questionnaire may be (slightly) adapted year by 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 investigating also whether the uncovered costs depend on the percentage of private services. Figure 3.5 shows that uncovered costs do not depend on the percentage of private services carrying out waste management.

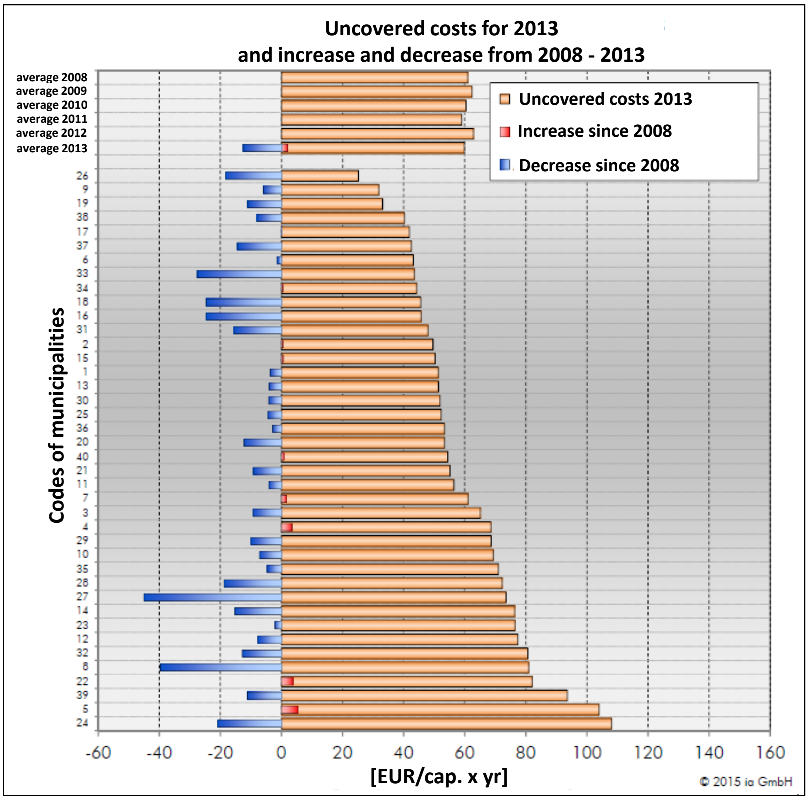


Figure 3.4.Increases and decreases of uncovered costs in 33 counties and 11 cities in Germany from 2008 – 2013, based on ia GmbH (2015)

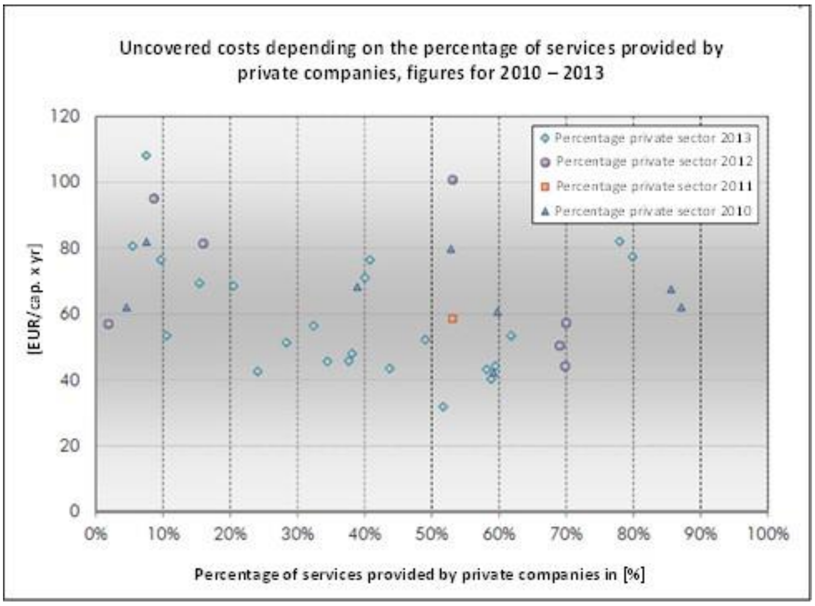


Figure 3.5.Uncovered costs and percentage of services provided by private companies in 33 counties and 11 cities in Germany, 2010 – 2013, based on ia GmbH (2015)

Applicability

Cost benchmarking can be applied in a county/region or on a national level, where waste management conditions are comparable and where there is a uniform legal framework. 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/cap x yr] are significantly different; as a consequence, in this case, the cost indicator [EUR/t total waste] may be more appropriate. Cost benchmarking could be very useful when assessing existing waste management systems with low performance 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 outlays balances, and an appropriate allocation procedure should be applied. A detailed description of cost estimation and allocation procedures is included in section 2.5.

Economics

Municipalities taking part in the cost benchmarking performed by the independent third-party organisation ForumZ (presented in the operational data section) pay an annual fee to ForumZ that is organising the collection and evaluation of cost data. This fee is in the range of 1,000 and 4,000 EUR/yr, depending on the size of the municipality.

According to Figure 3.1, waste management costs of different cities, counties or municipalities vary up to factor 3. For individual services, the range can be bigger, e.g. up to factor 8 for waste collection. For instance, in 2013, the cost for waste collection with bio waste bin, paper bin and recycling stations in all municipalities for all citizens varied between 17 and 76 EUR/cap x yr. If the costs for 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 5 EUR/cap x yr thanks to cost benchmarking and the improvement of the waste management system, the total cost savings in that region could reach one million EUR per year. This can be achieved by cost benchmarking for which the expenditure as a network member is 2-3 Cent/cap x yr.

Driving forces 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, so far, is the only one which has been carrying out cost benchmarking for six 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 both for technical and cost aspects, but not as regular and specific 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 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 does allow cost benchmarking (ADEME, 2015). However, detailed evaluations could not be identified or are not available yet. The same is true for the Paris Region Waste Observatory (ORDIF) which is also 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, accessed 5 November 2015

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, accessed on 10 December 2015

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)). Das Benchmarking-Projekt (The Benchmarking Project). http://www.vksimvku-benchmarking.de/das_projekt.php?thema=projekt, accessed in may 2016

3.5.2. Waste monitoring

Description

An efficient and effective waste management strategy is based on the detailed knowledge of statistical data for the waste streams treated and collected at local level. The collection and management of data can be carried out in detail, firstly defining which information should be collected and then keeping a good and updated database which allows the extraction and processing of the required data, in order to implement a number of analyses on waste management e.g. for improving the waste management strategy (see BEMP on cost benchmarking), to see the improvements due to a new measure implemented. It is therefore important that waste management companies and/or waste authorities monitor the waste streams at single stream level, between the different steps of the collection, recycling and disposal processes. Data collected can then be processed internally/externally and shared with the relevant public administration and citizens.

Waste collected at household level can be classified as:

- Recyclables like paper/cardboard, glass, plastic (mainly packaging), scrap metal, waste wood;
- bio waste;
- green cuttings;
- textiles and shoes;
- bulky waste;
- residual waste;
- demolition waste;
- hazardous waste.

For the monitored waste fractions, the capture rate and the fate, as far as the information is known (for instance, concerning recyclables, the municipalities, cities or regions do often not know the final disposal), is recorded and documented. With respect to residual or mixed waste, it is of advantage to perform composition analysis from time to time, a reasonable frequency is three to five years, in order to determine the potential of additional recycling and recovery to be explored in the future by optimisation collection, awareness raising etc.

Figure 3.6 illustrates important waste streams, also called waste fractions, derived from households and household-type commercial waste.

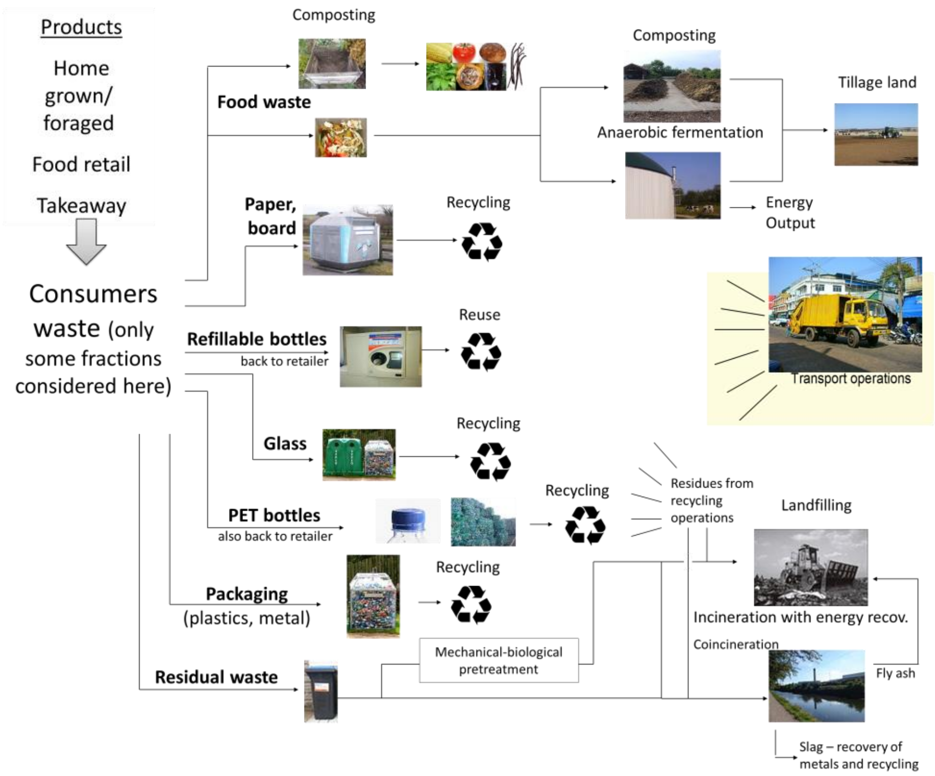


Figure 3.6. Important waste streams concerning municipal waste

Each of the separately collected waste streams can be quantified by weighting the amount collected, i.e. weight any collection truck and any container when entering and leaving each facility (storage/sorting/recycling/treatment/disposal facilities). Collected data from waste monitoring can also keep track of the area of collection, time of the year in order to improve the quality and amount of useful data recorded. Figure 3.7 shows an example of a truck scales.

Figure 3.7.Example of a truck scales

Waste monitoring then can also include the collection of data e.g. on the amount of fractions recycled after waste has been treated, amount of recovered materials, residual waste sent for disposal.

Data collected from waste monitoring can be easily recorded and managed by IT software which can also produce statistics.

When defining which data to collect and how, as well as how to store and process it, it is also important that waste management companies and/or waste authorities are fully aware of their reporting obligations to e.g. local or national statistical bodies or other public authorities. Indeed, it is very important that the way data is collected and stored complies with those requirements (e.g. the same definitions should be used) in order to allow the compilation of meaningful statistics also at higher level.

A good step towards agreeing on a methodology including a common definition of municipal solid waste (MSW), the waste fractions that are included in MSW and the waste fractions that are considered as "recycled" depending on their destination, is the R4R project (R4R, 2014). Waste authorities setting up their own waste monitoring system are recommended to refer to this methodology to build up a compatible system.

For a better comparison of waste monitoring data between different cities, municipalities, counties or regions, a common approach may be helpful (R4R, 2014b).

Achieved Environmental Benefit

On the basis of exact quantities of the different waste streams, the efficiency of measures can be determined and optimised or adjusted when required, i.e. 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. Against this background, the exact determination of the quantity of the different waste streams has no direct environmental benefit, but is likely to have an indirect environmental benefit as the knowledge of the quantities is a prerequisite to follow the continuous improvement process.

Appropriate environmental indicators

The appropriate environmental indicator is the determination of the quantity of all relevant waste streams (yes/no criterion).

Moreover, the quantities of the different waste streams and materials monitored can be assessed based on their mass per person and year [kg/person yr]. The reporting of specific waste quantities in [kg/person yr] enables a better comparison between different cities and counties.

Cross-media effects

There are no significant cross-media effects known.

Operational data

Table 3.2 shows an excellent example of monitoring the quantity of the different waste fractions collected in a German county (Aschaffenburg) from 1989 to 2013 (Aschaffenburg, 2013). 17 recyclables streams, 4 non-recyclable, bulky, hazardous and commercial waste streams have been systematically recorded over the last 20 years.

Table 3.2. Example for the determination and documentation of the quantity of the different waste fractions of a county (County of Aschaffenburg in Germany) from 1989 – 2013, the quantities are given in [kg/cap x yr], (Aschaffenburg, 2013)

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 allows recognizing the drastic change in the waste management system of the county in the last 20 years. The quantity of residual waste drastically decreased and the quantities of recyclables sharply increased. The county introduced a weight-based pay-as-you-throw system for residual waste, bio waste 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 residual waste is about 50 kg/cap x yr. These analyses and the successfulness of the waste management system implemented would have not 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, 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 3.5.3), it is easily possible to monitor which citizens do have individual bins and which use common bins. Then, it can be investigated where the collection and capture rates can be optimized most. The same is true for the collection frequency for the citizens as each collection is recorded and documented for all the citizens. In this case, the data availability is very short, practically just-in-time, and an evaluation and assessment is possible with a few weeks or months (Aschaffenburg, 2014).

Applicability

The determination of the quantity of the different waste fractions is applicable to all municipalities, cities and counties. However, when starting the systematic quantification, the quantities of the most relevant waste fractions may be determined first and may be extended to all fractions step by step.

Economics

So far, no detailed information about costs is available concerning the expenses for the scales (investment and operation) and personnel in charge of collecting and analysing data.

Driving forces for implementation

The legal requirements concerning recycling rates for packaging and the diverting rates for organic waste from landfills as well as the need to determine the efficiency of waste management systems are the driving forces for the quantification of the different waste streams.

Reference Organisations

Many cities and counties throughout Europe, for instance Copenhagen, Hamburg, Barcelona, Bristol, Milano, Aschaffenburg and Schweinfurt, and many more, have a detailed determination of the quantities of the waste fractions.

In terms of methodology, the R4R project (R4R, 2014a) can be considered a reference. The same may be true for 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) which 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

Reference literature

Landkreis Aschaffenburg (County of Aschaffenburg) (2014). Abfallwirtschaftsbericht 2013 (Waste Management Report 2013) (in German). http://opus.kobv.de/zlb/volltexte/2014/24230/pdf/AWB_2013.pdf

ORSo (2015). O.R.So – Osservatorio Rifiuti Sovraregionale (Regional Waste Monitoring Centre). Information on waste management, also on waste monitoring. http://www2.arpalombardia.it/siti/arpalombardia/imprese/rifiuti/Pagine/ORSO.aspx, accessed on 5 December 2015

Regions for Recycling (R4R) (2014a). Regions For Recycling – R4R Methodology [on-line]. http://www.regions4recycling.eu/R4R_toolkit/R4R_methodology, accessed 7 September 2015

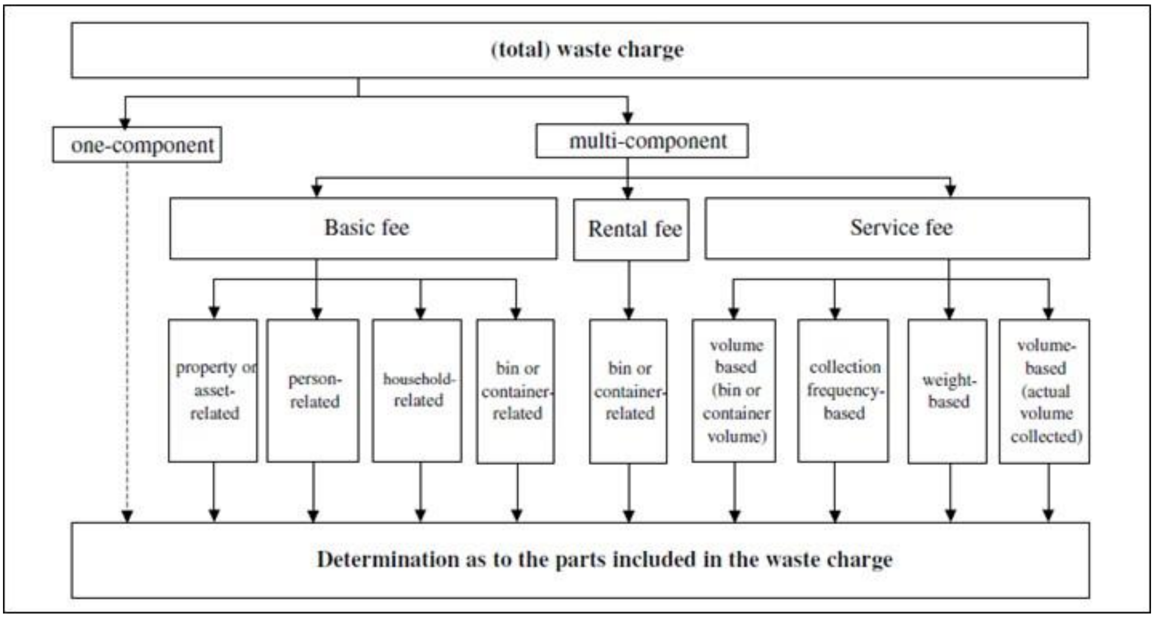
Regions for Recycling (R4R) (2014b). Data comparison – main findings, February 2014. http://www.regions4recycling.eu/upload/public/Reports/R4R_Data_comparison_main_findings.pdf, accessed 1 December 2015

3.5.3. Pay-As-You-Throw

Description

The approach of "Pay-As-You-Throw" (PAYT) (also known as unit pricing (Dijkgraaf/Gradus, 2009), differential and variables rates (OECD, 2006; van Beukering et al., 2009) and variable fee or charge systems) is to realise 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 revealed that the waste fee should not only comprise of the single component "amount of waste generated" but should best consist of basic 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 case the fee is only charged for the amount of waste collected (Reichenbach, 2008; Puig-Ventosa, 2008). Figure 3.8 shows the different possible components of a waste fee.



Source: Bilitewski (2008)

Figure 3.8. Different suitable components for the design of waste fees

In Figure 3.8 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.

Against this background, the PAYT approach can be implemented in different ways as illustrated in Figure 3.9.

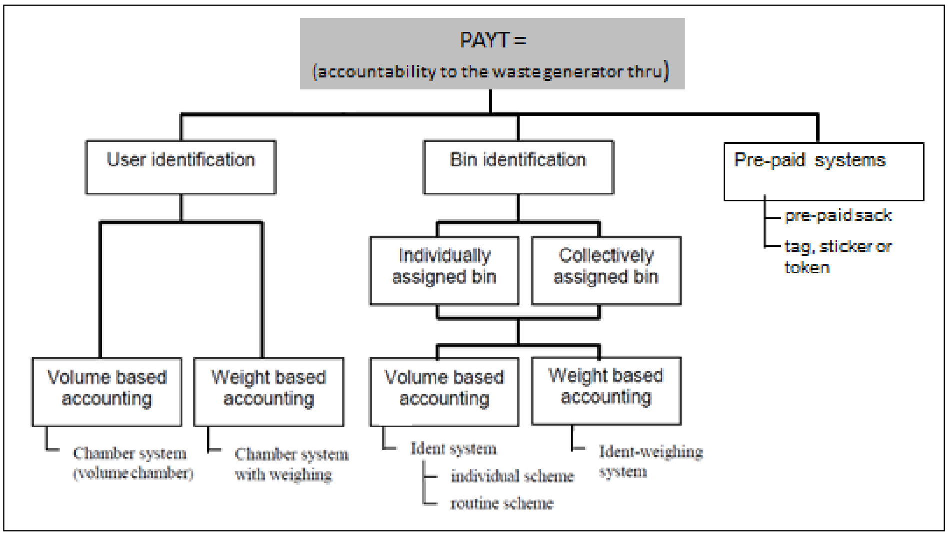


Figure 3.9. Overview of the different possibilities to implement the PAYT approach (based on Reichenbach, 2008)

The pre-paid sack system was also considered to belong to the volume-based systems but here is it 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. So, 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 set 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 set out for collection – this approach can be combined with volume- and weight-based schemes).

Best 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 a well-developed infrastructure to collect different fractions of waste by individual bins (paper/cardboard/board, organic waste, eventually waste plastic and organic waste), glass containers and recycling facilities for ferrous metals, non-ferrous metals, end-of-life electrical and electronic equipment, refrigerators and other white goods, waste plastic, waste polystyrene, waste wood, green cuttings, non-commercial construction and demolition waste, waste tyres, printer cartridges, vegetable fat, textiles, shoes, cork, CDs, etc. in order to offer the citizens a comfortable way to get rid of materials which they do not need anymore. The collection system for all these waste fractions is described in the BEMP on collection systems (see Section 3.9). In addition, awareness raising is also a key element for well-performing PAYT systems; if the citizens are aware and convinced of the system, they will actively support it.

The experience shows that with weight-based schemes best results can be achieved but also with pre-paid sack schemes good performances are reached whereas volume-based systems impart the weakest incentive for waste prevention and recycling (OECD, 2006; Watkins et al., 2012). In contrast, highest recycling rates and lowest residual waste quantities respectively are achieved with weight-based systems accompanied with well-developed infrastructure and citizens with high awareness. As a consequence, a respective case study is presented in more detail. For such a system, the technical PAYT approach is based on the following three pillars:

- Identification (for reasons of accountability to the waste generator)
- Measurement (of the collected or delivered amount of waste and/or services obtained for it), and
- Unit pricing (for individual charging according to the availed service)

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 more (Reichenbach, 2008), up to 86 % in case of weight-based systems (Aschaffenburg, 2013b), are achieved. Figure 3.10 shows the development of the quantities per capita for the total waste, the disposed and the recycled waste from 1991 – 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 bio waste), collected door-to-door, was introduced in 1997 and the subsequent increase in the recycled waste and the decrease in disposed waste are obvious. In principal this example is representative, as the weight-based system is applied, the recycling rates are particularly high.

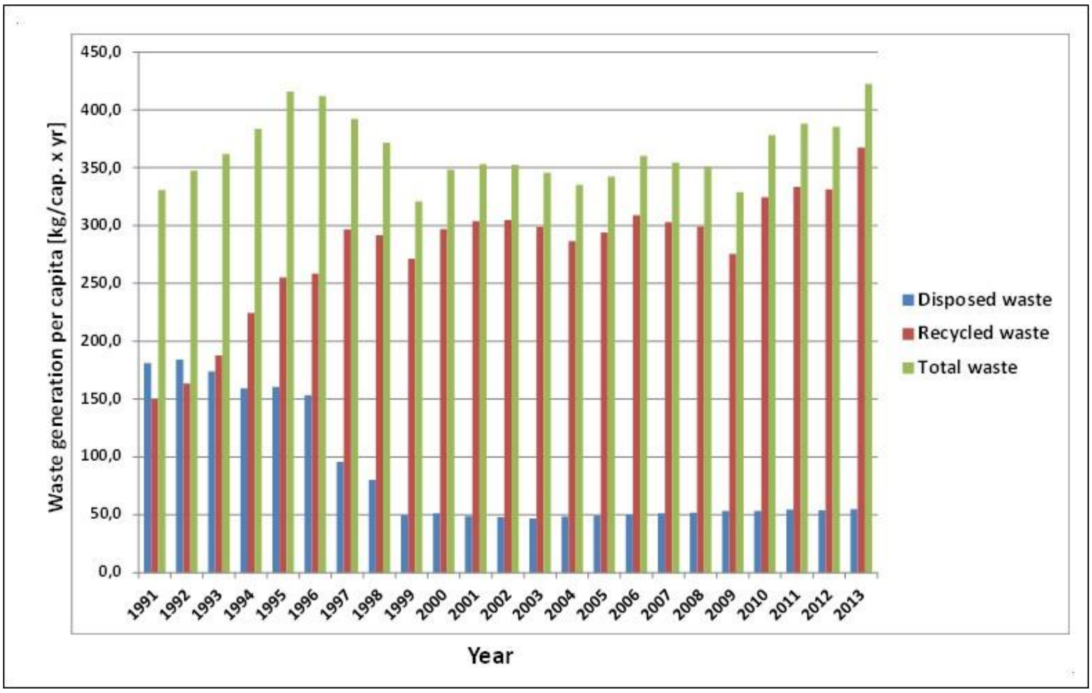


Figure 3.10. Development of the quantities of total waste, disposed and recycled waste from 1991 – 2013 of the County of Aschaffenburg/Germany (County Aschaffenburg, 2013b)

The reported recycling rates for weight-based systems vary significantly due to different levels of waste collection infrastructure and public awareness. Another example with 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 year, the residual waste quantity was 102 kg/capita x yr (see Figure 3.11).

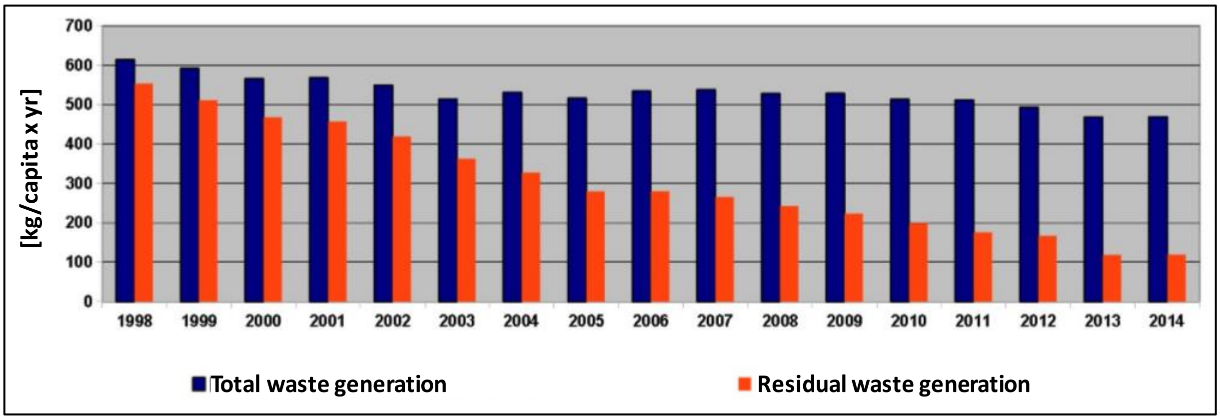


Figure 3.11. Development of the total and residual waste quantity in the municipality of Trento from 1998-2014 (Fedrizzi, 2015)

The same is true for Flanders, a region of Belgium, where first pre-paid 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 x yr (R4R Flanders, 2014). The development is indicated in Figure 3.12.

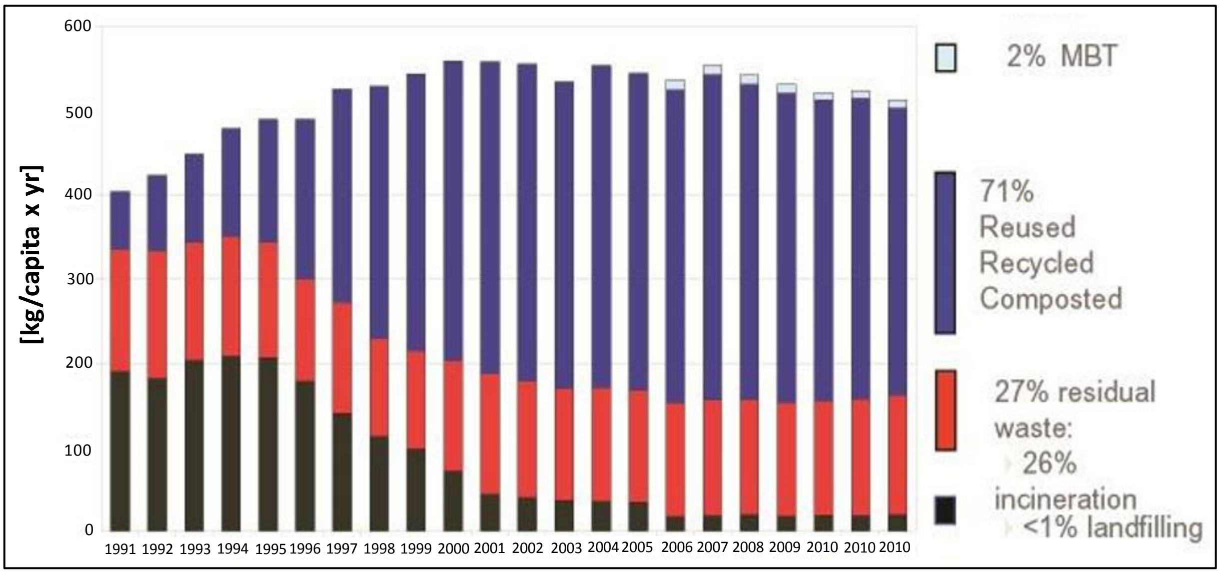


Figure 3.12. Development of recycled and residual waste as well as incinerated and landfilled waste in Flanders from 1991-2012 (R4R Flanders, 2014)

The pre-paid sack systems show also a significant decrease in the quantity of residual waste but the achievable figures are lower compared to optimum weight-based systems.

- In Switzerland, in average 391 kg/cap x yr are recycled which corresponds to 53.5 % of the total waste quantity (Switzerland, 2015).

- In Belgium, a reduction of residual waste of 44 % could be achieved (OECD, 2006).
- In Spain, a reduction of residual waste of 38 % could be achieved (OECD, 2006).

Appropriate environmental indicators

The most appropriate environmental indicator to assess a PAYT approach is the percentage of recycled waste compared to the total quantity of waste or the quantity of residual waste respectively. In addition, the capita-specific quantity of residual waste (kg residual waste (cap x yr) and the quantity of waste recycled (kg recycled waste/cap x yr) together with the quantity of total waste generated (kg total waste generated/cap x yr) are appropriate environmental indicators.

Cross-media effects

There is additional energy consumption for the separate collection of organic waste, paper and other waste fractions but this is by far outweighed by the savings gained from re-use or recycling of paper, glass, plastics and other materials.

When considering the implementation of PAYT, there is the argument that illegal dumping in the countryside or littering in the cities may increase. This is not the case where the infrastructure for the collection of waste is well-developed and easy to use and where citizens have an adequate environmental awareness. As a consequence, public awareness campaigns are very much required when introducing a PAYT system.

Operational data

The principal scheme of the weight-based system is illustrated in Figure 3.13.

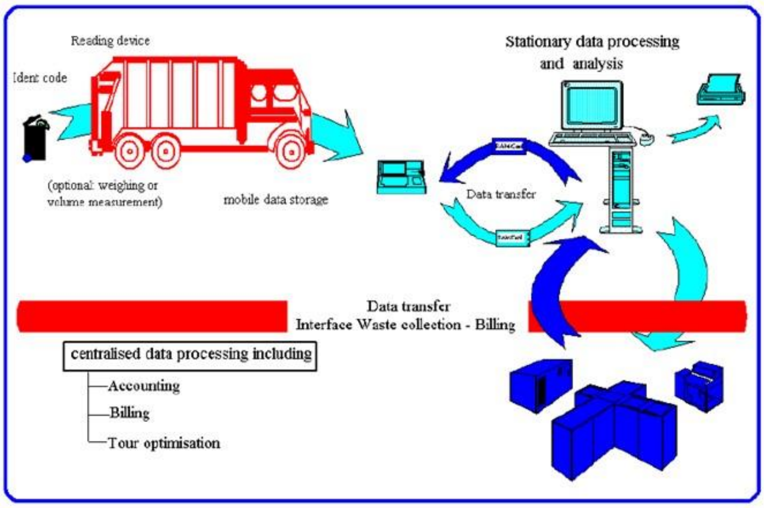


Figure 3.13. Process chart for the electronic identification and data transfer in a bin identification scheme (Bilitewski et al., 2004)

Figure 3.13 also indicates the option of a volume-based system but this is not further considered as the reduction rates with this system are very low. In contrary, the weight-based system, accompanied with well-developed infrastructure and citizen' awareness can achieve highest recycling rates and lowest residual waste quantity respectively.

In the example of weight-based PAYT schemes, all the waste bins are equipped with a chip and a bar code that can be read by a transponder or bar code reader. An example for a bar code is given in Figure 3.14 and examples for chips in Figure 3.15.

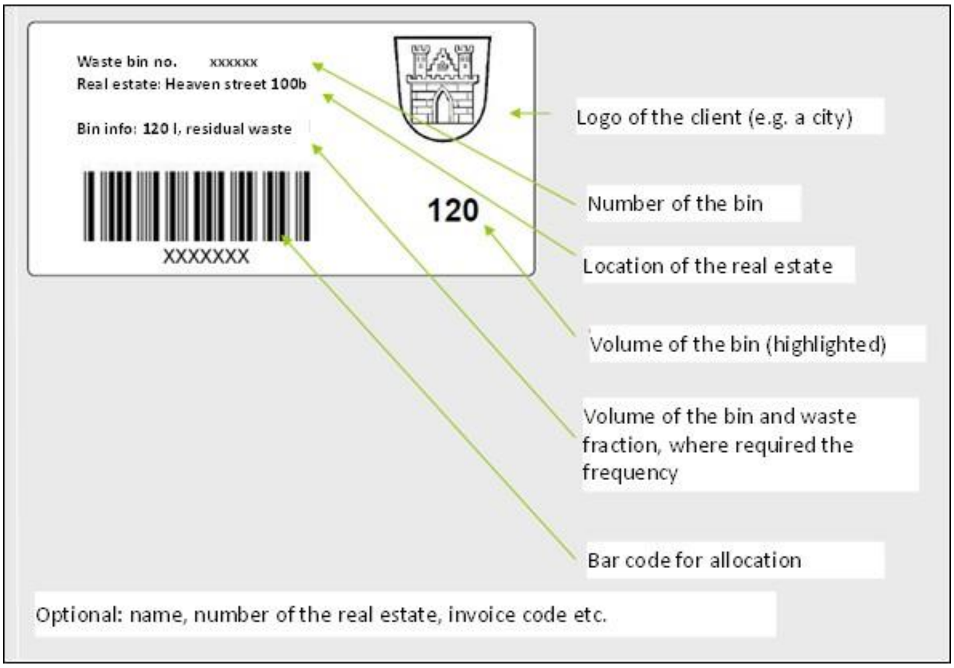
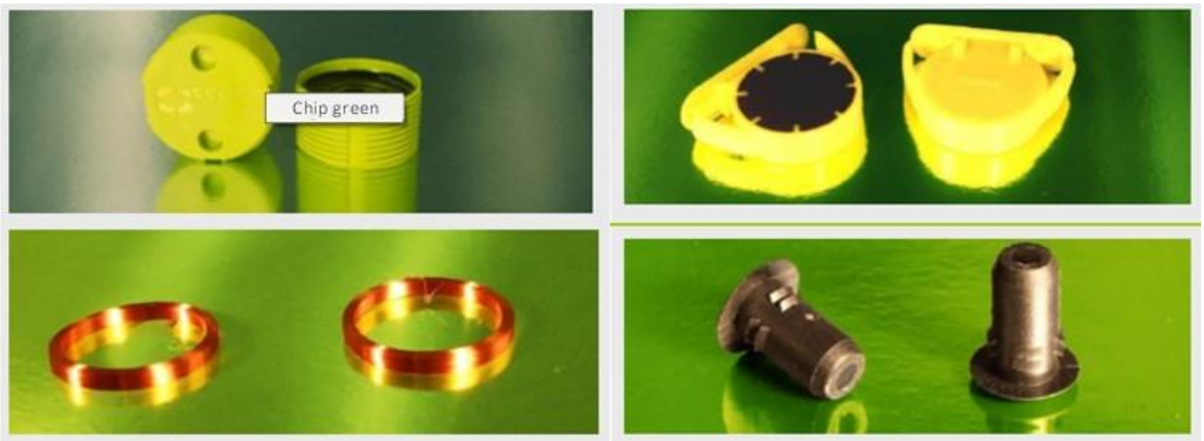


Figure 3.14. Example for the information automatically read by an identification system (c-trace, 2014[20])



Standard chip for new bins Chips for the retrofit of existing bins

Figure 3.15. Examples for chips for new bins (on the left) and for retrofitting existing bins (on the right) (c-trace, 2014[37])

[20] C-trace, requested information from the company c-trace , Bielefeld/Germany

Figure 3.16 shows a waste collection truck which is equipped with a waste identification system and a weighing system. The latter cannot be seen from the figure.

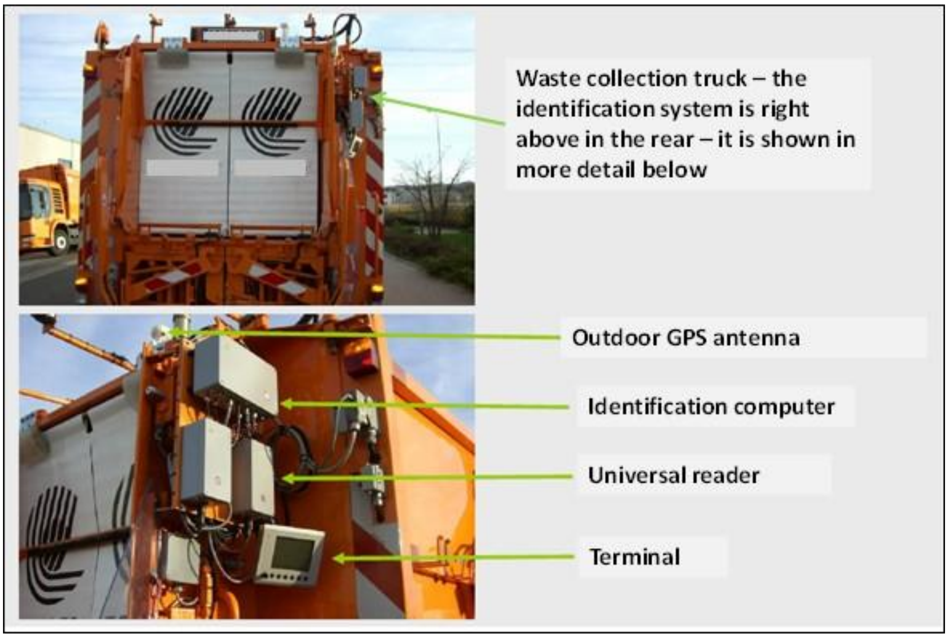


Figure 3.16. Waste collection truck which is equipped with a waste identification system (c-trace, 2014)

The weight-based system requires higher 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 (e.g. in Germany), the difference in reduction of residual waste between the identification and the weight-based system can be significant. In this case, also illegal dumping is negligible (County Aschaffenburg, 2013a).

For densely populated areas and high-rise buildings, container systems are in use to which only assigned people have access. In Figure 3.17, two examples for such container systems are presented.

Source: (Bilitewski et al., 2004)

Figure 3.17. Large bins or containers to which only defined persons have access to

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 the 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 chambers. Confidentially aspects can be managed and did not limit the application of the techniques so far; the privacy fears in the UK (e.g. Holmes et al., 2014) can be managed. At the time of introducing the system, there is a significant peak of work load for the municipality, city or county concerned as well as for the service provider (collector of the bins and chambers).

Further, as already stressed, a well-established infrastructure for the collection of the different waste fractions is required in order the citizens can get rid of certain waste fractions in an easy and comfortable way.

The environmental awareness of the 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 2.5, Operational Data, for more information). However, as indicated, this is not a big issue where the environmental awareness of citizens is well developed.

Economics

After implementation of the whole current waste management system the 2013 waste fee was lower compared to the initial situation 16 years before in the County of Aschaffenburg (PAYT approach with weight-based waste collection of residual waste and bio waste as well as separate collection of paper from all households, the operation of recycling facilities and composting of green cuttings (partly they are incinerated in a biomass-fired power plant) in all bigger municipalities, 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 bio waste, subsidies for composting on household level, for the use of re-usable nappies, and for families with incontinent persons). The calculation of the fee just before and just after introducing the weight-based system are publicly available (County of Aschaffenburg, 1995; County of Aschaffenburg, 1997). Although 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/Pavel, 2013) do not proof true. However, the extent of 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 cost; 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 3.3) although the county further invested in anaerobic digestion of the bio waste, 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 respectively) and also has low waste fees; the average waste fee is about 27 % lower as the average waste fee in Italy (Contó, 2015; Contarina, 2015).

In principle, these developments can be transferred to other municipalities.

Table 3.3. 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 4 persons household, 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, 2013a)

The fee after the introduction of the weight-based system represent an average value as all the bill are individual due to the variable fee for the weight

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 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

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, 2013a):

	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 bio waste, for which the basic fee is zero, and the residual waste.

Landkreis Aschaffenburg
- Müllgebührenstelle -

Landratsamt Aschaffenburg, Bayernstr. 18, 63739 Aschaffenburg
1

MUSTERMANN MAX
BEISPIELSTR. 35 1/2
38542 LEIFERDE

Öffnungszeiten Müllgebührenstelle
Mo.-Mi. 8.00-16.00
Do. 08.00-17.00, Fr. 08.00-12.00
Kommunikation
Tel. (06021) 394-396
Fax (06021) 394-944
eMail: abfallwirtschaft@Lra-ab.bayern.de
Gläubiger ID
DE7610000000010338
Müllsonderkonto des Landkreises:
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), bio waste (60 l) and paper/cardboard

In a country with a hot climate, the collection frequency for bio waste 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 forces for implementation

In many cases, waste managers in municipalities were motivated to implement the PAYT approach where landfills were exhausted, where fees were high and/or the public environmental awareness called for a change. Further, in some Member States, the landfill of untreated municipal waste was already banned before the EU-wide restrictions came into force[21].

Reference organisations

About 10 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) and in France (City of Besançon). It is also practised in the US (Skumatz, 2002, 2008; Hall et al., 2009).

The pre-paid sack system is widespread in Switzerland (Bilitewski et al., 2004, Switzerland, 2015) and is applied in Belgium, the Netherlands, Denmark and in few cases in Italy and Spain (Catalunya, 2010).

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3.5.4. Awareness raising

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 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 environmental benefits associated with that (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, etc.).
- Attitudes and perceptions: not accepting there is a need to recycle, being insufficiently motivated to sort waste and recycle.

A particularly effective way to improve attitudes towards waste re-use 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 school children 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 including third sector 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 "Responsibility Organisations" (PROs). PROs are collective entities set up by producers or through legislation with responsibility for meeting recovery and recycling obligations of the individual producers.

Best practice measures

Best practice in awareness raising is to effectively encourage waste prevention, re-use 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 multi-lingual or pictorial communication and via school activities.

The following critical elements of effective awareness raising should be embedded in all awareness raising campaigns (Zero Waste Scotland, 2012):

- Ensure consistency and clarity of communications with well-defined aims and objectives.
- Create clear messages appropriate to, and directed at, well-defined target audiences.
- Add impact through continuity and consistency, ensuring that communication activities build on each other.
- 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 3.4. Examples of how some of these channels have been used are provided under *Operational data* and *Reference organisations*, below.

Table 3.4. 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 re-use in section 3.8). Also roadshows, seminars and door-to-door campaigns.
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)

Table 3.4. Communication channels appropriate to various methods of awareness raising

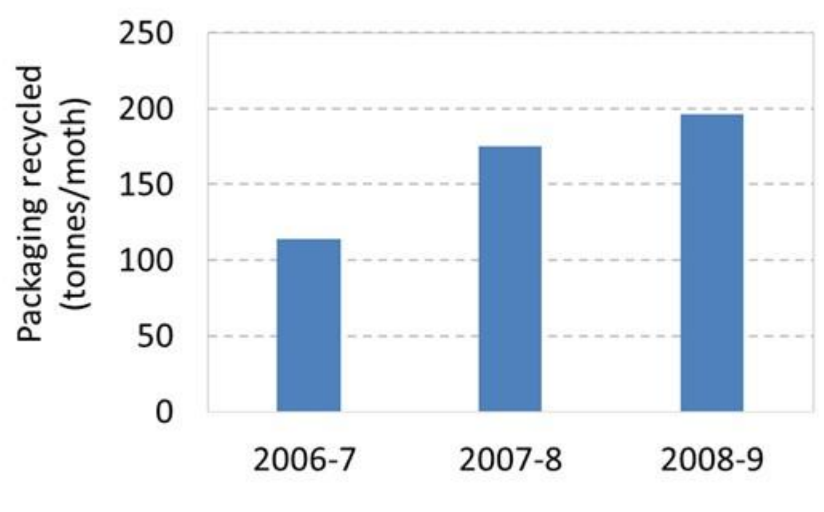
Methods	Communication channels
Product labelling	Producers may engage with other stakeholders, especially waste management organisations, to deliver communication to consumers via all of the above pathways within extended producer responsibility schemes. In addition, producers may clarify use-by dates, 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 WMO staff http://www.zerowastepro.eu/publications/

Source: Zero Waste Scotland (2012), Vienna City Council (2013), R4R (2014a), (EC 2014), own elaboration.

Achieved Environmental Benefit

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 re-use 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, bio-waste, 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 3.18).



Source: Derived from data in R4R (2014b)

Figure 3.18. 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 -68 % and -38 %, respectively, disposed cigarette butts -31 %.

They also reported that 1,100 tonnes of dog poo is collected every year in disposable bags provided from street dispensers.

Appropriate environmental indicator

Ultimately, effective awareness raising should reduce waste generation and increase recycling rates over time, reflected in the following key performance indicator at the relevant level (e.g. municipality, national):

- Residual waste generated, kg per capita per yr

The most practical definition of "residual waste" from the perspective of WMOs 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 WMO is involved in subsequent sorting (e.g. in sorting plants following co-mingled collection, or in mechanical and biological treatment plants).

A complementary indicator that reflects the efficiency of citizen waste sorting is:

- Contamination rate of individual waste streams (% weight of individual waste streams collected that is rejected for the intended recycling or recovery purpose)

However, it may be difficult for WMOs to obtain accurate data on contamination rate in cases where downstream processors do not report back to the WMO, which is likely to be the case where waste is shipped long distances for processing.

The effectiveness of a particular campaign could be monitored by comparing performance before and after the campaign, and reported as e.g. percentage reduction in residual waste generation, or percentage increase in the rate of separately collected recyclables, etc. However, residual waste generation rates can vary throughout the year and in response to many other factors, so that it is rarely possible to attribute short-term changes to a particular campaign (Zero Waste Scotland, 2012).

More specific metrics may be relevant for campaigns targeting specific waste fractions. For example, a campaign targeting the correct return of WEEE items could record the volume of WEEE returned to relevant collect points before and after the campaign, and the percentage change.

The size of the direct audience for a particular campaign may be estimated by recording e.g. the number of "hits" on a website or social media forum, or through knowledge of readership or audience numbers for relevant print media and television stations where advertising is undertaken, or from viewing statistics provided by advertising companies managing particular campaigns. Based on this, a direct indicator of dissemination associated with awareness raising could be:

- Percentage of citizens in the waste management catchment area receiving awareness raising messages over a given time period, (e.g. % 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 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 3.5).

Table 3.5. 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 – light-hearted 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 3.19, 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 3.19. Screenshot of a guide produced by WRAP providing an overview of various activities, highlighting suitability for different audiences and topics addressed

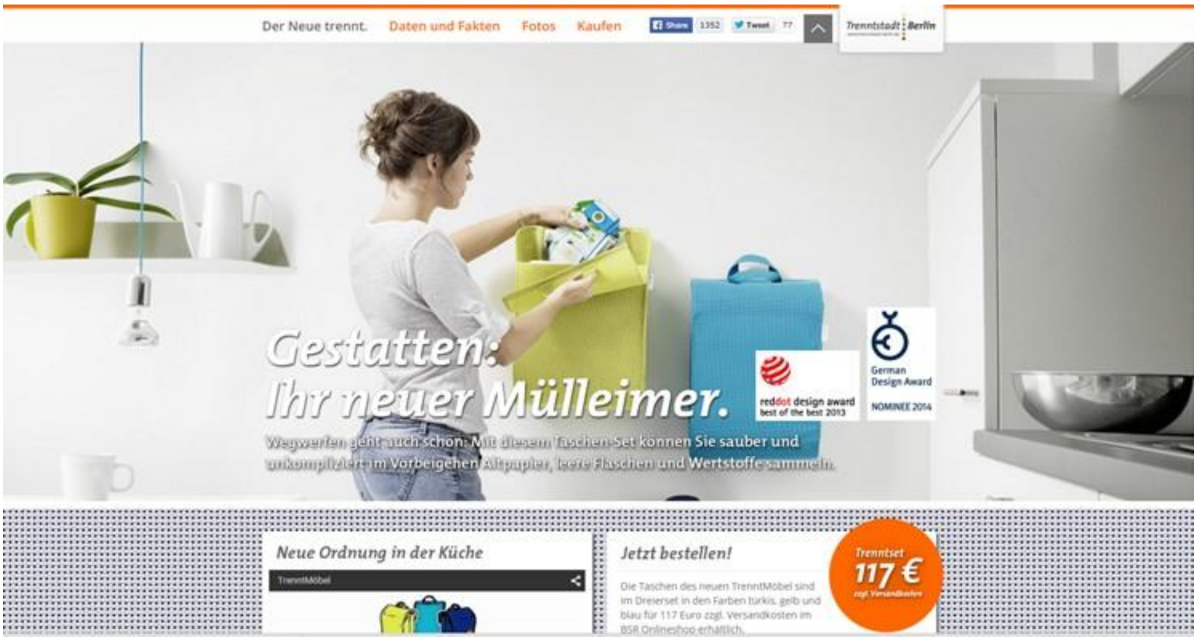
The ZerowastePro project has produced similar guidance, in the form of a recommended educational programme template that can be implemented by schools or other public organisations:

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 the bully pulpit
- 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*[22] campaign makes extensive use of social media sites, and includes the marketing of attractive recycling storage bags as "fashion accessories" (Figure 3.20).



Source: http://www.trenntmoebel.de/

Figure 3.20. Screen shot of the online shop marketing fashionable recycling storage bags and bins as part of BSR's *Trenntstadt* campaign for Berlin residents

[22] 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 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.

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 post code to locate their nearest WEEE collection point (Figure 3.21).

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 re-use 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 3.4.

Source: Recycle Now (2015).

Figure 3.21. 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 (1^{st} to 2^{nd} 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 organizations 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 a 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 in 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, made 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 deadline for completion of the project – which is exclusively associated with the fulfillment of objectives, not compromising the normal activity development of these institutions.

The PLG+ currently involves 141 institutions in an intervention universe that will reach over 40,000 citizens directly, consolidating LIPOR's regional strategy. So far, LIPOR have undertaken 137 environmental audits, covering 1,215 activities and 23,489 persons. 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 store 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) suggest 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 batteries 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 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 EUR 1.00 and EUR 1.50 per household, and for intensive communications activities for "hard to engage" residents costs may increase up to almost to EUR 3.00 per household (Zero Waste Scotland, 2012).

School activities and events may be paid out of national, regional or local government education budgets.

Producer Responsibility Organisations (PRO)

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 PROs own communication initiatives), to ensure participation of consumers with in 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 incurred 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 its efforts in educating school children 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, bio-waste, batteries and WEEE.
- SYBERT, France, has an extensive campaign educating citizens on waste management using various media, including theatre and videos.
- Tallinn City Council, Estonie, 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.

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WRAP (2015b). Introducing Love Food, Hate Waste - Save More. Available at: http://england.lovefoodhatewaste.com/sites/files/lfhw/LFHW %20Save %20More %20Introductory %20pack %201 %20-%20Introducing %20Love %20Food %20Hate %20Waste %20Save %20More.pdf Last access June 2015.

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3.5.5. Municipal waste advisors – practical work, qualification, role, impact

Description

This technique is described on the basis of the experience made in Austria, and drafted by the actual manager of the scheme.

Municipal waste advisors were first established in Austria in 1986 and are permanent full time employees of regional or local public waste authorities with the main focus on awareness building, public education of the population, PR and communication work on waste prevention, re-use, separate waste collection and sustainable consumption and lifestyles in general within the local or regional context. Their target groups are private households and small enterprises of their region. Additionally, they consult their regional waste management organisations in planning and implementing collection schemes, projects, campaigns, co-operations with private waste management companies and provincial and federal authorities.

The underlying idea is 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 approach is: "educate the population to prevent and separate waste instead of paying for expensive techniques to sort mixed waste or to dispose it".

Until 1995, the qualification consisted of a six months of permanent training, the trainers were experts mostly from public and private waste management organisations, authorities, NGOs, scientific institutions and communication and other experts. A large proportion of the training were site visits, mainly of waste treatment facilities and innovative production facilities which have introduced environmentally-friendly techniques. After 2000, the system changed due to financial restrictions (see operational data).

Achieved Environmental Benefit

The waste advisors certainly contributed to the high waste management performance in Austria where the recycling rates increased from around zero (1980) up to over 70 % in some regions like Styria today. Of course, to reach such rates, comprehensive federal waste legislation since 1990 and landfill taxes that significantly increased since the early nineties were also relevant factors. Against this background, it is not possible to quantify the contribution of the waste advisors to the increase of recycling rates. However, despite the introduction of waste advisors, the total waste quantity generated was increasing in the past 30 years but, most probably, the increase would have been much higher. However, it is not possible to quantify their influence on waste prevention.

Appropriate environmental indicator

The contribution to waste prevention cannot be quantified generally, but specific waste prevention projects with measurable effects like broad regional implementation of washable baby diapers, home composting, rejection of postal advertising by private households are still present today. A compared to other countries significantly high (although shrinking) proportion of refillable bottles in beverage retail and many more waste prevention examples would not have been possible without the permanent work of waste advisors. The biggest challenges of municipal waste advisors today are the prevention of food waste, prevention of littering and the broad implementation of re-use and preparation for re-use activities.

Cross-media effects

No cross-media effects are known.

Operational data

Since 1986, the first year of qualification, about 20 to 30 persons per year passed the training programme, with lower numbers in later years.

In the following years the duration was more and more reduced parallel to shrinking public funding, until it stopped completely in the year 2000. From the beginning, other institutions, especially other Austrian provinces, conducted their own qualification programs, which led first to rapidly rising numbers of advisors in the respective regions, but later had the effect, that after a few years there was not enough demand for this qualification any more. This resulted in the present situation, that there is no more specific qualification for municipal waste advisors at all.

Presently, the yearly low fluctuation of advisors of 10 to 20 persons (out of 410), lack of funding and a reluctance of municipalities to invest in qualification of their newly employed advisors makes any attempt of re-establishing a special qualification economically unfeasible in Austria. The present „newcomers" in the job usually are graduates of environmentally or education-related studies, some pass a 3-weeks training in environmental and waste legislation. All of them have to obtain the special knowledge and skills for municipal waste advising „on the job" by their own initiative, without standards or regulations.

Currently there are talks on federal level to reestablish at least some form of specific vocational in service training programme and a rough standard or guideline at least for those receiving staff cost contributions from packaging waste collection schemes.

As of 2016, meanwhile 410 municipal waste advisors are the backbone of public waste management communication and PR work. This means an average of one advisor for 20.000 inhabitants. More than 50 % of these advisors still are the graduates from the original training programme of ARGE Müllvermeidung, showing a high level of continuity within the profession.

Applicability

The implementation of municipal waste advisors requires an initial commitment of at least one region (province, big city) of more than 1 million inhabitants, to ensure economic feasibility of the development and implementation of a qualification and training programme as well as continuity of step by step implementation of waste advisors in all regions and municipalities. One of the success factors of the programme in Austria was that the main load of funding came from the federal labour agency (AMS) within a broad national initiative for the creation of new and innovative jobs on the background of rapidly rising unemployment rates (also within well qualified groups!) in the 1980s and early 1990s.

Environmental funding budgets would never have allowed such extensive funding. Since Austria at that time was not a member of the EU, no EU-funds were available. The funding was a long term political commitment of AMS within a long term general national funding programme for the creation of new jobs („Aktion 8000", the so called „experimental labour market policy"), which facilitated funding application for municipalities intending to employ waste advisors and send them to the training programme.

Economics

Initial funding in the first years

- Concept & qualification measures: funded by national Labour agency (AMS)
- Staff costs for consultants during training and employment in municipalities in starting period: either 1 year 50 % or 2 years 30 % of total staff costs funded by Labour agency (until 2000)
- Total costs after AMS funding expired: municipalities which employ advisors, provincial subsidies in one province (Styria, about 10 %, until 2008), staff cost contributions of packaging waste collection scheme(s) (about 20 – 30 %).

Present funding

The financing of the staff costs comes from the overall municipal waste management budget which in Austria consists of residual waste fees from the 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 advisors.

Since 1993 the packaging collection scheme(s) partly contribute to the staff costs, in return the municipalities provide the service of covering also the communication work for prevention and collection of packaging waste which legally is the obligation of the scheme(s).

The idea of financing education for waste prevention and recycling out of residual waste fees and packaging schemes may look like a contradiction but is the key factor for stable funding and continuity of public responsibility.

As a summary, the system of waste advisors is associated with

- new jobs created for concept and qualification (1984 – 2000): 4
- new permanent jobs created for waste advisors since 1986: 410
- additional permanent follow-up jobs in waste management and recycling industry since 1986: 10.000 – 20.000 (rough estimate compared to countries with only little separate waste collection)
- Initial labour market agency investment of EUR 15 million for waste advisors over a period of 15 years created permanent yearly wage tax income of over EUR 30 million from the additional jobs in waste management and recycling since then. This shows that investment in human resources and public education in the waste sector creates a net profit for the national economy.

Driving force for implementation

As an innovative solution to severe waste problems (lack of landfill capacities and treatment facilities) of the 1980s, leading to broad public political discontent in the context of the ecological/"green" movement of that time, „ARGE Müllvermeidung", a small environmental NGO promoting waste prevention, invented the concept of "municipal environment & waste advisors" and implemented it between 1986 and 1995 step by step all over Austria with labour agency funding for training and employment and with the support of the provincial government of Styria and the federal ministry for environment.

Within only a few years, the waste advisors successfully transformed the public discontent into highly motivated action and contributions of the majority of citizens to separate waste collection which subsequently led to political acceptance.

Between 1990 and 1993 some provincial waste laws (Styria, Salzburg, Tirol, Upper Austria) integrated obligations for municipalities or regional municipal associations to provide waste advising 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 advising.

Reference organisations

ARGE Müllvermeidung (association for waste avoidance), Puchstrasse 41, A-8020 Graz (phone: +43 316 71 23 09-0; e-mail: office@arge.at; www.arge.at)

VABÖ – Verband Abfallberatung Österreich (association of Austrian waste advisors), Trappelgasse 3/1/18, A-1040 Wien (phone: +43 699 100 51 038; e-mail: neitsch@repanet.at; www.vaboe.at)

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3.6. Enabling Techniques on Strategies for MSW

3.6.1. Performance-based waste management contracting

Description

The inefficiency of municipality contracted services usually happens 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. 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 this document, performance-based contracts in waste management are considered **an enabling technique** since they may facilitate the implementation of techniques leading to best performance. But this link is not obvious. It may be a financial instrument created to ease the implementation of techniques considered best practices, but not a best practice itself. In contrast to energy contracting, not many exemplary approaches for performance-based contracts for waste could be found. 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 Recycling for Regions (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 shifts the public sector to an only evaluation or measuring role. Also, Hogg et al., 2014, performed a theoretical study of the plausible impact of performance-based contracts and some conclusions were derived:

- Performance based contract are likely to work better to improve the performance of the system as incentives at collection, and less at the treatment.
- The municipality needs to develop a full set of indicators and develop monitoring practices
- Baseline has to be defined, and the influence of the variation in external conditions (economic, social, regulations, etc) has to be well taken into account in the benchmark mechanism.

The study does not include any example of its application, but plausible scenarios analysis in a theoretical perspective. In the light of these conclusions, it is concluded that the application of best environmental management practice (e.g. waste monitoring, PAYT, etc) enables the use of performance-based contracts. For systems with outstanding performance and a solid strategy, performance-based contracts would be a tool for optimisation. Unfortunately, no example is derived on this regard.

The key is to create a *win-win* situation for both the customer and the contractor, since both participate from 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 to 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 in three premises (U.S. EPA, 2004): (i) cost-effective opportunities to reduce waste, (ii) financial incentives to contractors to pursue recycling and reducing 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 spectra of techniques (prevention, re-use, 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 provides technical and/or administrative tasks (e.g. repairs, meters, etc.).
- Management contract: the private organisation takes over operation and management, although the user or client remains legally as responsibility of 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, finances a new project that 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 co-operator 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, being the private organisation bearing the risks. Public sector and independent regulatory agencies are in charge of supervision of the performance.

Table 3.6 shows how these contractual arrangements distribute responsibilities in the different stages of a performance-based contract.

Table 3.6. 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 Benefit

As an enabling technique, performance-based contracting eases the implementation of best environmental management practices, and, therefore, may result in a better environmental performance by:

- 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 burdens.
- 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 opposed to conventional contracts, paid per volume collected or treated, so the reduction of waste volume generated is against the economic performance of the service, while recycling sometimes is even not considered in terms of the contractor performance.

Appropriate environmental indicator

This is one of the key aspects of a performance-based contract and it is directly linked to waste performance monitoring (section 3.5.2). The U.S. Environmental Protection Agency identified that a very first benefit of a performance-based contract is the improved data tracking and reporting (U.S. EPA, 2004). The indicators to be used in a performance-based contract for a waste service provider should be those of a frontrunner approach. See waste monitoring for practitioner examples (section 3.5.2).

Objectives setting

Based on the indicators and metrics agreed for the contract, the regular monitoring and revision of the system is a responsibility of the waste authority. The performance should be benchmarked against certain objectives. In Bristol, UK, a minimum carbon footprint reduction of 25 % (from the baseline calculation before the contract) was established. However, benchmarks should be in accordance with the feasibility of the application of best environmental management practice. In terms of best practice, performance-based contracting provides the economic drivers and aligns the interests of the waste management contractor with those from the authority.

The indicators and objectives system can be based in the conventional performance monitoring of waste management contracts. The Chartered Institution for Waste management (CIWM) in the UK provides an exemplary contract for performance monitoring, establishing a list of performance standards to be set up in the schedules of contracts (see Table 3.7).

Table 3.7. Example of indicators used in a waste management contract (adapted from CIWM, 2009)

#	Performance indicator	Monitoring frequency
1a	Missed Collections % of missed household waste collections (per 100,000 collections)	Monthly
1b	Rectification of missed collections % of missed household waste collections rectified within 24 hr	Monthly
2a	Missed collections – non residential % of missed non-residential collections	Monthly
2b	Rectification of missed collections – non residential % of missed non-residential collections rectified within 24 hr	Monthly
3a	Recycling tonnage Tonnage of Recyclables collected	Annually
3b	Recycling rate Measured in accordance with the Audit Commission requirements for reporting Best Value performance indicators	Annually
3c	Recycling participation rate Measured in accordance with official WRAP guidance	Quarterly
4	Customer satisfaction The percentage of residents who are satisfied with - Household waste collection - Recyclables collection service (Results taken from an independent survey agreed with the Contractor)	Annually

Cross-media effects

Performance-based contracts are designed to remove cross-media effects from conventional contracting. The environmental beneficial performance of performance-based contracts is not always ensured and their benefit against conventional contracts can be in dispute: 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 program, shows that resource management contracting is quite applicable to business, institutions and municipalities.

In terms of waste management, clear differences are established between performance-based and conventional services (Table 3.8).

Table 3.8. 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 maximize waste service and volume.	Contractor seeks profitable resource efficiency 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 EPA detected through the analysis of several case studies is that traditional waste contracts typically pay a unit price based on the weight of trash collected, number of pick-ups and container rental fees, while recycling is not considered as a driver for any contractor. In terms of performance-based contracts, the contractors' profitability depends directly on e.g. recycling rates, diversion from landfill, and other indicators. This is done by establishing a fixed price to the waste management service and introducing bonuses to good performance and penalties to 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 best environmental performance of the contractor in charge of collection is directly linked to the profits.

Conventional waste contracting also results in little communication between 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 the 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, re-use, 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 by 2020. As the duration of the contract is 2011-2017, a pro-rated basis of 25 % was defined in the call for tender, using as a baseline the emissions data from the previous contractor in the period 2009-2010. 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 reduce 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 has 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-performance monitoring system is a certain pre-requisite 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 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 a higher price of the service that would then be charged to the citizens.

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 be then as many compensation options as contracts signed under performance-based clauses.

- Option 1. Pass-through of service costs with shared savings and performance bonus. Costs are established from the basic financial proposal in the bid, then, costs savings are shared between the waste authority and the contractor. Examples of savings opportunities are diversion of materials towards recycling, handling and hauling more efficient 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 e.g. 30/70 % for the contractor if the overall savings are over 5 %. Below 5 %, all savings go to the public authority. Then performance bonus/penalties can be given through the increase/reduction of 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 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 a year-to-year accounting.

Examples of implementation

The case in Bristol, UK, showed that the time to prepare the tender and the dialogue and negotiation took twice the time of a conventional contract, although its evaluation is not more complex. This factor adds an extra administrative difficulty 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 performance-based contracts could be found. However, these examples have been successfully implemented to other areas of public procurement, as energy efficiency of buildings, information technologies, 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 of the total amount of waste can be assumed as positive from the contractor perspective, however, performance-based contracts would link waste prevention actions or programs 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 that eventually would reduce costs.

Reference organisations

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Bristol City Council, bristol.gov.uk

European Commission, Green Public Procurement, http://ec.europa.eu/environment/gpp/index_en.htm

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3.7. BEMPs on Waste Prevention

3.7.1. Local waste prevention programmes

Description

The term `waste prevention` is defined in the Waste Framework Directive (WFD, 2008), and on top of the waste hierarchy, prevention measures that lead to the reduction in the amount of waste are of first 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 materials inputs and reduction of waste outputs (dematerialisation) (Kranert, 2009; Grooterhorst, 2010a, 2010b; van Ewijk and Stegemann, 2015; Gharfalkar et al., 2015). 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; EC Guidance, 2012). In this document, the focus is laid on waste prevention measures that can be implemented at the regional and local level.

Following the definition of waste prevention, the measures include those to avoid waste and those to re-use waste products or waste materials. For the identification of these measures,

- 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), and
- waste prevention plans of regions, cities or counties

have been considered. In many cases, in these documents the focus is laid on general strategies and recommendations and only a few concrete measures are mentioned. The proposed approach for the development of a waste prevention programmes is shown in Figure 3.22.

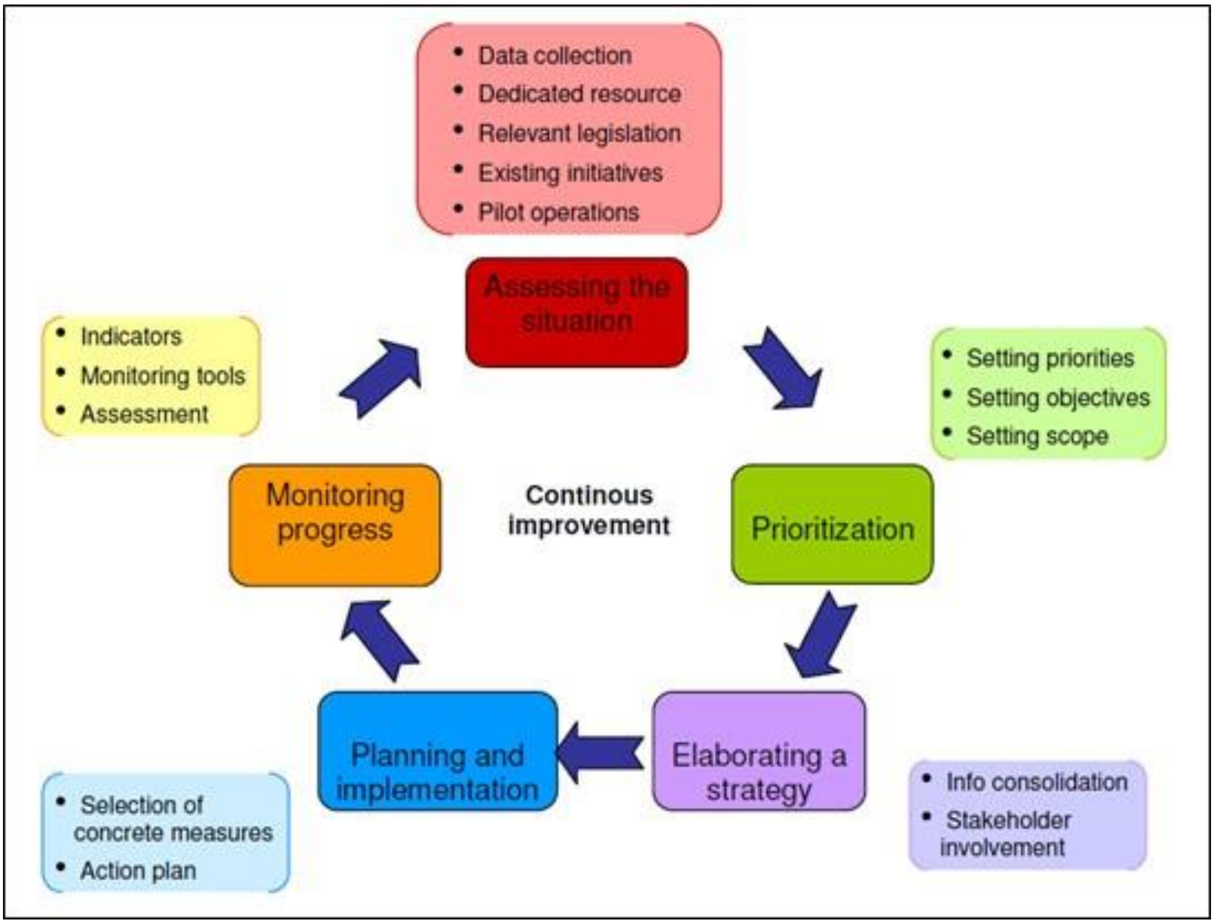


Figure 3.22.Developing a waste prevention programme (EEB, 2012)

When starting to identify measures of waste prevention at the regional and local level, it may be appropriate to focus on most relevant waste streams, such as food and bio-waste, paper/cardboard, plastic (packaging), glass, and textiles (see for instance Welsh Government, 2013, or Barcelona City Council, 2013). In the past years, specifically, the prevention of food waste has been discussed (Sharp et al., 2010a; Cox et al., 2010; European Commission, 2010, 2011a, 2011b). In Table 3.9, specific prevention measures are presented. They are grouped into measures for individuals and families and for municipalities, cities and counties or private organisations.

Table 3.9. Known waste prevention measures

Measure	Short description	Reference
For individuals and families (consumers)		
Little package	To buy things that are produced with as little package 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 package	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
Repair	To try to repair things before buying new items	Kuriso/Bortelo, 2011; 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, especially advertising, mail, for instance by a sticker on the mail box "no junk mail"	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 cacao 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 one-way bottles	Kuriso/Bortelo, 2011
My cup	To bring my own cup, e.g. to school or office	Kuriso/Bortelo, 2011
Needless package avoidance	To refuse needless package	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 re-selling	Kuriso/Bortelo, 2011; Graz, 2015
Bottled water avoidance	To try not to buy bottled drinking water	Kuriso/Bortelo, 2011; Florence, 2014
Reduction of food waste	To try to buy only the quantity of food To can consume, correctly store purchased food, cook adequate portions and use leftovers	European Commission, 2010, 2011a, 2011b and 2015; Sharp et al. 2010a; Cox/Giorig et al., 2010
Reusable nappies	To use reusable nappies (supported by the county or city)	Aschaffenburg, 2015

Table 3.9. Known waste prevention measures

Measure	**Short description**	**Reference**
Mobile dishwasher for festivals	To use a mobile dishwasher (provided by the county or city) for festivals to avoid one-way dishes and cutlery	e.g. Vienna, 2015; Rems-Murr County, 2015
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, 2015; Rems-Murr County, 2015; BMU, 2013; Graz, 2015
Reduction of canteen waste	To provide reusable dishes, cutleries, napkins and tablecloths as well as tap water and draught beverages in our canteens	
Reusable nappies	To financially support the use of reusable nappies	e.g. Enfield Council, County of Aschaffenburg, Besançon region
Lunch boxes	To provide school kids with reusable lunch boxes	e.g. Rems-Murr County, Barcelona, 2013
Repair shops	To support the set-up of repair shops	e.g. City of Vienna, Wales, 2013; BMU, 2013; or City of Graz (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 etc.)	Graz, 2015
Reduction of food waste	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 that citizens shall try to buy only the quantity of food they can consume	e.g. City of Vienna, Wales, 2013; BMU, 2013)
Pay-as-you-throw system (PAYT)	To introduce pay-as-you-throw-systems	see the BEMP on PAYT (Aschaffen-burg, 2015; Schweinfurt, 2015; BMUB, 2013)

Source: Own elaboration from different sources

Many of the measures mentioned in Table 3.9 are for consumers. The change of consumption patterns requires targeted awareness campaigns taking into account psychological mechanisms and the multi-faceted 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 re-use of products, such as furniture, electrical and electronic equipment, clothes and home textiles, books, bicycles, carpets, plants, toys, dishes, equipment for animals, etc., there are a number of web-based platforms established by public or private organisations to exchange products and goods for free (public websites) or on costs (a number of privately organized websites), e.g.

http://www.verschenkboerse-lk-aschaffenburg.de/list.asp

http://mainz.freeyourstuff.eu/

http://www.wien.gv.at/webflohmarkt/internet/

www.tauschticket.de

www.dietauschboerse.de

https://11870.com/pro/buy-recicle

Achieved Environmental Benefit

Although waste prevention has high priority, the prevention potentials appear 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 3.23, which shows the development of total municipal waste amount in Germany, consisting of the fractions: light packaging/plastic, glass, paper/cardboard, bio-waste and residual waste. Despite the fact that waste prevention was always given high importance in Germany, total waste quantity slightly increased. Probably, the increase would be even higher without prevention measures but their impact does not seem to be significant. Thereby, quantitative measurement of waste prevention is notoriously difficult as there is the basic problem to measure something that is not there (Sharp et al., 2010b; Zorpas and Lasaridi, 2013).

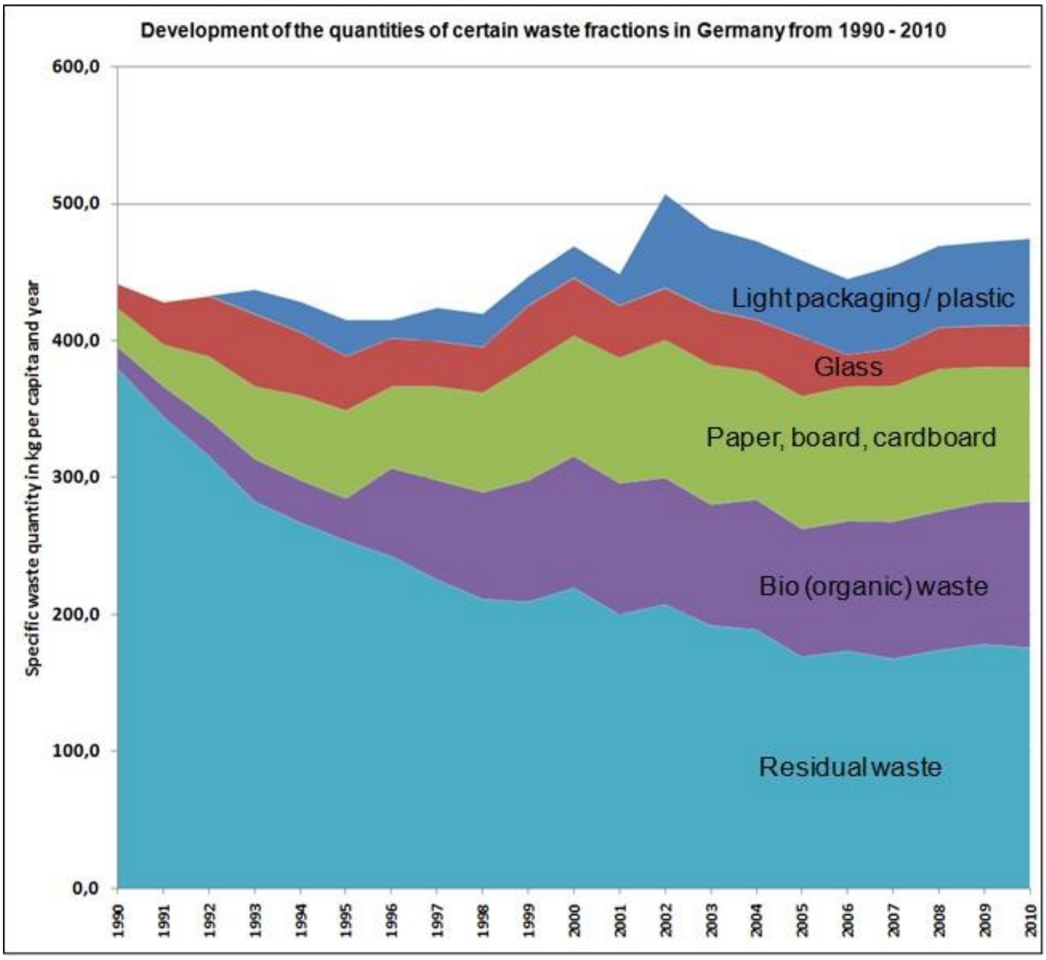
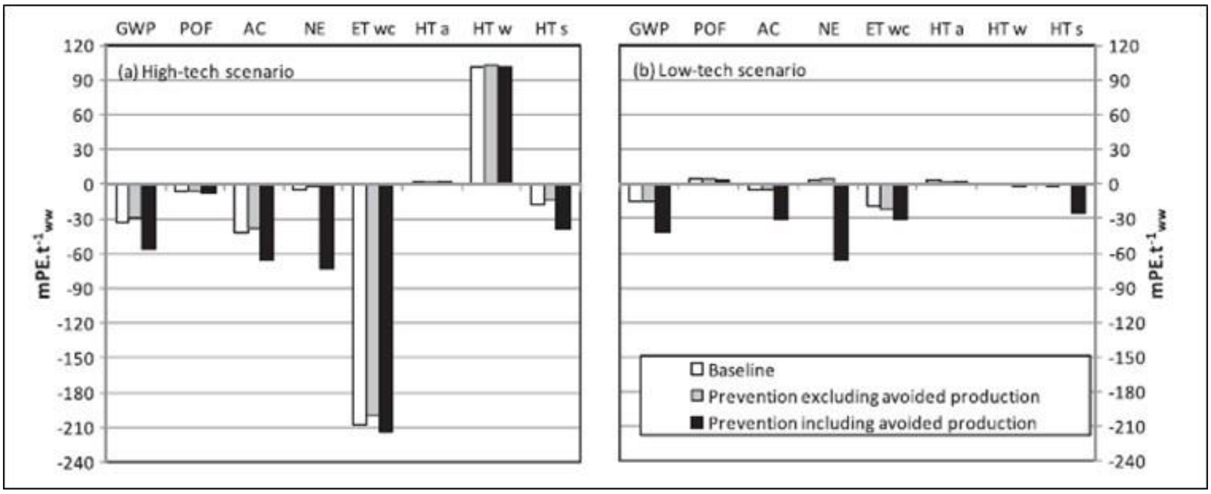


Figure 3.23. Development of the quantities of certain waste fractions in Germany from 1990 - 2010

The expectation that prevention means reduction of input mass streams and thus reduction of the environmental impact can be confirmed. Figure 3.24 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 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

Figure 3.24.Comparison of integrated waste prevention on two waste management systems. 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)

Appropriate environmental indicators

Appropriate environmental indicators are: reduction rate for the total municipal waste as well as for the different waste streams considered, expressed in kg per capita and year.

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 3.25.

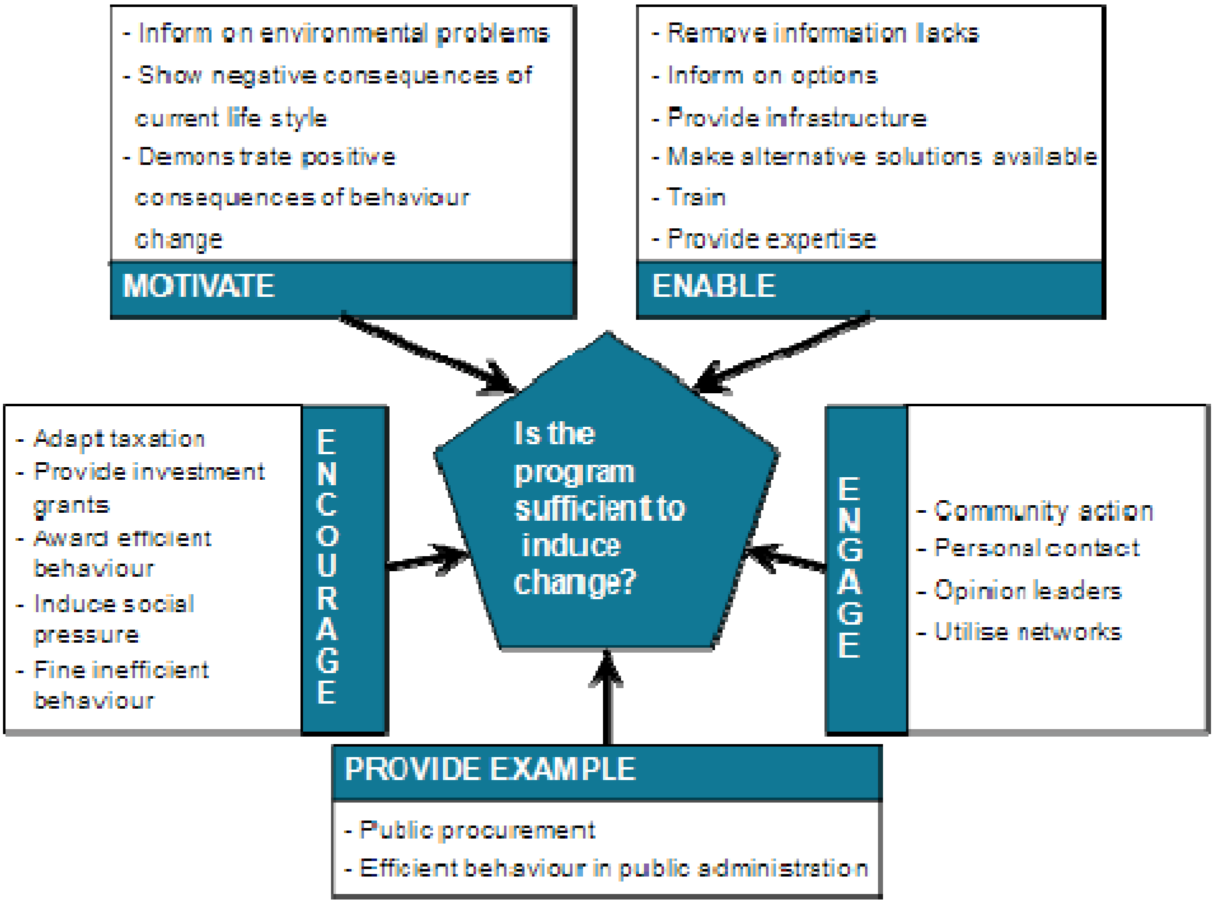


Figure 3.25. Aspects and steps to consider when developing a waste prevention programme (European Commission, 2011b)

It is important to develop a waste prevention programme/project specific for certain waste streams such as bio-waste, food waste, packaging, paper/cardboard, etc. The determination of the efficiency of waste prevention can be measured best for such waste streams. It can be expected that for food waste, the highest reduction rates can be achieved 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 questions of them.

Applicability

Waste prevention measures as such can technically be applied without limitations. However, the current economic system does not stimulate waste prevention. Thus, the application of measures should be supported by financial incentives.

Economics

There is not much 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 forces 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 and Copenhagen and the counties/region Besançon, Aschaffenburg, Schweinfurt and Île-de-France are references with regard to waste prevention (programmes).

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3.8. BEMPs on Product Re-Use

3.8.1. Product re-use schemes

Description

Background

Product re-use comes at the top of the waste hierarchy as a waste prevention measure that avoids environmental burdens associated with product manufacture and disposal or recycling. This BEMP addresses the implementation of re-use 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 re-use in second hand shops, considering the new product replacement factor associated with re-use of different types of product. They found that the greatest environmental savings arise from re-use of apparel products, due to the volume of items sold, followed by re-use of furniture products, owing to high environmental burdens from production of new items.

Best practice measures

Best practice is for waste management organisations to encourage diversion of re-usable end-of-life products away from waste streams and into re-use streams, through the active establishment or facilitation of second hand and municipal exchange markets (via repair workshops where necessary) or charity collections.

There are two key measures covered by this BEMP:

- Waste management organisations collect items for re-use and distribute to organisations, including charities, for sale or onward distribution.
- Waste management organisations establish effective information exchanges to advertise the demand for, and market the availability of, re-usable "waste" products.

Forming partnerships with third sector organisations and other stakeholders is an important aspect of best practice, as elaborated in case studies below.

Scope definition

In relation to electronic items, this BEMP covers re-use schemes that complement and go beyond the Waste of Electrical and Electronic Equipment (WEEE) regulations established under Directive 2012/19/EU. In particular, the WEEE Directive requires Member States to: (i) promote product design measures that facilitate re-use, 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 re-use and recycling targets established for national mass streams of WEEE. In particular, this technique encourages re-use streams that bypass WEEE collection, but is also relevant to the sale or provision for re-use of items collected under such collection.

Waste prevention by encouraging businesses and consumers to select re-usable products in favour of disposable products is addressed in Chapter 3.7.

Achieved Environmental Benefit

Product category benefits

WRAP's Benefits of re-use tool (WRAP, 2014) indicates the life cycle benefits for re-use of different waste categories within a UK context (Table 3.10).

Table 3.10. Environmental benefits achieved per tonne of product category re-used 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 (2014).			

Re-use network benefits

The Surrey re-use 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 re-use, 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 re-usable items in an Italian second hand shop:

- 160 t CO_2e/yr
- 7,000,000 MJe/yr
- 170 kg PM2.5e/yr.

National benefits

WRAP (2015) estimate that, during 2012, the emission of 1.5 Mt CO_2e was avoided in the UK through product re-use. This translates into a CO_2e saving from re-use of 23 kg per capita per yr.

Appropriate environmental indicator

It is difficult to measure re-use rates because of the multitude of pathways to re-use, many of which avoid product classification as "waste" and bypass waste management authorities. It may be possible for waste management authorities to monitor the quantities of potential waste re-use occurring via schemes they manage or facilitate, as indicated in the case studies below. In that case, an appropriate indicator would be:

- Mass of potential waste stream diverted to re-use (tonnes per annum), ideally expressed as kg per capita per yr in the waste management catchment or as a percentage of the baseline waste stream flow, and disaggregated by main product category (e.g. clothing, furniture, electrical equipment, transport equipment).

Alternatively, the quantities of relevant waste categories, such as textiles and bulky wastes, collected for recycling or disposal, expressed as kg per capita per year, will reflect changes in re-use rates over time, alongside other factors such as overall consumption rate.

Another important indicator is economic turnover realised through re-use, expressed as EUR per tonne of material re-used.

Related to this, the total economic value of re-used goods (EUR per year) sold within a municipality (DG ENV, 2009) is another potentially useful indicator that integrates the rate of re-use with the local economic value of re-use.

Watson et al. (2013) propose the share of second-hand products in total sales of textiles as a useful measure of progress in textile re-use, but note that data on total consumption of textiles by value would also need to be collated at the appropriate geographic level (e.g. municipalities), which may be challenging.

Cross-media effects

Re-use of most products is not associated with any significant cross-media effects. Transport distances for collection of re-usable items are unlikely to be greater than life cycle transport distances associated with production and disposal or recycling of new products.

However, for some kind 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 re-using 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 re-use 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 re-use, available at the following link: http://www.wrap.org.uk/content/how-guides-0. They describe how to:

- make re-use a strategic priority
- establish a re-use baseline for your area
- set up and run a re-use forum
- produce a re-use action plan
- write a communications plan to boost re-use
- provide for re-use on household waste collection centres (Figure 3.26)
- provide a re-use focused bulky waste collection service

Figure 3.26. Clearly identified bins accepting clothing for re-use at a community waste collection centre in Aschaffenburg, Germany

Establishing collaborative re-use networks

Successful re-use schemes involved multiple stakeholders, including third sector organisations that sell re-usable items to raise money for charitable causes or that distribute re-usable items to people in need, businesses, local authorities and government agencies. Coordination among stakeholders can reduce the costs of collecting and distributing re-usable items. Consequently, the establishment of local re-use 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 (2014) describes the role of a local authority in catalysing the establishment of a successful re-use 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 re-use. SCC worked with numerous independent local furniture re-use organisations with Surrey County, and realised that they could become more efficient if they pooled their resources. SCC therefore embarked on a project to increase furniture re-use across the county by:

- Enabling furniture re-use organisations to work as a county-wide network, delivering co-ordinated, high-quality services,
- Building capacity of furniture re-use organisations to handle greater volumes of furniture and white goods,
- Raising public awareness of the potential for re-use, and improving access to it.

Key steps and actions in the development of the re-use network are summarised below, based on information described in WRAP (2014).

Table 3.11: Key steps and actions in the development of the re-use network

Step	Actions
Engaging furniture re-use organisations	SCC offered grants to build capacity and quarterly furniture re-use credits, as well as funding a county-wide communications campaign, providing marketing support, and part-funding an interim manager. In return, each furniture re-use 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 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 re-use 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 re-usable items from household waste 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 (2014).

Training citizens in re-use and promoting re-use markets

Managing schemes that directly engage with citizens is also an important component of best practice. Training in basic repair work, and advertising repair services, are two simple measures that could increase re-use rates. Area Metropolitana de Barcelona (AMB) provides an example of collaboration among different administrations and organizations, 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 re-use. 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 city, re-use 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 up-cycling and re-use 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 re-use 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 re-use discarded miscellaneous waste streams such as books, clothes and small electronic appliances. Purpose built vehicles pick up goods from those novel 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 technique applies to all waste management organisations that handle any type of re-usable "waste" products, in particular garments, furniture and electrical appliances.

Economics

Waste management organisation economics

Local authorities or waste management organisations may work in partnership with each other, and with third sector re-use organisations, to efficiently design and implement re-use schemes. Such re-use 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 re-usable items.

Budgetary constraints may decrease opportunities for local authorities to organise and advertise re-use schemes, and to commission agreements with third sector re-use organisations (Ricardo-AEA, 2015).

Re-use schemes avoid recycling or disposal costs, and may even generate income if re-usable items are sold on.

Societal cost benefit analysis

In 2012, the third sector in the UK benefited by an estimated GBP 430 million through re-use, and re-use 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 re-use in 2012. The Surrey re-use network described above provides goods to approximately 5,000 low-income household families each year (WRAP, 2014).

Re-use of materials can generate turnover of up to EUR 1,500 per tonne, over ten times more than the turnover generated by recycled materials (TWG, 2015).

Driving force for implementation

Waste re-use schemes can significantly reduce waste handling and disposal costs for waste management organisations, facilitating compliance with various waste-related Directives.

Re-use and recycling targets are set out for Member States within the WEEE Directive.

Consumer demand creates a market for used products that are often considerably cheaper, and offer comparable functionality, compared with new products.

Reference organisations

In addition to the examples elaborated below, WRAP has compiled a number of video and downloadable pdf case studies of local-authority-led waste re-use 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 & Re-use 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 will last for 3.5 years. During this time the consortium of 9 partners from 7 different Central European countries will carry out evaluation, quality management and dissemination activities in the field of re-use and repair of waste products as a new form of waste treatment, at national and transnational levels. The Municipal Waste Management Association Mid-Tyrol (ATM) in Austria is 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 re-use, 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 3.1. Establishment of Leicestershire and Rutland Re-use Network

WRAP contracted Ricardo-AEA to assist in the development of a re-use plan for Leicestershire County Council, Leicester City Council, Rutland County Council and local third-sector re-use organisations (TSROs). The objective was to support the development of a financially sustainable re-use 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 collaborative working within a re-use network were identified.

A re-use mapping exercise quantified current levels of re-use for items within the bulky waste stream, and estimated the potential for increasing re-use across major material streams.

A four-year action plan for the delivery of the re-use network was devised, based around eight service options to improve rates of re-use 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 Re-use 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 3.2. Example of the London re-use network

Waste re-use is prioritised within London's Municipal and Business Waste Strategy plans, which identify the third sector as an important growth area and the London Re-use Network as a lead delivery partner to drive re-use targets.

The London Re-use Network comprises various re-use 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 Re-use 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, re-use, repair and remanufacturing will have ever greater roles to play in our lives".

Source: Waste Management World (2014).

Box 3.3. Waste prevention and re-use 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 re-use shop. The photo on the left below shows the repair desk and on the right repaired mobiles for reuse.

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The company is 30 % funded by Styria, the municipal waste management organisation.

Source: Schoenberger (2015).

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3.9. BEMPs on Waste Collection

3.9.1. Introduction

Waste collection is one of the primary functions of waste management organisations, and accounts for the largest share of Gross Value Added (GVA) within the waste management sector. In the UK in 2013, waste collection accounted for GBP 2,642 mill (EUR 3,540 mill), waste treatment and disposal accounted for GBP 1,434 mill (EUR 1,922 mill) and materials recovery accounted for GBP 1,354 mill (EUR 1,814 mill) of GVA (Defra, 2015). Whilst the latter two figures are volatile, varying with commodity prices and likely to increase over time with increasing scarcity of resources, the value of waste collection services may be expected to steadily increase as more waste is collected selectively. A shift in GVA away from the extraction or import of natural resources towards waste collection and materials recovery activities will contribute to the development of a circular economy.

3.9.2. Environmental burdens of waste collection

Municipal waste collection from residential areas involves inefficient start-stop driving of large waste collection trucks, leading to traffic, noise, GHG emissions and emissions that damage health and contribute to ozone formation including NO_x, PM and VOCs. Multiple collections of separated waste fractions can increase these environmental burdens compared with single collections for non-separated MSW. For example, in Denmark, diesel consumption for separated organic waste collection and transport to biogas plants has been estimated at 7.2 litres per tonne, compared with 3.3 litres per tonne when collected in a single MSW fraction for incineration (Fruergaard and Astrup, 2011). From a life cycle perspective, the environmental benefits of recycling outweigh the additional transport burdens, additional fuel consumption for collection of separated waste fractions is typically constrained by economic factors before it reaches a critical level with respect to the environmental balance of recycling. It is therefore essential that waste collection strategies are optimised within the context of an integrated waste management strategy (Section 3.5) that maximises recycling rates. Nonetheless, there is considerable scope to improve the environmental, and often the economic, efficiency of MSW collection operations.

3.9.3. Best practice technique portfolio

Municipal solid waste collection strategies must address multiple objectives and multiple waste streams, and are an integral component of overarching integrated waste management strategies (Sections 2.3 and 3.5). Developing an optimised MSW collection strategy requires holistic systems thinking that considers a wide range of related best practice measures (Table 3.12). For example, there is a strong interdependence between waste collection strategy (Section 3.9.5) and type of waste collection vehicle, influencing the opportunity for deployment of certain low-emission vehicle types (Section 3.9.8). Given the dominance of raw material extraction and waste disposal in the life cycle environmental burden profiles of most materials (Chapter 1), maximising recycling rates should be the key overarching objective of waste collection strategies (Section 3.9.5). Then, implementation of the selected waste collection strategy can be optimised in terms of environmental and economic efficiency through infrastructure to support take-back obligations (3.9.6), logistics optimisation (3.9.7) and selection of low emission vehicles (BEMP 3.9.8). Collaboration among waste management organisations and local authorities (Section 3.10.1) can play an important role in such optimisation.

Table 3.12. Key measures involved in the establishment of an efficient MSW collection strategy, and overlap with other BEMP techniques described in this document

Phase	Measure	Key points	BEMP
Waste management strategy	Integrated waste management plan (IWMP)	Decide management and fate of waste streams to minimise environmental impact.	2.3
	Benchmark performance	Benchmark effectiveness (residual waste and recycling rate) and cost efficiency of service.	3.5.1 & 3.5.2
	PAYT	Include micro-chipped bins with weigh scales on lorries, or alternative methods of PAYT	3.5.3
Efficient collection strategy	Collection strategy optimisation	Ensure delivery of quality separated waste streams to point of recycling as per IWMP, in maximum quantities possible (citizen convenience), with minimum environmental burdens and cost. Integrate logistics optimisation and low-emission vehicle deployment into planning.	3.9.5 , 3.9.6 & 3.9.7
	Waste sorting	If waste is not sorted by households or at the kerbside during collection (this BEMP), co-mingled dry recyclable waste streams must be sent to sorting centres.	3.11
	Waste collection centres	Establish accessible and user-friendly waste collection centres.	3.9.5
	Decentralised composting	Establish decentralised community composting centres if collection of organic waste for centralised anaerobic digestion or bioenergy recovery is not possible.	3.11.2
Efficient collection operations	Logistics optimisation	Hub location, route planning and driver training to minimise distance travelled, congestion and transport energy requirements per tonne waste collected.	3.9.7
	Low emission vehicles	Select vehicle drive trains and modifications to minimise life cycle environmental burdens per tonne waste collected.	3.9.8

3.9.4. Reference literature

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3.9.5. Waste Collection Strategy

Description

Background

Collection of MSW can be undertaken via kerbside ("door-to-door") 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: bio-wastes[23], glass, paper and card, plastics, metals and residual waste (where "residual waste" refers to unsorted waste at the point of collection destined for 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 the "selective collection" as the separation of waste materials at source with the intention of recycling them, and have benchmarked performance across European cities (Figure 3.27). 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.

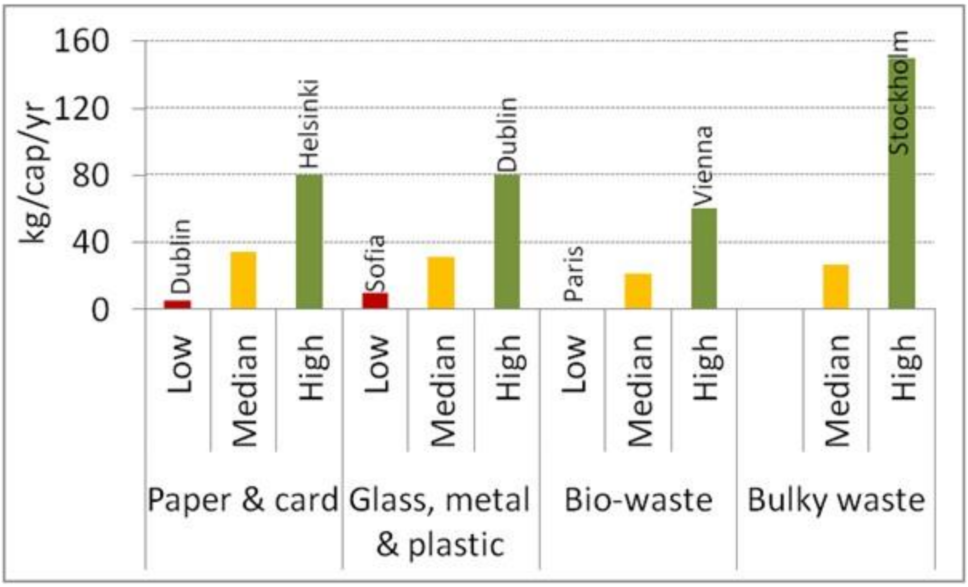


Figure 3.27. Range of quantities of different waste fractions selectively collected across European cities, according to ACR+ (2014).

Benchmarking, such as that undertaken by ACR+ (2014) can be a powerful driver to improve effectiveness and cost-efficiency. BEMP 3.5.2 on monitoring highlights the wide range of performance across waste management organisations, and the potential to simultaneously achieve a high rate of effectiveness (i.e. low residual waste sent for

[23] 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).

disposal) and a high rate of cost efficiency. Information is provided on the benchmarking of kerbside dry recycling and residual collections in UK, offered for free in the "*Local Authority Waste and Recycling Information Portal*", under Operational data, below.

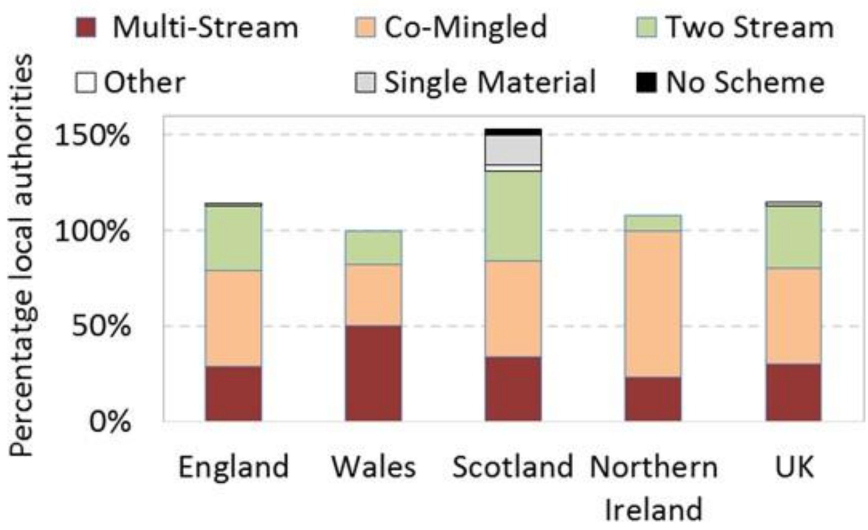
Types of selective waste collection

Various models of waste collection exist to deliver separated fractions for recycling, including separate kerbside 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 3.13).

Table 3.13. Definitions of waste collection strategies for dry recyclables provided by WRAP (2015)

Strategy name	Definition
Multi-stream	Materials are separated by householder or on collection at the kerbside into multiple material streams. Streams may include a selected mix of a few materials, typically cans and plastics, which are then separated using basic sorting facilities at the operating depot or sold to re-processors as a mixed commodity.
Co-mingled	Materials are all collected in one compartment on the same vehicle and require sorting at a materials recycling facility.
Two-stream	Materials are collected as two material streams, typically fibres (textiles) and containers, at least one of which requires sorting at a materials recycling facility.
Single material	One material is collected and requires no sorting.
Co-mingled plus textiles and two stream plus textiles	Textiles are collected as a separate stream alongside single-stream, co-mingled or two-stream collections. The textiles stream is not included in the post collection sorting of materials at the materials recovery facility.

Figure 3.28 shows the frequency of different waste collection strategies 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 3.28. 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 3.14). Similarly, multi-stream collection systems such as *Optibag* and the *Quattro System* 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) cites 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 re-processor 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 EUR spend 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 poor overall economic performance owing to reduced revenue for low quality material streams. Table 3.14 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 based explains 42 % of the variation in kerbside recyclable collection performance (kg/hh/yr) 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 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 citizen consciousness of the need to reduce residual waste. Meanwhile, the number of recyclable fractions collected, and recyclable fraction containment volume and frequency of collection, were all positivity associated with 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 bio-waste,
- minimises 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) note that European cities with the highest rates of selective 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 highest 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 per household per year more separated recyclable waste streams compared with kerbside sort collections, across the affluence / deprivation spectrum (WYG Environment, 2011). Although co-mingled collections have been found to be more expensive than kerbside sort 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 materials recycling 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 2.3). Each local authority must decide on the most appropriate strategy for their area and residents, and under local conditions. Common elements of best practice for an optimised waste collection strategy include:

- At least weekly kerbside collection of separated food waste (frequency may need to be higher in warmer climates) – see BEMP 2.3 on Integrated Waste Management planning.
- Reduced frequency of residual waste collection (e.g. every two weeks or depending upon the touristic period, etc.)
- Kerbside collection of dry recyclables (paper, cardboard, can, plastics, glass), source separated where public acceptability allows and enables high recycling rates, otherwise co-mingled and sorted at a materials recovery facility.
- A convenient network of waste collection points that accept all waste fractions not collected on-site from households, including hazardous waste, and that may substitute under certain conditions kerbside collection of dry recyclables and green waste depending on public acceptability.

Effective communication and awareness raising (BEMP 3.5.4) to encourage waste separation by citizens are key to successful implementation of waste collection strategies.

Achieved environmental benefit

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 bio-waste for anaerobic digestion leads to avoided fossil fuel combustion and fertiliser production, and avoids significant GHG emission associated with the landfilling of bio-waste. 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 yr (Contarina, 2014).

Appropriate environmental indicators

One of the simplest indicators of separated waste collection is the quantity of separated waste streams collected, expressed per household per year (kg/hh/yr) or per person per year (kg per capita per yr). However, this is not a particularly indicator for benchmarking the effectiveness of waste collection strategies in terms of separation efficiency owing to the wide variation in quantities of waste generated across different municipalities. A more accurate reflection of collection efficiency with respect to separation is:

- Percentage of MSW generated that is selectively collected (% weight)

The above indicator does not account for cleanliness and recyclability of separately collected waste streams, which is another important indicator of collection strategy efficacy:

- Contamination rate of individual waste streams (% weight of individual waste streams collected that is rejected for the intended recycling or recovery purpose)

Although it would be ideal from a performance benchmarking perspective to report on the % weight recycled for each waste stream, WMOs often do not routinely record the necessary data on municipal waste composition. Where these data are available, then the following indicator would represent best practice in collection to maximise recycling:

- Capture rate for individual waste streams (% weight of waste stream generated that is separated out for recycling)

The indicator below reflects both the quantity of material delivered to sorting and recycling plants, and also the separation and recycling efficiency at those sorting plants, which will partly depend the cleanliness of collected fractions:

- Percentage of MSW generated that is recycled (% weight *exiting* material recovery facilities in separated fractions)

However, the above indicator is only possible to derive when data are available to the WMO from the relevant material recovery facilities.

Cross media effects

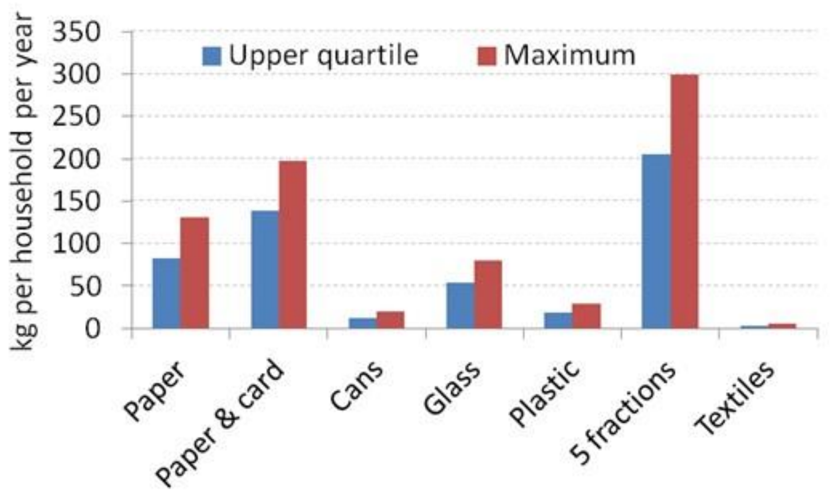
There may be a trade-off for waste collection strategies between maximising material recovery and minimising fuel consumption and emissions associated with collection. For PET plastic, for example, Bing et al. (2014) conclude that post-separation of co-mingled dry-recyclable collections is associated with higher costs and 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"* that 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 3.29).



Source: Based on data from WRAP (2010).

Figure 3.29. Top quartile and maximum achieved kerbside collection rates, expressed as kg per household per year, for waste management authorities throughout the UK in 2008/2009

These figures above correspond well with data on maximum selective collection rates across European cities provided by ACR+ (2014) and presented in Figure 3.27.

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 citizens, because it minimises contamination. Two examples of kerbside multi-stream collection of separated waste fractions are the Opitbag 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 onsite. Twenty-five Optibag plants are currently in operation across Europe (Optibag, 2015). The modular approach maximises logistical efficiency, but relies on a high level of householder motivation and engagement, which may not be achievable in some regions with less history of household waste sorting.

The *Quattro Select* system is based on householder separation of waste into eight separate fractions, stored in small containers that fit within two separate wheelie bins collected in two collection rounds using vehicles with four separate compartment (Figure 3.30). The *Quattro System* has been in use in Sweden since 2004, and has met with a high public acceptance, resulting in 90 % of all waste being recyclable (LAPV, 2012). The high wheelie-bin format for the separated fractions improves health and safety for both householders and WMO employees by minimising the need to pick up heavy collection containers.

Source: Björk (2015).

Figure 3.30. Quattro Select bins

Collection centres

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. But bring banks may 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 with household collection services owing to source separation (Table 3.14). However, an important criterion missing from Table 3.14 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 3.31. 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 operated by local citizens for limited opening hours, paid for by the County, including:

- Eight fractions of non-Fe metals
- Fe metal
- Batteries
- Glass
- Paper and card
- Plastics
- Non-impregnated woods
- Impregnated woods
- Three fractions of green cuttings (grass and leaves; wood, leaves and needles; trees without leaves)
- Cooking oils
- Residual and bio-waste (charged: EUR 0.18 per kg)

In addition to the 29 village collection centres, there are 131 mostly smaller waste collection centres in Aschaffenburg, including a few large centres where hazardous wastes, such as paints and solvents, can be brought. Hazardous wastes are also collected twice per year from households using a mobile hazardous waste collection vehicle.

Important factors to maximise efficient use of bring centres are:

- Accessibility – centres should be distributed as widely as possible so that most of the population has one in close proximity, and located conveniently to major roads or near 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 extended opening hours, including out-of-office hours and weekends, maximise social acceptance and use of collection centres. Waste management authorities may need to make a trade-off between

the duration of opening times and the number (accessibility) of centres, especially in rural areas.

- Clear signage – clear signage is essential to improve ease-of-use and minimise contamination/maximise quality of separated materials.

Mobile collection centres

The LIFE EMaRES project demonstrated application of the *Dynamic Ecopoint* concept in Italy; 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 (a more frequent version of the service provided in Aschaffenburg, mentioned above). 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 3.32. The Dynamic Ecopoint "Ricimobile", a 7.5 tonne vehicle for the collection of small WEEE, used cooking oil and batteries

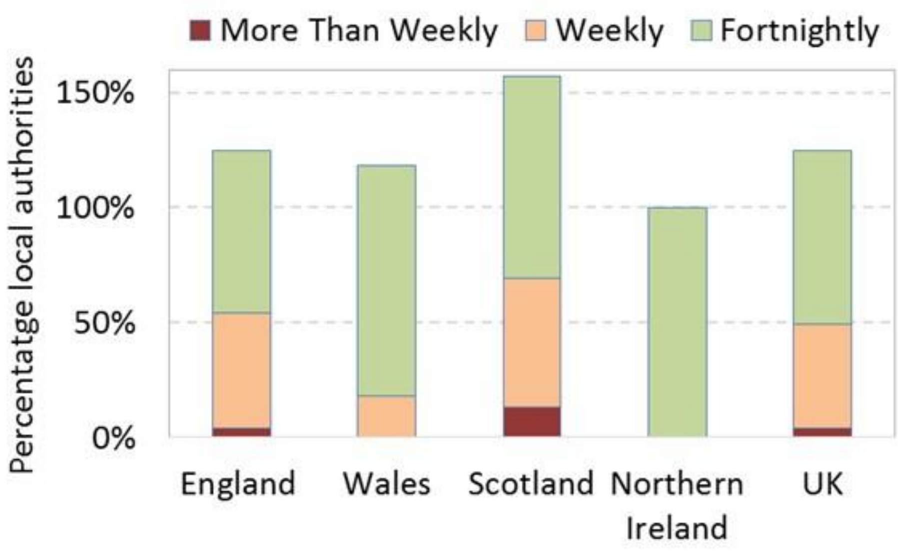
Preliminary activity as of September 2015, since the Ricimobile dynamic Ecopoint started operating in May 2015, indicate an annual collection rate of about 2,000 kg/year. This represent 2 % of the static Ecopoint collection rate in the region for

WEE, 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 3.33). 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[24] to "squeeze" waste (WYG Environment, 2011)
- Separate collection of bio-waste, especially in warmer climates



Source: WRAP (2015).

NB: Values over 100 % owing to multiple collection frequencies across zones with local authorities

Figure 3.33. 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. Figure 3.34 displays information leaflets produced by Worcester County Council in the UK regarding mixed dry recyclable bin and sack collections.

[24] 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.

The authority also provides a website with full information about kerbside collections times (based on a post code search) and alternative options[25]. Household calendars of collection dates are useful to remind citizens when to put out bins for collection.

[25] http://www.worcester.gov.uk/recycling

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 3.34. Information leaflets provided by Worcester County Council (UK)

Table 3.14 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 3.14. 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 sort 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 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 re-processors 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. Extra effort of separation at the kerbside would result in less energy 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 coloured 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 sort 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 on a kerbside sort service.	Medium. A lower revenue per tonne will be received for the glass compared to colour-sort options. Cost impacts for a kerbside sort 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)

Materials 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 materials recovery facility (MRF) for sorting and onward shipment to production facilities for final recycling into products. Modern materials 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. Some examples are listed below:

- ALBA-plants in Walldürn, Leipzig, Berlin and Braunschweig, http://www.alba.info/en/recycling/plant-technology/plastics.html,
- Jakob Becker plant in Mannheim, http://www.jakob-becker.de/index.php?id=88&uid=131
- Migros-plant in Zürich for packaging waste, http://www.industrie.de/industrie/live/index2.php?menu=1&submenu=4&type=news&object_id=33711881
- SYBERT plant in Besançon, see BEMP 3.12.1.

Applicability

The optimum approach to maximise recycling rate whilst minimising costs will vary considerably depending on local circumstances, including human behaviour that is partly related to socio-economic situation. WRAP (2010) found that prevailing socio-economic status within local authority areas was an important factor determining recycling rate, with lower recycling rates associated with 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 of 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 3.5), may be required to achieve acceptable levels of recycling across the major dry recyclable fractions in such areas.

Less frequent (e.g. two-weekly) residual waste collection may not be practical in warmer climates owing to odour and hygiene issues if it contains bio-waste. The separate collection of bio-waste 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 bio-waste collection is twice a week, in Germany it is usually once a week in summer and two times a month in winter.

Driving forces 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 bio-waste 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 air pollution reduction) of collection.

Economics

Costs for the staff, for 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 3.9.6). 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.

Although best practice is to send wet bio-waste to anaerobic digestion, necessitating collection, some authorities have ruled out separated bio-waste collection on cost grounds. In such cases, the next best alternative is decentralised composting (see decentralised composting BEMP). In Besançon, eastern France, collection of dry mixed recyclables and residual waste is managed by seven "communautés de communes" within the "Grand Besançon", but kitchen and garden waste is composted in decentralised composting centres (SYBERT, personal communication).

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 covers 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 WMOs by minimising residual waste disposal costs.

Reference organisations

A number of WMOs have adopted the *Optibag* system, with 25 plants already in operation: http://www.optibag.com/reference-projects

Box 3.4. Gwynedd County Council waste collection strategy, involving separate bio-waste collection and kerbside sort

The UK has only recently begun to recycle food waste in composting and anaerobic digestion plants, food waste recycling has increased from 1 % in 2006 to 12 % in 2012 (Defra, 2014). Gwynedd County Council collects food waste separately once per 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.

© E³ Environmental Consultants Ltd

Gwynedd County Council collects the following dry mixed recyclable fractions in blue boxes (right) once per week, on the same day as food waste collection, using a kerbside sort 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 or butter pots, plastic containers for fruit and vegetables and meat trays, food and drink cartons, fruit juice or soup cartons, cardboard.

© E³ Environmental Consultants Ltd

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: Gywnedd Council (2015).

Box 3.5. Example of twice-weekly bio-waste collection in Milan

The municipality of Milan-Amsa comprises 1.281 million citizens, and first introduced door-to-door collection of household bio-waste 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. 120 litre brown bins and compostable bags 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 minimize odours and anaerobic decomposition, are used in apartments. Bio-waste is collected twice per week.

The waste management organisation coordinated activities with the City of Milan. Census data from the area were used to prepare the service set-up. A software model was used to determine logistical 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 has risen 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: Di Monaco (personal communication) and R4R (2014c).

Box 3.6. Example of waste collection strategy operated by the County of Aschaffenburg near Frankfurt 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 3.7. 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 urbanization, 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 3.8. Initiating door-to-door collection in Lisbon, Portugal

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 bio-waste 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 bio-waste 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 answers 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 total MSW generated.

Source: R4R (2014b).

Box 3.9. Contarina SPA integrated waste management collection strategy

Contarina is a publically-owned WMO 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 population 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 3.5.3). 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 yr.

Source: ZeroWasteEurope (2015).

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3.9.6. Infrastructure to recycle or to recover waste streams and to dispose of hazardous compounds

Description

The 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 fractions paper/cardboard, bio-waste, packaging and eventually glass. In addition, at its best, every bigger municipality (> 1,000 inhabitants) has at a least one collection centre (also called 'container park' or 'civic amenity sites') where citizens can drop off as many as possible waste fractions which can be recycled or recovered at reasonable costs.

The county, city or region are identifying the numbers and locations of collection centres and provide a standard layout for them. The latter can be applied by municipalities. In addition, staff is trained to operate the centres in a way that all fractions are well separated and dropped in the correct container, drum, box etc. Concerning the location, it is important that there is easy to access to citizens, well connected to the road network and not disturbing the neighbourhood. The area must be water-tight 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 the citizens to drop off different waste fractions, an example is shown in Figure 3.35. In spring, summer and autumn, the opening hours are longer compared to winter when less material is delivered, especially green cuttings.

Figure 3.35. Opening hours of a collection centre of a German village with about 8300 inhabitants, the opening hours are adapted to day light and season, specifically, there are extended opening hours in November to increase the reception of green cuttings

The different fractions which are least collected are described under 'operational data'.

Achieved Environmental Benefit

The recycling of the manifold mentioned waste fractions corresponds with savings of 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 indicator

For a county or a city, the number of collection centres per 100,000 capita can be used as an indicator. The weight of the different waste fractions per capita collected via collection centres can also be used as an appropriate environmental indicator.

Table 3.15 provides an indication for the number of collection centres of German cities and counties with a well-developed network of collection centres.

Table 3.15. Number of inhabitants per collection centre in German municipalities

County/City	Inhabitants	No. of collection centres	Capita per centre
Aschaffenburg (County)	172,000	30	5,700
Enzkreis (County)	200,000	11	18,182
Bad Homburg (City)	53,000	2	26,500
Rems-Murr-Kreis (County)	416,000	13	32,000
Aschaffenburg (City)	68,000	2	34,000
Schweinfurt (County)	113,000	2	56,500
Neumünster (City)	77,000	1	77,000
München (City)	1,400,000	11	127,273
Hamburg (City)	1,800,000	12	150,000
Berlin (City)	3,500,000	15	233,333

The table shows that the capita-specific density of collection centres in smaller cities as well as in counties is higher compared to big cities. It reveals that the operation of collection centres here is cheaper compared to door-to-door collection.

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

At the collection centre, at least the following fractions can be dropped off:

- Green cuttings (with low structure, branches with leaves or needles, woody material without leaves or needles – see photos below). The green cuttings with low structure can be shredded and classified on demand. The fine fraction is usually composted. 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. The woody green cuttings without leaves and needles (preferably in winter and spring) are shredded and used for energy recovery, the shredded material is incinerated in a biomass power plant, partly without prior sieving to save costs.
- 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 has to be disposed of for about 170 EUR per tonne.

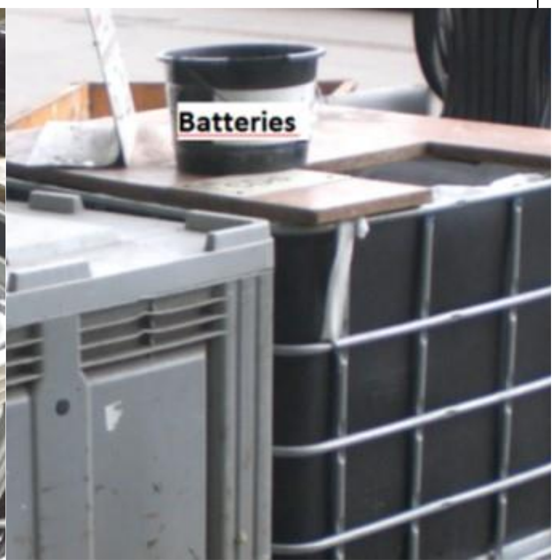
- Scrap metal and different non-ferrous metals (e.g. copper, aluminium, brass) as well as stainless steel, lead or lead-containing materials are also collected – see photos below)

- Paper, board and cardboard is separately collected at household level (door-to-door/kerbside). Nevertheless, a collection centre is also equipped with a container concerned. 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 for paper/cardboard and glass containers at a collection centre.

- Metal tins are also separately collected at collection centres as well as white clean packaging polystyrene (see photo below), there is a special agreement with the "Duales System" to collect metal bins separately. Polystyrene is separately collected to enable high quality recycling polystyrene chips can even be re-used.

- In order to support take-back obligations, waste of electrical and electronic equipment (WEEE)[26] 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).

[26] Directive 2002/96/EC and Directive 2912/19/EC



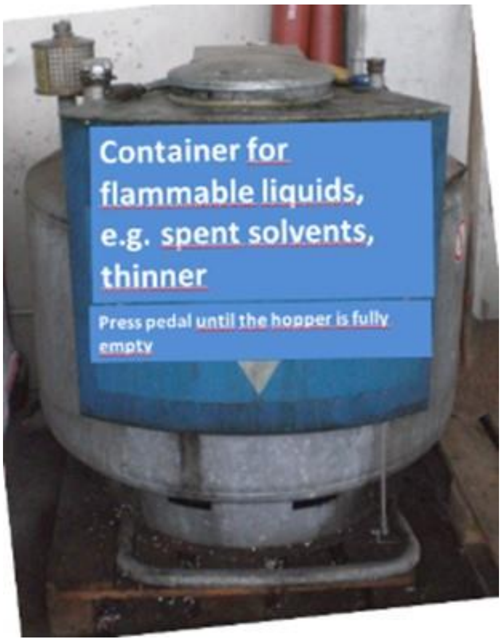
- Bulbs and fluorescent tubes are additional fractions that are separately collected and delivered to recycling according to legal regulations, no revenues are gained for them.

- 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 (furnish with wood preservatives, such as window frames, exterior doors, wood from palisades and other outdoor applications – see photos below).

- More and more, also small items are 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 yr, it is financially attractive) and items mainly made of lead (see photo below). Taken-back shoes, textiles and hand bags can be recycled and the revenues can be donated to social projects (see photo below).

- It is very important to separately collect waste containing relevant amounts of hazardous compounds, 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). Time and location of their stops in all municipalities, city quarters, etc. are adequately communicated to the citizens. Mercury containing waste is strictly kept 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 on roads have to be met (see photo below showing a special container).
  Also solar panels are separately collected (see photo below).

Figure 3.35 provides an example for the opening hours of a collection centre of a village having 8300 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 per week, preferably on Saturday, whereas in cities with high frequencies, the opening time is more than 20 hours, in some cases even more than 40 hours. Figure 3.36 indicates the distribution of opening hours of almost 100 collection centres in Germany.

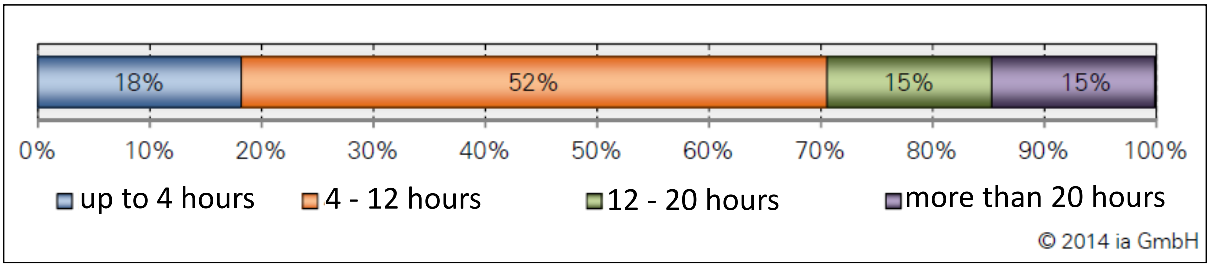


Figure 3.36. Distribution of opening hours of collection centres in Germany (ia GmbH / UMSICHT, 2015, p 15)

Ideally, the average catchment area of collection centres in city areas is at 34 km², in rural areas at 43 km² and in individual municipalities at 16 km². Thus, as an average, the distance of the inhabitants to 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 connected citizens are important parameters.

To improve the user-friendliness, with respect to bigger items (scrap metal, cardboards, green cuttings, etc.), it is of advantage to go for so-called two-level solutions where the levels of the delivering persons and the level of the container bottom are different (see Figure 3.37).

Figure 3.37. Two-level solutions for the delivery of materials (ia GmbH / UMSICHT, 2015, p 16)

Putting a roof over the collection centre makes deliveries more comfortable (see an example in Figure 3.38) but is much more expensive compared to open space facilities (see economics).

Figure 3.38. Example for a roofed collection centre, (ia GmbH / UMSICHT, 2015, p 17)

It is important and required that skilled personnel of the municipality, county or city controls the deliveries of the citizens in order to avoid cross-contamination of the different fractions. They are also instructed 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 municipality, cities or counties. The introduction of collection centres in cities can be limited due to space constraints. The recyclability also depends of 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 municipality of a county vary considerably. According to Figure 3.39, in 2013, the range for counties or cities collecting bio-waste, paper/cardboard and residual waste in specific bins as well as operating collection centres in all municipalities (upper part of the figure) is between 17 and 76 EUR per capita per yr.

This indicates that an efficient system can be operated at reasonable costs and that there can be significant room for cost optimisation. The cost figures already include the revenues gained from some of the recycled fractions.

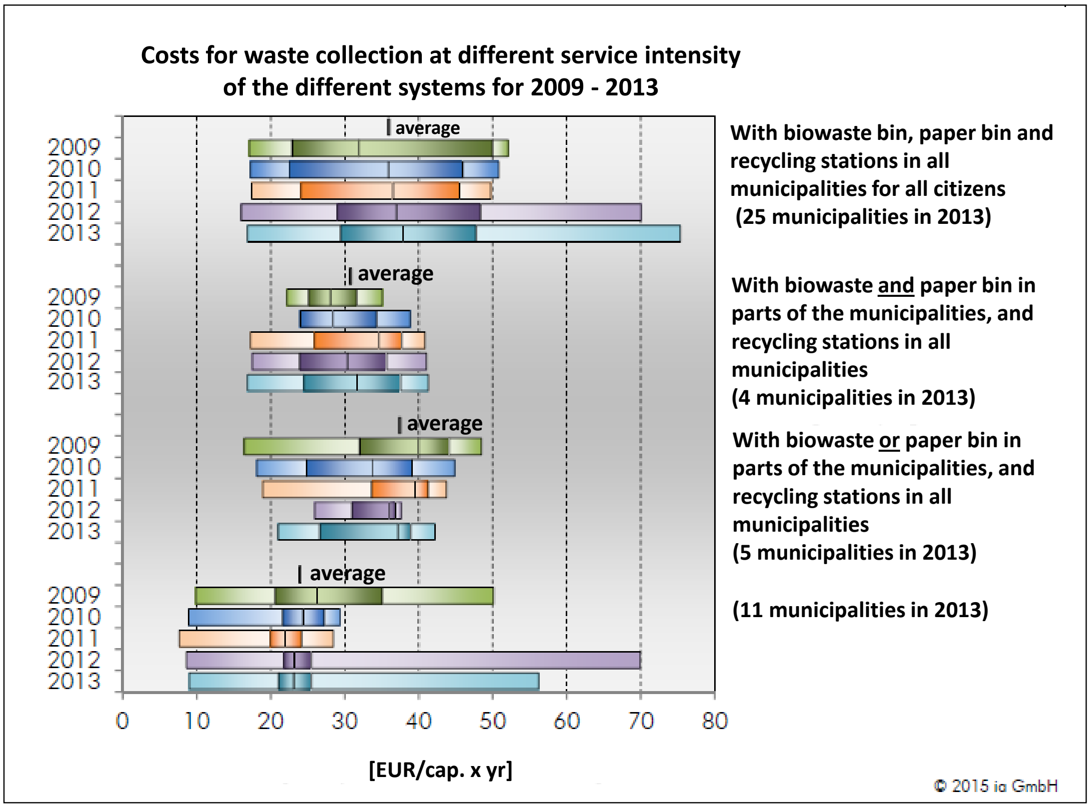


Figure 3.39. Costs for waste collection at different service intensity of the different systems for 2009 – 2013, based on ia GmbH (2015), see BEMP on cost benchmarking, see explanations in the footnote[27]

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 quarters).

Considering the collection centres only, the cost range is also large (Figure 3.40). In most of the counties, cities and municipalities (about 100 in total), the costs are between less than four and ten EUR per capita and year. For the evaluated cities, the average cost figure is 7.8 EUR per capita per yr, for counties 5.1 EUR per capita per yr and for individual municipalities 6.6 EUR per capita per year.

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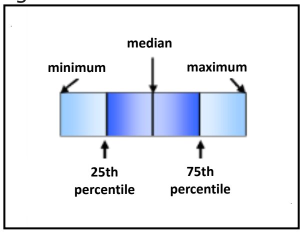
[27] The values are presented as median, minimum, maximum and 25th/75th percentiles as indicated in the figure below.



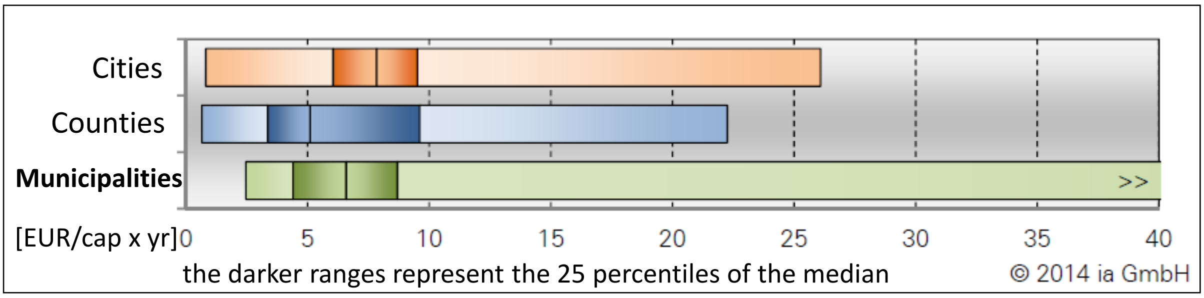


Figure 3.40. Costs for the operation of collection centres (ia GmbH / UMSICHT, 2015, p 32), see footnote 27 at the previous side

The composition of the cost for collection centres is illustrated in Figure 3.41. Almost two third 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 3.37), etc. will not significantly influence the total costs.

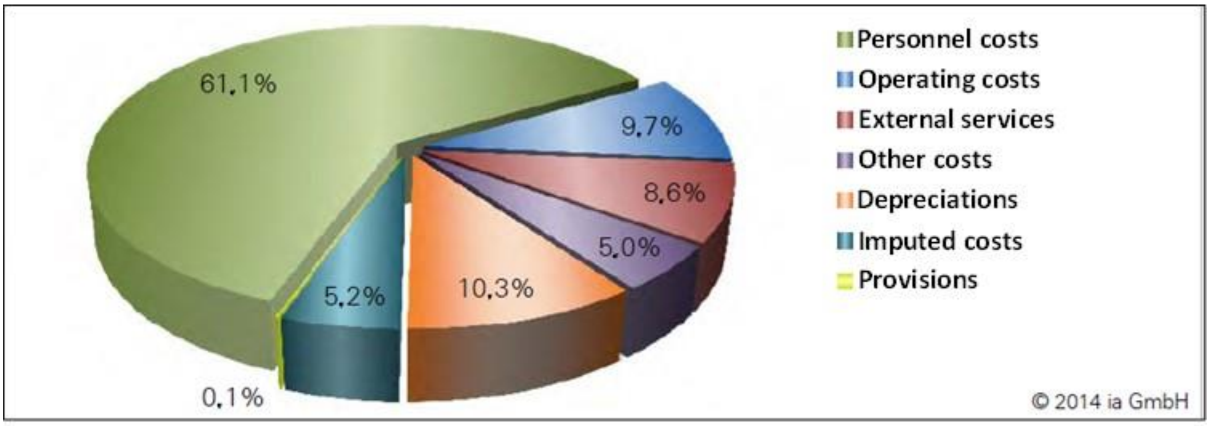


Figure 3.41. Composition of the costs for operating collection centres (ia GmbH / UMSICHT, 2015, p 32)

In connection with Figure 3.38, it is already indicated that investment costs for collection centres depend on the standard that can be grouped in to 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 2-level solution
- Category II: investment costs between EUR 50,000 and 150,000 – container or roofing as "operating building", flatly asphaltic area
- Category III: investment costs between EUR 150,000 and 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, eventually reception of hazardous waste, levelled area with ramps

Considering about 100 collection centres in Germany, about half of them fall into category II, about one fifth each into categories I and III and only a few into category IV (see Figure 3.42).

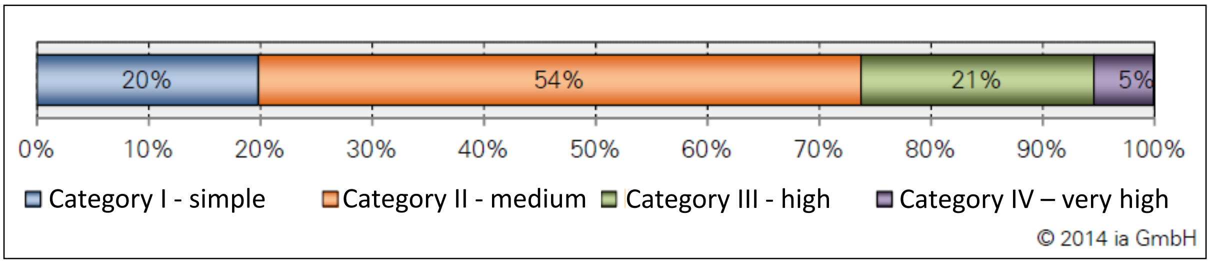


Figure 3.42. Different categories of collection centres (ia GmbH / UMSICHT, 2015, p. 26)

Driving force for implementation

The rising awareness to go for a circular economy is a major driving force for establishing and operating collection centres. The awareness was often driven by the limited availability of landfills, and, in some Member States, by the legal stop of landfilling untreated waste. For instance in Germany, Austria and the Netherlands, the awareness started to significantly increase already more than 30 years ago.

Reference organisations

Germany: Counties of Aschaffenburg, Rems-Murr, Schweinfurt, Enzkreis. Cities: Munich, Hamburg, Berlin, Neumünster.

Austria: see http://www.altstoffsammelzentrum.at/ and http://de.wikipedia.org/wiki/Altstoffsammelzentrum

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3.9.7. Logistics optimisation for waste collection

Description

Overview

Once WMOs have designed an effective waste collection strategy to maximise recyclability, there is often scope for significant logistics optimisation in order to reduce 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, 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 3.16 summarises the key measures to optimise logistic operations for waste collection, and their underpinning rationale.

Table 3.16. 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 opportunities with neighbouring waste management organisations.	Collaboration offers considerable scope for improvement through efficiency savings, such as route optimisation and depot rationalisation (AMEC, no date).
Benchmarking 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 into 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.

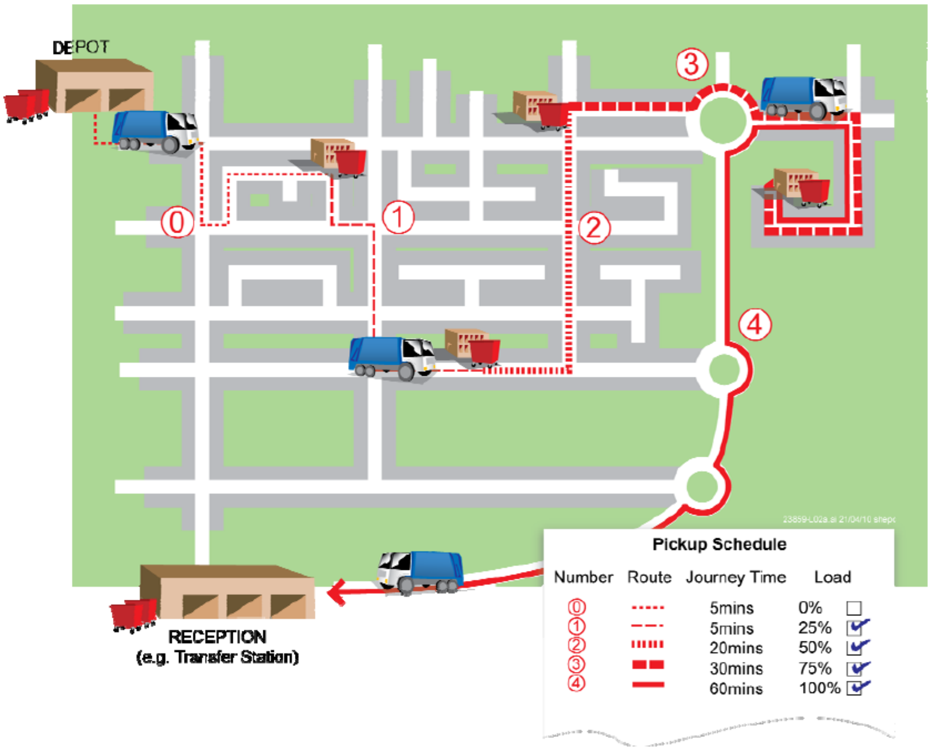
Route optimisation

Logistics operations for waste collection can be optimised with respect to[28]: (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 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 through integration of zone (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 3.43). 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.

[28] All the choices should take into account the local traffic conditions and the architecture of the examined area.



Source: WRAP (2010).

Figure 3.43. 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 of 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. Note that alternative road-transport vehicles are described in the next BEMP.

Achieved environmental benefits

Pneumatic systems can lead to significant savings in fuel use, and reduce noise, visual impact, odours and traffic associated with conventional waste collection systems. Installation of a pneumatic system in the Hammarby Sjöstadis district of Stockholm is estimated to have reduced waste collection traffic 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) report 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) report 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 (2-4 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) report a trial with SITA UK in which they used telematics to target driver training, this allowed reducing fuel consumption by 12 %. AEA-Ricardo (2009) suggests that more efficient driving can reduce fuel consumption by up to 10 %.

Appropriate environmental indicators

It is crucial to place logistics performance and burdens in the context of whole-chain waste management efficiency, considering life cycle performance indicators (BEMPs 2.3 & 2.4). It is imperative that logistics optimisation does not compromise performance in the key performance indicators for waste collection strategy outlined in BEMP 3.9.5), especially overall performance indicated by:

- Percentage of MSW generated that is recycled (% weight exiting material recovery facilities in separated fractions)

Some pertinent environmental indicators specifically relating to the efficiency of logistics operations include:

- Fuel consumption per tonne of waste fraction collected (L/tonne)
- Average fuel consumption of waste collection vehicles (L/100 km)
- Cumulative Energy Demand (CED) per tonne of waste fraction collected (MJ/tonne)
- GHG emissions per tonne of waste fraction collected (kg CO_2e/tonne)

Dekker et al. (2012) cite studies using CED as a measure of environmental impact for logistics operations owing to high correlation with many LCA impact categories, whilst Aronsson and Huge-Brodin (2006) propose GHG emissions as a useful indicator for environmental impact. These indicators are necessary to compare the performance of alternatively-fuelled vehicles (BEMP 3.9.8), and to place transport and logistic operations in the context of whole-waste-chain system performance.

Dekker et al. (2012) report that route optimisation in general can reduce transport distances and associated environmental burdens by 3-5 %.

When it comes to optimising the life cycle of specific waste fractions or reporting environmental burdens associated with logistics operations for specific waste fractions, allocation procedures will be required for all transport involving multiple materials – e.g. collection of co-mingled dry recyclables. Subjective judgement will be required to decide whether to allocate on a mass, energy or economic basis, for example (ISO, 2006). Whether or not trucks backhaul materials when transporting recyclable materials to production facilities can have an important effect on the fuel consumption and emissions attributed to the recycled materials.

In order to capture the health and urban environmental benefits of alternative (e.g. pneumatic collection systems), the following indicator is proposed:

- Waste is collected from densely populated urban areas using an alternative (e.g. pneumatic) waste collection system that minimises the use of refuse collection vehicles (yes/no)

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 3.9.5).

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 emissions and SO_x emissions per tonne of waste transported, compared with 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 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 re-designing and optimising collection rounds using CVRS technology. Data generated by PAYT systems (BEMP 3.5.3) 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, highlighted the use of micro-chip 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) note that waste management organisations are sometimes sceptical to 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) report on a trial with CVRS optimisation across three waste management organisations in the UK. Round data was supplied with post code 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 time,
- pick 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 routes were newly designed. Through the project, the collection logistics were streamlined, and ten waste collection routes were saved, leading to a reduction in truck traffic, and a saving on fuel 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 adapted 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 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 Jätkäsaari residential area of Helsinki will be completed by 2023, and will house 16,000 residents and 6,000 workplaces. 350 pneumatic collection points will be installed to handle 22,000 kg/day of waste (6,400 tonnes of residential waste plus 550 tonnes of commercial waste annually) (Waste Management World, 2009).

At Hammarby Sjöstadis, a neighbourhood of Stockholm, four pneumatic systems have been installed since 1997, operational as of 2000. 457 inlets and 12.5 km of pipes manage 11 tonnes of waste per day, split into four fractions:

- Bio-waste,
- 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, 2015a).

Figure 3.44. Pneumatic system inlets in Hammarby Sjöstadis

Source: Envac (2015)

Economics

WRAP (2010) quote costs in the range of GBP 5,000 to GBP 10,000 (EUR 7,042 to 14,084) to model and optimise existing collections rounds for a waste management organisation running 12 collection vehicles. Adding alternative future scenarios costs GBP 2,000 to GBP 6,000 (EUR 2,817 to 8,451) per scenario. In the case study example, WRAP (2010) estimate a fuel saving of up to GBP 36,208 (EUR 51,000) 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 likely to realise greater savings).

The outsourcing of waste collection activities by WMOs can reduce incentives for both separation efficacy and logistics optimisation, depending on how contracts are structured. In the absence of specific performance-related clauses, sub-contracted collection companies may maximise revenue by maintaining high frequency bin collections, justifying higher charges to the WMOs. 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

In terms of collection strategy optimisation, logistics optimisation is a secondary consideration to be implemented after identification of the most effective collection strategy to maximise waste prevention and separation efficiency.

Logistics optimisation can be implemented at different levels of system and technological sophistication by any WMO, often saving on operational costs.

Alternative pneumatic systems are intended to alleviate problems associated with waste collection in densely-populated areas, and are easier to install in new developments.

Driving forces 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

SITA UK

Sefton Metropolitan Borough Council l, UK

Multi-council collaboration in Hampshire, UK

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 3.10. SITA UK telematics and driving 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 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 per cent, which were extrapolated up to an annual GHG emission reduction of 3,000 tonnes per annum. 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 3.11. Optimisation of collection rounds for a new waste collection strategy by Sefton Council, UK

Sefton Metropolitan Borough Council is a local authority comprising 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 Council 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 Council 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 3.12. Logistics optimisation through multi-council collaboration and depot rationalisation in Hampshire, UK

Background

Project Integra is a partnership of the fifteen 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 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 annum, 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 annum. 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 annum.

Source: AMEC (no date)

Box 3.13. 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 content below 40 % to 50 %.
- 75 to 80 % of bins are collected at least once per year with 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, to allow for collection every 15 days.
- 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 ca. 5,000 litres per year, whilst workforce savings are in estimated to be 40-50 %, switching from weekly to biweekly collection.

Source: Wellness Smart Cities and Solutions (2015).

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3.9.8. Low emission vehicles

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 and 141 L/100 km of diesel, reflecting inefficient low-speed and stop-start driving. Priority measures identified by Ricardo-AEA to reduce GHG emissions from municipal HGV use are summarised in Table 3.17.

Table 3.17 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 yrs	Small air quality and noise reduction benefits in congested urban areas. Marginal increase in life cycle impact due to additional components.
2=	Hybrid electric / hydraulic hybrid vehicles	15-25 %	4-16 yrs	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 yrs	Significant particulate emission & noise reduction benefits, requires additional refuelling infrastructure. Substantially larger CO_2e reduction benefits with biomethane.
3	Electrically-powered truck bodies	10-12 %	9 yrs +	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 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 of 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 that 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 multipoint injection system, enabling 100 % gas fuelling and superior overall environmental performance. Smaller petrol-driven collection vehicles can be converted to run on either 100 % gas, or as bi-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 Czechia (82) (metanoauto.com, 2015).

Biomethane is becoming more common as a transport fuel in Germany and Sweden. The technology for the utilization 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 sub-optimal 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 a cycling of energy and carbon 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) report on the testing of hybrid (Figure 3.45) 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.

Figure 3.45. A "Rotopress Dualpower" refuse collection truck during testing in Bremen, Germany (*Source:* Nehlsen, 2013)

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 fraction.

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 waste collected, as described above, although Nelsen (2013) note that there may be routes where a truck's full capacity is not required and on which hybrid trucks would not require an additional stop-over. Carefully integrating hybrid vehicles into optimised collection tours is therefore essential to obtain maximum efficiency savings.

Achieved environmental benefit

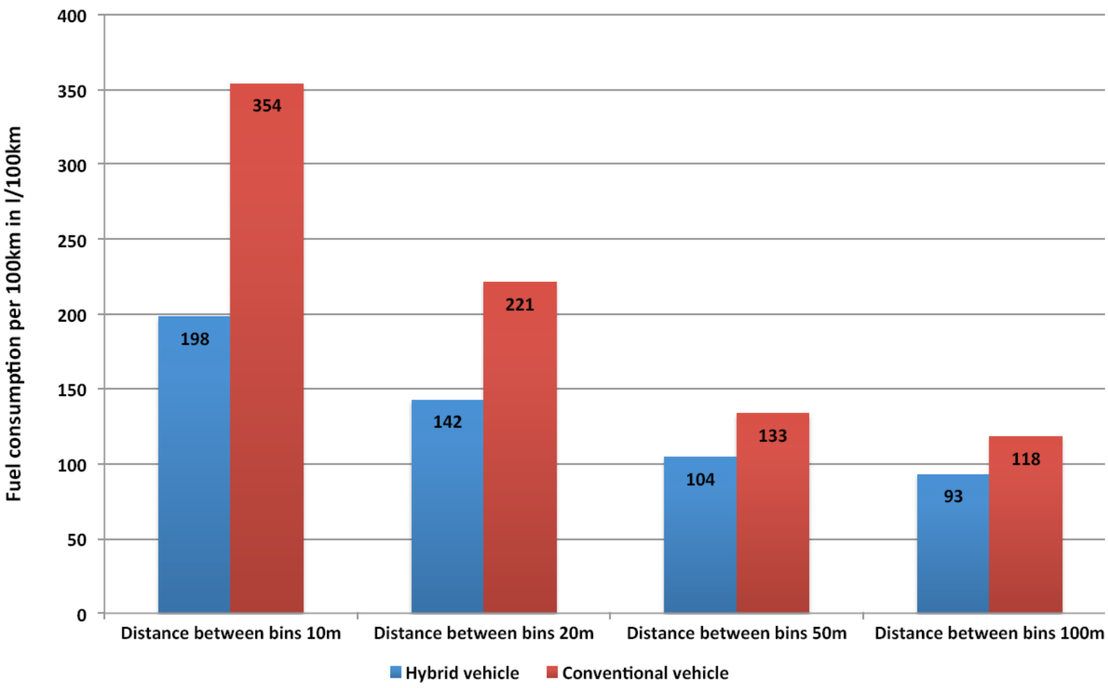
GHG emissions

Direct CO_2 emissions from combustion are significantly lower for CNG- compared with diesel-powered trucks, 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) report that overall fuel consumption per Mg 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 tours, achieving reductions in fuel consumption of up to 40 % in case of bin stops being separated by short distances of 10 m (i.e. urban areas) (Figure 3.46).



Source: Nehlsen (2013).

Figure 3.46. Fuel consumption for a hybrid and 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 3.18; Figure 3.47).

Table 3.18 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.

Figure 3.47 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.

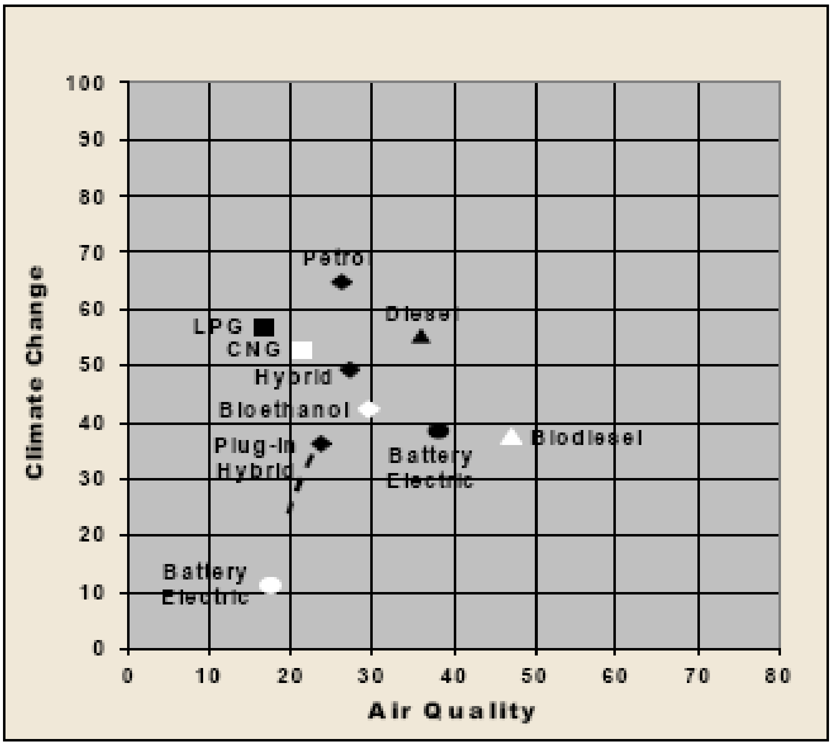


Figure 3.47. Performance of different vehicle propulsion options in terms of GHG emissions (y-axis) and emissions affecting air quality (x-axis). Source: LES (2011).

Appropriate environmental indicators

Technical indicators

The most appropriate environmental indicators are life cycle emissions of GHGs, fossil resource depletion, and emissions affecting health and air quality. These are ideally expressed as life cycle burdens for global warming potential (CO_2e), fossil resource depletion potential (e.g. MJe) photochemical ozone formation potential (e.g. kg VOCe), acidification potential (e.g. kg SO_2e), human toxicity potential (e.g. kg 1,4-DCBe) per tonne-km (tkm) of transport to compare the efficiency of different fuel types.

However, more readily available indicators from the European Test Cycle include:

- Transport rated CO_2 emissions (g CO_2e/km)
- Engine PM, NO_x, VOC emissions (g/kWh)
- Percentage vehicles that are EURO VI compliant

Ideally, life cycle CO_2e/km should be used based on fuel carbon content or relevant (national) electricity GHG intensities and life cycle assessment of biogas feedstock types.

Management indicators and possible benchmarks

The proportion of trucks in the waste collection fleet operating on alternative fuels provides a good indication of performance in this technique, and provides a useful basis for benchmarking. The following benchmarks of excellence are proposed:

- All new waste collection vehicles purchased or leased are fitted with stop-start and idle shut-off technology and electrically operated bodies.

- All either new waste collection vehicles purchased or leased are dual-fuelled or fully fuelled with natural gas, biogas where available, or are hybrid electric vehicles.
- Existing vehicles with sufficient remaining planned years of service to justify the cost are retrofitted to run on natural gas, or biomethane where available.

The following management performance indicator could be used to reflect the above targets:

- Percentage vehicles that are hybrid-electric or natural gas/biomethane powered

Information on the prevalence of alternative-fuelled refuse collection trucks is provided in the case studies section.

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 c.1-2%, which can have an important affect 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 onsite combined heat and power plant fuelled by biogas, or imported from the grid. Chesshire (2014) reported electricity consumption of 1.06 and 0.6 kWh per kg 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 case study of transport biomethane production at 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 3.19).

Table 3.19 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	0-4000	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 3.20 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 10 ppm maximum concentration, 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 3.20 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 a., 2013).

Dedicated engine technology

Natural gas dedicated engine technology (e.g. Otto cycle stoichiometric combustion with 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 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 e.g. (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).

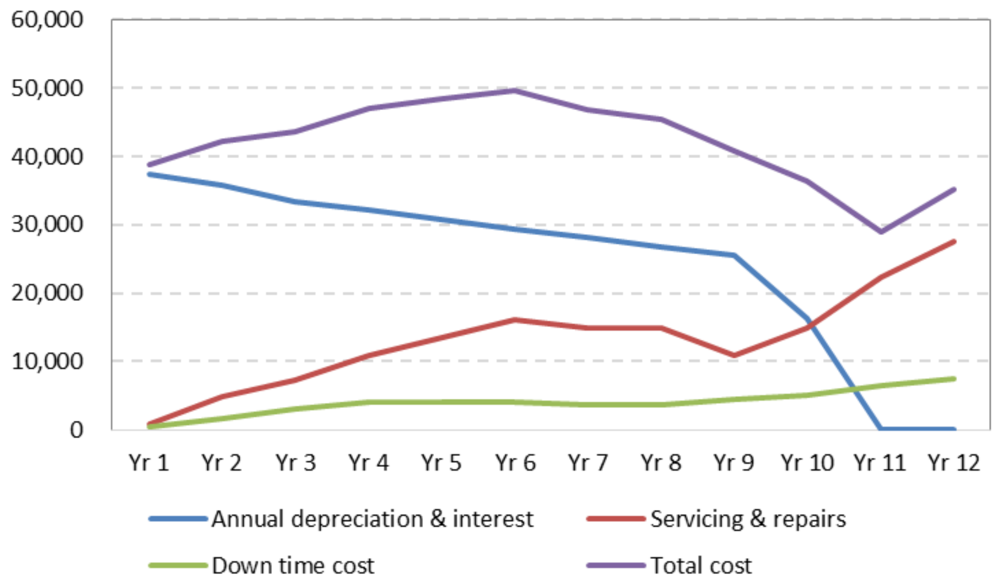
Applicability

The prevalence of filling stations is less of an issue for refuse collection than other types of transport because vehicles usually operated on limited distance run from a centralised waste depot where refuelling can take place.

CNG is available in all countries. Biomethane may not be available in many regions, although using wet organic waste (e.g. food waste) to produce biogas that is upgraded to transport biomethane is best practice within an integrated waste management strategy (section 2.3).

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 400k (EUR 563,000) to GBP 1m (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 3.48 shows average annual running costs, excluding fuel, for a fleet of 150 CNG refuse collection vehicles. BSR (2015b) note that maintenance costs are only slightly higher for CNG compared with EURO VI diesel trucks.



Source: BSR (2015b).

Figure 3.48. 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 diesel (8 mpg = 35 L/100km), costs GBP 0.62 per mile (EUR 0.54/km), while natural gas costs approximately GBP 0.39 (EUR 0.34/km). WRAP (2010) recorded fuel efficiency of between 6 and 10 mpg for a single modal refuse collection vehicle (skip carrier), and 3.5 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 over 200,000 km, at least off-setting the higher purchase cost.

Stricter vehicle emission standards are associated with higher operating and maintenance costs for HGVs. When converting a HGV to run on gas, the removal of the diesel after treatment 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 forces 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 sub-contracting of waste management to companies that use low emission vehicles to reduce their environmental footprint.

Reference organisations

Renova, Sweden. 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 München, Nürnberg, Offenbach, Baden-Baden and Darmstadt have tested electro-diesel hybrid vehicles over the past 4-5 years (AWM, 2014).

Box 3.14. BRS, Berlin, biomethane case study.

BSR process approximately 60,000 tonnes per year of organic waste from Berlin households in a biogas plant. The produced biogas is cleaned, processed, concentrated and fed into the city gas network as biomethane. 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 on three BSR depots. As a result, annual savings of around 2.5 million litres of diesel are achieved (BSR, 2015a).

Box 3.15. 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 specializing in collection and cleaning equipment, and Li-lon, 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 3.16. 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. Usability, technical, environmental and economic performance of these vehicles is 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 "Operational data", above.

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3.10. Enabling Techniques on Waste Collection

3.10.1. Best practice in the application of inter-municipal cooperation (IMC) for waste management in small municipalities

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 of the economy of waste management for small municipalities, as derived 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 duplicities of work and creation of synergies. IMC improves resource efficiency and leads to improved services and less associated costs to public services, conventionally with high intensity of cost, as waste management.

The empirical evidence shows that for small municipalities, the collaboration with other municipalities reduces the total cost of management. For higher 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 out a very interesting and, somehow, unexpected effect of inter-municipal cooperation in small municipalities, under certain conditions, a high rate of collection frequency does not increase waste management cost. This is directly opposite to any other empirical observation but the authors identified this effect coming from the same concept of economy of scale, as e.g. the same truck serves several municipalities. On the management side, inter-municipal cooperation is not necessarily a saving money 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 (only for small municipalities),
- 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 co-ordinated planning and environmental protection.

The crucial point for this BEMP is: What is the definition of a best practice 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):

- build central waste disposal or treatment plants,
- develop joint policies for solid waste management and
- establish recycling to achieve better environment protection.

Municipalities collaborating in the management of waste are quite well established in Europe. A survey of the mayors 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 the best environmental management practice that directly leads 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 size municipalities. 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 good performing inter-municipal cooperation scheme (Table 3.21).

Table 3.21. Inter-Municipal Cooperation (IMC) basic structure (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 co-operation 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 Benefit

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 possible to apply. The borderline of the applicability of a best practice to small municipalities is never clear, but some examples on the performance of cooperations are shown in Table 3.22.

Table 3.22. 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 following year of the implementation of a volume-based PAYT. 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 the current situation.	This is an estimation of performance after a proposed route for IMC implementation.	Põldnurk, 2015

Appropriate Environmental Indicators

As an administrative measure, no environmental indicator is directly linked to the implementation of an IMC agreement, but of the best environmental practices or best available techniques facilitated by the IMC. For example, the implementation of a PAYT system in the Grand Besançon area allowed small municipalities to participate of the benefits of a BEMP, which recommended indicator is the amount of residual waste per capita and year or the percentage of recycled waste with respect to the total waste.

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 more reference), avoiding the overlapping of responsibilities or a distortion of the primary objectives of the scheme (e.g. the IMC can improve the 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 been only recently 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 taking place 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, organizational, dimensional and expertise problems, as well as have more problems to fulfil challenging objectives in the delivery of public services. So, IMC has been looked recently as the most promising solution for them (Bolgherini, 2011).

Higher tier local government structures are usually the responsible for the implementation of cooperation agreements, as e.g. *comarcas* or *Mancomunidades* in Spain, *communauté 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 on county level. Thus, IMC has a legally binding character.

In the following boxes below, several European case studies[29] for IMC for waste are described. Not much detail is given on the specific administrative arrangement, as the regulatory framework is dependent on national and regional legislation, but to the specific outcomes and benefits obtained in terms of waste management performance.

Box 3.17. 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 with 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 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 and 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 prophase and 7 % in the actual implementation (Sybert, 2015; Pre-waste, 2012).

Box 3.18. 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 bio-waste and paper separate collection in rural areas. The study identified the administrative, economic and logistic benefits of the adoption of inter-municipal cooperation. In rural areas, the main cost source comes from 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 analysed area, there are 23 officials or more in charge of waste management in the 23 municipalities. However, the multiplicity of tasks of these

[29] Only three examples are shown in this current version of the text. More examples will be included as a result of the research exercise.

officials, with a very low specific dedication to waste, could be easily solved with only four officials in charge of a waste supramunicipal structure. In total, 70 % of the municipalities in Estonia have less than 4,000 inhabitants and would benefit from such schemes (Põldnurk, 2015).

Box 3.19. City of Friedberg (Germany) and Waste management company of the Wetterau county (AWB, 2015)

In 2005, 209 tons of bulky wastes were collected in Friedberg by street collection. In 2010, there were only 125 tons. In the same period, the bulky waste delivered at the recycling center in Friedberg increased from 121 to 604 tons. Waste fees could be reduced by 2011.

Applicability

There are no specific barriers for the application of IMC in waste management. However, benefits from the economy of scale are only evident for small municipalities (Bel and Fageda, 2010). Some other barriers for the application of this BEMP are the insufficient legal framework, weak incentive system or the lack of capacities to develop and manage contracts.

Economics

In rural areas, there is an increased probability of administrative and logistical inefficiencies affecting waste management service. High waste transportation costs, multiplicity of tasks, different pricing and lower control over the collection service are only some of the few 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 in:

- **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 increases.

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 sole influence of IMC 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 an IMC in small municipalities: 16 % cost reduction in municipalities under 5,000 inhabitants, while the difference was not statistically significant for municipalities over that size. Djemaci (2009) attributed a cost reduction of EUR 5.25 per capita per year due to the application of

IMC in the area of Grand Besançon, although the fee system had to change to a PAYT system. In the Estonian region of Harju, the establishment of IMC would save around EUR 28 per inhabitant and year (including a raise in the residual waste fee) in an optimistic scenario and EUR 10 per inhabitant in a more realistic projection (Põldnurk, 2015). In Germany, e.g. the cities of Dreieich and Neu-Isenburg reduced their garbage fees from 1.1.2015 by 10 % as a result of inter-municipal cooperation. This was possible because the expenditures on material resources decreased due to IMC. E.g. the 120-liter residual waste bin is priced at EUR 20.20 instead of EUR 22.60 per month at 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 and year (Werwitzke, 2013).

Driving forces for implementation

The existence of a 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 costs 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 require of techniques and technologies that require higher capital investment and would be unaffordable for a single, small municipality.

Reference Organisations

The *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 reference, see http://sybert.fr/presentation.html.

In addition, the establishment of new IMC schemes has been and will 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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3.11. BEMPs on Waste Treatments

3.11.1. Sorting of co-mingled packaging waste

Description

In many parts of Europe[30], lightweight packaging waste (i.e. packaging made of plastic, composites, aluminium and steel, sometimes including also paper and cardboard packaging) is collected together in order to ease the waste separation task for consumers and reduce collection costs.

When that is the case, in order to guarantee a high level of recycling, an advanced sorting of the co-mingled packaging waste 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) allow sorting and achieving the high level of segregation that allows recycling of a very high share of the mixed packaging waste collected from households.

Figure 3.49 shows the scheme of a modern sorting plant to obtain nine main fractions of recyclables from co-mingled waste packaging. The input of this plant is packaging waste not containing the paper/cardboard fraction as this is collected separately. This plant is based in Germany, where lightweight packaging is collected in yellow bags and bins at kerbside (so called yellow bin or yellow bag) and, sometimes, also in on-street containers (ARGUS et al., 2001; Gerke and Pretz, 2004).

Thanks to a number of near infrared (NIR) sorting machines, this plant enables the separation of different types of plastic such as polyethylene (PE), polystyrene (PS) and polyethylene terephthalate (PET), in addition, there is also a mixed plastic, the Tetrapak and the foil fraction. Further, non-ferrous metals, especially aluminium, and ferrous metals are separated as well as paper/cardboard, the latter is present to a low extent as this fraction is separately collected.

[30] Co-mingled collection of lightweight packaging started in Germany in 1990 when the Duales System Deutschland GmbH (DSD GmbH) was founded by trade and industry in order to fulfil their legal obligation towards packaging waste. Following the implementation of the EU Packaging and Packaging Waste Directive (Directive 1994/62/EC) (EU Packaging, 1994), this EPR-based model is now followed with some degree of variation in the 28 EU Member States, plus Turkey, Serbia, Norway, Iceland, Ukraine and four provinces of Canada (Cimpan et al., 2015).

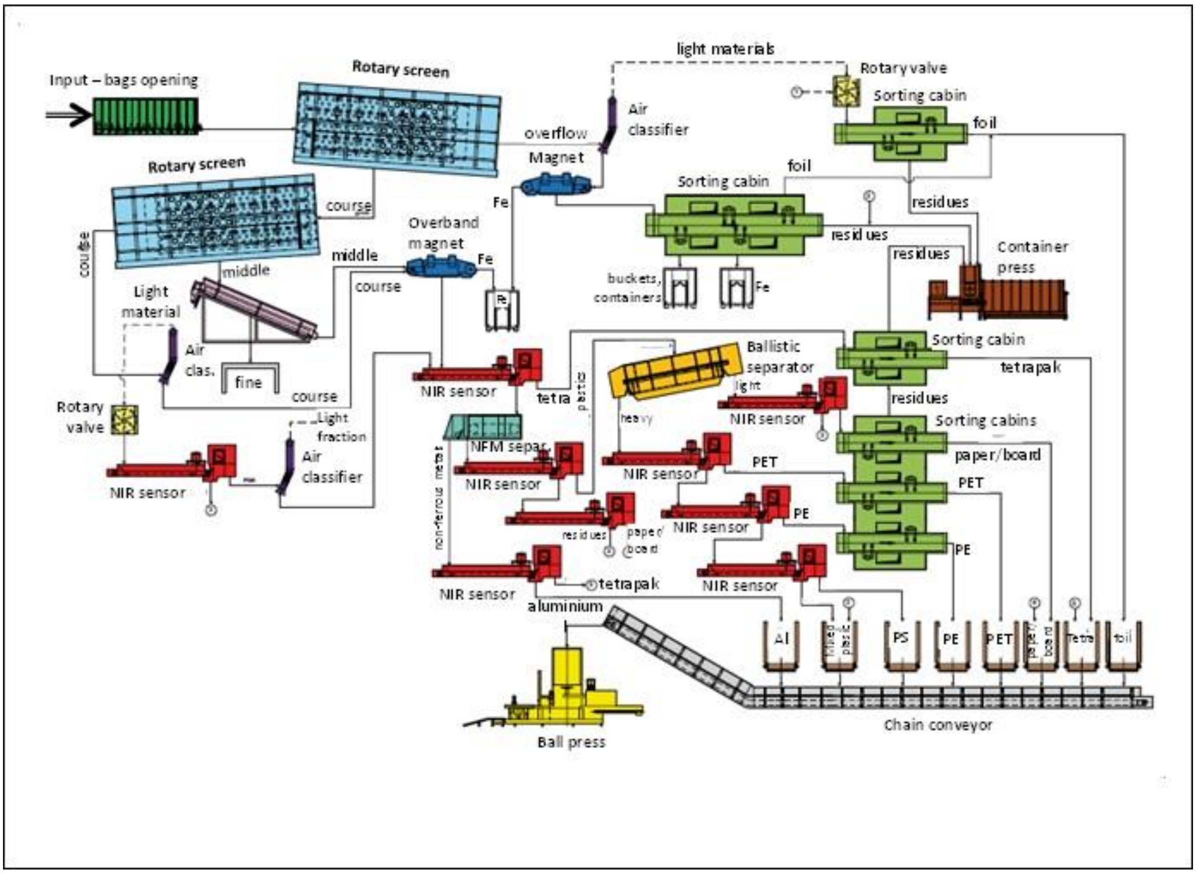

Figure 3.49. Scheme of a modern plant for sorting packaging waste (plastic, metal) to obtain nine main fractions of recyclables (Sutco, 2015)

Figure 3.50 shows the scheme of a similar plant for the sorting of co-mingled recyclables including the paper/cardboard fraction, i.e. paper/cardboard is not separately collected. There are less NIR sorting machines compared to the plant shown in Figure 3.49 but nevertheless different plastic fractions are generated.

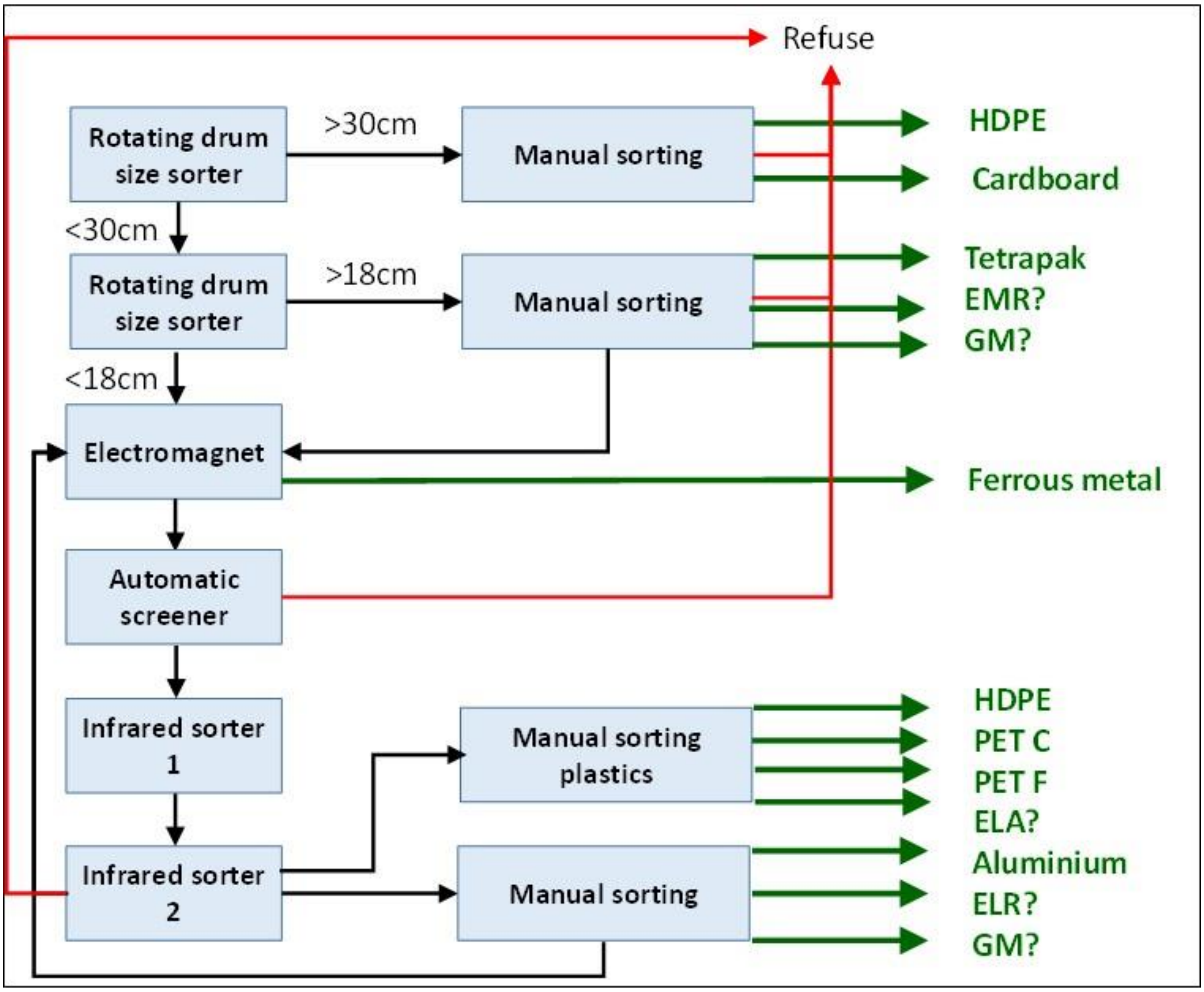

Figure 3.50. Scheme of the sorting plant in Besançon to process co-mingled recyclables (plastic, paper/cardboard, metals)

Achieved Environmental Benefit

The sorting of co-mingled lightweight packaging enables the recycling of plastic (different fractions), paper/cardboard, ferrous metals and non-ferrous metals. Thus, the material cycle can be closed or, when co-incinerated, the packaging waste can substitute fossil fuels.

Appropriate environmental indicators

The percentages of recycled/recovered materials contained in the material processed, specific for the different fractions, are appropriate environmental indicators, e.g. percentage of ferrous metal recovered from the material processed.

Cross-media effects

The operation of the waste sorting plant is associated with electricity consumption. Precise figures need to be identified. Emissions of dust and odour can occur but do not appear to be significant. 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

State-of-the-art plants can have up to a total of 20 NIR (near infrared) sorting machines. In addition to NIR, multi-sensor systems 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 help the plant operator to react to changes in the volumetric flow in the plant set up. Notwithstanding the high level of automation, these installations need to be complemented with some manual quality control in order to correct for systematic sorting errors and achieve some refining tasks before products are ready for the market (Bünemann et al., 2011; Christiani, 2009). As a consequence, high recycling rates can be achieved (Table 3.23).

Table 3.23. Material recovery in state-of-the-art lightweight packagingplants, (Cimpan et al., 2015), based on (Bünemann et al., 2011)

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	>80 %	Mechanical recycling or energy recovery
Residues	-	-	Energy recovery

Applicability

In principle, there are no barriers to build and operate a packaging waste sorting plant. However, careful planning (especially considering the plant capacity) is required as part of an integrated waste management concept including awareness raising and information campaigns for citizens and efficient waste collection.

Economics

The economics of different collection and sorting systems vary widely depending on system specifics, such as location, size, whether they serve urban or rural communities and many other factors (Cimpan et al., 2015).

During a study that examined seven European Member States (Portugal, Belgium, France, Germany, Romania, the UK and Italy) (da Cruz et al., 2014; Marques et al., 2014), a balance of costs (packaging collection and processing costs) and benefits (extended producer responsibility financial support, sale of materials and other revenues) was compiled for each country. It revealed that, from a financial perspective, costs were fully balanced by benefits only in Germany and Belgium. Costs for collection and processing (sorting) of packaging waste varied considerably across these countries due to widely different management systems (collection and sorting systems) and background conditions (e.g. salary levels, landfill and incineration taxes and gate fees).

Driving forces for implementation

The European Packaging Directive has been the most important driving force for sorting packaging waste and/or co-mingled packaging waste and paper/cardboard.

Reference Organisations

There are about 50 sorting plants in Germany for sorting co-mingled packaging waste.

Other reference plants are:

- the sorting plant in Besançon (France) which processes co-mingled packaging waste including paper/cardboard;

- the Migros-plant in Zurich (Switzerland) treating packaging waste http://www.industrie.de/industrie/live/index2.php?menu=1&submenu=4&type=news&object_id=33711881.

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3.11.2. Decentralised composting

Description

Decentralised 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. Decentralised composting avoids the economic costs and environmental burdens associated with organic waste collection, and can represent best practice by diverting organic waste from landfill or incineration *in situations where the environmentally preferred options, anaerobic digestion or centralised composting, are not possible*.

A major advantage of decentralised 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 increasing overall recycling rates and thus decreasing residual waste volumes (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 decentralised composting is the replacement of peat used in hobby gardening (Andersen et al., 2012).

In this BEMP, best practice for implementation of decentralised composting is described for situations where anaerobic digestion or centralised composting is not possible. Key aspects of best practice are:

- Undertake a feasibility study for anaerobic digestion of wet organic waste before committing to decentralised composting (see BEMP on integrated waste management strategy).
- Provide information and equipment to households to encourage home composting.
- Establish and train citizens to manage community decentralised composting facilities in urban areas (Figure 3.51).

Figure 3.51. Example of a community composting point in Bescançon, France (© E3 Environmental Consultants Ltd)

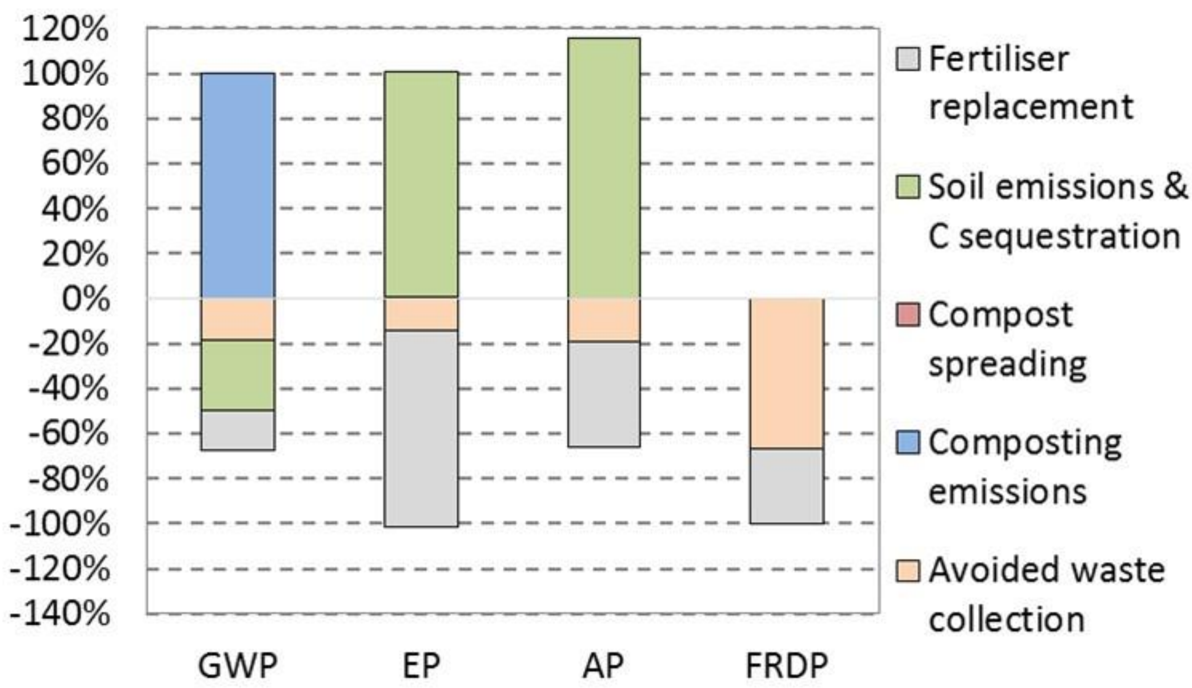
Achieved Environmental Benefit

Life cycle assessment of composting

Table 3.24 and Figure 3.52 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 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 (2 yrs), only 11 % of organic N is likely to be plant-available 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 3.24. Environmental burdens and credits calculated for home composting using life cycle assessment

Environmental burdens	Environmental credits
• Methane emissions during composting of 2.3 kg CH_4-C per tonne wet waste, median of 0.4 to 4.2 kg CH_4-C reported in Andersen et al. (2012). • Nitrous oxide emissions of 0.075 kg N_2O per tonne wet waste (Saer et al., 2013), which corresponds closely with 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.	• Avoided fertiliser manufacture and 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 yr) 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.: Represents best case where available nutrients in the compost replace fertiliser application.

Figure 3.52. 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 the assumptions described in Table 3.24, 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 credit 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 ten-fold to 331 kg CO_2e per wet tonne OHW.

However, Andersen et al. (2012) reported a modest additional GWP credit for OHW compost on the assumption that approximately 20 % of 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 a 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 low organic matter content, especially heavily cultivated soils on arable farms. Although compost produced by decentralised 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 decentralised composting schemes could also provide compost (free or at a price) to local horticulture enterprises.

Appropriate environmental indicator

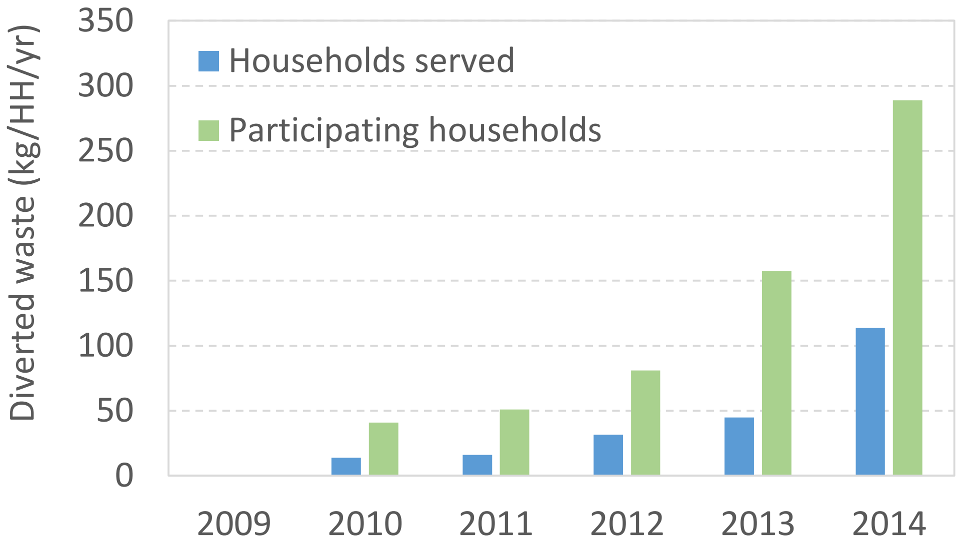
WRAP (2008) proposes the use of an indicator devised by Parfitt (2005):

- Mass of organic waste diverted from landfill through decentralised composting, kg/household/yr

This may be adapted to account for organic waste diverted from incineration:

- Mass of organic waste diverted from landfill or incineration through decentralised composting, kg/household/yr

An example of the above indicator used to track performance of a new decentralised composting scheme initiated by Sybert in Besançon is provided below.



Source: SYBERT (2015).

Figure 3.53. Organic waste diverted from incineration from households served by, and participating in, a new decentralised composting scheme rolled out by Sybert in Besançon

An important related indicator that may be more directly measured by waste management organisations is:

- Percentage of organic waste present in collected residual waste (% mass)

WRAP (2008) applied regression modelling to survey data for home composting at the household and district level in order to estimate the effect of a WRAP scheme to promote home composting. They categorised households as "WRAP enhanced" where composting was already practised for more than a year, but where a WRAP compost bin had been bought within the past six months, and "WRAP new recruits", where composting was initiated upon purchase of a new WRAP composting bin. "WRAP enhanced households" were estimated to divert a total of 115 kg/household/yr from kerbside collection, of which around 112 kg/household/yr was attributable to the enhanced effect of participating in the WRAP home composting scheme, and of which 72 kg/household/yr was diverted from residual waste collection and therefore probable landfill. "WRAP new recruits" were estimated to divert a total of 97 kg/household/yr, of which approximately 47 kg/household/yr was diverted from residual waste collection, suggesting that households new to home composting can achieve levels of diversion comparable to established home composters within just six months of initiating composting.

Cross-media effects

Composting may give rise to emissions of methane, nitrous oxide and ammonia if not adequately aerated.

Poor household separation of organic waste could potentially lead to soil contamination with plastics and potentially toxic compounds such as heavy materials (e.g. from batteries).

When promoting decentralised composting, the risk of soil eutrophication has to be taken into account. For example, in Germany, around 50 % of gardens are smaller than 300 m^2, around 29 % are smaller than 140 m^2 (Oetjen-Dehne et al., 2015). A lot of these gardens are used as decorative gardens. Here, the "soil-plant-compost-soil"-cycle is probably not closed, in which case compost applications to restricted areas (e.g. no application on lawns) may supplement mineral fertiliser applications, leading to an excess of imported nutrients and consequently leaching and eutrophication of local water bodies. Following Oetjen-Dehne et al. (2015), some nutritional studies have shown that between 60 % and 80 % of private gardens in Germany are oversupplied with nutrients, including phosphorus.

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 online 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 3.54).

Figure 3.54. Screenshot of one page from the WRAP guide to composting at home

Communal decentralised 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 alongside diverting waste from the residual waste stream.

Sybert is a waste management company located in the Besançon region of 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:

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As of 2015, 11 composting sheds were installed in very dense areas, with ten 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 2-3 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 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 at the foot of 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 at the foot of apartment buildings. Since 2012, 740 tonnes of organic waste have been diverted from residual waste incineration through these facilities.

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One automatic rotating drum composter 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 decentralised composting.

Figure 3.55 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.

Figure 3.55. Screenshot of district composting locations in Brussels.

Source: WORMS (2015).

Training

The example of Sybert (above) included significant efforts in training of local volunteers to manage decentralised 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 under "Reference organisations", it can be seen that Vlaco train volunteer 'Master Composters' to inform local citizens on management of home compost 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 acceptable quality. Appropriate training is essential to avoid some of negative environmental outcomes that can arise from poorly-managed composting (see "Cross-media effects").

Applicability

Anaerobic digestion and incineration with energy recovery are preferred options for "wet" (e.g. food waste) and "dry" (e.g. wood cuttings) organic waste respectively. Composting may be considered best practice only where the aforementioned options are not possible for those waste fractions. In such cases, there are no major restrictions to implementation. However, the success of decentralised composting as an environmental management strategy is highly dependent on management of the waste separation and composting process by citizens. Citizens 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 decentralised composting in urban areas, but it is possible, as demonstrated for Besançon by Sybert, among other cities.

Economics

Costs

Eunomia (2007) estimated the costs for the waste management company/authority for instigating household recycling (Table 3.25). 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 decentralised composting, assuming a bin lifespan of 10 years.

Table 3.25. 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

Decentralised composting avoids a number of costs for waste management organisations, most notably:

- Avoided waste collection costs
- Avoided waste management or disposal (landfill) costs.

According to cost benchmarking data presented in the BEMP on cost benchmarking (section 3.5.1) 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 decentralised 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 3.26.

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 3.26. 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 food waste)	Avoided fertiliser costs (EUR per tonne 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 decentralised composting as the lowest-cost option to divert organic waste from landfill.

Another important factor driving decentralised composting is the fact that it counts towards "waste prevention" under statistical accounting rules, because it avoids the collection and classification of "waste". Thus decentralised composting may count towards waste prevention targets established by local authorities and/or WMOs, 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 3.20. 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 3.21. Example of decentralised composting implemented by Sybert in Besançon, France

Sybert is a waste management company in Besançon, France, that is pursuing a strategy of decentralised home and community composting. Owing to absence of high feed-in-tariff subsidies for bio-electricity and high cost of collection, and possibly reflecting small local agricultural areas for digestate disposal, Sybert did not pursue anaerobic digestion. Sybert 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 into 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 decentralised composting that Sybert know of in France.

Source: Sybert (2015).

Box 3.22. Example of home composting and organic waste management promoted by Vlaco npo in Flanders

In Flanders, Vlaco npo supports and implements sustainable bio-waste 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 thousands of 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 the last 20 years. For the moment, 2,700 of those 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: organizing courses (about the preventing and processing of organic waste); (co-)organising campaigns and events (Closed Loop Weekend, Closed Loop Festival, Floralies 2016 …); 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 intermunicipal 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 chipper, school games, compost information box…).

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 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 has an acceptable quality. 91 % of respondents that are composting at home are not experiencing problems with the composting itself or with the quality of the home compost. Almost all the compost produced is used at home.

Source: Vandenbroucke (2015).

Box 3.23. 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 to 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 target any citizen that intends to replicate the practices at their own household. The 3 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 trained on home composting course. The farm has received over 26,500 visitors since 2002.

Source: Lopes (2015).

Communal decentralised composting or district composting is realized 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 place. Examples of leaflets in the links below (German language only):

- Mustervertrag_Betreuung_Quartierkompostplatz.pdf (pdf, 21.6 kB)
- Infoblatt_Quartierkompost_Seemaettli.pdf (pdf, 55.8 kB)
- Infoblatt_Pflichtenheft_Quartierkompost.pdf (pdf, 48.7 kB)
- Infoblatt_Leitfaden_Quartierkompost.pdf (pdf, 104.0 kB)

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4. Construction and Demolition Waste (CDW)

4.1. Scope

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 during the last years, 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 proposal (EC, 2015). The industry, however, has pointed out that national circumstances are heterogeneous in European Member States and that the Waste Framework Directive is not an incentive any more for the industry of those countries or regions were 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 report on best environmental management practice of 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 logistics 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. Main identified gaps for public administration, and waste management and treatment organisations are listed below:

- Formulation of local, county and regional specific plans for construction and demolition waste, including the quantification of generated waste, required treatments and the integration with final users.
- The implementation of these plans (or part of them) through the participation in voluntary agreements at different scales for the achievement of recycling targets by e.g. arranging commitments on use and the establishment and participation in quality assurance schemes.[31]
- The management of hazardous substances, with a specific focus on PCBs-containing wastes, where new approaches are being developed.

[31] These two points are elaborated under two best environmental management practices oriented to develop strategies for construction and demolition waste management, and the quality assurance of the recycled product. These practices are dependent on the national and regional environmental policies rather than local, but its implementation and best use is quite dependent on the performance of waste authorities and waste management companies. The description of these best practices takes into account this rationale.

These issues are addressed in five BEMPs where plasterboard recycling represents an outstanding example of the implementation of best practices along the whole supply chain for a single material stream.

A full list of techniques is provided in Table 4.1. This list is comprehensive and considers also the current techniques considered in the Building and Construction Sector. A summary of the techniques already covered in the Building and Construction sector document are provided in more detail in section 4.2.

Table 4.1.Techniques Portfolio for the management of Construction and Demolition Waste

Management aspect	**Techniques in this document**	**Section in the Building and Construction document**[32]
Strategy	1. Construction and Demolition Waste Planning	Section on site waste management plans
Prevention	-	Section on designing out waste, Section on site waste prevention and management Section on material use efficiency
Collection	-	Section on site waste prevention and management Section on selective deconstruction of buildings Section on selection of environmentally friendly deconstruction / demolition techniques
Re-use	-	Section on Re-use of materials
Treatment	2. Quality assurance schemes 3. Acceptability of recycled aggregates 4. Recovery of plasterboard 5. PCBs release prevention	Section on Construction and demolition waste sorting and processing Section on Use of recycled materials

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[32] Direct link to the sections in the Building and Construction Document will be inserted when the final version is uploaded to the European Commission JRC website.

4.2. Best Environmental Management Practice for wastes in the Building and Construction Sectoral Reference Document

The Technical Report on Best Environmental Management Practice in 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 table is provided below (Table 4.2).

Table 4.2. 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, etc), 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 reuses 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.
	Materials 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 materials delivery (and in some cases for waste handling) are also considered under this BEMP.

Table 4.2. 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 over materials, products or auxiliary materials that are harvested at site. In the case of construction materials, it refers to bricks, tiles, slaps, beams, etc., as 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, its well-known economic performance and the environmental benefit of materials reclamation achieved from 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 background report. The first document is oriented to all construction stakeholders (designers, developers, contractors, etc.). However, there are wide fields of overlapping; 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 affects directly to the recovery of materials and to its quality. A good example is the separation of plasterboard, which, if not separated, is 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 quite heterogeneous map of practices for CDW in Europe. With that in mind, the best practices shown in this document and in the Building and Construction sector technical report try to draw a general picture of frontrunners achievements.

4.3. Best Environmental Management Practice for Construction and Demolition Waste

4.3.1. Integrated Construction and Demolition Waste Plans

Description

The elaboration of Integrated Waste Management Plans or Strategies is a common approach in local, county, and regional governments. However, waste authorities are not the only responsible for its 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 under 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. After waste is transported, 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 tools to the industry for the safe use of recycled aggregates. The most important is the existence of quality assurance schemes for secondary materials, which has demonstrated its ability of opening markets to recycled materials in the construction sector.

The elaboration of a local plan or a strategy to manage with 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:

- Involves stakeholders from the local construction industry, main developers, associations, NGOs and relevant public administration departments, including regional organisations.
- Prioritises waste prevention in construction projects through several instruments, both oriented 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 buildings re-use schemes (ICE, 2008), and other tools oriented to the avoidance in origin of the construction waste. 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 integrated plan for demolition is the Dutch demolition code of practice developed by VERAS (VERAS, 2014). The Code for Responsible Commissioning and Contracting during the Tendering and Execution of Demolition works describes what best practice is 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.

- Establishes minimum waste sorting and management requirements in construction sites of 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. 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, hazardous wood materials (such as sound absorbers, medium-density fiber boards, and glued laminated timber) (Frankfurt, 2013)
- Defines a performance baseline, based on actual quantifiable data and empirical observations.
- Identifies and quantifies future flows of wastes and establishes monitoring mechanisms. 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 (identify waste flows within a site) but have helped to 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.
- Calculates total costs and the impact of its implementation.
- Establishes objectives far beyond 70 % recycling in 2020 with appropriate monitoring mechanisms and, in some cases, enforcement mechanisms. Two examples were identified by Gradman et al., 2013, for the Committee of Regions: Region of Wales, with a recycling target of 90%, and the City of Copenhagen with an achieved recycling rate of 88%.
- Aims to clear guidance, especially for SME and very small producers. Clear guidance and communication campaigns, along with reduced prices for small generators or municipal collection points would help to avoid poor management, sorting or illegal dumping.

Planning of CDW management should consider all the stages in a construction project (WRAP, 2011a). In Figure 4.1, the relationship between waste minimisation and management strategies and the construction activity is shown.

Source: Adapted from WRAP, 2011a

Figure 4.1. Construction and Demolition Waste Strategies in relation to Construction Projects Life Cycle.

As stated in the EMAS sectoral reference document on Best Environmental Management 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 onsite construction activity. Construction companies usually manage and transport an important amount of wastes, and usually need to get a waste manager permit to operate their own sites.

Achieved Environmental Benefit

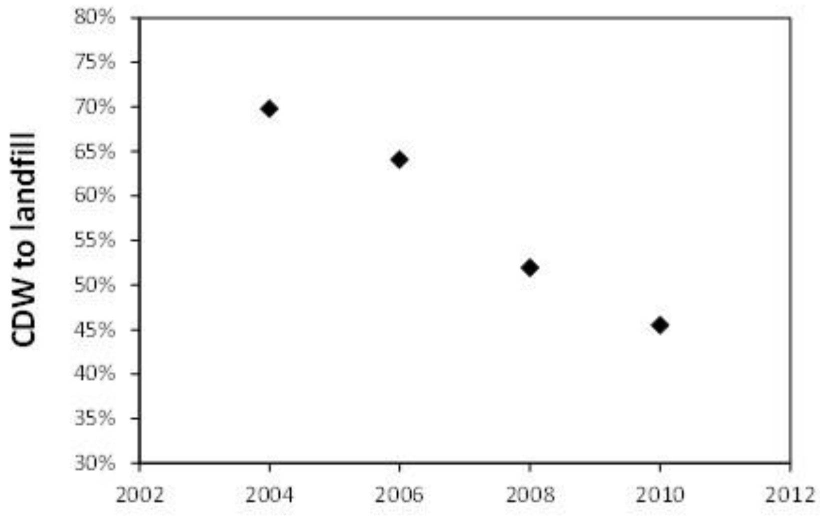
The impact of plans is not easily quantifiable, as it enables a number of techniques, which are applied with different degrees of success, and the influence of those plans in its application is uncertain. As an example, the avoided impact on the proper application of waste sorting techniques, through 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 4.3.

Table 4.3. 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 as 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 on the amount of CDW going to landfill can be observed in Figure 4.2 (see *Implementation of national strategies at local level* in Operational data for details on the applied instruments).



Source: Data from Defra (2011)

Figure 4.2.Construction and demolition waste going to landfill in England

However, it is challenging to differentiate the impact of isolated waste management plans from 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 in the 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 on the concrete formulations, plus smaller amounts of recycled rubber and asphalt for landscaping purposes. One of the main benefits, however, was considered the increased awareness and the availability of more sustainable materials in the local construction centre (Environcentre, 2015).

Appropriate Environmental Indicator

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:

> *the percentage (%) of total generated waste, correctly segregated and managed towards materials recovery, re-use or any other type of valorisation.*

The development of this indicator should the real amount of waste for the calculation and not estimations. The efficiency of materials 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 to monitor local authorities, 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 4.4 shows reference values calculated by BRE in its SMART Waste model (BRE, 2010).

Table 4.4. Environmental Performance Indicator: reference values 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)

Cross-media effects

An important observed fact is the increase of illegal dumping of CDW as a consequence of the (i) increase of waste management fees and other economic instruments, especially in the case of small producers and (ii) the increased requirement of waste sorting. Although better regulation enforcement is required on 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 planning estimations of 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 forecasting. In Europe, average CDW generation is around 1 tonne / person / year (McBean and Fortin, 1993; BioIS, 2011).
- **Financial Value Extrapolation.** It is proven that, given the specific value of buildings or construction projects, certain wastes 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 4.5 shows average CDW generation rates.

Table 4.5. 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 retrieving can become an endless procedure full of 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 main stakeholders, it is important to establish mechanisms for their mobilisation and participation into the planning process, not only as a reactive process (complaints, opinions, etc) but also active through e.g. data provision, early participation in committees, etc. This would provide 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 and partial views are avoided. According to ISWA, 2012, the best-functioning SWM systems should involve all the stakeholders in planning, implementing, and monitoring the changes. In this sense it is crucial the waste authority demonstrate a range of good practices in issues such as:

- **Consultation, communication and involvement of users**. Usually achieved through information campaigns, targeted letters, social media, etc.
- **Participatory and inclusive planning**. Those stakeholders that expressed interest would become part of a local steering committee that meet 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 meet regularly and have decision making attributions is an recommended practice (ISWA, 2012) .

WRAP can be considered a frontrunner in the implementation of best practices in stakeholder management. A good example is considered 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 commitment (WRAP, 2011bc).

At a more practical level, Copenhagen developed an exemplary bricks reuse system, still in pilot phase, with the help of local collection centres, 'recycling hubs', that the city manages, involving construction companies, builders and other stakeholders for the re-use of bricks from construction sites (Copenhagen, 2014). Also, bricks can be sold in local stores (second hand or construction materials supplies).

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 on waste and was considered a best practice by the European Commission (EC, 2009). However, it failed on achieving its main objective, to reduce by half CDW going to landfill (CPA, 2012) due to an unexpected increase of 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 them were actual big players in construction) from the whole supply chain of construction, 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.
- Implement site waste management plans and monitor waste and its treatment.

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 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

The minimum content of waste management studies for licensing are:

- 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 4.6).

Table 4.6. Ratio of waste generation, total and per material, assumed for permitting purposes in the Basque Country

	New, residential building	New, 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	1 %	1 %	0.5 %	0.5 %
Hazardous waste	2.3 %	2.3 %	2.5 %	2.5 %

Source: Basque Country Government (2012)

Applicability

The formulation of local waste management plans for CDW is a well extended instrument for larger counties and municipalities, by local authorities expecting a large impact from construction and with the collaboration of waste managers. A good example of a waste management plan for CDW at county level can be found for Hastings Borough Council (UK), which establishes clear objectives for CDW, since they observed that half of the total waste going to landfill was actually CDW (HBC, 2015). The size of the municipality would however have a high influence in the commitment of resources and on the development of plans and its implementation. The enforcement of the regulatory measures should be oriented to avoid illegal dumping, but awareness instruments and municipal collection centres for CDW have also shown to be effective.

Economics

Some examples of waste management fees applied by waste management companies are shown in the report on best environmental management practice in the building and construction sector (EC, 2012). Prices range between EUR 6/ton (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 the 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 extensively been included in national CDW strategy plans, but that is out of the scope of this document.

Driving force for implementation

Given the small economic savings of best practice in waste management, implementation driving forces are regulations, mandatory schemes, green credentials through enhanced environmental performance and awareness.

Reference organisations

Organisation providing best practice guidance on CDW management strategies: WRAP (UK), BRBL Recycling (NL), GERD (ES), RUMBA Guidelines (AT), Bundesverband der Deutschen Recycling-Baustoff-Industrie resp. Kreislaufwirtschaft Bau (2015) (DE), International Solid Waste Association (ISWA, 2012)

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4.3.2. Quality assurance schemes

Description

Quality assurance schemes are a key element for the marketing of secondary materials produced from CDW. Construction industry has had a very conservative approach to innovation, partly due to its traditional behaviour, partly due to the legal liability of architects, engineers, developers and contractors on the quality of the final building, making the sector to minimise risks. That is why construction relies heavily on standardisation, and any new technique or material needs to fulfil existing standards or to be supported by new ones. Also, construction in Europe has low profit margins and every decision is influenced by its economic results. Traditionally, recycled aggregates, RA, produced from CDW have had very few applications, usually as backfilling materials for exhausted quarries, some road sub-base applications and as inert cover of landfills. These applications do not require a high quality of aggregate, and save natural aggregates for other applications, as concrete manufacturing.

In order to improve the confidence of the sector on secondary materials, quality assurance schemes are in place for recycled CDW and are usually steered by waste managers in collaboration with the final users. For instance, in the next section on plasterboard recycling, it is described how the UK prescribed End-of-Waste (EoW) criteria for the plasterboard industry, which can make recycling rates to increase over 60 % (WRAP, 2011).

So, a waste manager would apply a Best Environmental Management Practice when produces recycled products under a quality assurance scheme that:

- Aims for an increased uptake of recycled aggregates by the industry. For that, the strategy follows a voluntary agreement approach or similar, being highly inclusive.
- Encourages waste segregation and diversion from landfill and, at the same time, includes environmental-related criteria e.g. for their leaching characteristics, with the achievement of EoW character or similar to the secondary material produced.

The market on recycled aggregates for higher quality applications, however, still needs support. Concrete manufactures do not feel comfortable using RA or recycled concrete aggregates (RCA) even if it is proven that their performance is as good as the natural aggregate. On that regard, usually at regional level, waste managers, construction companies and, up to a certain point, the public administration have established quality assurance schemes of recycled aggregates, using voluntary agreements rather than regulations. For example, a voluntary quality requirement for construction and demolition waste recycled products was established in the Baden-Württemberg region in Germany (QRB, 2009) and in other German states as Berlin or others (APPRICOD, 2006). Three levels of quality are proposed based on the leaching characteristics towards certain pollutants, establishing the suitable applications of the aggregates (EC, 2012; QRB, 2009 – see operational data for more information).

There are other similar approaches in Europe, usually initiated due to the need for acceptance criteria of the industry and based on voluntary agreements. For instance, a non-exhaustive list was reported by Delgado et al. (2009):

- The Austrian construction materials recycling association developed guidelines for recycled aggregates with a quality certification fulfilling criteria for natural aggregates where environmental-related parameters are also included.
- In the region of Flanders, Belgium, recycled aggregates can only leave the waste status if they meet specific requirements on chemical composition (both for solid content and leaching properties)
- In Finland, the SFS standard 5884 sets a technical protocol for the acceptance of crushed concrete products, including an environmental set of parameters.
- In the UK, the WRAP aggregates programme, Aggregain, was established, with some quality specific protocols developed for demolition practices, CDW management, and recycled aggregates.

Methodologies for the development of EoW criteria are, however, far to have a harmonised approach in Europe. Saveyn et al. (2014) noted that current requirements in many Member States are less stringent for natural or manufactured aggregates than for those coming from waste.

This technique is strongly linked to the section on "Use of Recycled Aggregates" in the Best Environmental Management Practice document for the construction sector (EC, 2012), aimed to inform the sector on the possibilities of recycled aggregates, for low and high quality applications. This section aims to show quality requirements for those applications and how they are articulated in recent standards and/or quality assurance schemes that waste managers can apply, and excludes the internal quality system at the waste manager. The existence of quality documentation standards, as ISO 9001, is an added commercial value that may help also to the identification of main inefficiencies (*quality of processes*). This BEMP only considers the application of standardised criteria to ensure the reliability of secondary products use (known as *quality assurance*).

Achieved Environmental Benefit

The main environmental benefit is derived from the increased use of secondary materials due to a higher confidence from the industry on the material they use, due to the reliability of the quality standard that backs up the performance of the material produced by the waste manager. The final aim, of course, would be the reduction of natural aggregates from quarries, which normally have a large impact on the local environment. Baden-Württemberg and Berlin recycling rates are higher than 90 %, higher than the German average. Experts attribute this to the existence of voluntary agreements on quality and acceptance criteria of the produced material.

In terms of greenhouse gases emissions and primary energy consumption, recycled aggregates have a proven lower footprint than their equivalent natural aggregate. The use of recycled aggregate supposes a net reduction in the CO_2 emissions and primary energy consumption, as the production and extraction of new raw material is avoided (Table 4.7). Nevertheless, the large influence of transport for the performance of recycled aggregates may produce different results depending on local circumstances.

Table 4.7. Life cycle environmental burdens for one tonne of Construction and Demolition Waste treated according to different methods

Treatment	Global warming potential, kg CO_2e/t	Primary Energy, MJ/t	Land Use PDF*, m^2a/t
Collection	6	100	0.15
Landfill	15	300	0.80
Recycling	2.5	45	0.18

Source: Blengini and Garbarino (2010)

*Potentially Disappeared Fraction, Ecoindicator 99 method

Appropriate environmental indicator

Several indicators can be used to monitor the achievements through quality assurance schemes, as the amount of recycled materials marketed, in absolute units (e.g. tonnes) or the percentage of natural materials substituted by recycled aggregates, e.g. for concrete manufacturing. Public administrations may also register the amount of recycled materials sold from recycling plants under the quality assurance scheme in place and can track the progress of the scheme.

Substitution of materials in new construction products seems to be one of the most important indicators in terms of measuring achievements. For applications in structural concrete, it is not recommended to substitute more than 20 % of aggregates in concrete manufacturing, while substitution rates of 100 % are achievable. Looking at total volumes of CDW and used inert materials as aggregates, the ratio is 1:5, so high recycling rates are fully achievable (and *achieved* in some Member States).

Cross-media effects

There are some trade-offs identified in the energy consumption and e.g. GHG emissions due to transport needs, however, recycled aggregates tend to travel less than natural aggregates, so the total balance would still remain positive for the recycling option. This may depend on the local or regional circumstances (BioIS, 2011).

Operational data

Quality requirements under EN 12620:2013

Among the desired characteristics of the recycled aggregates, the most important are related to the performance. Standard EN 12620:2013 (CEN, 2013) specifies the properties of aggregates, obtained by processing natural, manufactured or recycled materials and mixtures of these aggregates for use in concrete. It is an attempt to standardise, under the construction products regulation, the quality requirements for aggregates. It also includes a general requirement that aggregates should not release any dangerous substances, in excess of the maximum permitted levels, specified in a relevant European standard or regulation. Table 4.8 shows general recommendations for coarse recycled aggregates according to the standard.

Table 4.8. Technical parameters according to EN 12620:2013

Property	Description
Flakiness Index	Measures size and shape of the aggregate
Resistance to fragmentation	Measure of the aggregate quality to resist fragmentation during handling and mixing, usually lower in recycled aggregates
Oven dried particle density	Usually, recycled aggregates tend to have less density than naturally sourced materials
Water absorption	Water absorption capacity of RAs is higher than that of natural aggregates; it is an indication of the workability of the final mix
Constituents	Crushed concrete, unbound stone, crusher brick, asphalt, glass and others
Water soluble sulfate content	Aggregates sourced from concrete subjected to marine environments may have a high soluble chloride and sulfate content and then a specific approach is needed to guarantee a sufficiently low concentration of chloride or sulfate ions
Acid-soluble sulfate content	
Acid-soluble chloride content	
Drying shrinkage	Quite relevant for recycled materials, although the shrinkage caused by a 20 % substitution is negligible

This standard is of high relevance for the development of quality schemes and also labels. EQAR has developed a labelling scheme for aggregates from mineral wastes from construction and demolition activities. Other standards would be applicable in the case of aggregates for roads (EN 13242), for asphalts (EN 13043), etc. (EQAR, 2013).

Extra quality requirements for recycled aggregates

These requirements are mainly oriented to environmental protection and are especially focused on the leaching capacity of certain pollutants. These are not included in the EN 12620 standard and may be added in voluntary agreements or quality assurance schemes. In the Baden-Württemberg region (QRB, 2009), three levels of quality are foreseen: quality Z 1 is for material lying under non water tight layer. Z 1.1, which is the more restrictive, is for layers of materials placed at least 1 m above the water table. Quality Z 1.2. is for layers above at least 2 m above the water table and over compact material. The less demanding quality is Z 2, which is placed under water tight layers (concrete or asphalt) and above 1 m of the water table. These quality requirements, to be measured under leachability with water, DIN 38414, are shown in Table 4.9. These are requirements to add to those for any other aggregate.

Table 4.9. Quality levels according to different leachability tests for Recycled Aggregates in Baden-Württemberg

Quality level	Z 1.1.	Z 1.2.	Z 2
Organic material, C_{10}-C_{22} (C_{10}-C_{40}), mg/kg	300 (600)	300 (600)	1,000 (2,000)
PAH, mg/kg (EPA method)	10	15	35
Extractable organic halogens, mg/kg	3	5	10
PCB_6, mg/kg	0.15	0.5	1
As, μg/L	15	30	60
Pb, μg/L	40	100	200
Cd, μg/L	2	5	6
Cr, μg/L	30	75	100
Cu, μg/L	50	150	200
Ni, μg/L	50	100	100
Hg, μg/L	0.5	1	2
Zn, μg/L	150	300	400
Phenols, μg/L	20	50	100
Chloride, μg/L	100	200	300
Sulfate, μg/L	250	400	600
pH	6.5-12.5	6-12.5	5.5-12.5
Conductivity, μS/cm	2,500	3,000	5,000
N.B. Leachability with water tests made under DIN 38414			

Similar requirements can be found in other regions in Europe. An example is the regional regulation of the Basque Country on the use of recycled aggregates (Basque Country government, 2015).

Applicability

There is no main technical concern on the acceptance of recycled aggregates if they perform according to the standards and quality assurance schemes as shown in Operational Data. Generally, it is recommended to study on a case-by-case basis the applicability of a recycled product, especially when it comes to sensitive aspects, as structural concrete. But, as a matter of fact, the applicability of quality assurance schemes is ensured for all European regions with recycling facilities.

Economics

A previous study showed that the selling price of aggregates is around EUR 3 to 12 per tonne in Europe, which is quite competitive to natural aggregates (EC, 2012). The implementation of quality assurance schemes can ease a wider use of recycled aggregates from construction waste, but it is well known that the availability of low cost natural materials is a great disadvantage, as these materials are still preferred against secondary materials.

Driving force for implementation

The most important driving forces for implementation are environmental, as the reduction of CDW landfilled and the use of natural aggregates, which also have an economic dimension, as in most of the Member States, secondary materials tend to be cheaper than the natural material, sometimes due to taxes and levies.

Reference organisations

WRAP, Waste Resources Action Programme

European Commission, JRC-IPTS, is in charge of the development of the End-of-Waste criteria for aggregates. Many technical aspects of recycled aggregates from construction waste are well described at http://susproc.jrc.ec.europa.eu/activities/waste/, which gathers information on national approaches.

CEN, European Committee for Standardisation, TC 154 on aggregates for construction.

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WRAP (2011). Recycled Gypsum from Waste Plasterboard. End of waste criteria for the production and use of recycled gypsum from waste plasterboard. Report available at wrap.org.uk, last access in June 2015.

4.3.3. Improving the acceptability of recycled aggregates

Description

This BEMP summarises the main outcomes from the Building and Construction Document regarding recycling practices for concrete (EC, 2012). The focus of this section is on best practice on the selection of the products portfolio of recyclers of CDW, based on 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 aggregates, 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.

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 managers and other stakeholders. This section describes the range of products per application that waste treatment recyclers may consider, as they have been proven to achieve maximum recovery rates.

Recyclability of the inert elements of construction and demolition waste depends on the level of segregation at site where they are generated. Poor segregation leads to cost inefficient situations for waste recyclers, since the range of products would be heavily 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 recycling plant usually consists of:

- Reception, weighing and visual inspection.
- Manual preselection and rejecting to other treatments (depending on acceptability criteria, if 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 in origin is poor, manual separation of plastic, wood and other waste typologies may be needed.
- Crushing.
- Screening and secondary crushing (depending on produced aggregates and marketing of products).

A CDW manager has mainly to deal with the inert fraction (concrete plus masonry). From well sorted waste, waste managers are able to produce high quality aggregates 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 4.10.

Table 4.10. Classification of aggregates according to DIN 4226-100

DIN Classification	Type 1	Type 2	Type 3	Type 4
Recycled aggregates	Concrete and 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-pored 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, with a consideration to 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 % of 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 4.11 shows applicable solutions for the two main products produced in recycling plants, i.e. concrete aggregates and mixed aggregates.

Table 4.11. 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, sulfate 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 would be 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 % maximum 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 % of application 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 % of 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 main benefits, but the chemistry of the mixture does not allow using a substitution rate more than 10 % (Hauer, 2007). Nevertheless, 100 % substitution is allowed if technical specifications are met.

Table 4.11. Possibilities for recycled construction materials.

Material	Use	Applicability	Specifications/restrictions
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 Benefit

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. Table 4.7 in the previous section describes well these benefits.

In terms of life cycle environmental performance, generalisation is not possible, and each separated 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 balancing benefits from primary aggregate substitution, the type of application and the type and origin of the natural aggregate strongly influences the life cycle performance.
- However, washing, which is applied when site segregation is poor, can count 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 indicator

As stated before, the most important indicator that readily shows the environmental performance is the amount of waste diverted from landfill. This indicator is expressed as:

> *percentage (%) of total generated waste, correctly segregated and managed towards materials recovery, re-use or any other type of valorisation.*

The development of this indicator includes the real amount of waste for the calculation and not estimations. The efficiency of materials recovery at plants should be considered (e.g. rejects from recycling plant are not considered to be diverted). For a waste management organisation claiming the benefits of recycling, the substitution of primary aggregates, and therefore, its main environmental impact, can be estimated through the following indicator:

> *amount of recycled materials marketed, in absolute units (e.g. tonnes) or the percentage of natural materials substituted by recycled aggregates, e.g. for concrete manufacturing.*

This is not a straightforward indicator to calculate for recyclers. Although the waste treatment plant manufactures different qualities of products, some of them for high-grade applications, there is no proof of the actual substitution rate achieved at site.

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 4.12). This is the case with recycled aggregates, as they come from waste, whose composition is likely to contain some of the hazardous materials shown in Table 4.12, but also for those recycled products to be used for construction (e.g. slags, ashes, etc.). The Commission made 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, ground water, surface water and indoor air (Delgado et al., 2009). Actually, several (preliminary) technical standards and rules are under drafting/approval or have been published.[33]

Table 4.12. 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

[33] http://standards.cen.eu/dyn/www/f?p=204:32:0::::FSP_ORG_ID,FSP_LANG_ID:510793,25&cs=135BD767027D4B4E081006EF46B5E957C

Table 4.12. Hazardous materials in construction and demolition waste

Product / Material	Potentially hazardous components	Hazardous properties
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 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 leaching limit values. An assessment made by DHI (2011) on the leachability of some aggregates, is shown in Table 4.13.

Table 4.13. Recycled aggregates leachability: elements close to, partially exceeding or consistently exceeding the EU leaching limit values for acceptance of waste at inter 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

Table 4.13. Recycled aggregates leachability: elements close to, partially exceeding or consistently exceeding the EU leaching limit values for acceptance of waste at inter waste landfill

Product	Close to the limit	Partially exceeding	Consistently exceeding
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 to 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 de-dusting 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 emission from the commonly used diesel engines.

Operational data

Recycling plants

Recycling plants can be mobile, semi-mobile or stationery. 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. 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 4.14.

Table 4.14. Comparison of different crusher types in mobile, semi-mobile and stationary plants

Type	Advantages	Disadvantages	Applications
Semi-mobile and mobile with jaw crusher	Simple, rugged construction Low wear 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

Table 4.14. Comparison of different crusher types in mobile, semi-mobile and stationary plants

Type	Advantages	Disadvantages	Applications
Semi-mobile and mobile 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 size of 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 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 Best Environmental Management Practice in the Building and Construction sector document (EC, 2012), where a full description is provided. According to the managers of the plant, there is a healthy demand on recycled aggregates, especially for those coming from concrete. The mass balance for years 2009, 2010 and 2011 can be observed in Table 4.15. As shown, the total input matches the total output of materials, being the amount accumulated to be negligible (or even negative). Main fraction is concrete, followed by excavated materials and asphalt and bituminous materials.

Table 4.15: Input-output balance of the FEBA recycling plant

Waste Input	LoW Number	2009 Tonnes	2010 Tonnes	2011 Tonnes	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 Tonnes	2010 Tonnes	2011 Tonnes	2009+2010+2011
Waste for disposal	50	30	40	130
Sold scrap	260	170	420	850
Total	**310**	**210**	**460**	**980**
Product Output	**2009 Tonnes**	**2010 Tonnes**	**2011 Tonnes**	**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 up to 100 % are achievable, if some 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 4.16 and Table 4.17). For more information on the quality requirements, see section 4.3.2.

Table 4.16. 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 4.17. Proposed technical specifications to fulfil mechanical properties for non-structural concrete

Parameter	Value
Water absorption	< 12 %
Total S content	< 1 %
Sulfates (acid soluble)	< 1 %
Other materials (glass, plastic, …)	< 1 %
LA value (Los Angeles abrasion coefficient)	< 50 %
Fines content	< 4 %
Ceramics content	< 50 %
Gypsum content	< 2 %

Source: CEDEX, 2010

Further applicability is dependent on the level of waste segregation. For instance, as described above and also in section 4.3.4, gypsum content of CDW is extremely important on the applicability of recycled aggregates produced from them (Table 4.18).

Table 4.18. 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 sulfates.
>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 substantially not different from the natural aggregate cost and, in some circumstances, can even be lower. Selling price varies from EUR 3 to 12 and depends on many local circumstances, especially on transport costs (WBCSD, 2009) and quality. The high share of transport costs on total costs is highlighted by Hiete (2013) as a very decisive factor for C&D waste 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 tons per year, an utilisation factor of 80 % and with an 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 surroundings. Good segregation of construction waste at 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 main barrier for the uptake of recycled aggregates in most cases.

Main factors for 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)
- 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 driver for the application of concrete recycling is costs and the marketability of the final product, both induced through economic instrument affecting to 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

Organisation providing best practice guidance on CDW recycling and application of recycled aggregates: WRAP (UK), BRBL Recycling (NL), GERD (ES), CEDEX (ES), RUMBA Guidelines (AT), Bundesverband der Deutschen Recycling-Baustoff-Industrie resp. Kreislaufwirtschaft Bau (DE)

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4.3.4. Improving the recovery of plasterboard

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. Gypsum plasterboard life cycle has become an example of how a circular economy can work effectively. In Europe, 2.35 million tonnes of waste per year plasterboard from construction and demolition projects are produced and an extra 0.6 million tonnes are produced during its manufacturing and installation (GTG, 2015). However, almost all the waste plasterboard can be successfully fed into the manufacture of new plasterboard, as raw material for other uses and plasterboard itself can incorporate wastes from other industrial processes. Plasterboard produced with 89 % recycled material was achieved by Knauf in 2013 (Knauf, 2013). The importance of plasterboard segregation and its impact on the whole CDW reprocessing is of high relevance. A separate thematic area was set up by WRAP in the UK, where several local authorities introduced waste plasterboard collection at their Household Waste Collection centres, as e.g. Sheffield (WRAP, 2009). Also, at European level, Eurogypsum is currently coordinating the LIFE+ project GypsumToGypsum (GTG, 2015; Eurogypsum, 2014), aimed to integrate better the supply chain of gypsum-based products by closing the loop and to increase the quantity of gypsum based waste being diverted from landfill for recycling. Europe demands around 15 million tonnes of plasterboard, and the annual production of its waste is around 2.35 million tonnes. So, therefore, there is more than enough capacity for recycling.

Although it involves all actors of the supply chain, the description of this BEMP is oriented to inform waste authorities, with a more practical perspective based on the options for waste collectors and recyclers, among other actors of the whole supply chain of gypsum plasterboard. However, waste authorities have successfully developed pilot schemes for the collection of waste plasterboard in municipal collection centres (WRAP, 2009). Several best environmental management practices are identified around gypsum plasterboard supply chain and end-of-life:

1. Plasterboard is the main subject of designing-out waste practices in the construction industry, the right sizing and design of plasterboard panels, and just-in-time practices would reduce the amount of wasted plasterboard considerably.
2. Plasterboard is a durable product, so panels and tiles made of plasterboard, with no damage, can easily be reinstalled.
3. The product itself can incorporate secondary material up to virtually 100 % of the raw material, although industry still tends to use primary material. Knauf reported 89 % recycled gypsum coming from recycled plasterboard and flue gas desulphurisation process. So, in the definition of secondary material, flue gas desulphurisation is also taken into account (See operational data for more information).
4. Reprocessing waste plasterboard can produce gypsum of high quality, according to certain standards, with a variety of potential uses apart from new plasterboard: raw material for cement manufacture, roads sub-base, and soil improvement for agriculture. So, quality assurance schemes are required for gypsum produced from waste.
5. Waste plasterboard segregation benefits other CDW recycling, as sulfates, generally coming from plasterboard, are mixed with other CDW fractions in

unsorted waste management, which prevents the application of the recycled aggregate (see Operational Data in 4.3.3).

Main indicators and benchmarks for these best practices are anticipated in Table 4.19, as addressed in the document on Best Environmental Management Practice of the Building and Construction sector of the European Commission (EC, 2012). Best practices 3 and 4, however, involve waste managers, public authorities and private companies of the construction sector willing to reduce the environmental impact through the use of more sustainable materials. In this document, therefore, the main focus is laid on technological options for waste recovery and the B2B practices on the standardisation of the quality of reprocessed gypsum in waste collection, developed through the case studies shown in Operational Data.

Table 4.19. Best environmental management practice related to gypsum plasterboard

BEMP	Appropriate environmental indicator	Benchmark of Excellence	Reference
1 Waste prevention: Designing-out gypsum plasterboard products	Amount of waste reduction, %	Up to 60 %	Building and Construction Document (EC, 2012),
2 Waste re-use: Re-use of dismantled panels	Waste diverted from landfill*, %	95 %	Building and Construction Document (EC, 2012),
3,4 Waste recovery: Fully recyclable and able to incorporate virtually 100 % recycled materials	Reprocessed materials use rate, % Recycled gypsum from waste plasterboard incorporated to the product, %	100 % 25 %	Current BEMP
5 Waste collection: Segregation of plasterboard waste	Waste diverted from landfill, %	95 %	Building and Construction Document (EC, 2012),

*Includes re-used materials

Achieved Environmental Benefit

From the environmental point of view, gypsum plasterboard recovery is not highly advantageous compared to the manufacture from conventional raw materials, natural gypsum and synthetic gypsum (calcium sulfate from Flue Gas Desulphurisation (FGD)), however, its segregation from other streams of CDW is highly beneficial and pollution from other treatment options, as landfill, are avoided.

The main 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, coming from the lining of panels, has a negative effect on the product performance. For this level of recycling, the difference on the environmental performance of plasterboard production under several scenarios is relatively small, less than 10 % (see Table 4.20).

For instance, the reduction in GHG emissions from e.g. incorporating 15 % recycled materials would be only 2 %, in a low transport scenario, or 1.4 % in a high transport scenario, 4.5 % and 3.8 % for a 25 % recycling. These very low reductions are due to two main factors:

- The environmental impact is mainly allocated to the thermal stages: *calcination*, as defined in dehydration of gypsum to produce hemihydrate, $CaSO_4½H_2O$, the process 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 of 25 % of recycled content is assumed. Production of natural gypsum ready for the process (extraction, transport and pre-processing), is associated to the emission of 120 kg of CO_2e per tonne (84 kg in production), while collection, transport and pre-processing of recycled gypsum is 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 content of fibres, the higher the recyclability (see operational data for more information).

Table 4.20. LCA results for 1 tonne of Plasterboard, adapted from WRAP, 2008a

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 (100yr)	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

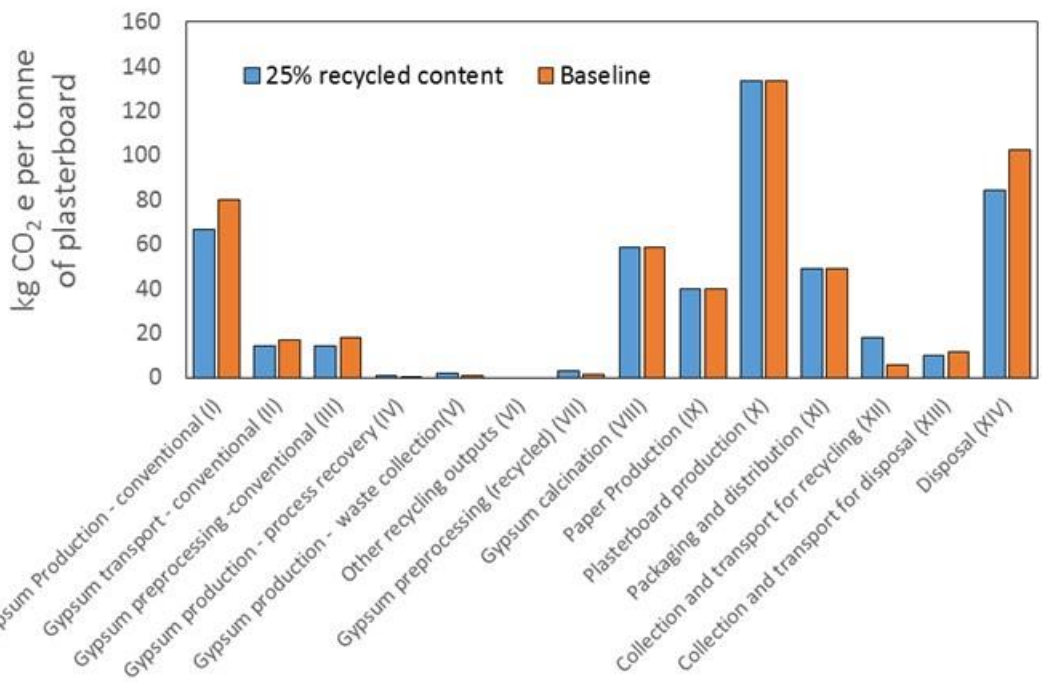
LT: Low Transport scenario, HT: High Transport scenario.

Regarding to the process contribution, Figure 4.3 shows the contribution to greenhouse gases emissions for each different stage and the life cycle flow chart reflecting all stages assumed in the study by WRAP (2008a). As shown, main contributors are plasterboard production (mainly drying), calcination, natural gypsum production and disposal. Disposal and production of natural materials, of course, are reduced once recycled materials are incorporated, but the extension of the benefit needs to be further optimised 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.

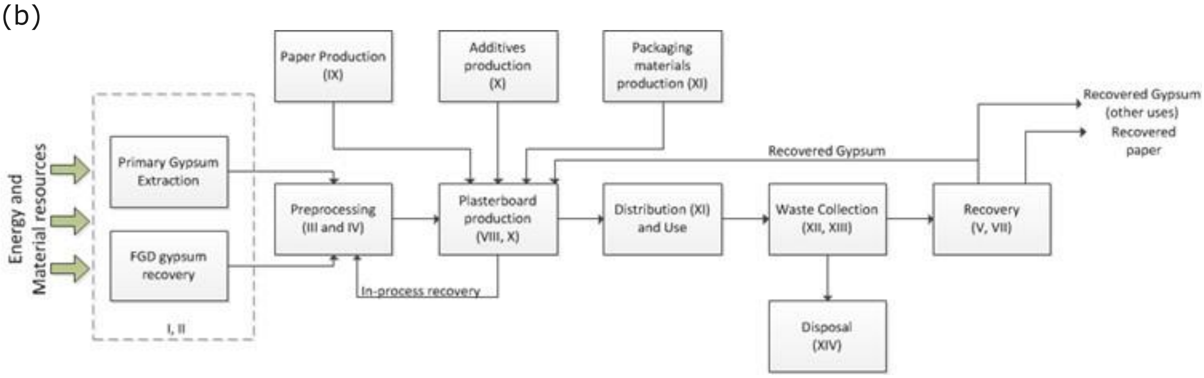
Another environmental benefit of gypsum plasterboard segregation and recycling is the removal of sulfates from the main bulk of construction and demolition waste, mainly composed by concrete. Gypsum in CDW is found to be around 5 to 10 % (Asakura, 2013), while the threshold value for the acceptability 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 sulfate-containing waste are being developed and successfully applied (Vegas et al., 2015)

Plasterboard waste can be problematic in landfill conditions due to the sulfate content of gypsum. When mixed with biodegradable municipal waste in a landfill, sulfate reducing bacteria form hydrogen sulphide in anaerobic conditions, which dissolves in the leachate in wet conditions or generates bad odours. Life cycle assessment confirmed that H_2S reduction up to 17 % in the low transport scenario can be achieved when 25 % of recycled gypsum is used in the manufacture of new plasterboard.

(a)



(b)



Source: Adapted and modified from WRAP, 2008a

Figure 4.3. Greenhouse gases emissions per process stage (a) and assumed supply chain for plasterboard (b)

Appropriate environmental indicator

Several indicators may be used in connection with the whole supply chain of gypsum plasterboard. A simpler way to understand some of the indicators is to represent the material flows of gypsum or plasterboard in the life cycle diagram (Figure 4.4)

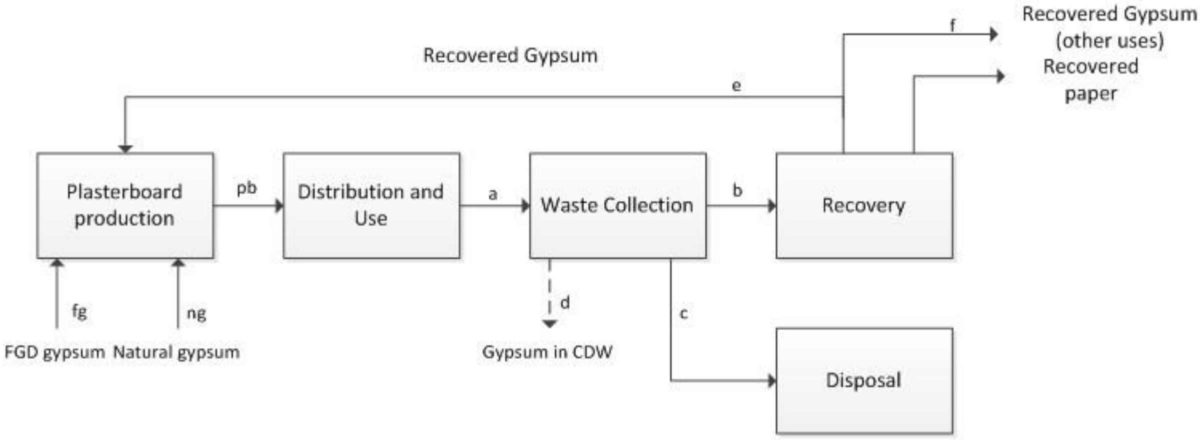


Figure 4.4. Supply chain simplification for the calculation of the indicators

The main indicators can be divided in the following categories, according to the mass flows from Figure 4.4:

- Industry indicators:
    - Reprocessed materials use rate, % = $100\frac{fg+e}{fg+ng+e}$. It indicates the amount of non-natural gypsum used in the production of plasterboard. An industrial reference for this value can be 100 %, that means ng = 0.
    - Internal recycling rate % = 100 $\frac{pb-(fg+ng+e)}{fg+ng+e}$. It indicates the amount of internal recycling required in a certain mill. It gives an indication on the acceptability of external recycled gypsum from waste plasterboard, because of the fibre content. It is usually around 5 %, as the early strength of plasterboard products is not high and breakages are frequent.
    - Recycled gypsum from waste plasterboard incorporated to the product, % = 100 $\frac{e}{fg+ng+e}$. The fraction *e/pb* is frequently used instead, however, it does

not reflect material losses in the manufacture, and it is higher than the real value. The industrial reference on this indicator is 25 %.

- Construction indicators
    - Waste plasterboard diverted from landfill, % = 100 $\frac{a}{a+c}$. This indicator reflects the amount of waste not being sent to a landfill but to recovery operations. A benchmark of 95 % was proposed for this indicator (EC, 2012).
- Waste Management indicators
    - Waste plasterboard collection efficiency, % = 100 $\frac{b}{a}$. The indicator evaluates the percentage of plasterboard sent to recycling operations. A benchmark of 95 % is proposed for the construction industry (EC, 2012).
    - Waste plasterboard recovery efficiency, % = 100 $\frac{e+f}{b}$. This value indicates waste plasterboard diverted from landfill and segregated from other CDW fractions. This parameter refers to gypsum only, an overall mass balance should also include recovered paper from the panels.
    - Sulfate content in plasterboard, % = 100 $\frac{d}{CDW}$. SO_3 content is a standardised quality measurement in all recycled aggregates and CDW fractions. A limit of 3 % is set by the industry standards, although it may need to be lower, depending on the final application of the recycled aggregate. When plasterboard is not segregated, values up to 10 % have been detected (Asakura, 2013).

Cross-media effects

No cross media effects are expected.

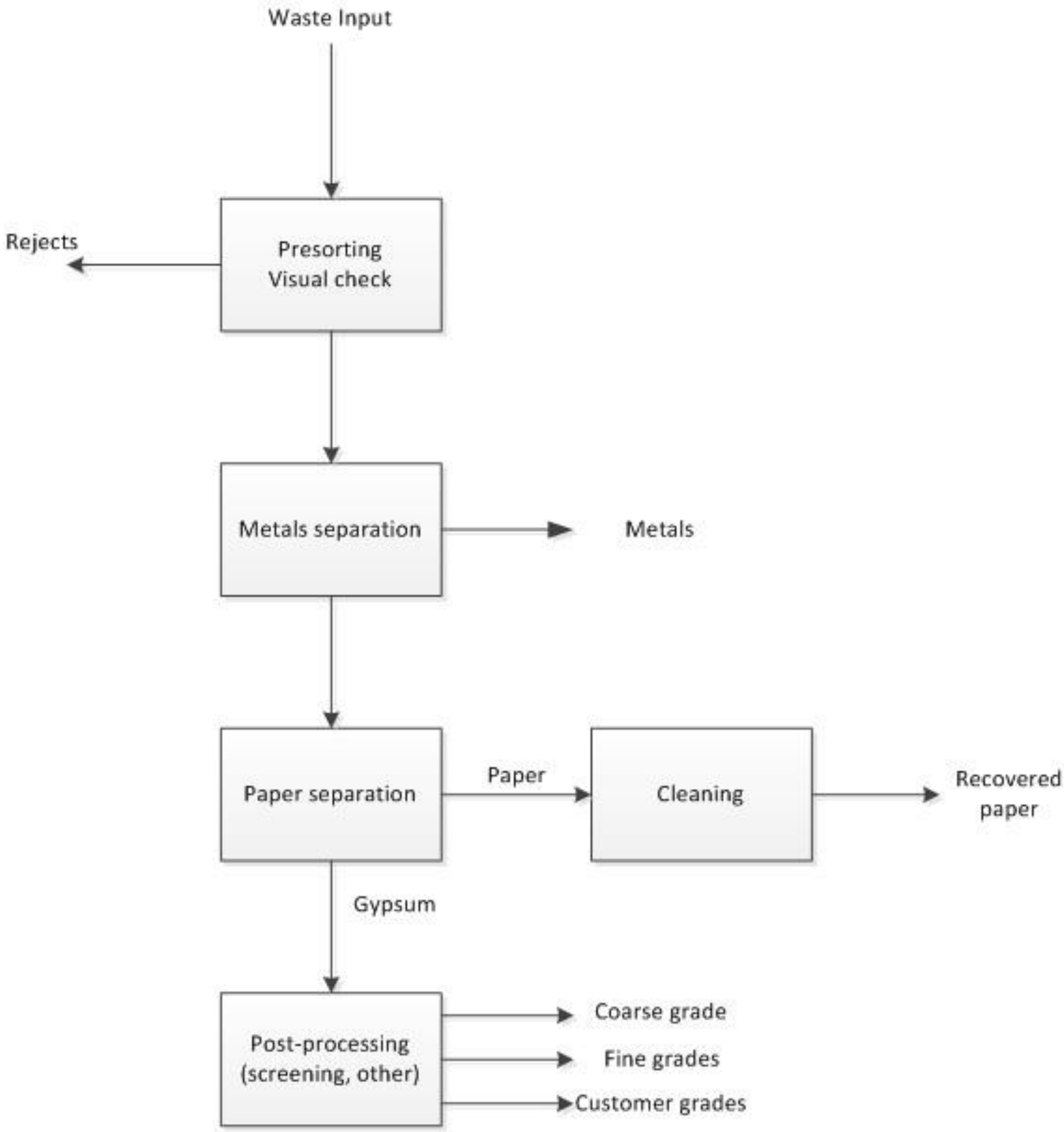
Operational data

Recycling process

Gypsum from waste plasterboard can be fully recycled as new plasterboard. Chemically, the production of gypsum consists of a dehydration-rehydration process. Natural or synthetic calcium sulfate 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 sulfate dihydrate are formed in an interlocked net. The material is of low density, has low thermal conductivity and develops enough strength, so it can be used in a wide range of construction products.

The source of raw gypsum can consist of recovered plasterboard, selectively collected from construction or demolition sites, and the so-called synthetic gypsum (calcium sulfate dehydrate as byproduct of industrial processes), usually coming from Flue Gas Desulphurisation (FGD), also called FGD gypsum. The reprocessing of calcium sulfate wastes, therefore, can be considered a high-grade recycling, quite rare in the recycling of CDW. The environmental advantage of the process itself is not high. A plasterboard panel made with 25 % recycled waste plasterboard, in a low-transport scenario, saves an average of 33 kg CO_2e of associated GHG emissions per tonne compared to conventional gypsum plasterboard, i.e. around 10 % savings (WRAP, 2008). But, in addition, when waste plasterboard is incorporated in the manufacture of recovered gypsum, it has indirect benefits on the recycling of CDW, as the segregated recovery of plasterboard would remove sulfate contamination in the matrix of recycled aggregates from clean concrete wastes, increasing its recyclability and applicability (EC, 2012; Asakura, 2013).

The production process of recycled gypsum from waste plasterboard is straightforward and very similar to any process for construction and demolition waste treatment. An example from Roy Hatfield is shown in Figure 4.5. At the entrance, the waste materials are visually checked and classified per 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 pre-treated and packed for its recycling. Gypsum is sieved (or even crushed again) according to the grades to be produced.

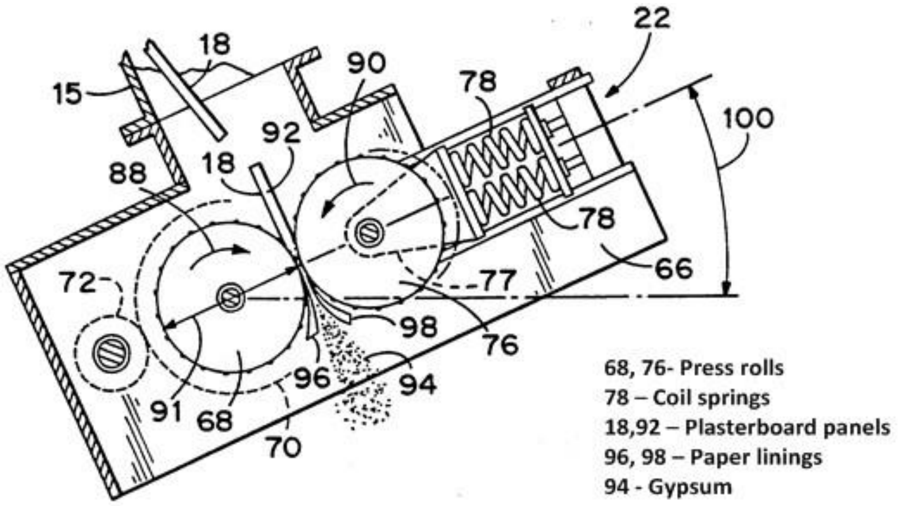


Source: Adapted from Roy Hatfield, (2013)

Figure 4.5. Waste plasterboard processing

The example from Roy Hatfield factory takes waste plasterboard from a variety of sources, such as construction waste managers or household waste. 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 drywall manufacturers, where it is combined with virgin rock or synthetic gypsum to make new wallboard. The recycler claims low fibre content in the recycled process and a use rate over 25 % in the making of new plasterboard.

In any plasterboard recycling facility, the key step for the quality of the final product 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 4.6 (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 4.6. Waste plasterboard crusher

Quality assurance

In the United Kingdom, an agreement on “End of Waste” criteria was established in England, Wales and Northern Ireland for reprocessed gypsum from waste (WRAP, 2011). The so-called Quality Protocol identifies the criteria when waste plasterboard is no more a waste and when waste management controls do or do not apply. 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 on new plasterboard, raw material for cement and soil treatment for agriculture, although this last option should avoid its spreading within 50 metres of potable groundwater, due to the risk of pollution.

Approved specifications under the UK example of the 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 4.21.

Table 4.21. 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 standard PAS 109:2013 defines three grades of recycled plasterboard and a minimum quality specification, depending also on the final use, for agriculture and as a raw material (Table 4.22). The standard also defines the minimum requirements on the Quality Management system of the re-processor 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 4.22. Specification 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 (% w/w of $CaSO_4$ $2H_2O$)	> 85 %					
Physical Contaminants	< 2mm, upper limit 0.25, of which 0.12 is plastic (% w/w dry sample)					
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: PAS 109:2013 (WRAP, 2013)

The quality restrictions for recycled gypsum from waste plasterboard and for that coming from other industrial process, as flue gas desulphurisation, are very similar. Table 4.23 gives an overview of quality parameters of recycled gypsum and flue-gas desulphurisation gypsum, as shown in EC (2012). In Germany, no end-of-waste criteria have been agreed 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 4.23. 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 sulfate 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 sulfite-hemihydrate	$CaSO_4½ H_2O$	Mass %	< 0.50	< 0.50
pH	--	--	5-9	5-9
Colour		%	white	white
Odor	--	--	neutral	neutral
Toxic compounds	--	--	harmless	harmless
Grain size	--	mm	--	< 5

Source: LFU (2007) as cited in EC (2012)

Applicability

The exemplary schemes applied in the UK are applicable with almost no limitation. Similar examples to the UK approach for gypsum recycling have started, for instance, in Germany, where the German gypsum association, BV Gips, is willing to re-process 150,000 tonnes per year, but, for that, end-of-waste criteria are required to be developed (BV Gips, 2012). Main barriers are economical, as the cost of a segregated plasterboard collection increases waste management costs in construction sites, but compensated by gate fees for wastes with no plasterboard segregated. In fact, segregation at constructions sites is required to be extensively applied in construction sites in the area where the recycler operates, since plasterboard separation from CDW is not technically possible in the treatment plant.

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 with extensive natural sources.

Economics

Three trials in the UK were reported by WRAP. Sheffield implemented a waste plasterboard collection system in one of the city's Household Waste Collection centres (WRAP, 2009). Instructions were given to the staff not to contaminate it with other waste materials from users of the centres. A legal limitation had to be established, as waste plasterboards are mainly trade wastes (produced by construction sector companies and have to be managed by specific managers). During five weeks, two tonnes of plasterboard were collected and the total costs for the city was around GBP 69 per tonne of plasterboard (at a gate fee of GBP 22.5). Landfilling cost of the same amount of material would have cost GBP 71 per tonne. The main reason for this is the cost of the haulage of the waste, while landfilled materials are transported in large skips (8 tonnes), the lack of space available for waste plasterboard forced the collection centre to use small skips, which were transported every week to the recycling facility.

In Islington (WRAP, 2012), haulage and transport costs were even higher and increased the cost of management for the recycling option (GBP 118 per tonne) compared to the landfill option (GBP 46 per tonne). In this case, gate fees per tonne are probably too high for the recycling facility (GBP 37) compared to landfill (GBP 44.80). Stafford (WRAP, 2008b) also detected high transportation and haulage costs, making the case uneconomic for the municipality, although it is cost-efficient for the industry.

Driving force for implementation

Main driving forces for the implementation of recycling schemes for plasterboard are, of course, the environmental performance of the process, which is favourable to the use of recycled materials. Also limited landfill capacities, the protection of natural resources, the expectable declining of FGD gypsum quantities due to the phasing out of coal-based power generation lead to a rethinking towards the gypsum recycling of construction and demolition materials. In addition, in some countries like Germany there is currently a debate regarding stricter sulfate limit values for elution in the recovery of secondary materials, and this could lead to significant restrictions on the use of recycled construction materials in the future.

The economics of gypsum recycling is favourable for the manufacturer, but there may be restrictions on the application of a recycling scheme at e.g. municipal level. However, due to the differences in the volume of waste and production rates of plasterboard, a 100 % recycling rate is virtually possible.

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.

German gypsum association, Bundesverband der Gipsindustrie

Gypsum Re-processors Association UK and Ireland (GRAUKI)

Gypsum industry

KNAUF

British Gypsum

Roy Hatfield UK

Regyp recycling solutions

New West Gypsum Recycling

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4.3.5. Management of PCB contaminated CDW

Description

The presence of hazardous substances in building materials and the consequent construction and demolition waste has been a relevant issue in the management of waste coming from buildings of certain age. This is the case for asbestos-containing materials used from the early 20th century until its toxic character as carcinogenic revealed the need of banning it and to establish specific procedures for waste management. The same also happens for fluorescents, lead, certain types of paints, etc. In a similar way, polychlorinated biphenyls, PCBs, have become also an important aspect to manage in some construction or demolition sites. PCBs is 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. Although in the past they were used quite frequently in the construction industry, e.g. in sealants, PCBs were banned in the 1970s due to their environmental toxicity and their classification as a persistent organic pollutant, POP.

Although there is always a certain risk of PCBs presence in the built environment, due to more interest on air quality issues, recently it has been detected a rise in the content of PCB in the inert fractions of construction and demolition waste, CDW. This is a consequence of the demolition and refurbishment of buildings from the 1950, 1960 and 1970 decades (Butera et al., 2014), generating CDW with PCB-containing sealants. These waste streams, if exceeding a limit concentration of PCBs, would be considered as hazardous waste and cannot be re-used or recycled. However, this is far from real practice, since the determination of hazardous substances in recycled products from CDW is still poor.

In Denmark, there has been a great concern on the presence of PCBs in CDW during the last two years. It is regulated that, when demolishing or refurbishing a building from the period 1950 to 1977, a screening of the presence of PCBs has to be performed, especially in those parts were it is expected (e.g. double glazed windows). If PCB_7 (2,4-dichlorobiphenyl) concentration is higher than 50 mg per kg, the waste has to be considered hazardous and disposed safely. If the concentration is lower it may be considered non-hazardous, but still not suitable for recycling. Local authorities have to assess the suitability of CDW from concerned buildings, and use a limit concentration of 100 μg/kg (PCB total) as a reference value. A limit concentration will be legislated soon (BioIS, 2015). Also, the Danish government has published several guidelines for the best options on the management of PCBs containing waste, also extended to other sectors: www.pcb-guiden.org.

Following the Danish example, there are certain main principles that can be established as Best Environmental Management Practice for CDW:

- The waste management plan includes tasks, before or during demolition works, to identify all PCB containing sealant materials, remove and separate them. In general, the pre-audit of the building or the structure should identify any potential hazardous waste and its management should be part of the licensing process. For more information, see 'Selective deconstruction of Buildings' BEMP in the report on best environmental management practice for the building and construction sector (EC, 2012).

- After the screening of the building, bridge or structure, the constructor has to inform the public authority about the presence of PCBs and map their presence.
- In case that the future CDW will probably be catalogued as hazardous, an appropriate waste manager shall remove it, and dispose it safely (incineration or safe hazardous waste deposits). Also, fractions of CDW rejected due to their high content of PCB, but still not considered hazardous, must be safely deposited.

Achieved Environmental Benefit

This technique corresponds to an Environmental Sound Management technique, ESM, according to the PCBs elimination network established in the Conference of the Parties of the Stockholm Convention on Persistent Organic Pollutants (UNEP, 2009). The benefits from PCB control and appropriate management have to be considered a priority. The control of PCBs releases from CDW is, anyway, extremely important. In 2006, high levels of PCBs were detected in San Francisco Bay in the U.S., which were linked to the demolition activity and consequent landfill of huge amount of PCB-containing waste. PCB was washed off with stormwater runoffs. As a consequence, marine species from the bay accumulated PCBs and increased cancer risk for those eating fish (Lee et al., 2010).

Appropriate environmental indicator

Concentration levels of PCB are usually required to be determined according to EN 15308:2008. The limit values are specified for PCB total and the standard asks for the measurement of 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 congener, which is defined as each of the existing chlorinated biphenyl, number from 1 to 209.

The concentration of PCBs in waste, according to the standard, has to be reported in ng, μg, or mg per kg of waste (see Operational Data for examples on the issue).

Cross-media effects

In Denmark, the application of the PCB action plan, along with stricter requirements of RA as sub-base material, has reduced the recycling rate of CDW in the period 2013-2014, however, it is expected that the rate will increase again after an adaptation period (BioIS, 2015). Similar behaviour is to be expected from similar approaches in other countries.

Operational data

Study on Danish construction sites

Butera et al. (2014) conducted a study on the presence of inorganic elements (by leaching) and organic compounds in CDW. They determined the concentration of different PCBs in CDW from different sites and from different segregation practice. Table 4.24 below shows the obtained results. Butera et al. (2014) analysed those PCBs according to the EN 15308:2008 standard.

A statistical analysis indicates that PCB_{total} content does not have a significant variance among sites and that the only relevant variation is observed between mixed and clean concrete. "New concrete" CDW is waste coming from buildings built after 1977, where PCBs-containing sealants were not used.

Table 4.25 and Table 4.26 show the statistics and the comparison, which is only significant between mixed aggregates and clean concrete. The lack of significant difference between clean concrete and new concrete made the research team to conclude that a background level of PCBs in construction raw material is present. In any case, the average PCB total of sampled waste is still lower than the benchmark used by Danish authorities (100 μg/kg CDW).

Table 4.24. Analysis of 33 samples of CDW from different sites, in μg per kg of CDW (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
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 4.25. Statistics for PCB total per nature of waste (Data from Butera et al., 2014)

Waste nature	Average, μg / kg CDW	Standard deviation μg / kg CDW
Clean Concrete	10.3	9.4
Mixed aggregates	26.7	17.7
New concrete	10.7	4.6

Table 4.26. Statistical comparison of waste nature for PCB total (Data from Butera et al., 2014)

Comparison	Difference of means μg / kg CDW	t- student	p-value
Mixed aggregate vs. Clean Concrete	16.337	3.314	0.007
Mixed aggregate 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 4.24, 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 aggregate (composed mainly by 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

Although the concern for the contamination of CDW with PCBs from sealant and other sources is relatively recent, the procedures of screening, identification, removal and separation of hazardous waste have been always an issue in the demolition of buildings of certain age. The most striking example is asbestos, which removal requires skilled labour and long operations. Other examples of hazardous waste in old buildings are: lead, paints, chlorofluorocarbons (CFCs), hydrochlorofluorocarbons (HCFCs), halons, pesticides, pentachlorophenol-treated timber, lindane, tributyltin, PCBs and polychlorinated terphenyls (PCTs) (Lend Lease, 2012).

In addition, PCBs are also present in other main components of buildings, for instance, mineral-oil filled electrical equipment, capacitors, plastics, paints, adhesives, some fluorescent ballasts, etc. Table 4.27 shows primary sources (materials manufactured with PCBs), secondary materials (not manufactured with PCBs but easily contaminated due to their physical characteristics, e.g. porosity), non-porous surfaces and concentrations from exposed media.

Table 4.27. PCB containing building materials and exposure media

Material	**Range resp. 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

Applicability

There is no restriction on the applicability of this technique. Small works, producing less than 1 tonne of CDW or affecting to less than 10 m^2 of the floor area of the building are not considered under the scope of the Danish regulation.

Economics

The costs associated to the screening, identification, removal and separation of PCB-containing CDW have to be assumed by the producer of the waste, usually the developer and/or the construction contractor. In the short term, these costs are assumed by the new building owner as an increased investment. However, it is expected that PCB wastes management will increase the cost of the recycled aggregate.

Driving force for implementation

The hazardous character and the health risks associated to PCBs constitute a main priority in the management of CDW. Strong regulations are in place and should suffice as driving force.

Reference organisations

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5. Healthcare Waste (HCW)

5.1. Introduction

The management of healthcare[34] waste, HCW, is under strict control but differently regulated within EU Member States, due to its hazardous characteristics. 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

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). The waste 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

[34] Healthcare Waste (HCW) is the waste generated at a medical institution, hazardous and non-hazardous (including MSW-like), 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 corresponds is HCW, as recommended by the Technical Working Group.

- 20 01 99 other fractions not otherwise specified

The classification 20 01 99 is used in the case of *offensive*[35] waste, common definition in the UK for waste that is usually separated from the rest of the fractions. 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 category 18 wastes. For instance, in the Greek regulation (EPTA, 2006), HCW is classified according to these categories for its management:

a. Non-hazardous HCW (MSW-like)
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):

- Alternative treatments, as chemical or thermal sterilisation (autoclaving)
- Thermal treatment, as high temperature incineration, incineration and landfilling of incineration residues
- 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 applies to each of them, as shown in Table 5.1.

Table 5.1. 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		Re-use, recycle, energy recovery, incineration

Source: Adapted from CIWM (2014)

[35] *Offensive* waste is a term used for non-hazardous healthcare waste that causes *offence* due to its appearance, odour or wetness.

In terms of HCW management, 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)
- Enabling techniques (may help to the implementation of best environmental management practices):
    - Waste management plans in hospitals and other heavy producers
    - Traceability (compulsory in some Member States)
- Best Environmental Management practices:
    - Prevention measures at source
    - Integrated segregation and collection of wastes, including non-mandatory fractions and MSW-like waste
    - Extended Producer Responsibility schemes for pharmaceuticals
    - Alternative treatments

While the mandatory measures are oriented to public health protection and are strongly regulated, the enabling techniques are usually oriented to organisational measures oriented to minimise costs. For instance, main producers as hospitals need a waste management plan for their daily operations that would allow the implementation of other best environmental management practices. Traceability consists of the use of labels that can trace the source of the waste in order to investigate non-conformance situations.

Best environmental management practices are those oriented to minimise the environmental impact produced by HCW generation. Prevention measures are the most important but excluded from this document, as they exclusively affect to the activities of health care sector and not to the waste management sector as well as the application of integrated segregation in healthcare institutions. Alternative treatments and its applicability are the main focus of this chapter.

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5.2. Management of HCW in health-care institutions

5.2.1. Waste segregation

The segregation of waste at the point of production is strongly regulated in the EU Member States and regions and under different regulation approaches. For instance, segregation of wastes according to the categories shown in Table 5.1 is mandatory in England and Wales, but a best practice recommendation in Scotland and North Ireland – the four regions share the same health service (DH, 2013). The World Health Organisation publishes regularly guidelines for the safe management of healthcare waste and recommends a basic segregation scheme (Table 5.2).

Table 5.2. 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 health-care waste	Black	Plastic bag

Source: WHO (2014)

Beyond the basic segregation, successfully implemented in Europe, the use of a unique black plastic bag for non-hazardous waste (MSW-like and others) prevents further recycling and materials 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 on several streams:

- Recyclables: paper, plastic and cans, usually generated by patients and visitors in common areas
- Food waste: generated by the kitchens

Hazardous waste, other than healthcare waste, is generated at higher rates by health-care activities than households, e.g. chemicals, solvents, batteries, light bulbs, batteries, etc.

However, HCW management should ensure hygiene and infection control as a top priority. All measures of prevention, re-use 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 only in second place. A higher segregation rate of HCW would eventually reduce the amount of waste incinerated at high temperature. As a consequence, less waste fuel would be supplied to the incinerator that would require an extra fossil fuel amount 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:

- Segregating HCW at least according to the minimum recommendations of WHO, but minimise the amount of waste leading to highest environmental impact treatment methods (landfill or high-temperature incineration)
- Segregating food waste and recyclables from the black plastic bag.
- Training all the personal handling HCW and any other type 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, protocols and monitoring the performance, according to a similar standardised system to ISO 14001 or EMAS.

However, segregation of HCW is dependent on the size of the healthcare institution. While small labs, clinics, dental practice, etc. generate a rather small amount of waste with 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, etc.).

A high rate of diversion of offensive waste, which is not hazardous, is feasible due to the high costs derived of 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: 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 (sic) firefighting exercise, where waste is assumed to exist and the cost of its management minimised as much as possible. Waste minimisation through prevention or re-use 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 e.g. thermometers and sphygmomanometers only USD 6,000. However, the main management of waste-containing devices or materials is segregation at origin. 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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5.2.2. 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 meters long, up to 6 meters of 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 quite 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 side, 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 5.3. 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 pair Time-Temperature of 20 min – 121 °C, although it will always depend on the type of installation.

Table 5.3. Levels of microbial inactivation efficacy

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 a more 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 an indicated non-incineration method to render safe and unrecognisable HCW consisting of body parts. This is done by heating the waste to a temperature between 110 and 127 °C in an alkaline solution (water plus sodium or potassium hydroxide) in a stirred tank during six to eight hours (WHO, 2014), removing any pathogenic microorganism. 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 improves considerably 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 during 30 minutes, which would correspond to a pressure of 2 bar. However, 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 particle size of waste 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 from 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 during a certain period of time in a closed chamber under a certain pressure of air. Pressures, temperatures and times are usually higher than 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, that generates a steam pressure in the system during 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 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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5.3. Best Environmental Management Practice for the treatment of Healthcare waste

5.3.1. Selection of alternative treatments of healthcare waste

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 to prioritise, with more or less success, the implementation of technologies that prevent the release and formation of dioxins and furans, scaling-up the so-called alternative treatments of healthcare waste consisting on non-incineration technologies (HCWH, 2007). Although Table 5.1 (see page 367) establishes the suitability of several waste streams to different treatments, the lack of proper segregation increases considerably the fraction of HCW that needs to be incinerated at high temperature.

An important part of the environmental impact of incineration can be avoided through the use of alternative treatment methods that remove e.g. the infectious character and, therefore, can be treated as MSW-like waste streams. But, the use of alternative HCW treatments, described in the previous section, should comply with certain requirements in order to be considered as a suitable treatment. 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 risks 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 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 it becomes *unusable* and *unrecognisable*.
- For pharmaceuticals waste: destroy 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 taken, 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 as BEMPs is shown in Table 5.4. Admission criteria, in this table, 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 5.4. Admission criteria, minimum environmental and best practice criteria for HCW alternative treatments

	Autoclaving	Microwave	Chemical Treatment
Admission criteria	Render safe treatment. Non-bulky wastes; or bulky wastes suitable for shredding operations, if applicable. Not applicable to mercury- or heavy metals- 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 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 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 to recycling. Waste-to-energy applied to the output when incineration is admissible.	Optimal segregation at source. Water addition at the inlet. Drying step after treatment. Output separated per material stream when possible and sent to recycling. Waste-to-energy applied to the output when incineration is admissible.	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 to recycling. Waste-to-energy applied to the output when incineration is admissible.

Achieved Environmental Benefit

Townend and Cheeseman (2005) reported some of the environmental impacts from alternative treatment of HCW in comparison with incineration practices (Table 5.5).

Table 5.5. 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 wastewater.	Toxic volatile organic compounds and odours. Generates wastewater.	May generate toxic volatile organic compounds and odours. Generates liquid hazardous waste and/or wastewater.	High volume of air emissions that require an appropriate pollution abatement system. High risks of dioxin and mercury emissions.

Source: Townend and Cheeseman (2005)

Figure 5.1 shows the aggregated value associated to the environmental impact of different alternative treatments 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 less environmental impact associated is microwave, followed by autoclaving. The environmental impact of both techniques is associated to the production and use of energy (so greenhouse gases emissions and fossil fuel depletion are the main categories of impact considered), being the consumption for microwave much less than for pressurised steam, as anticipated by the authors of the study (Soares et al., 2013). In the case of the chemical disinfection, the assumption of alkaline hydrolysis with lime makes the main life cycle environmental hotspot to be the production of the chemical agent.

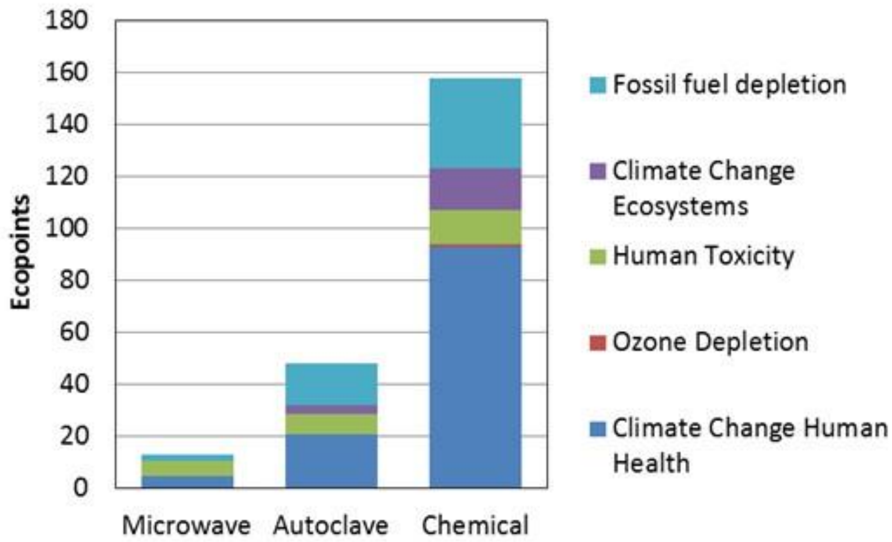

Figure 5.1. Environmental impacts of different alternative HCW treatment techniques. (Data from Soares et al., 2013)

Zhao et al. (2009) published an LCA comparison of incineration and alternative treatments for HCW (Figure 5.2). 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. The definition of the systems can be considered, however, out of a best practice approach, as sterilised waste is sent to a 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 MSWI, that operates at lower temperature but with an efficient waste heat and electricity cogeneration. If, for instance, the results from Figure 5.2 are adjusted for the energy balance of MSWI incineration, 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 very affected, with reductions of up to 60 %.

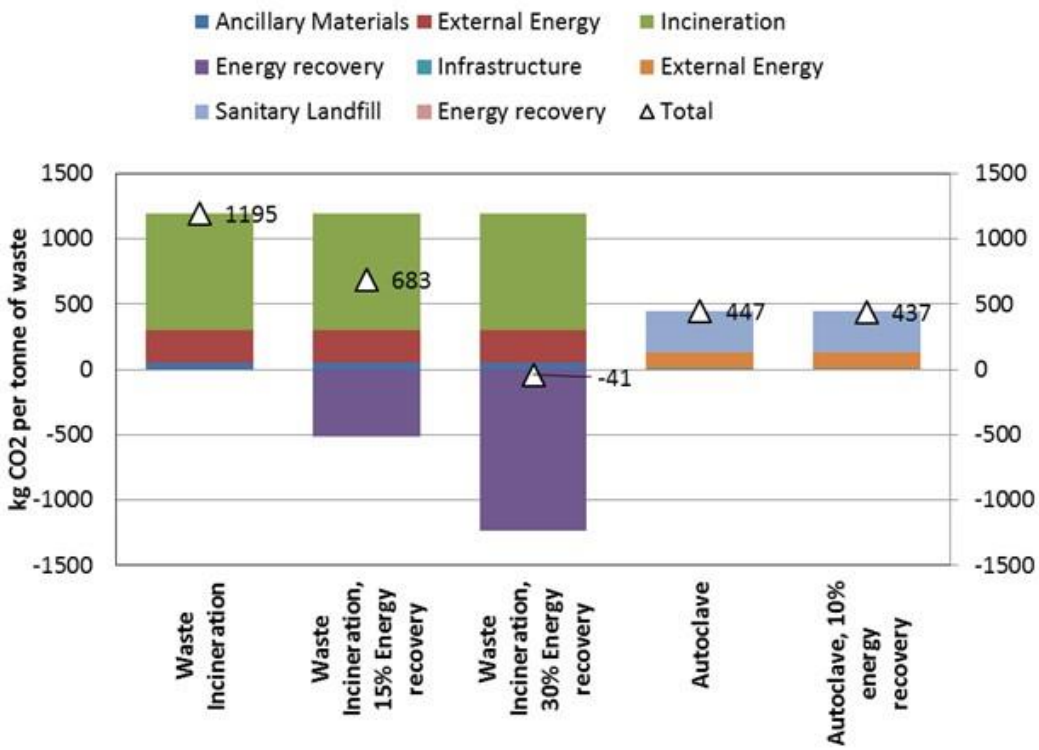

Figure 5.2. GHG emissions from several alternatives of HCW high temperature incineration and autoclaving

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 indicator

Several indicators can be useful to monitor HCW flows:

- % of total managed waste diverted to alternative treatments

For each of the alternative treatments, there is a number of useful technical parameters that help to understand the performance of the technique:

- Temperature – time
- Batch/Continuous/Semi-continuous operation
- Throughput, in kg waste per hour, day or cycle
- Water consumption per kg of waste
- Water treatment and recovery (y/n)
- Suitable for bulky materials (y/n)
- Pre/Post/Non shredding and final particle size distribution
- Production of hazardous waste

These technical parameters (most of them, design specifications) are translated in several technical indicators that provide a very valuable information for the post-processing of the final waste:

- Level of disinfection (Level I/II/III/IV/V of the STAATT), counts of microorganisms according to STAATT standards

- Volume reduction/increase (%)
- Weight reduction/increase (%)
- Waste water generated (kg/kg of waste)

Example values are shown in Operational Data.

Cross-media effects

As shown in the Achieved Environmental Benefit section, the trade-offs of environmental impacts can be quite complex. While some techniques are oriented to avoid the burning of highly recyclable materials, they produce wastewater effluents that, in many cases, need to be treated or, as in the case of chemical disinfection, are considered a hazardous waste.

In addition to the need of a safe treatment of the waste, it is required to remark that separate waste collection in healthcare is far from optimal; (e.g. chemicals and pharmaceuticals are still not well segregated) and should not be treated in autoclaves or microwave systems and then incinerated in MSW incinerators. This is the case of HCW with high content in PVC, which may require a safer treatment than incineration (Simon, 2015).

Operational data

The decision-making for waste treatment and management not only belongs to the waste contractor and the service provider, but to the waste producer, as the legal liability of rendering safely a waste stream lays on 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. But, 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:

<u>1. Sourcing better segregated waste.</u> Segregation at origin is key for the whole treatment to achieve an optimal performance, including or not the application of alternative treatments. Both can be achieved through the availability of resources for segregation (e.g. differentiated containers for the categories shown in Table 5.2), pre-acceptance audits and through awareness campaigns and regular and update training. Some case studies are described in Table 5.6, which were applied to hospitals, usually with the collaboration of a waste contractor or consultant.

Table 5.6. Achieved environmental benefit reported for several case studies

Case Study	Description	Quantifiable benefit	Reference
University College London (UK)	Implementation of a segregation scheme for HCW waste 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 recycling scheme Reduction of waste collection journeys by half Stakeholder engagement programme	18 % diversion from high temperature incineration Savings of max 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 training to staff to avoid inefficiencies	Infectious waste reduction by 50 %, 14.7 tonnes of waste sent to recycling (approx. 29.4 kg/bed yr)	HCWH (2007)
Freiburg Hospital (DE)	Phase out and re-use programmes for paper towels, dishes, baby bottles, shoe protectors	Reduction of 577 tonnes of waste per yr	HCWH (2007)
Gloucestershire Hospitals NHS Foundation Trust, UK	Implementation of an offensive waste stream in hospitals to divert 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 to waste contractors for a HCW treatment permit, in order to check for compliance of the minimum standards of segregation, composition and amount of waste streams and, in the case of the HC organisation, to monitor the 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, identifying easily no-cost or low cost opportunities, and can recover costs through the implementation of improved practices at source (DH, 2013).

Training programs and awareness campaigns are of extreme importance 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 on 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, Figure 5.3 shows the poster supplied by Econix Ltd. for their disposable waste bins in the UK, which is freely downloadable from the internet.

Figure 5.3. Example of information poster supplied by waste contractor. (http://www.bio-bin.org/assets/img/YELLOW-POSTER.png)

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 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 among the other options.
- Treatment within clusters. A major hospital has a scaled-up facility for the treatment of waste generated on site plus those generated in the area or district.

3. Selecting vendors and technologies.

Some examples of vendors per technology are shown in Table 5.7, as extracted from HCWH, 2007.

Table 5.7. Vendors example, 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 market of alternative treatments is quite innovative and is moving fast to achieve tailored solutions. As an example, microwave sterilisation, which is usually a discontinuous or semicontinuous operation, can be however redesigned to offer a continuous process that includes pre-shredding, continuous screw-driven feed with no water addition, and storage. The final product is shown in Figure 5.4. 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 5.4. Shredded and unrecognisable microwaved healthcare waste

4. Understanding and reporting on the applied technology. As stated under in the indicators section, technical information on the technology is key to understand its economic and environmental potential. Table 5.8 below shows the performance of alternative treatments.

Table 5.8. 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, in 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 key for the treatment of a significant fraction of HCW (Tudor et al., 2009). It is, then, 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 are costs, as high temperature incineration is reported to be highly expensive due to the use of support fuels and pollution abatement, while alternative treatments have reported up to 60 % savings in a very optimal scenarios. In 1990, the US already reported a cost 2 to 5 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 USD 1.90 per kg of waste to incineration, while maximum costs would be then USD 0.40 per kg of waste to alternative treatment (post-treatment not included). More recently, Tudor et al. (2009) reported a cost of GBP 500 – 800 for incineration, which maximum corresponds to USD 1.30 per kg of waste. In this regard, there was a shift from local, small incinerators installed 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, the calculations by Soares et al. reflected a cost of USD 0.12/kg for microwave treatments, USD 1.10/kg for autoclaves, and USD 1.53/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 of HCW.

Driving force for implementation

Risks 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 shown also 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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6. Applicability to Micro-, Small- and Medium-sized Enterprises

The purpose of this chapter is to facilitate use of this document by small and medium sized enterprises (SMEs). As described in Chapter 1, waste management is mainly undertaken by micro companies of less than 10 employees, often specialising in collection and materials recovery. According to Eurostat, 77 % of NACE 38 companies are classified as micro, and 99.7 % as SMEs. Similarly, the construction sector is composed of more than 95 % SMEs. However, these data fail to reflect the importance of a few large waste management companies in Europe that manage, directly or indirectly, a large share of MSW.

Most of the best environmental management practice techniques described in this document are of direct relevance to SMEs, and will either be directly applicable to them, or will have implications for them via implementation by larger waste management organisations (WMOs). Certainly, most of the proposed indicators and benchmarks can be used by SMEs to monitor environmental performance. Being an SME is not a reason to avoid responsibility for monitoring and improving environmental performance. Nonetheless, some best practice techniques relating to e.g. waste management strategies and economic instruments may be of more direct relevance to municipal authorities than to SMEs. In addition, some techniques requiring high upfront investment may not be applicable to smaller SMEs that typically have little capacity to invest in environmental technologies. Finally, management structures and capabilities are usually more limited for SMEs than for larger companies or municipalities.

This section therefore describes which practices are most applicable and affordable to SMEs. Table 6.2 characterises all the BEMPs described in this document in relation to three aspects: costs, applicability and achieved environmental benefit, using a user-friendly coloured-coded "traffic light" assessment, as described in Table 6.1

Table 6.1. Colour coding for the assessment of the applicability of best environmental management practices for SMEs

Symbol	● (orange)	● (light yellow)	● (green)
Cost (initial investment)	High	Medium	Low
Applicability to SME	Not applicable	Applicable with restrictions	Full applicable
Environmental benefit	Low	Significant	High

Table 6.2. Cost and applicability to SMEs, and environmental benefit, of best environmental management practices described in this document

Topic	Best Environmental Management Practice	Sect-ion	Cost	Appl. to SMEs	Env. benefit	Comments
Cross-cutting issues	Integrated waste management strategies	2.3				Development of an integrated waste management strategy is integral to best environmental management practice for any WMO that has strategic control over the flow and treatment of waste streams, though in most cases this won't be an SME (this technique primarily applies to municipalities).
	Life cycle assessment of waste management options	2.4				Life cycle thinking and review of relevant LCA studies is central to the development of integrated waste management strategies, and is applicable to any SMEs that have strategic control over the flow and treatment of waste streams. Buying bespoke LCA services and/or paying for staff training in LCA is applicable to larger organisations.
	Economic instruments	2.5				Economic instruments are costly and in many cases its implementation has required governmental subsidies. They are complex to manage, but highly beneficial for smaller municipalities in rural environments
Municipal solid waste strategies	Cost benchmarking	3.5.1				SMEs may pay a modest annual fee to benchmark their costs against similar waste management organisations.
	Waste monitoring	3.5.2				Applicable to all SMEs engaged in developing waste management strategies, but useful for most WMOs.
	Pay-as-you-throw	3.5.3				Establishing a weight-based PAYT scheme requires significant investment, preferably at the municipality level. However, PAYT can be operated by SMEs.

	Awareness raising	3.5.4				Elements of this technique such as staff training are applicable to all SMEs, and partnerships across SMEs can enable delivery of effective advertising and awareness campaigns.
	Municipal waste advisers	3.5.5				Waste advisers tend to be implemented at local level in big cities in the initial phases, and then spread regionally. High investment in labour costs means that small municipalities may not achieve regional implementation of this BEMP.
Waste prevention	Local waste prevention	3.7.1				Waste prevention measures are most effectively implemented by municipal authorities, but some measures can be implemented by SMEs.
Waste re-use	Product re-use schemes	3.8.1				Product re-use schemes often require coordination and financial support from municipal authorities, but SMEs play a key role in delivery within re-use collaborative networks.
Waste collection	Waste collection strategy	3.9.5				The development of optimised waste collection strategies is best undertaken at the municipality level, in the context of integrated waste strategies. However, SMEs have an important role to play as deliverers of waste collection services, and may identify specific options to reduce burdens of waste collection.
	Infrastructure to recycle or to recover waste streams and to dispose of hazardous compounds	3.9.6				Collection centres may be deployed as part of an integrated management strategy at municipality level, though SMEs may be involved in their operation.
	Logistics optimisation	3.9.7				All SMEs involved in waste collection can take actions to optimise the logistics of their operations, buying in expert advice and software with short payback times.

	Low emission vehicles	3.9.8				All SMEs involved in waste collection may select lower emission vehicles. High upfront costs for some vehicles, and for installation of biomethane infrastructure, mean that some elements may not be viable for all SMEs.
Waste collection enabling technique	Inter-municipal cooperation (IMC) for waste management in small municipalities	3.10.1				This enabling technique primarily relates to municipalities, although SMEs managing waste collection in neighbouring localities could also implement certain measures.
Waste treatment	Sorting of co-mingled packing waste	3.11.1				This technique is fully applicable to SMEs that may operate sorting plants.
	Decentralised composting	3.11.2				This technique is primarily applicable to municipalities, and in circumstances where anaerobic digestion of organic waste is not possible, but may also be implemented by SMEs that have control over waste strategies.
Construction and demolition waste	Integrated construction and demolition waste plans	4.3.1				As with overarching integrated waste management strategies, this technique is integral to best environmental management practice for any WMO that has strategic control over the flow and treatment of CDW streams, though in most cases this won't be an SME.
	Quality assurance schemes	4.3.2				The application of quality assurance schemes to the production of CDW is usually applied by SME WMOs.
	Acceptability of aggregates	4.3.3.				A sole, independent recycling plant would enter in the definition of SME; therefore, no technical disadvantage is foreseen. Marketing issues related to recycled products are more difficult for small producers, and depend on the awareness and traditional behaviour of construction companies.
	Plasterboard recovery	4.3.4				There are no major barriers to the widespread implementation of this technique by SMEs; collectors are usually SMEs, although big players dominate the market in many countries.

	Management of PCB contaminated CDW	4.3.5				This technique is applicable to all SMEs managing CDW, high costs may be incurred, but ultimately these should be borne by the waste generator, depending on local regulations.
Healthcare waste	Selection of alternative treatments of healthcare waste	5.3.1.				This technique is primarily applicable to waste contractors managing the waste from larger medical institutions and having an influence on the segregation strategy of waste. This technique may be implemented by SMEs, but requires large investment.

7. Conclusions

The conclusions from this report are summarised in the proposed list of best environmental management practices, indicators and some aspects of their applicability and economics, as shown in the Table 7.1 below.

Table 7.1. Summary of Best Environmental Management Practices in the Waste Management Sector

BEMP description	Section	Environmental Performance Indicator	Remarks
CROSS-CUTTING ISSUES			
Based on mass stream and life cycle thinking, **define a short-term and long-term strategy for all the different waste streams** in order to increase prevention, recycling rates for the different recyclables and to minimise residual waste quantity by means of an appropriate mix of different approaches, including technical, economic and psychological aspects.	2.3	Recycling rates for the different waste streams which can be recycled such as paper/cardboard, glass, waste plastic and composite packaging, bio waste, green cuttings, etc. and residual waste to be disposed of (kg per capita per yr).	The careful analysis of the existing situation, including the quantities and composition of all waste streams, and the development of the waste strategy may require external expertise and, most important, highly competent and motivated staff as well as the full support of the top management. The long-term strategy includes a systematic step-by-step approach.
Apply **life cycle thinking** throughout waste management strategy design and implementation, informed by relevant published studies of comparable systems, and/or undertaking (or commissioning) bespoke life cycle assessment studies where necessary to identify the optimum strategy for a particular waste stream.	2.4	Systematic application of life cycle thinking, and where necessary undertaking of life cycle assessment, throughout waste management strategy design and implementation (Y/N) Management strategies for all waste streams are supported by documented life cycle environmental performance data.	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.
Use economic instruments to encourage and maximise the environmental performance of the system and save costs, by recycling incentive schemes, pay-as-you-throw, local refund schemes, and B2B approaches for industrial wastes.	2.5	The waste authority participates, regulates or manages deposit refund schemes of e.g. waste beverage containers at local level (Y/N) Percentage of MSW generated that is selectively collected (% weight) Percentage of MSW generated that is recycled (% weight exiting material recovery facilities in separated fractions)	Only economic instruments applied or regulated at the local or county level are described. For commercial and industrial waste, the *local* approach involves B2B best practices, as consolidation centres, which involve recirculation of materials by waste managers.
MUNICIPAL SOLID WASTE: WASTE MANAGEMENT STRATEGY			
Perform **cost-benchmarking** with the help of an independent third-party organisation whereas cost figures for all waste streams (paper/cardboard, glass, plastics including	3.5.1.	It does not appear to be possible to quantify the participation in cost benchmarking. So, the regular participation (YES/NO) is the appropriate environmental indicator.	Cost benchmarking needs the setting up of a system with a clear definition of the costs considered, such as costs for collecting the different waste streams/fractions, for the

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BEMP description	**Section**	**Environmental Performance Indicator**	**Remarks**
composite packaging, bio waste, green cuttings, scrap metal, non-ferrous metals, hazardous waste, etc.) comprise costs for waste management services, for disposal of certain waste streams as well as revenues gained from marketing of recyclables.			treatment/disposal of residual waste and recycling/energy recovery of waste fractions, for the after-care of existing landfills, for staff and administration as well as miscellaneous costs. Also, all revenues gained due to recycling / recovery activities are taken into account.
As indicated in the waste management strategy, the composition and quantities of the different waste streams/fractions need to be known as well as the fate of them. For this purpose, **monitor the different waste streams** which includes the determination of the quantities and the composition.	3.5.2.	The determination of the quantities and also of the composition and the fate of all relevant waste streams/fractions (YES/NO) is the appropriate environmental indicator.	As waste management deals with a considerable number of waste streams (different waste fractions), the monitoring of them is indispensable. The systematic long-term monitoring allows the evaluation of the success of waste management practices.
Introduce a system where citizens pay per weight or bag of residual waste generated and where bio waste and bulky waste is also weighted.	3.5.3.	Recycling rates for the different waste streams which can be recycled such as paper/cardboard, glass, waste plastic and composite packaging, bio waste, green cuttings, etc. and residual waste to be disposed of in (kg per capita per yr).	Pay-as-you-throw is an important economic instrument that significantly contributes to minimise residual waste quantity and to increase the recycling rates. In many countries, the fear that waste is increasingly illegally dumped could not be confirmed. The system is most successful if a well-developed infrastructure and level of awareness of citizens is given.
Educate citizens on waste prevention and management, clearly advertise waste management services, engage staff in best practice.	3.5.4.	Residual waste generated (kg per capita per yr) Contamination rate of individual waste streams (% weight of individual waste streams collected that is rejected for the intended recycling or recovery purpose) Percentage of citizens in the waste management catchment area receiving awareness raising messages over a given time period, (e.g. % population per month)	Applicable to all waste management organisations (WMO). Level of effort and costs across each aspect will vary depending on primary role and size of WMO. Partnerships with other organisations can improve the efficacy of advertising and awareness campaigns.

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Full-time employees of regional or local public waste authorities as **waste advisers**, with the main focus on awareness building, public education of the population, PR and communication work on waste prevention, re-use, separate waste collection and sustainable consumption and lifestyles in general within the local or regional context.	3.5.5	Use of waste advisers in the awareness raising campaigns (Y/N)	Requires an initial commitment of at least one region (province, big city) of more than 1 million inhabitants, to ensure economic feasibility of the development and implementation of a qualification and training program as well as continuity of step by step implementation of waste advisors in all regions and municipalities.
MUNICIPAL SOLID WASTE: PREVENTION			
Here, only waste prevention measures on the local and regional level are considered. **Set up and perform or stimulate waste prevention measures** for individuals and families (little package, my bag and my cup, reusable package, repair, refillable products, donation, reduction of food waste, reusable nappies, etc.) as well as for municipalities, cities, counties or private organisations (mobile dishwasher for festivals, lunch boxes, repair shops, pay-as-you-throw system, etc.).	3.7.1.	Recycling rates for the different waste streams which can be recycled such as paper/cardboard, glass, waste plastic and composite packaging, bio waste, green cuttings, etc. and residual waste to be disposed of in (kg per capita per yr).	On the local and regional level, the achievable reduction rate by waste prevention measures for residual waste is very limited (few kg per capita per yr) as significant rates can only be achieved by product policies and other measures on the European or national level.
MUNICIPAL SOLID WASTE: RE-USE			
Collect items for re-use and distribute to organisations, including charities, for sale or onward distribution, and establish effective information exchanges to advertise the demand for, and market the availability of, re-usable "waste" products.	3.8.1.	Mass of potential waste stream diverted to re-use in the waste management catchment, expressed as: tonnes/yr kg per capita per yr percentage of the baseline waste stream mass flow (disaggregated by main product category, e.g. clothing, furniture, electrical equipment, transport equipment)	Applicable to all WMOs that handle re-usable "waste" products, in particular garments, furniture and electrical appliances. WMOs may work in partnership with each other, and with third sector re-use organisations, to efficiently design and implement re-use schemes, realising economies of scale and "critical mass" with respect to effective advertising and awareness campaigns.

Table 7.1. Summary of Best Environmental Management Practices in the Waste Management Sector

BEMP description	Section	Environmental Performance Indicator	Remarks
MUNICIPAL SOLID WASTE: WASTE COLLECTION			
Separate out biological wastes so that residual waste can be collected less frequently, and to devise **a collection strategy** that cost-effectively maximises the rate of selective collection.	3.9.5	Percentage of MSW generated that is selectively collected (% weight) Contamination rate of individual waste streams (% weight of individual waste streams collected that is rejected for the intended recycling or recovery purpose) Capture rate for individual waste streams (% weight of waste stream generated that is separated out for recycling) Percentage of MSW generated that is recycled (% weight exiting material recovery facilities in separated fractions)	The optimum approach to maximise recycling at acceptable cost will vary depending on local circumstances, including citizen behaviour. Bring centres can be an effective and cost-efficient strategy where recycling is well established in the public psyche, in other areas more costly strategies (e.g. door-to-door collections) may be required. Less frequent (e.g. two-weekly) residual waste collection may not be practical in warmer climates as it still contains some organic waste.
Provide the required infrastructure to collect for recycling and recovery of a considerable number of waste streams / fractions. In addition to door-to-door collection, this means the installation of collection centres where the different wastes are received and kept separate for efficient recycling and, in some cases, for recovery.	3.9.6	For a county or a city, the number of collection centres per 100,000 capita can be used as an indicator or the weight of the different waste fractions per capita collected via collection centres.	The infrastructure for waste recycling is very much required but the achievement of high recycling rates also needs further instruments such as awareness raising campaigns and regular adequate information of the citizens. Further, the municipalities of a county need to be supported by the county or region in terms of expertise and financial assistance.
Optimise logistics operations using Computerised Vehicle Routing and Scheduling (CVRS) technology or equivalent software, and performance is benchmarked using appropriate efficiency indicators.	3.9.7	Percentage of MSW generated that is recycled (% weight exiting material recovery facilities in separated fractions) Fuel consumption per tonne of waste fraction collected (L/tonne) Average fuel consumption of waste collection vehicles (L/100 km) Cumulative Energy Demand (CED) per tonne of waste fraction collected (MJ/tonne) GHG emissions per tonne of waste fraction collected (kg CO_2e/tonne)	Applicable to all WMOs undertaking waste collection. Costs of undertaking CVRS in-house or outsourced are paid back quickly by fuel and time cost savings in the region of 15 %.

Table 7.1. Summary of Best Environmental Management Practices in the Waste Management Sector

BEMP description	Section	Environmental Performance Indicator	Remarks
Purchase or lease refuse collection vehicles that are: (i) fitted with stop-start and idle shut-off technology and electrically operated bodies, (ii) dual-fuelled or fully fuelled with natural gas, biogas where available, or hybrid electric vehicles. Retrofit existing refuse collection vehicles with sufficient remaining planned years of service to justify the cost to run on natural gas, or biomethane where available.	3.9.8	Vehicle rated CO_2 emissions (g CO_2e/km) Engine PM, NO_x, VOC emissions (g/kWh) Percentage vehicles that are EURO VI compliant Percentage new vehicles that are hybrid-electric or natural gas/biomethane powered	Applicable to all WMOs undertaking waste collection. Higher vehicle purchase costs or conversion costs are offset by reduced fuel costs, and higher vehicle maintenance costs are offset by reduced maintenance costs for increasingly expensive diesel exhaust treatment systems. Compressed natural gas is readily available in many EU Member States. Biomethane can be produced from waste biogas, but installation of necessary infrastructure may be expensive.
MUNICIPAL SOLID WASTE: TREATMENT			
It is one option **to collect and to sort co-mingled packaging waste** to recycle and to recover as much as possible plastic, composite packaging, paper/cardboard, ferrous and non-ferrous metals.	3.11.1	Recycling rates for plastic such as PET and polyethylene, composite packaging, paper/cardboard, ferrous and non-ferrous metals.	Depending on the sorting technique, the quality of the different fractions may be not high enough for recycling and the separate collection of the different fractions is an option to improve the recycling rates.
Evaluate the feasibility of anaerobic digestion of wet organic waste before pursuing **a decentralised composting strategy**, provide information and equipment to households to support home composting, and establish community-run decentralised composting facilities in urban areas.	3.11.2	Mass of organic waste diverted from landfill or incineration through decentralised composting (kg/household/yr) Percentage of organic waste present in collected residual waste (% annual mass relative to annual mass generated)	Anaerobic digestion and incineration with energy recovery are preferred options for “wet” (e.g. food waste) and “dry” (e.g. wood cuttings) organic waste respectively. Composting may be considered best practice only where the aforementioned options are not possible. Additional effort is required to organise community decentralised composting schemes in urban areas.

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BEMP description	Section	Environmental Performance Indicator	Remarks
CONSTRUCTION AND DEMOLITION WASTE			
Develop local or supra-local CDW management plans that involve main stakeholders, prioritise waste prevention and re-use, establish minimum sorting and management requirements, identify and quantify amounts of CDW and treatment needs, drive innovation on recycling opportunities, regulate or standardise the management of hazardous materials within CDW. It fulfils the strategic established at national and regional levels.	4.3.1.	The percentage (%) of total generated waste, correctly segregated and managed towards materials recovery, re-use or any other type of valorisation. Avoided waste to landfill (tons) or as percentage of the total (%)	Requires instruments for SME and small producers of CDW. It is difficult to differentiate the impact of isolated waste management plans from cities or communities, since the statistics are usually generated at treatment centres, without any differentiation of the waste origin. There is a wide variation of costs on CDW management in Europe. However, management costs induce appropriately proper management and sorting always produces cost reduction for producers.
A waste manager produces recycled products under a **quality assurance scheme** that aims for an increased uptake of recycled aggregates by the industry and encourages waste segregation and diversion from landfill and, at the same time, includes environmental-related criteria e.g. for their leaching characteristics, with the achievement of EoW character or similar to the secondary material produced.	4.3.2.	Total amount of recycled materials used by the industry (e.g. tonnes) Percentage of substitution of natural aggregates by recycled aggregates (%)	Application of recycled aggregates needs a case-by-case approach for use. In some European countries there is a high competition with natural aggregates, usually, costs are favourable to the use of recycled aggregates.
CDW composed by at least 50 % concrete is recycled in an optimised process were product applications aim for high-grade recycling, as **recycled concrete aggregates** for filling operations and structural and non-structural concrete applications, and is produced under certain quality criteria that ensures its applicability.	4.3.3.	Amount of marketed recycled materials, in absolute units (e.g. tonnes) Percentage of natural materials substituted by recycled aggregates, e.g. for concrete manufacturing (high grade recycling) (%)	The applicability of recycled aggregates is dependent on the quality criteria and technical specifications required for structural and non-structural concrete, plus recycled materials specific restrictions, as gypsum and salt contents and leachability.

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BEMP description	Section	Environmental Performance Indicator	Remarks
Recycle waste plasterboard and other sources of waste gypsum to the manufacture of new plasterboard, according, if available, to a quality assurance scheme or industrial agreement.	4.3.4.	Percentage of recovered materials that are reprocessed as raw materials in plasterboard manufacturing (%) Percentage of reprocessed materials incorporated to the product (%) Percentage of waste plasterboard diverted from landfill (%)	The approach requires the engagement of the industry, as realised in the UK or Denmark. Collection from municipal recycling points is expensive, so it requires segregation in the origin (demolition and construction sites).
PCBs-containing wastes are well managed through the identification of PCB containing materials, removing and separating them, where the public authority is informed about the presence of these substances, and establish standard criteria for its management.	4.3.5.	Concentration levels of PCB are usually required to be determined according to EN 15308:2008, and expressed in mg or μg per kg of waste.	No restriction on applicability. Not regulated under certain thresholds of waste volume or floor area of construction. Costs of screening, identification, removal and management of PCB containing waste, hazardous or not, have to be assumed by the construction company and/or the developer.
HEALTHCARE WASTE			
Alternative techniques may constitute Best Environmental Management Practice if described environmental 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.	5.2.1.	Percentage of waste to alternative treatment (%)	Many applications are not suitable for bulky waste and certain fractions of HCW. Diversion of waste from high temperature incineration creates trade-offs on the consumption of fossil fuel. Cost of alternative treatment is lower, but the overall impact of these treatments does not show a better economic performance. Certain instruments must be also in place by the WMO, as awareness raising, training on the segregation of waste at origin.