Wrocław University of Technology

Environmental Quality Management

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SOLID WASTE MANAGEMENT

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Preface

This handbook presents current European waste management practices and is based on European law and waste management policy. Specifically, this handbook presents the up-to-date state of waste management organization and technologies. The material presented here is intended for students of environmental engineering.

The authors of this handbook are European experts in waste management with international experience in waste management planning and optimization.

The handbook consists of 14 chapters which were dedicated to various aspects of waste management, including legal, technological, environmental and economic ones.

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List of Abbreviations

AI	Administrative instruments
ANC	Acid Neutralization Capacity
AOX	Adsorbable Organic Halogen Compounds
AT4	Respiration Index within 4 days
BOD	Biological Oxygen Demand
BREF	Reference Document on the Best Available Techniques
BTEX	Benzene, Toluene, Ethylbenzene, and Xylenes
C&D	Construction and Demolition
CHP	Combined Heat and Power unit
COD	Chemical Oxygen Demand
DM	Dry Mass
EC	European Commission
EEA	European Environment Agency
EI	Economic instruments EI
ELV	End of Life Vehicles
EoL	End of Life
EPS	Electrostatic Precipitators
EWC	European Waste Catalogue
GB21	Gas Building Potential within 21 days
IE	Inhabitant Equivalent
II	Informative instruments
Inh.	Inhabitant
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
LOI	Loss on Ignition
LoW	List of Waste
MBP	Mechanical-Biological Pretreatment
MDR	Material Dry Recyclables or Mixed Dry Recyclables
mio	million
MRF	Material Recycling Facility
MSW	Municipal Solid Waste

Mt	million ton
NIMBY	Not In My Backyard
ODM	Organic Dry Mass
OLR	Organic Loading Rate
OU	Odor Unit
PAH	Polycyclic Aromatic Hydrocarbons
PAYT	Pay as You Throw
PCB	Polychlorinated Biphenyls
PCDD	Polychlorinated dibenzodioxins
PCDF	Polychlorinated dibenzo-furans
PDF	Packaging Derived Fuel
PEF	Processed Engineered Fuel
PEHD	Polyethylene High-Density
POPs	Persistent Organic Pollutants
PPF	Paper and Plastic Fraction
RDF	Refuse Derived Fuel
REF	Recovered Fuel
RTO	Regenerative Thermal Oxidation
S/S	Solidification and Stabilization
SCR	Selective Catalytic Reduction
SEA	Strategic Environmental Assessment
SNCR	Selective Non Catalytic Reduction
SRF	Solid Recovered Fuel or Specified Recovered Fuel
TEQ	International Toxic Equivalents
TOC	Total Organic Carbon
VFA	Volatile Fatty Acids
VOC	Volatile Organic Compounds
WEEE	Waste Electrical and Electronic Equipment
WFD	Waste Framework Directive
WWTP	Wastewater Treatment Plant

1 Introduction to waste management

Waste is generally regarded as material which has lost its possibility for use or value (McDougall, 2001). It is a by-product of various human activities occurring in the home, garden, workplace, public areas and in many other places. In most cases, the physical state and chemical composition of waste are the same as useful products with the only difference being the lack of value. This lack of value can often be related to the mixed material structure of waste and hence the simple separation of materials leads to an increase in their value. Increasingly, waste is being considered as a valuable source of secondary materials and renewable energy.

There are various criteria for the classification of waste:
- physical state (solid, liquid and semi-liquid or semi-solid waste),
- physical properties (combustible, biodegradable, recyclable etc.),
- origin of generation (household, municipal, rural, industrial, agricultural etc.)
- material composition (paper, glass, metals, kitchen waste etc.),
- safety level (hazardous, non-hazardous, inert).

1.1 Legal definitions of wastes and waste avoidance

Directive 2008/98/EC of 19 November 2008 on waste and repealing certain directives (referred to subsequently as the waste framework directive – WFD) is the primary and the most general European waste management regulation, containing
- legal definitions:
        of waste and waste types,
        of main terms of waste management, and
- main rules governing waste management.

Waste means any substance or object which the holder discards or intends or is required to discard. The meaning of the term "discarding" is not colloquial, and is limited to the discarding of waste to a recovery or disposal system.
The WFD introduces a new term "by-product". A substance or object, resulting from a production process, the primary aim of which is not the production of that item, may be regarded as not being waste but as being a **by-product** only if the following conditions are met:

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(a) further use of the substance or object is certain;

(b) the substance or object can be used directly without any further processing other than normal industrial practice;

(c) the substance or object is produced as an integral part of a production process; and

(d) further use is lawful, i.e. the substance or object fulfils all relevant product, environmental and health protection requirements for the specific use and will not lead to overall adverse environmental or human health impacts.

Two terms defined by the WFD are very important for avoiding waste generation: prevention and re-use.

Prevention means measures taken before a substance, material or product has become waste that reduce

(a) the quantity of waste, including through the re-use of products or the extension of the lifespan of products;

(b) the adverse impacts of the generated waste on the environment and human health; or

(c) the content of harmful substances in materials and products.

Re-use means any operation by which products or components that are not waste (e.g. by-products) are used again for the same purpose for which they were conceived.

Mutual connections of the main definitions of waste and processes of dealing with waste are presented in Figure 1-1.

The WFD also introduces definitions for specific types of waste such as biowaste, hazardous waste and waste oils which will be discussed later in this chapter.

Other directives contain definitions of additional waste streams with the following being those given in directive 1999/31/EC of the 26 April 1999 on the landfilling of waste.

Municipal waste means waste from households, as well as other waste which, because of its nature or composition, is similar to waste from households.

Biodegradable waste is regarded any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and garden waste, and paper and paperboard.

Inert waste means waste that does not undergo any significant physical, chemical or biological transformations. Inert waste will not dissolve, burn or otherwise physically or chemically react, biodegrade or adversely affect other matter with which it comes into contact in a way likely to give rise to environmental pollution

or to harm human health. The total leachability and pollutant content of the waste and the ecotoxicity of the leachate must be insignificant, and in particular not endanger the quality of surface water and/or groundwater.
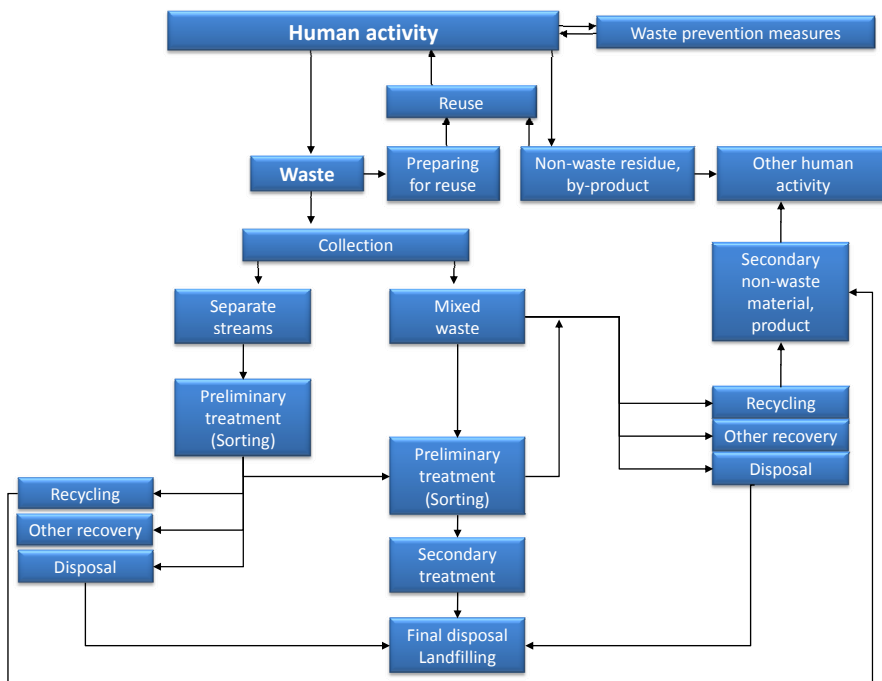
Figure 1-1. Connections among main definitions of waste and processes of dealing with waste

1.2 *Dealing with waste and end of waste status*

Dealing with waste is defined as **waste management** meaning the collection, transport, recovery and disposal of waste, including the supervision of such operations and the aftercare of disposal sites, and including actions taken as a dealer or broker. Any original waste producer or the other holder has the obligation to carry out the treatment of waste himself or to have the treatment handled by a dealer or an establishment or undertaking which carries out waste treatment operations or arranged by a private or public waste collector. The responsibility for arranging waste management is to be borne partly or wholly by

the producer of the product from which the waste came and the distributors of such a product may share the responsibility.

Waste holder means the producer of the waste or the natural or legal person who is in possession of the waste. **Waste producer** is defined as anyone whose activities produce waste ("original waste producer") and/or anyone who carries out pre-processing, mixing or other operations resulting in a change in the nature or composition of this waste.

The WFD introduces two new entities into waste management: dealers and brokers. **Dealer** means any undertaking which acts in the role of principal to purchase and subsequently sell waste, including such dealers who do not take physical possession of the waste. A **broker** is regarded as any undertaking arranging the recovery or disposal of waste on behalf of others, including such brokers who do not take physical possession of the waste.

Collection is the first step of dealing with waste and means the gathering of waste, including the preliminary sorting and preliminary storage of waste for the purpose of transport to a waste treatment facility. **Separate collection** means the collection where a waste stream is kept separately by type and nature so as to facilitate a specific treatment.

Preparing for re-use means checking, cleaning or repairing recovery operations, by which products or components of products that have become waste are prepared so that they can be re-used without any other pre-processing.

Treatment means recovery or disposal operations, including preparation prior to recovery or disposal.

Recovery means any operation the principal result of which is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfill a particular function, or waste being prepared to fulfill that function, in the plant or in the wider economy. **Recycling** is defined as any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes. It includes the reprocessing of organic material but does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations. Annex II to the WFD sets out a non-exhaustive list of recovery operations, provided as follows:

R 1 Use principally as a fuel or other means to generate energy, including
    incineration facilities dedicated to the processing of municipal solid waste
    only when their energy efficiency is equal to or above:

- 0,60 for installations in operation and permitted in accordance with applicable Community legislation before 1 January 2009,
- 0,65 for installations permitted after 31 December 2008,

R 2 Solvent reclamation/regeneration

R 3 Recycling/reclamation of organic substances which are not used as solvents (including composting and other biological transformation processes as well as gasification and pyrolisis using the components as chemicals)

R 4 Recycling/reclamation of metals and metal compounds

R 5 Recycling/reclamation of other inorganic materials (including soil cleaning resulting in recovery of the soil and recycling of inorganic construction materials)

R 6 Regeneration of acids or bases

R 7 Recovery of components used for pollution abatement

R 8 Recovery of components from catalysts

R 9 Oil re-refining or other reuses of oil

R 10 Land treatment resulting in benefits to agriculture or ecological improvement

R 11 Use of wastes obtained from any of the operations numbered R 1 to R 10

R 12 Exchange of wastes for submission to any of the operations numbered R 1 to R 11 (if there is no other appropriate R code, this can include preliminary operations prior to recovery including pre-processing such as, inter alia, dismantling, sorting, crushing, compacting, pelletizing, drying, shredding, conditioning, repackaging, separating, blending or mixing prior to submission to any of the operations numbered R1 to R11

R 13 Storage of wastes pending any of the operations numbered R 1 to R 12 (excluding temporary storage, pending collection, on the site where it is produced - temporary storage means preliminary storage of waste for the purposes of transport to a waste treatment facility).

Disposal means any operation which is not recovery even where the operation has as a secondary consequence the reclamation of substances or energy. Annex I to the WFD sets out a non-exhaustive list of disposal operations, provided below:

D 1 Deposit into or onto land (e.g. landfill, etc.)

D 2 Land treatment (e.g. biodegradation of liquid or sludgy discards in soils, etc.)

D 3 Deep injection (e.g. injection of pumpable discards into wells, salt domes or naturally occurring repositories, etc.)

D 4 Surface impoundment (e.g. placement of liquid or sludgy discards into pits, ponds or lagoons, etc.)

D 5 Specially engineered landfill (e.g. placement into lined discrete cells which are capped and isolated from one another and the environment, etc.)

D 6 Release into a water body except seas/oceans

D 7 Release into seas/oceans including sea-bed insertion

D 8 Biological treatment not specified elsewhere in this Annex which results in final compounds or mixtures which are discarded by means of any of the operations numbered D 1 to D 12

D 9 Physico-chemical treatment not specified elsewhere in this Annex which results in final compounds or mixtures which are discarded by means of any of the operations numbered D 1 to D 12 (e.g. evaporation, drying, calcination, etc.)

D 10 Incineration on land

D 11 Incineration at sea (this operation is prohibited by EU legislation and international conventions

D 12 Permanent storage (e.g. emplacement of containers in a mine, etc.)

D 13 Blending or mixing prior to submission to any of the operations numbered D 1 to D 12 (if there is no other appropriate code, this can include preliminary operations prior to disposal including pre-processing such as, inter alia, sorting, crushing, compacting, pelletizing, drying, shredding, conditioning or separating prior to submission to any of the operations numbered D1 to D12)

D 14 Repackaging prior to submission to any of the operations numbered D 1 to D 13

D 15 Storage pending any of the operations numbered D 1 to D 14 (excluding temporary storage, pending collection, on the site where it is produced - temporary storage means preliminary storage of waste for the purposes of transport to a waste treatment facility)

Certain specified waste shall **cease to be waste** when it has undergone a recovery, including recycling, operation and complies with specific criteria to be developed in accordance with the following conditions:

(a) the substance or object is commonly used for specific purposes;

(b) a market or demand exists for such a substance or object;

(c) the substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products; and

(d) the use of the substance or object will not lead to overall adverse environmental or human health impacts.

The criteria shall include limit values for pollutants where necessary and shall take into account any possible adverse environmental effects of the substance or object.

End-of-waste specific criteria should be considered, among others, at least for aggregates, paper, glass, metal, tires and textiles.

Waste which ceases to be waste in accordance shall also ceases to be waste for the purpose of the recovery and recycling targets set out in Directives on packaging and packaging waste (94/62/EC), on end-of-life vehicles (2000/53/EC),

on waste electrical and electronic equipment (2002/96/EC) and on batteries and accumulators 2006/66/EC and other relevant Community legislation when the recycling or recovery requirements of that legislation are satisfied.

1.3 Waste hierarchy

The following waste hierarchy shall apply as a priority order in waste prevention and management legislation and policy:
(a) Prevention
(b) Preparing for re-use
(c) Recycling
(d) Other recovery, e.g. energy recovery,
(e) Disposal
When applying the waste hierarchy, Member States shall take measures to encourage the options that deliver the best overall environmental outcome.
This may require specific waste streams departing from the hierarchy where this is justified by life-cycle thinking on the overall impacts of the generation and management of such waste (see chapter 11).

The decision making should also take into account the general environmental protection principles of precaution and sustainability, technical feasibility and economic viability, protection of resources as well as the overall environmental, human health, economic and social impacts.

1.4 Hazardous waste

Hazardous waste means waste which displays one or more of the hazardous properties listed in Annex III of the WFD, presented below.

Hazard	Description of hazardous properties
H1 Explosive	substances and preparations which may explode under the effect of flame or which are more sensitive to shocks or friction than dinitrobenzene
H2 Oxidizing	substances and preparations which exhibit highly exothermic reactions when in contact with other substances, particularly flammable substances
H3-A Highly flammable	- liquid substances and preparations having a flash point below 21 °C (including extremely flammable liquids), or - substances and preparations which may become hot and finally catch fire in contact with air at ambient temperature without any application of energy, or - solid substances and preparations which may readily catch fire after brief contact with a source of ignition and which continue to burn or to be consumed after removal of the source of ignition, or

	- gaseous substances and preparations which are flammable in air at normal pressure, or - substances and preparations which, in contact with water or damp air, evolve highly flammable gases in dangerous quantities
H3-B Flammable	liquid substances and preparations having a flash point equal to or greater than 21 °C and less than or equal to 55 °C
H4 Irritant	non-corrosive substances and preparations which, through immediate, prolonged or repeated contact with the skin or mucous membrane, can cause inflammation
H5 Harmful	substances and preparations which, if they are inhaled or ingested or if they penetrate the skin, may involve limited health risks
H6 Toxic	substances and preparations (including very toxic substances and preparations) which, if they are inhaled or ingested or if they penetrate the skin, may involve serious, acute or chronic health risks and even death
H7 Carcinogenic	substances and preparations which, if they are inhaled or ingested or if they penetrate the skin, may induce cancer or increase its incidence
H8 Corrosive	substances and preparations which may destroy living tissue on contact
H9 Infectious	substances containing viable micro-organisms or their toxins which are known or reliably believed to cause disease in man or other living organisms
H10 Toxic for reproduction	substances and preparations which, if they are inhaled or ingested or if they penetrate the skin, may induce non-hereditary congenital malformations or increase their incidence
H11 Mutagenic	substances and preparations which, if they are inhaled or ingested or if they penetrate the skin, may induce hereditary genetic defects or increase their incidence
H12 Waste which release toxic or very toxic gases in contact with water, air or an acid	
H13 Sensitizing	substances and preparations which, if they are inhaled or if they penetrate the skin, are capable of eliciting a reaction of hypersensitization such that on further exposure to the substance or preparation, characteristic adverse effects are produced
H14 Ecotoxic	substances and preparations which present or may present immediate or delayed risks for one or more sectors of the environment
H15 Waste capable by any means, after disposal, **of yielding another substance,** e.g. a leachate, which possesses any of the characteristics listed above	

The production, collection and transportation of hazardous waste, as well as its storage and treatment, should be carried out in conditions providing protection for the environment and human health including action to ensure the traceability from production to final destination and control of hazardous waste. Hazardous

waste must not be mixed, either with other categories of hazardous waste or with other waste, substances or materials. Mixing shall include the dilution of a hazardous substance. By the way of derogation, the mixing of waste may be allowed provided that the mixing operation is carried out by an establishment or undertaking which has obtained a permit and the adverse impact of waste management on human health and the environment is not increased.

In the course of collection, transport and temporary storage, hazardous waste is packed and labeled in accordance with the international and Community standards in force. Whenever hazardous waste is transferred within a Member State, it shall be accompanied by an identification document, which may be in electronic format.

Rules concerning the control of hazardous waste, ban on mixing of hazardous waste and its labeling shall not apply to mixed waste produced by households. Rules concerning the labeling of hazardous waste shall not apply to separate fractions of hazardous waste produced by households until they are accepted for collection, disposal or recovery by an establishment or undertaking which has obtained a permit or has been registered.

Special regulations were provided in the WFD for the management of specific hazardous waste - waste oils. **Waste oils** mean any mineral or synthetic lubrication or industrial oils which have become unfit for the use for which they were originally intended, such as used combustion engine oils and gearbox oils, lubricating oils, oils for turbines and hydraulic oils. One of the possible treatment operations is the r**egeneration of waste oils** regarded as any recycling operation whereby base oils can be produced by refining waste oils, in particular by removing the contaminants, the oxidation products and the additives contained in such oils. Waste oils are to be collected separately, where this is technically feasible and treated accordingly to the waste hierarchy without endangering human health and without harming the environment. Where this is technically feasible and economically viable, waste oils of different characteristics should not be mixed and also waste oils should not be mixed with other kinds of waste or substances, if such mixing impedes their treatment.

For the purposes of separate collection of waste oils and their proper treatment, additional measures may be applied such as technical requirements, producer responsibility, economic instruments or voluntary agreements. For example, Poland applies producer responsibility measures relying on obligation to ensure given recycling targets.

1.5 Extended producer responsibility

In order to strengthen the re-use and the prevention, recycling and other recovery of waste, Member States may take legislative or non-legislative measures to ensure that any natural or legal person who professionally develops,

manufactures, processes, treats, sells or imports products (producer of the product) has extended producer responsibility.

Such measures may include an acceptance of returned products and of the waste that remains after those products have been used, as well as the subsequent management of the waste and financial responsibility for such activities. These measures may include the obligation to provide publicly available information as to the extent to which the product is re-usable and recyclable.

Member States may take appropriate measures to encourage the design of products in order to reduce their environmental impacts and the generation of waste in the course of the production and subsequent use of products, and in order to ensure that the recovery and disposal of products that have become waste take place in accordance with the waste hierarchy and the protection of human health and the environment. Such measures may encourage, inter alia, the development, production and marketing of products that are suitable for multiple use, that are technically durable and that are, after having become waste, suitable for proper and safe recovery and environmentally compatible disposal.

1.6 List of Wastes

The list of waste (LoW - List of Wastes) was established by Decision 2000/532/EC and amended by the Decision 2001/118/EC. The inclusion of a substance or object in the list shall not mean that it is waste in all circumstances. A substance or object shall be considered to be waste only where the definition is met. The different types of waste in the list are defined by the six-digit code for the waste and the respective two-digit and four–digit chapter headings. There are 20 two-digit chapters of the list where wastes are classified according to the place of their origin.

The List of Wastes includes hazardous waste marked with an asterisk (*) and takes into account the origin and composition of the waste and, where necessary, the limit values of concentration of hazardous substances.

Wastes classified as hazardous are considered to display one or more of the properties listed in Annex III to the WFD and, as regards H3 to H8, H10 and H11 of that Annex, one or more of the following:
- flash point ≤ 55 °C,
- one or more substances classified as very toxic at a total concentration ≥ 0,1 %,
- one or more substances classified as toxic at a total concentration ≥ 3 %,
- one or more substances classified as harmful at a total concentration ≥ 25 %,
- one or more corrosive substances classified as R35 at a total concentration ≥ 1 %,
- one or more corrosive substances classified as R34 at a total concentration ≥ 5 %,
- one or more irritant substances classified as R41 at a total concentration ≥ 10 %,

- one or more irritant substances classified as R36, R37, R38 at a total concentration ≥ 20%,
- one or more substances known to be carcinogenic of category 1 or 2 at a total concentration ≥ 0,1 %,
- one or more substances toxic for reproduction of category 1 or 2 classified as R60, R61 at a total concentration ≥ 0,5 %,
- one or more substances toxic for reproduction of category 3 classified as R62, R63 at a total concentration ≥ 5 %,
- one or more mutagenic substances of category 1 or 2 classified as R46 at a total concentration ≥ 0,1 %,
- one or more mutagenic substances of category 3 classified as R40 at a total concentration ≥ 1 %.

The List of Wastes is binding as regards determination of the waste which is to be considered as hazardous waste. A Member State may consider waste as hazardous where, even though it does not appear as such on the List of Wastes, it displays one or more of the properties listed in Annex III. Where a Member State has evidence to show that specific waste that appears on the list as hazardous waste does not display any of the properties listed in Annex III, it may consider that waste as non-hazardous waste. In both cases, the Member state should notify the European Commission.

The reclassification of hazardous waste as non-hazardous waste may not be achieved by diluting or mixing the waste with the aim of lowering the initial concentrations of hazardous substances to a level below the thresholds for defining waste as hazardous.

According to the WFD the Decision 2000/532/EC shall be updated.

The Regulation (EC) 2150/2002 on waste statistics obliges the Member States to report statistical data on waste amounts to the European Statistical Office (EUROSTAT). Annex III to this Regulation contains the transposition table between the statistical nomenclature (EWC-Stat) and the European List of Wastes (LoW). EWC-Stat is regarded as a "mainly substance oriented waste statistical nomenclature" and is in contrast to the European List of Wastes (LoW) which is mainly source oriented. The European List of Wastes is used in Europe mainly for controlling and monitoring waste streams. For waste statistics the EWC-Stat categories have to be used, but this does not necessarily mean that EWC-Stat has to be used directly at the level of companies. The majority of the countries intend to primarily use the LoW for data collection and to transform the collected information into EWC-Stat using the transposition table. Most of the countries use LoW, Lithuania uses EWC-Stat and some countries have extended the LoW with additional national codes, e.g. Poland (Guidance 2004).

1.7 Prevention, re-use and recovery of waste

Member States shall establish waste prevention programs no later than 12 December 2013. Such programs shall be integrated either into the waste management plans or into other environmental policy programs as appropriate, or shall function as separate programs. The programs shall set out the waste prevention objectives. Member States shall describe the existing prevention measures and evaluate the usefulness of the examples of measures indicated in annex IV of the WFD or other appropriate measures.

The aim of such objectives and measures shall be to break the link between economic growth and the environmental impacts associated with the generation of waste. The appropriate specific qualitative and quantitative benchmarks for waste prevention measures shall be adopted in order to monitor and assess the progress of the measures and specific qualitative and quantitative targets and indicators may be determined.

The European Commission shall submit to the European Parliament and the Council the following reports accompanied, if appropriate, by proposals for measures required in support of the prevention activities and the implementation of the waste prevention programs covering:

(a) by the end of 2011, an interim report on the evolution of waste generation and the scope of waste prevention, including the formulation of a product eco-design policy addressing both the generation of waste and the presence of hazardous substances in waste, with a view to promoting technologies focusing on durable, re-usable and recyclable products;

(b) by the end of 2011, the formulation of an action plan for further support measures at European level seeking, in particular, to change current consumption patterns;

(c) by the end of 2014, the setting of waste prevention and decoupling objectives for 2020, based on best available practices including, if necessary, a revision of the indicators.

The necessary measures shall be taken to ensure that waste undergoes recovery operations. Where necessary to facilitate or improve recovery, waste shall be collected separately if technically, environmentally and economically practicable and shall not be mixed with other waste or other material with different properties.

Member States shall take measures to promote:
- the re-use of products and preparing for re-use activities, notably by encouraging the establishment and support of re-use and repair networks, the use of economic instruments, procurement criteria, quantitative objectives or other measures.
- high quality recycling and, to this end, shall set up separate collections of waste where technically, environmentally and economically practicable

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and appropriate to meet the necessary quality standards for the relevant recycling sectors.

By 2015 separate collection shall be set up for at least the following: paper, metal, plastic and glass.

In order to comply with the objectives of the WFD and move towards a European recycling society with a high level of resource efficiency, Member States shall take the necessary measures designed to achieve the following targets:

(a) by 2020, the preparing for re-use and the recycling of waste materials such as at least paper, metal, plastic and glass from households and possibly from other origins as far as these waste streams are similar to waste from households, shall be increased to a minimum of overall 50 % by weight,

(b) by 2020, the preparing for re-use, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste excluding naturally occurring material defined in category 17 05 04 in the List of Wastes shall be increased to a minimum of 70 % by weight.

1.8 Self-sufficiency and proximity rules

According to the WFD:

- Member States shall take appropriate measures, in cooperation with other Member States where this is necessary or advisable, to establish an integrated and adequate network of waste disposal installations and of installations for recovery of mixed municipal waste collected from private households, including where such collection covers such waste from other waste producers, taking into account of the best available techniques.
- the network must enable the Community as a whole to become self-sufficient in waste disposal and in the waste recovery of mixed municipal waste and to enable Member States to move towards that aim individually, taking into account geographical circumstances or the need for specialized installations for certain types of waste.
- the network of waste disposal installations and of installations for recovery of mixed municipal waste shall enable waste to be disposed of or mixed municipal waste to be recovered in one of the nearest appropriate installations, by means of the most appropriate methods and technologies in order to ensure a high level of protection for the environment and public health.

1.9 Waste management plans

Competent authorities in each Member State of the EU must establish one or more waste management plans which cover the entire territory of the state concerned.

Such plans shall contain at least the following:
(a) the type, quantity and source of waste generated within the territory and the waste likely to be shipped from or to the national territory;
(b) existing waste collection schemes and major waste disposal and recovery installations;
(c) an assessment of the need for new collection schemes, the closure of existing installations, additional waste installation infrastructure;
(d) sufficient information on the location criteria for site identification and on the capacity of future disposal or major recovery installations, if necessary;
(e) general waste management policies including planned waste management technologies and methods, or policies for waste posing specific management problems.

1.10 Permits and registrations

Any establishment or undertaking intending to carry out waste treatment must obtain a permit from the competent authority. Such permits shall specify at least the following:
- the types and quantities of waste that may be treated,
- for each type of operation permitted, the technical and any other requirements relevant to the site concerned,
- the safety and precautionary measures to be taken,
- the method to be used for each type or operation,
- such monitoring and control operations as may be necessary,
- such closure and aftercare provisions as may be necessary.

Establishments or undertakings for the disposal of their own non-hazardous waste at the place of production or for recovery of waste may be exempted from obtaining such permits.
Some entities such as
- establishments or undertakings which collect or transport waste on a professional basis,
- dealers or brokers,
- establishments or undertakings which are subject to exemptions from permit requirements

may be not subject to permit requirements, but in this case the competent authority should keep their register.

2 European waste legislation and policy

2.1 *Waste legislation*

European Union law (historically called European Community law) consists of treaties as well as law and court judgments which operate alongside the legal systems of the EU Member States. It has direct effect within the EU Member States and, where conflict occurs, takes precedence over national law. The primary source of EU law is the treaties which set broad policy goals and establish institutions that, amongst other things, can enact legislation in order to achieve those goals. The legislative acts of the EU come in three forms: regulations, directives and decisions. Regulations become law in all Member States the moment they come into force, without the requirement for any implementing measures, and automatically override conflicting domestic provisions. Directives require Member States to achieve a certain result while leaving them discretion as to how to achieve the result. The details of how they are to be implemented are left to Member States. A decision is binding on the person or entity to which it is addressed. Decisions may be addressed to Member States or individuals.

EU legislation derives from decisions taken at the EU level, yet implementation largely occurs at a national level. The principle of uniformity is therefore a central theme in all decisions by the European Court of Justice, which aims to ensure the application and interpretation of EU laws does not differ between Member States.

The directives are the predominantly used legislation instrument regulating waste management in the EU (Figure 2-1). The regulations concern only matters of waste statistics and transboundary shipment of waste which should be uniformly managed across Europe. The European framework of waste legislation has evolved along the following three lines: general waste law, specific waste streams and specific waste treatment methods.

General waste law
Directive 2008/98/EC of the European Parliament and of the Council of November 2008 on waste and repealing certain Directives;

Regulation (EC)No 1013/2006 of the European Parliament and of the Council, of 14 June 2006 on shipments of waste;

Regulation (EC) No 2150/2002 of the European Parliament and of the Council of November 2002 on waste statistics;

Directive 2008/1/EC of the European Parliament and of the Council of 15 January 2008 concerning integrated pollution prevention and control;

Directives on specific waste streams
Directive 86/278/EEC on use of sewage sludge in agriculture;

European Parliament and Council Directive 94/62/EC of 20 December 1994 on packaging and packaging waste, amended by Directive 2004/12/EC;

Directive 2000/53/EC of the European Parliament and of the Council of 18 September 2000 on end of life vehicles;

Directive 2002/96/EC of the European Parliament and of the Council of 27 January 2003 on waste electrical and electronic equipment (WEEE);

Directive 2006/21/EC of the European Parliament and of the Council of 15 March 2006 on the management of waste from the extractive industries;

Directive 2006/66/EC of the European Parliament and the Council of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91 157 EEC;

Directive 96/59/EC of the Council of 16 September 1996 on the disposal of polychlorinated biphenyls and polychlorinated terphenyls (PCB/PCT);

Directive 2000/59/EC of the European Parliament and of the Council of 27 November 2000 on port reception facilities for ship-generated waste and cargo residues

Directives on specific waste treatment methods
Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste;

Council Decision 2003/33 of 19 December 2002 on criteria for the acceptance of waste at landfills;

Directive of the Council and the European Parliament 2000/76/EC of 4 December 2000 on the incineration of waste.

Besides the legislation instruments, there are also some policy actions and documents undertaken, such as environmental actions programs and thematic strategies. The 6th Environmental Action Program introduced the concept of Thematic Strategies, which were developed by the European Commission. The Thematic Strategies are a modernization of EU environmental policy-making, taking a broader, strategic approach. The Thematic Strategies build on the existing EU legal/regulatory framework and include new knowledge on threats to human health and the environment. They focus on an integrated approach (the effects of decisions in one policy area which has consequences on the others) and on implementation issues. The Thematic Strategies can be seen as key elements of the Commission's Better Regulation strategy: they are all accompanied by a

thorough impact assessment, assessing the economic, social and environmental impacts of different policy options, extensive stakeholder consultations were held and they aim, where possible, at simplifying the existing regulatory framework. Thematic Strategies cover the following fields: air, waste prevention and recycling, marine environment, soil, pesticides, natural resources, urban environment. The Thematic Strategies were to be reviewed in 2010.
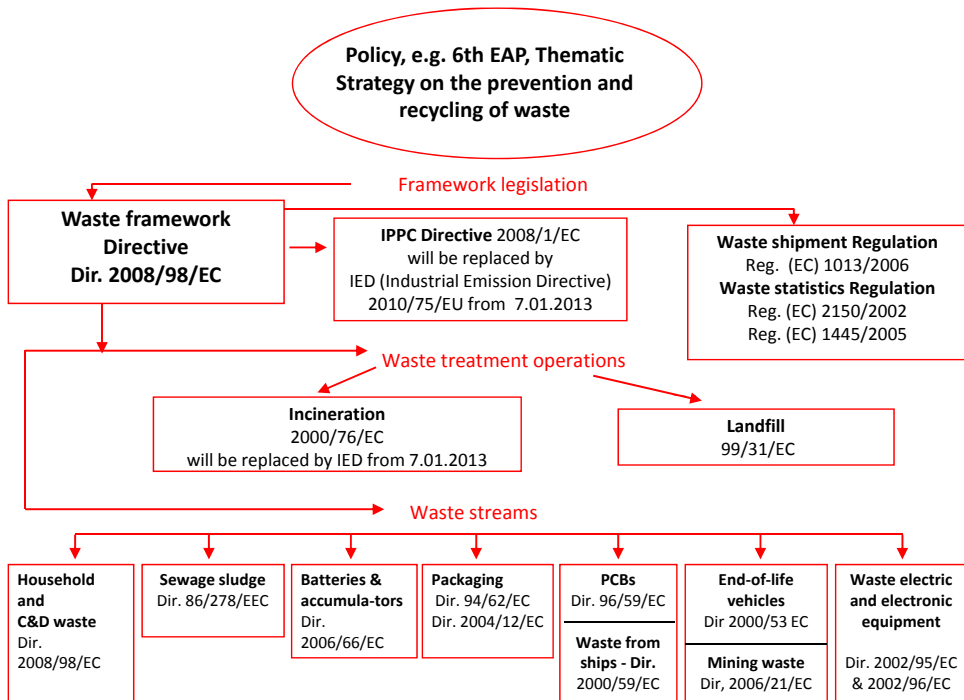
Figure 2-1. General structure of the EU waste management law

The waste directives introduce some quantitative targets to be reached in the EU countries by the provided deadlines. Those targets are presented in the Table 2-1.

Table 2-1. Some targets for recycling, recovery and separate collection of waste

Targets, waste streams	Year	min. recovery	min. recycling	collection rate
Packaging waste	2008/2014	60%	55%	
ELV (used cars)	2015	95%	85%	100%
WEEE	2006/2008	70%	50%	4 kg/inh. year
Batteries	2011		50% to 70% efficiency	
	2012			25%
	2016			45%
Used tires	2006	Ban on landfilling		
Biodegradable waste diversion from landfills	2006/2010	reduction to 75% of the 1995 level		
	2009/2013	reduction to 50% of the 1995 level		
	2016/2020	reduction to 35% of the 1995 level		
NEW targets WFD	2015	Separate collection paper/metals/plastics/glass		
	2020	50% of recycling or preparation for re-use of paper/metals/plastics/glass from households		
	2020	50% of recycling or preparation for re-use of C&D waste		

2.2 Waste policy

According to the Thematic strategy on the prevention and recycling of waste, waste is an environmental, social and economic challenge for Europeans, showing two different faces:

1. negative images: rubbish bags, litter and toxic waste dumps.
2. positive - waste as an opportunity – Europe's drive to deal with waste in environmentally sound ways has generated jobs and business opportunities.

The waste management and recycling sector shows a high growth rate, an estimated turnover of over €100 billion for EU-25 and provides between 1,2 and 1,5 million jobs. The recycling industry can be characterized as providing an increasing amount of resources to the manufacturing industry: at least 50% of the paper and steel, 43% of the glass and 40% of the non-ferrous metal produced in the EU are currently derived from recycled materials. The results of 30 years of European waste management policy should be summarized by the 5 main fields of activities undertaken:

- waste treatment facilities under the control of environmental authorities,

- waste disposal is subject to environmental standards,
- disposal of hazardous waste is monitored,
- shipment of waste is monitored,
- recycling and recovery of waste is increased.

However, the waste prevention measures are still not satisfactory. EU waste policy has the potential to contribute to reducing the overall negative environmental impact of resource use. Preventing waste generation and promoting recycling and the recovery of waste will increase the resource efficiency of the European economy and reduce the negative environmental impact of using natural resources. This will contribute to maintaining the resource base, essential for sustained economic growth. The basic objectives of current EU waste policy – to prevent waste and promote re-use, recycling and recovery so as to reduce the negative environmental impact – are still valid and will be supported by the impact-based approach proposed in the Thematic Strategy. The long-term goal is for the EU to become a recycling society that seeks to avoid waste and uses waste as a resource. With high environmental reference standards in place, the internal market will facilitate recycling and recovery activities.
The main objectives of EU waste policy can be expressed as
- decoupling economic growth from environmental impacts, e.g. growth of waste generation,
- prevention of waste,
- moving towards a recycling society,
- promoting the use of waste for energy production,
- better implementation of waste legislation.
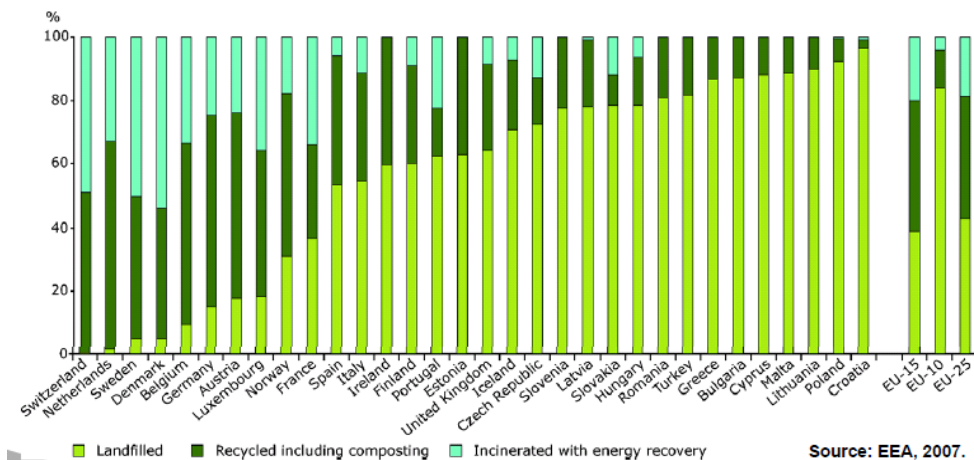
Figure 2-2. Recycling, incineration and landfilling of municipal waste in the EU in 2007

There are still many differences in the level of development of municipal waste management among the EU-15 and EU-12 Member States (Fig. 2-2). Fig 2-3 shows that EU countries can be categorized under three waste management 'groupings', according to the strategies for diversion of municipal waste away from landfills and the relative shares of landfilling, material recovery (mainly recycling and composting) and incineration. The first grouping comprises countries with high levels of both material recovery and incineration and relatively low landfill levels. The second grouping brings together countries with high material recovery rates and medium incineration levels as well as a medium dependence on landfilling. The third grouping contains those countries whose material recovery and incineration levels are both low and whose dependence on landfilling is relatively high.
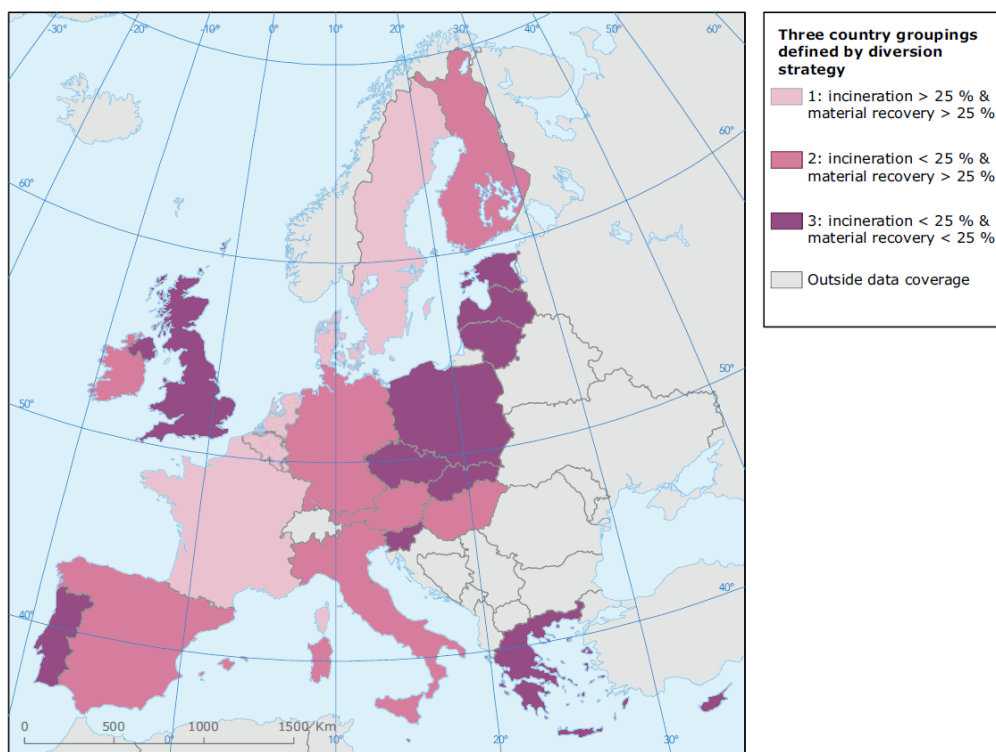
Figure 2-3. Three country groupings defined by the landfill diversion strategy (EEA 2007)

Figure 2-4 presents the general sketch of municipal waste management and landfill policy objectives. These objectives can be summarized as less waste generation, more recycling and treatment, less environmental impact. Different policy instruments may be adopted by individual countries to achieve the above mentioned targets.

The waste policy instruments can be divided into three main groups (Tojo, 2007):

- **Administrative instruments AI -** *You may!   You may not!*

- **Economic instruments EI -**   *If you do not, you will have to pay! If you do, you will be subsidized!*

- **Informative instruments II -**   *based on the notion that people and organizations would act differently if they had more understanding and knowledge*
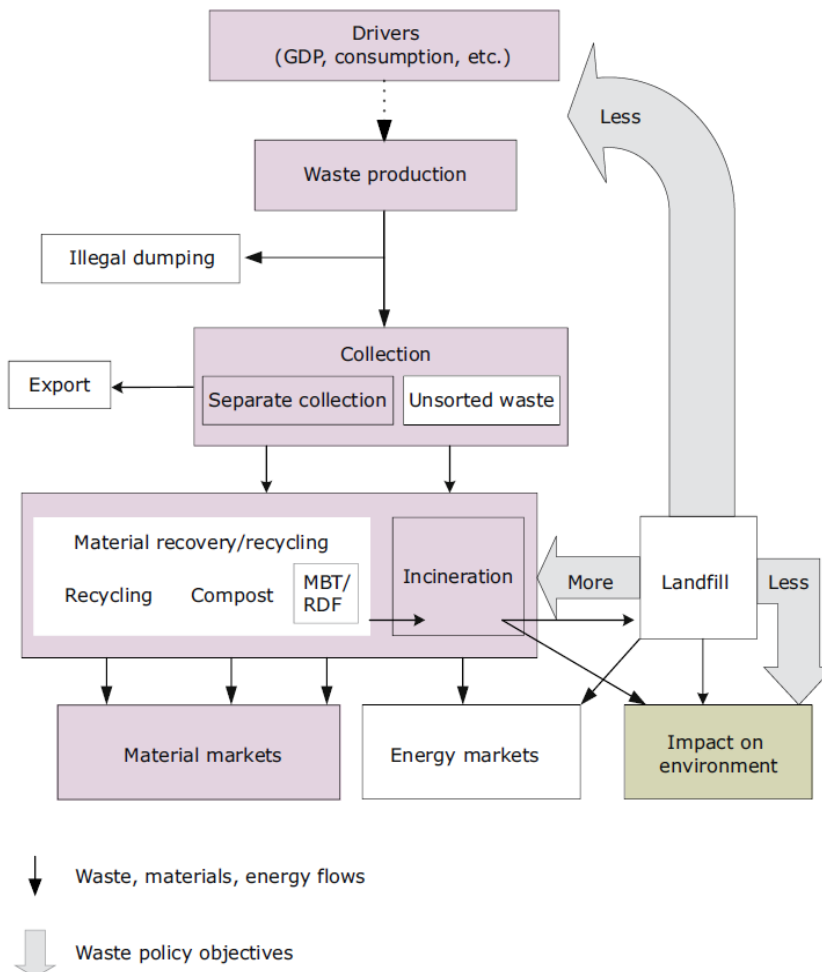
Figure 2-4. Simplified sketch of municipal waste management and landfill policy objectives (Diverting, 2009)

Various criteria may be used for evaluation of instruments, such as
- Content
- Objectives (Waste Hierarchy)
- Addresses
- Environmental effectiveness (goal attainment, attributability assessment)
- Potential introduction at the local level (AI,II)
- Frequency of application (EI)
- Political applicability/enforceability (EI)

As the main administrative instruments, the following should be listed:
- Substance restriction
- Source separation
- Producer's take-back of specific discarded products
- Collection, reuse/refill and recycling targets
- Minimum recycled material content standards
- Landfill restriction
- Environmentally sound treatment standards

All these instruments are addressed to different elements of the waste hierarchy as presented below.

Prevention	Reuse/recycling	Env'lly sound treatment/disposal
Substance restriction		
Reuse/recycling target		
Minimum recycled material content standard		
Producer take-back		
	Source separation	
	Collection/landfill diversion target	
	Landfill restriction	
		Treatment/disposal standard

Among the various economic instruments the following are used most frequently:
- Landfill tax
- Waste disposal tax
- Recycling credit scheme
- Subsidy on secondary products
- Waste-pricing: "pay-as-you-throw" scheme
- Deposit refund system
- Green public procurement.

They have various influences on the waste management hierarchy as shown on the diagram below.

Prevention	Reuse/recycling	Env'lly sound treatment/disposal
Subsidy on secondary product		
Pay-as-you-throw scheme		
Deposit-refund system		
Green public procurement		
	Recycling credit scheme	
	Landfill / disposal tax	

The following informative instruments enumerated below:
- Eco-labeling scheme
- Green shopping guide
- Marking of products and components
- Information campaign to citizens
- Information provision to treatment facilities

are mostly used. They primarily address prevention, reuse and recycling of waste.

Prevention	Reuse/recycling	Env'lly sound treatment/disposal
Eco-labeling scheme		
Green shopping guides		
Information campaign to citizens		
	Marking of products and components	
		Information to treatment facilities

There is a lack of general evaluations regarding the effectiveness of particular instruments in achieving the main objectives of European waste management policy. Various instruments address different parts of the waste hierarchy and are introduced simultaneously. There are also various cultural and demographic contexts and limited periods of time since various instruments in the different EU countries have been introduced.

3 Quantity and composition of waste

This book is centered around the theme of municipal waste management. In this chapter, characteristics of generated and collected amounts of municipal waste are provided. However, municipal waste is only one of many types of waste. Figure 3-1 displays the amounts of waste generated in the European Union.
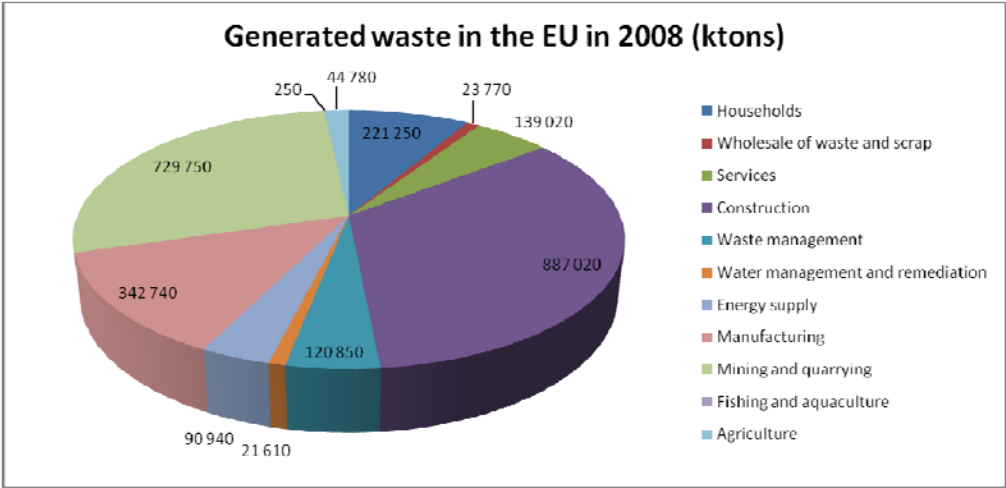
Figure 3-1. Generated waste in the European Union (EU27) in 2008 (Eurostat 2010a)

From the above figure, it can be seen that municipal waste only constitutes a relatively small share of the total waste generation, less than 10%. Also, the figure shows the irregularity of using the terms 'municipal' and 'household' waste.

Intermezzo: household vs. municipal

The OECD/Eurostat Joint Questionnaire defines household waste as (Eionet 2010)

> *"waste from households as well as other waste, which, because of its nature or composition, is similar to waste from households."*

This can be seen as the waste that is collected during collection activities aimed at collecting waste generated by households. These may also include waste from small shops and other enterprises. It refers both to separated fractions and residual or mixed waste.

For municipal waste various definitions are in use. In section 20 of the European List of Wastes (generally referred to as municipal waste) it is defined as (EC 2000)

> *"Municipal wastes and similar commercial, industrial and institutional wastes including separately collected fractions"*

The Landfill Directive mentions municipal waste as (EC 1999)

> *"…waste from households, as well as other waste, which, because of its nature or composition, is similar to waste from households"*

These definitions, however, would also include industrial waste fractions and sludge from municipal wastewater treatment. Therefore, the description here of municipal waste in the OECD/Eurostat Joint Questionnaire is followed. According to this, municipal waste is collected by or on behalf of municipalities and includes (Eionet 2010)

- waste from households;

- waste from commerce and trade, small businesses, office buildings and institutions (schools, hospitals, government buildings);

- waste from selected municipal services, i.e. waste from park and garden maintenance

- waste from street cleaning services (street sweepings, the content of litter containers, market cleansing waste;

- both separately collected fractions, mixed/residual waste as well as bulky waste;

- similar waste that is generated, but not collected, e.g. in rural areas not serviced by waste collection.

Explicitly excluded are

- similar waste generated by other sectors;

- waste from the municipal sewage network and treatment, municipal construction and demolition waste.

In the following, the above definitions of municipal and household waste as provided by the OECD/Eurostat Joint Questionnaire will be followed.

The European Union keeps statistics of waste generated and treated by the Member States. Because of differences in nomenclature and statistical categories, these are not necessarily identical with the national figures. In Figure 3-2 the data for the year 2008 is shown.
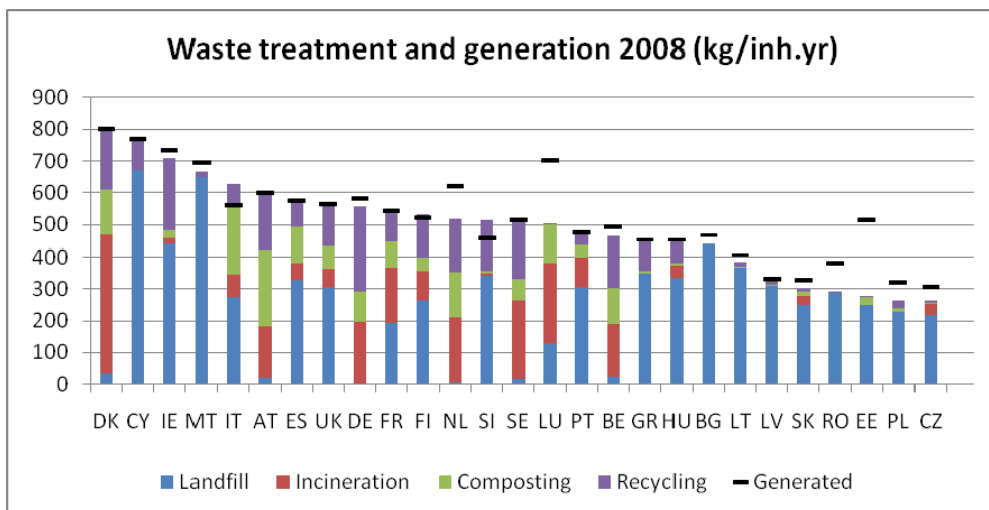
Figure 3-2. Generated and treated municipal waste in the European Union (EU27) in 2008 (Eurostat 2010a)

The above figure shows a clear division between the new and old Member States. In the former, most of the waste is disposed at landfills. The total amount generated, however, is significantly lower than in the old Member States. The old Member States show diversion rates from landfilling of up to 100%.

The total generation number does not always equal the sum of the provided treatment and disposal options. Reason for this deviation are

- exports and imports that are not always accounted for, e.g. most of Luxembourg's waste for recycling is exported and therefore does not show up in its recycling figure;
- some treatment options are not available in the data system. In some countries mechanical-biological treatment of waste is applied in varying levels. It is not clear whether the input and output flows of these facilities are included in one of the treatment options;
- different sources for generation and treatment data. This may lead to a different proportion of waste being attributed as municipal.

Nevertheless, these figures give a good indication of the recent state of municipal waste management in Europe. Figure 3-3 shows the development of the amounts generated, treated and disposed.
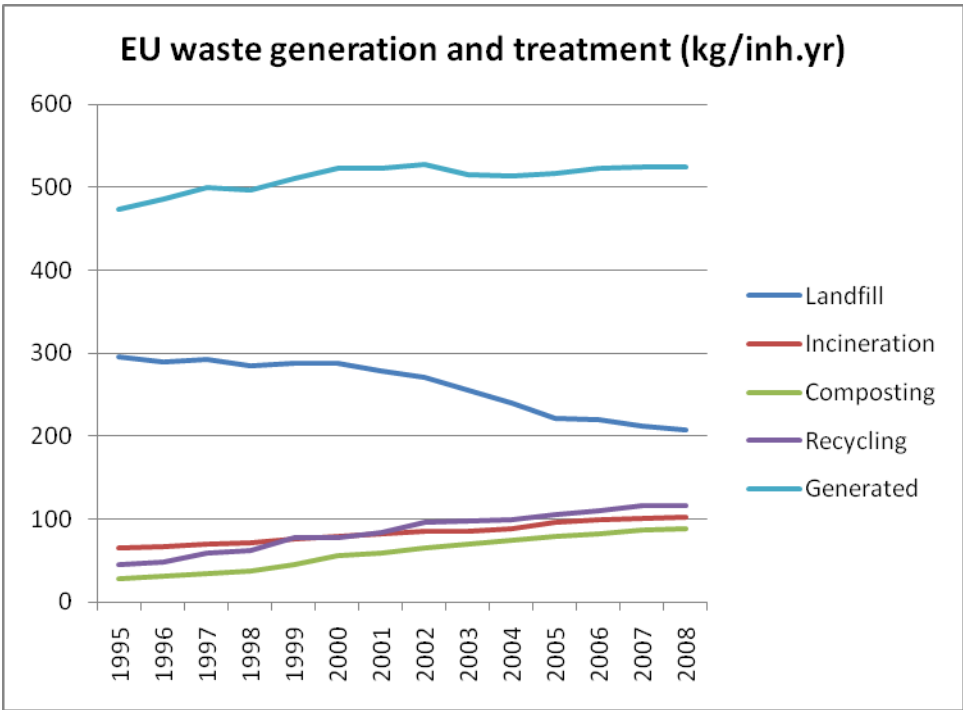
Figure 3-3. Development of generated and treated municipal waste in the European Union (EU27) 1995-2008 (Eurostat 2010a)

In the above figure, it can be seen that the average European generation of municipal waste lies just over 500 kg/inh.yr. This amount does not show an increasing tendency anymore. The amount of waste that was landfilled was reduced by a third in the considered 13-year period. Recycling, composting and incineration are at similar levels, whereas the growth rate of incineration is lower than for composting and recycling.

Especially for the waste under the heading 'recycling' there is a wide variety of waste fractions and treatment processes. In the following Table 3-1, the collection results for household waste in two example countries, Poland and The Netherlands, are presented[1]. In The Netherlands, apart from a number for the whole country, a division into **municipality classes** is provided by the national statistical bureau (CBS 2010):

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[1] It should be noted that whereas in some countries the amount of waste collected is assumed to be equal to the amount of waste generated. In other countries, e.g. in Poland, waste statistics do differentiate. Part of the waste is illegally disposed of in wild dumps and a fair share of the households (21% in 2009) does not have a contract with a collection company (GUS 2010). In The Netherlands, 5% of households do not get a waste tax bill.
It is not known whether this is a deliberate policy for low-income families (Agentschap NL 2010a)

I. very urban. Over 2.500 households per km^2. This applies normally only for cities over 100.000 inhabitants;
II. urban: 1.500 - 2.500 households per km^2;
III. intermediate: 1.000 - 1.500 households per km^2;
IV. rural: 500 - 1.000 households per km^2;
V. very rural. Less than 500 households per km^2.

The advantage of this division is that it incorporates, in a coarse manner, the type of housing or the social-economic status, which has an impact on the generation and source-separation of waste.

In Table 3-1, it becomes clear that the level of source separation in Poland is still far behind that of the level achieved in The Netherlands. This is especially true for biowaste (here for the year 2009, the 2008 figure was assumed) and paper. Glass, textiles and other fractions are currently at a level of 20-60% of that in The Netherlands, whereas plastics and metals show higher results. Despite the low level of source separation, however, the amount of collected residual waste is still lower in Poland than in The Netherlands.

For Poland no information is available on the share of bulky waste that is disposed of as residual waste. In the Netherlands the level of separately collected bulky waste is about 60% (varying from one third for the very urban municipalities to three quarters for the rural ones).

Considering the municipality classes in the Netherlands, it can be concluded that an increase in urbanity leads to a decrease in the source-separation of waste (only in the case of glass the effect is moderate), whereas the level of residual waste increases as well.

As the next step, it is interesting to see what still remains within the residual waste. For Poland, data is available for three different areas: cities (over 200.000 inhabitants), towns and rural areas. In order to match the available Dutch data, the municipality class I (very urban) was considered 'city', the classes II and III as 'towns' and the classes IV and V as 'rural'. This results in the residual waste composition in the year 2008 shown in Figure 3-4.

Table 3-1. Collected amounts of waste in Poland and The Netherlands in kg/inh.yr (GUS 2009, 2010; CBS 2010)

Waste fraction	Poland		NL total		NL I		NL II		NL III		NL IV		NL V	
	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009
Residual	169	167	241	235	317	307	245	238	227	220	202	195	211	207
Biowaste	1,1	0,0	79	79	15	18	70	73	92	92	106	105	118	120
Paper	1,8	2,1	69	65	39	37	65	62	74	72	81	78	88	83
Glass	3,6	4,1	21	21	18	17	21	21	22	22	23	23	23	22
Plastics	1,6	2,0	0,8	1,9	0,3	0,4	0,9	1,7	0,6	1,8	0,9	2,9	1,1	3,2
Metals	0,1	0,2	0,1	0,1	-	-	0,1	0,1	0,1	0,1	0,2	0,2	0,3	0,2
Textiles	1,0	0,9	4,2	4,0	2,7	2,4	4,1	3,9	4,4	4,2	5,1	4,9	4,8	4,4
other separate	1,4	2,5	4,1	3,2	2,0	1,0	3,2	3,3	4,1	4,1	5,2	5,3	4,1	4,1
Hazardous	0,03	0,00	1,3	1,3	1,0	1,0	1,3	1,3	1,4	1,4	1,4	1,4	1,4	1,5
WEEE	1,5	2,9	4,9	4,9	3,9	3,8	5,5	5,2	5,3	5,5	4,7	5,0	5,0	5,2
bulky waste	2,3	2,4	105	101	94	91	100	95	109	103	109	106	116	117
Total	**183**	**184**	**531**	**516**	**493**	**479**	**516**	**504**	**540**	**526**	**538**	**527**	**573**	**568**
Separation level	*7%*	*9%*	*47%*	*47%*	*23%*	*23%*	*44%*	*45%*	*51%*	*52%*	*56%*	*57%*	*58%*	*58%*
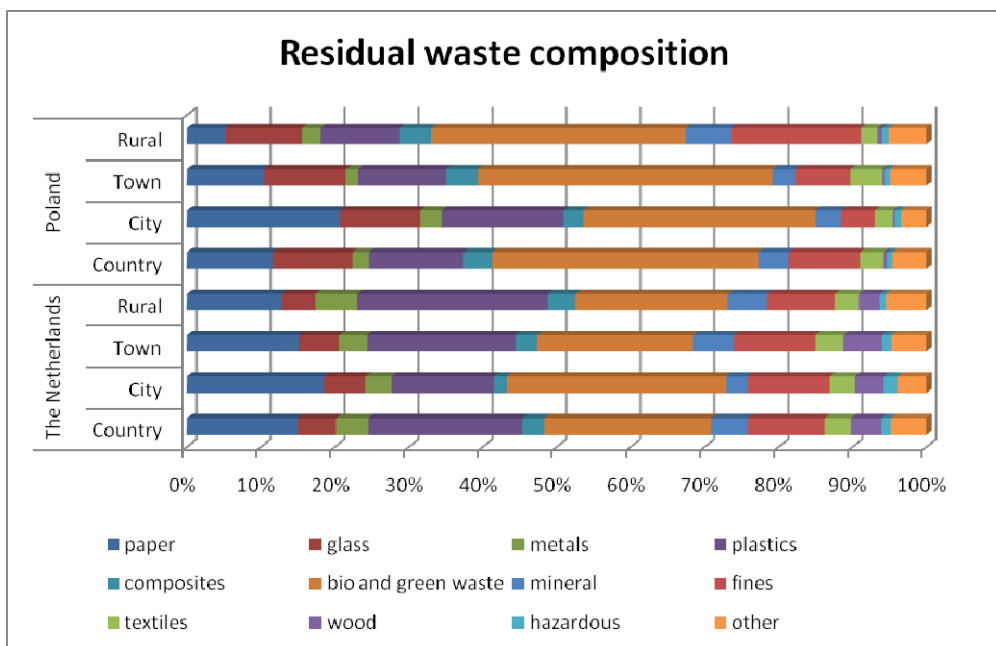
Figure 3-4. Residual waste composition[2] in Poland and The Netherlands in 2008 (Agentschap NL 2010, Szpadt 2010)

In the figure above, for both Poland and The Netherlands the average composition of residual waste is given for three different areas as well as for the country as a whole. Within Poland there is a striking difference in the share of paper and fines with the share of paper much higher in the urban areas compared to rural areas and the share of fines much higher in rural areas compared to urban areas. Apart from a general trend of higher paper generation in cities and better source separation in rural areas, this can be caused as well by the higher share of individual solid fuel heating systems on the countryside. Paper is frequently used as a fuel and the ashes show up in the fines fraction.

In The Netherlands it is peculiar that the share of plastics in the residual waste fraction is higher in rural areas. At the same time the collection results are also higher (see Table 3-1) which would indicate a significantly higher generation than in urban areas. Unlike other countries with advanced waste management systems, The Netherlands did not have a country-wide collection system for plastics or packaging materials. To meet the collection targets of the European Union, such a system has only very recently been introduced. In 2009, at the time

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[2] The fraction 'fines' consist for Poland of the fraction <10 mm, for The Netherlands <20mm.

of measurement, in three out of eleven sampling municipalities the separate collection of plastics was already introduced. This may have had an effect on the plastic amounts remaining in the residual waste.

Comparing the countries as a whole it can be observed that

- the paper content is somewhat higher in The Netherlands. This may be caused by higher generation levels (e.g. from old newspapers) and use as a fuel in Poland;
- the glass content is lower in The Netherlands. The separate collection from glass is well established for a long time already, reaching very high collection rates (of about 90%);
- plastics content is lower in Poland, whereas the biowaste content is higher. Poland shows higher collection results for plastics, whereas The Netherlands is clearly ahead in case of biodegradables; this leads to lower shared in remaining in the residual waste;
- the amount of metals and wood are very low in Poland. A reason for this may be the use as a fuel and the informal collection (waste pickers) and abundant presence of scrap metal shops.

4 Waste logistics

4.1 Introduction

A municipal waste management system can be, apart from the organizational aspects, divided into four (or three) stages:

- Temporary Storage of the waste
- Collection of the waste
- Transport of the waste to treatment or disposal (sometimes referred to in combination with the collection)
- Treatment and Disposal of the waste

Waste logistics comprises, after the waste has been generated, all following stages until the waste is finally delivered to treatment or disposal facilities. In Figure 4-1 a general overview of a municipal waste management system is shown with the logistical stages highlighted.
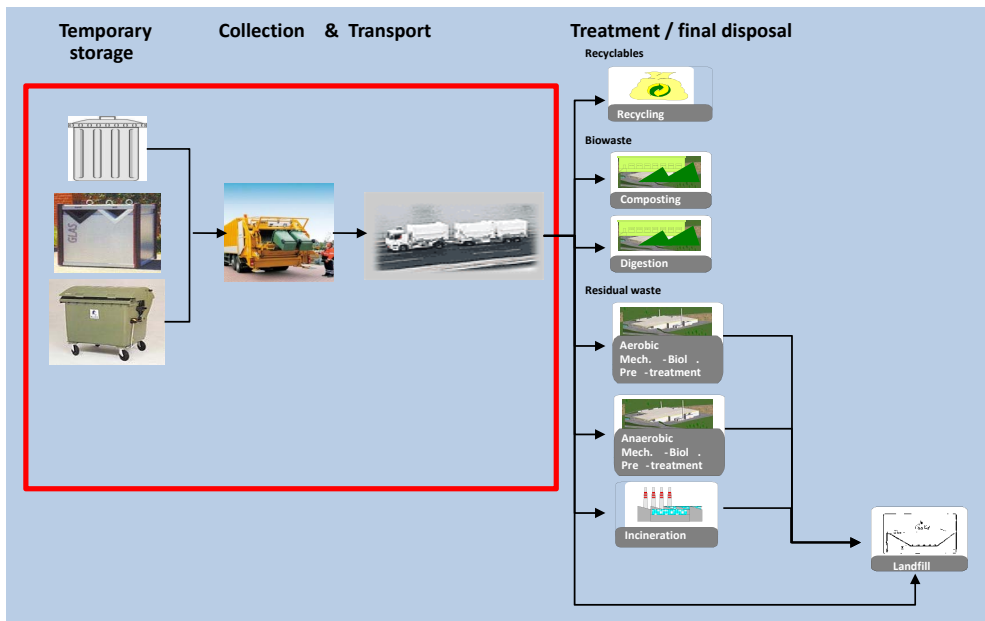
Figure 4-1. Simplified overview of a municipal waste management system (Den Boer 2007)

In this chapter the waste logistics stages up to treatment and disposal facilities are discussed. Products or wastes generated in these facilities generally are transported further. However, since these transports in their nature are often similar to the normal transport of bulk products, they are not considered here.

4.2 Temporary Storage

Temporary storage is the keeping of waste, after its generation, until its collection for recovery or disposal. Four different types of temporary storage can be distinguished:

- without containers
- with small containers <500 l
- with large containers >500 l
- underground containers

Temporary storage **without containers** can be observed for a variety of waste types. In such cases consumers do not have their own containers, nor are there common containers. The waste is either offered for collection in bags (e.g. residual/mixed waste, packaging waste (Gelbe Sack in Germany) or mixed dry recyclables (MDR)) or loose (garden waste, bundles paper, bulky waste or WEEE). For some waste flows the lacking of containers is rather exceptional (e.g. residual waste), for other flows it is the standard situation (e.g. bulky waste, WEEE).

Small containers (<500 l) are mostly used on an individual basis: one for every household. In the case of smaller multi-apartment houses they can be also commonly used by several families. For separately collected waste this is more often the case, e.g. biowaste at high rise buildings.

Individual containers bring an individual responsibility for the container. Results of separate collection of waste are better, both in amounts and in terms of separation quality, for individual containers. Small containers, being used individually, use private space. Especially when several fractions are collected separately in small containers, competition for the precious surface in a garden or garage may be fierce. An advantage of individual containers is the very limited walking distance to dispose of the waste, compared to commonly used containers. On the other hand, comfort may be limited because of the waste staying on the consumer's premises until collection. In systems with commonly used containers the waste can be disposed of 'out of the eye and out of the heart'.

Large containers (>500 l) are generally used commonly and thus also show a shared feeling of responsibility, resulting in worse results in separate collection.

Also street littering as a result of overfilled containers and bulky waste or bagged waste put aside the containers is clearly higher in the case of large containers. Large containers do not use the precious space of the consumers, but do stand in the public space, where they may be disturbing as well. A big advantage is that waste generators can dispose their waste out of their premises at all times.

Large containers are in use for virtually all kinds of waste, whereas small containers primarily are used for residual, biowaste and sometimes paper waste.

In Table 4-1 some technical aspects of the different container types are provided.

Table 4-1. Technical aspects of waste containers

Container	Sizes, liter	Wastes	Material	Pick-up system	Specialties
Small (<500 l)	60-80-120-**140**-**240**-380	**residual, bio,** paper	**HDPE,** galvan-ized steel	bar, diamond, grabber	odor filters, composting containers, locks, compartments (for two fractions)
Large (>500 l), wheeled	660-770-**1100**-2500-5000	**residual,** bio	**HDPE, galvan-ized steel**	bar, diamond	round or flat lids, (electronic) locks
Large (>500 l), wheel-less	up to 3000	**recyc-lables**	HDPE, **galvan-ized steel**	hook or mushroom	tailor made openings, 3 compartments for colored glass, sound isolation

An alternative for small and large containers is the use of **underground containers**. These consist of a maneuverable inner container, generally made of steel, which is placed in a fixed outer container, generally made of concrete. Some properties of underground containers are the following:

- expensive in investment costs. These include, apart from the inner and outer container, safety platform, insert unit, eventually electronic access and communication systems of substantial placement costs (groundwork, location finding, which may be complex, due to cables and pipes);
- for residual and biowaste, as well as for recyclables;
- large, up to 5 m^3, which leads to lower collection costs;

- less visually intrusive than over ground containers. Related to that, they are also less vulnerable to vandalism. For this reason they are often introduced to city centers, sometimes financed from budgets other than waste management (e.g. infrastructural renewal);
- have possibilities of automatic communication of filling rates, which enhances collection efficiency;
- can be attributed with user identification/access systems. It is easier to combine with Pay-As-You-Throw (PAYT) systems than over ground containers.

In Figure 4-2 an overview of an underground container is shown.

Figure 4-2. Schematic overview of a underground container (left) and the over ground insert unit (right) (Bammens 2010)

In the figure above (left), the separate parts of an underground container are shown. These are, from bottom upwards:

- outer container. Made of concrete, up to 6 tons for the large 5 m^3 variant. Can be equipped with a drainage system for dripping liquids from the inner container.
- safety floor. Moves up when the inner container is lifted out and prevents pedestrians from falling in.
- inner container. Made of steel, weighing up to 600 kg. They are opened by flow in the bottom. These can temporary store dripping water, which is

released together with the waste during the emptying. Sizes vary generally from 3 to 5 m^3. The inner container can be, as over ground containers, divided into different sections

- platform. Covers the inner container, to which and to the insert unit, it is attached. Can have a steel or rubber surface or be covered with e.g. street clinkers.
- insert unit. Can be equipped with a variety of openings. The residual waste variant has a lock, in which waste with the volume of one waste bag (about 80 l) can be entered at a time. For glass a double or triple unit may be applied. The inner container is lifted up for emptying by a hook (right) or mushroom (left) system, which is either installed on top of the insert unit or can be reached by opening the top of the insert unit.

4.3 Collection

4.3.1 Collection vehicles

A general definition of waste Collection is:

> "each activity, and especially gathering of waste, sorting, storage and/or mixing of waste in order to prepare it for the purpose of transport to the treatment or disposal site"

In Western countries, waste collection is generally conducted by specially designated **vehicles**. For compressible waste **compaction vehicles** are used. These can be a rotary drum compactor (in which the waste is placed in a rotating tube, compressing it by rotary movement) or a plate compactor (a moving plate pushing in the waste from a pre-chamber). Three types of vehicles, considering the loading technology can be differentiated:

- Rear loader. Can load both small and large containers as well as loose waste (e.g. bundled paper, bulky waste).
- Side loader / front-side loader. Is used for automatic loading, in which a maneuverable arm picks up the containers. It is mainly applied to small containers, not loose waste. The advantage of side loaders is that less personnel are needed for collecting. Automated collection implies, however, the properly putting to the street side of containers by inhabitants. Since the waste press is not mounted at the rear of the vehicle it is possible to have a transport container used instead of a fixed waste containing compartment. Thus, further transport by truck, train or

boat is possible without emptying the vehicle (instead the full transport container is exchanged by an empty one, see the next chapter).

- Front loader. Mainly used for bigger containers. Also for these vehicles transport containers can be mounted.

Apart from the above mentioned compaction vehicles that are mainly used for residual and biowaste and to a lesser extend for paper and packaging waste, vehicles with **cranes** can be operated. These can be equipped with a compaction unit as well and often they consist of simple open compartments or transport containers. Vehicles with cranes are often operated for the collection of glass and other recyclables. Both over ground and underground containers can be handled by hook or mushroom connections.

In water-rich areas with limited accessibility for garbage trucks, waste can be collected by **boat**. This is the case for instance in the city centre of Utrecht in The Netherlands or in Venice.

4.3.2 Collection systems

Collection systems can be divided into **curbside** and **bring** systems. In Figure 4-3 an overview of various gradations collection systems is shown.
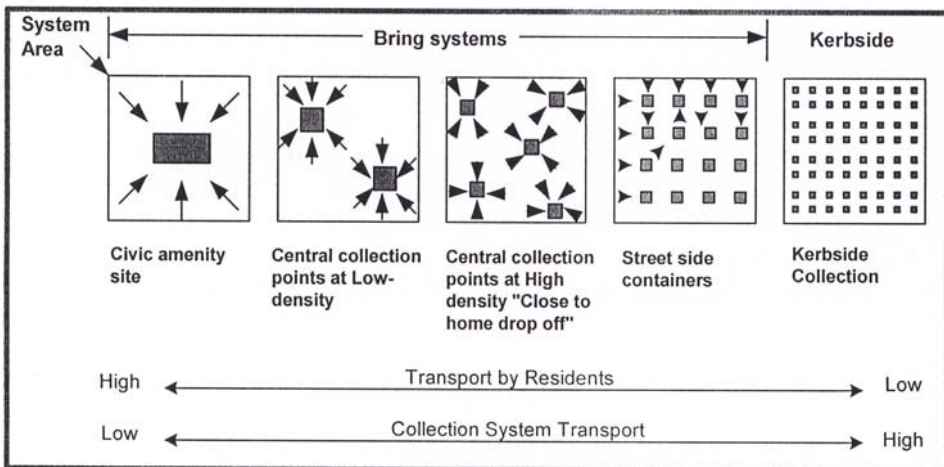
Figure 4-3. Collection systems of various densities (McDougall et al. 2001)

Curbside collection is mainly applied for residual and biowaste, sometimes also paper or packaging waste. It implies the use of individual containers or bags. Using curbside systems for source separated fractions (the fractions other than residual waste) lead to higher collection results in comparison to bring systems

(SenterNovem 2005, Zegwaard 2000). This is especially true for paper waste and textiles. For waste glass such a relation is disputed, here the strategic choice of the central collection point is important (e.g. close to shopping centers), rather than the density of the placed containers (SenterNovem 2005, Zegwaard 2000).

Commonly used containers are examples of **bring systems**. These systems are often applied for recyclables, but residual waste as well can be collected by bring systems, e.g. in case of high rise buildings or underground containers. In bring systems the households undertake a share of the total transport distance of the waste. The collection company is facing a smaller transport distance. Moreover, since mostly the containers in use in case of bring systems are larger, the number of stops per ton of collected waste is lower, thus making the collection more efficient, although there is more time needed for emptying a single large container than for a small container or bags.

4.3.3 Innovative systems

A waste management system in which the temporary storage and collection of waste are not clearly separated is the so-called **waste sewer**. In these systems, the waste is put in a chamber, either constructed within the resident's premises or outside. The chambers are emptied by opening them to a kind of sewer tubes, in which air is sucked out. In this way the waste is propelled to a central collection point, which may be a container, often with a press and an air cleaning system (stationary system) or to a collection vehicle (mobile system).

Waste sewers have large investment costs and are more easily implemented in newly developed areas. Currently implemented systems mostly are in areas with a limited accessibility for collection vehicles, such as airports, hospitals, shopping centers, but also in historic city centers. Examples can be found mainly in Nordic countries, Spain, Singapore, China, Malaysia, Dubai and Qatar (Envac 2010).

In Figure 4-3 an artist impression of a mobile system is shown.

4.3.4 Bulky waste

Bulky waste can be defined in many ways. The easiest and most practical definition of bulky waste is

> "all waste too large to fit in available containers for residual/mixed waste."

Figure 4-4. Waste sewer, mobile system (Envac 2010)

There are a number of options to collect bulky waste:

- **curbside collection**. Residents put their waste outside on the street from where it is picked up by the collection vehicle. This can be done on a regular basis, but more often routes are based on notification. The collection company/municipality creates a route based on the received notifications of bulky waste awaiting collection. The households that sent the notification get an announcement with the date of the collection. On that day they are allowed to put the bulky waste onto the street side.
- the bulky waste can, formally organized or in an informal manner, also be put **alongside commonly used containers.** In the case of a rear-loaded collection vehicle for the collection of residual or mixed waste, this type of bulky waste collection needs only little extra effort.
- central collection points with **large transport containers**. On a regular basis, e.g. twice per year large containers are placed at centrally located, well reachable points. After a few days the filled containers are picked up. This way of collection is easy to organize, but leads often to an overflow of the containers, which needs additional manpower and machinery to clean up.
- **civic amenity site**, **waste collection center, recycling center** are all synonyms for systems with specially designed sites where the waste is to be taken by the consumers. On these sites a wide variety of fractions can be disposed of separately, often directly into large transport containers. This leads to higher separation efficiencies than for the other systems. Costs for investment and personnel during opening hours on the other hand are higher.

All types of bulky waste collection are increasingly facing informal activities. Items of bulky waste that can still be used are taken by neighbors, people passing by, but also by so-called waste pickers or scavengers. The items are either for their own use, but especially in case of waste pickers, also traded as second-hand goods. For this reason they are often transported into countries or regions with lower economic standards (Obersteiner et al. 2010).

4.4 Transport

In municipal waste management the transport of waste encompasses a variety of pathways:

- the transport from the citizens to containers and collection site or directly to treatment and disposal plants;
- the transport of waste that was collected from households or containers to treatment and disposal plants or sites;
- the transport of residues from treatment plants to disposal plants or sites and
- the transport of products from treatment plants to customers.

In the following only the transport of collected wastes to treatment and disposal by collection companies or municipalities are considered. For this transport three main options are in use:

- transport by the **collection vehicle**. This is the most common transport form. The collection vehicles are unloaded directly at e.g. the incineration plant, landfill, composting plant or sorting site. No investments are needed in designated transfer stations or transport vehicles. At the same time, the collection crew and parts of the vehicle (press, soft- and hardware) during the transport are not doing what they are being paid for.
- transport by using **transfer stations**. At higher transport distances and waste amounts it becomes economically feasible to transfer the waste from the collection vehicle to designated transport vehicles (or rail or water transport). Transfer stations exist in various levels of complexity, from outdoor paved areas where the unloaded waste is filled into transport vehicles by shovels to in-house multi-modal complexes with press installations and air cleaning equipment. Often some sort of compaction is applied to the waste to enhance transport efficiency.
- transport by **intermodal transport systems**. Examples thereof are ACTS (Advanced Container Transport System) or MSTS (Multi Service Transport System). The difference from the aforementioned systems is that here the

transport of containers filled with waste is considered, not the transport of the waste itself. The use of these standardizing systems eases the transfer from collection to transport and from transport to treatment or disposal. In MSTS systems the collection vehicles are equipped with transport containers into which the collected waste is pressed. The full containers can be either directly transferred to transport vehicles or exchanged at a designated area (which can be a simple car park). For longer transport distances the containers can also be transferred to trains or ships. Waste delivered at treatment or disposal sites does not have to be unloaded to bunkers immediately; the containers constitute a kind of secondary temporary storage. In this way long waiting times, which may occur for collection vehicles unloading at incineration plants, are avoided.

4.5 Waste logistics management

4.5.1 Container management

Waste containers, also small containers, are generally not owned by their users, but by the local government or the collection company. This leads to a uniform streetscape and enables the collector to identify whether or not the waste provider is registered. This control can be as simple as the presence of the container (only handed out to registered waste providers), which is an advantage over bagged waste collection. Container management systems can identify specific users, e.g. by transponders fitted into the containers or bar code stickers. This enables the keeping of black-lists of defaulters, which can then be automatically blocked by the collection vehicle.

4.5.2 Collection logistics planning

For planning the collection routes **optimization software** is widely available. Yet, route-planning is often at least partly based on experience and personal knowledge of the local situation. Planned routes can be entered directly into GPS-software. The use of GPS systems is widely applied in Europe because is also allows the management to control the drivers (e.g. for too long breaks, private business with the vehicle, illegal dumping or collection, etc.).

To plan the collection routes and also the economics, two indicators are of special interest: the production per truck per day, both in number of containers and in waste amounts or collection trips covered. This is combined with the costs per truck per day, in which both the complete vehicle costs (investment and operation) and the personnel costs (driver and loaders) are contained.

Apart from what can be physically or technically collected on a daily basis, the **well-being** of the collection crew should be considered. Waste collectors face physical, psychological and physiological burdens and are exposed to chemical and biological hazards. Waste collection is also risky in terms of accidents. An example consideration for the well being of the collection crew is the so-called Dutch P-90 Norms. This norm stands for such working conditions that 90% of the workers can continue to do the work in good health for a period of 40 years.

The application of such norms means in practice that the collectors are only allowed to collect a maximum daily amount of waste, number of containers and time. This maximum is dependent on the workers' age and the type of temporary storage (e.g. more small containers allowed, but the total amount of collected waste is larger for large containers). Results from the application of waste collection related working norms are

- young working force (they are allowed to have a higher productivity than older employees);
- task shifts (morning collection, afternoon other task);
- automated collection (e.g. side loaders);
- containerization (the introduction of containers instead of bagged waste collection).

5 Dismantling and sorting

Dismantling is the operation of manual or mechanical decomposition of used objects and elements such as ELV, used electric electronic equipment (WEEE), furniture, buildings and constructions, etc. This process is the prerequisite for further detailed treatment of separated elements as spare parts and used materials for recycling and disposal purposes. Some EU directives regulate operations of dismantling and further recycling for ELV, WEEE, ships and other waste materials such as bulky wastes.

Sorting is the operation of division of mixed waste in various materials or removing impurities from separately collected waste streams. It is an inherent component in the life cycle of waste.

5.1 Sorting of packaging and municipal wastes

Wastes are collected in a mixed form as mixed municipal waste or collected separately as one waste material or as a mixture of defined waste streams. The separation of particular components of a waste mixture is one of the main aspects of almost every waste recovery facility. Material Recycling Facilities (MRF) are necessary when waste is collected in a mixed form.

MRF and sorting plants can receive various waste streams:
- mixed, unsorted fractions of municipal waste,
- mixed recyclables collected separately in a waste generation place,
- specific fractions of waste (e.g. from the Green Dot, light mixed packaging waste) collected separately,
- separately collected individual material fractions (paper, plastics etc.),
- commercial waste similar to municipal waste,
- waste from construction and demolition (C&D).

Due to this variety of waste material composition, there is no standard design of sorting plants. One can distinguish lines with simplified concepts and the domination of manual sorting and more complicated lines fully mechanized and automated equipped with various devices for waste treatment and sorting. Technical solutions of sorting plants depend on the waste streams to be sorted, capacity of the plant and the required quality of the sorted materials defined by the recyclers and other material users. Figures 5-1 to 5-4 present various solutions of sorting lines within a MRF.

Unit operations applied in sorting lines for separately collected and mixed wastes are designed for

- **modification of physical properties** of waste in order to facilitate the sorting of selected materials,
- **separation of specific components** and impurities from the waste stream,
- **treatment** of separated material before further usage.

Several appliances and machines are used for the separation of one or more materials from the waste stream. Their selection depends of many factors such as capacity, cost, location, environmental impact, financial conditions, etc. There are also many combinations and concepts of using these appliances as technological sorting lines and complete installations.
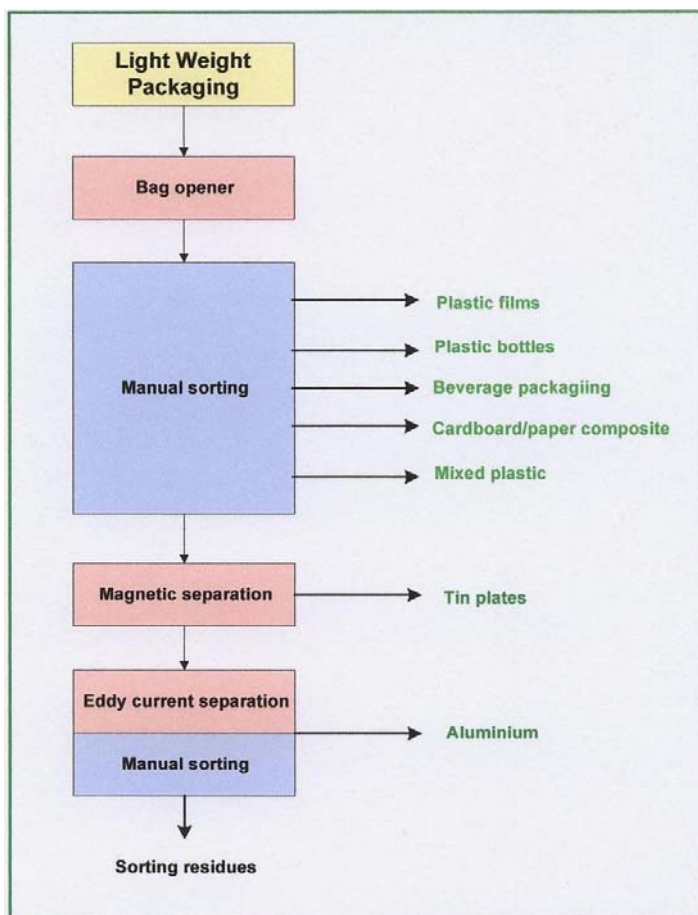
Figure 5-1. Example of a simple sorting line for mixed light weight packaging wastes (Kranert, 2004)

Mechanical treatment is used for the liberation of waste from bags, loosening the structure of waste, dividing it into various granulometric and material fractions and the separation of impurities and interfering elements. Depending on the place of treatment, preliminary treatment and fine (secondary) treatment are distinguished.

Preliminary treatment includes
• shredding (size reduction),
• screening (size separation),
• mixing,
• magnetic separation,
• separation of particular materials using advanced technologies,
• manual separation.

The fine treatment is carried out after completing the main waste treatment processes such as biological stabilization, composting, anaerobic digestion, etc. The extent of this treatment depends on the range of preliminary treatment as well as the types of wastes and technology of further treatment of sorted fractions. The main goal of fine treatment is the separation of fractions of defined granulometry and the removing of impurities according to the demands of the receiver and quality requirements, e.g. compost standards or other regulations.
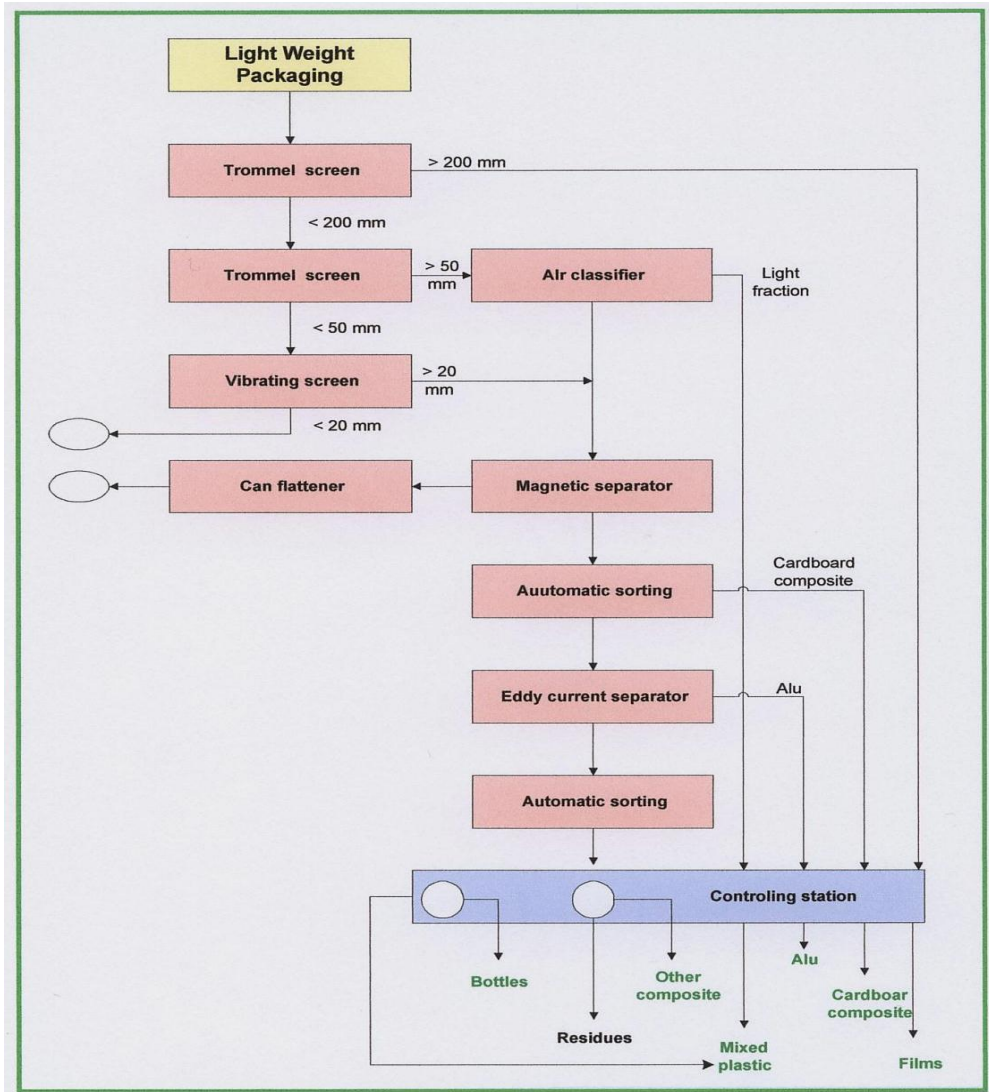
Figure 5-2. Example of a state-of-the-art sorting line for mixed light weight packaging wastes (Kranert, 2004)
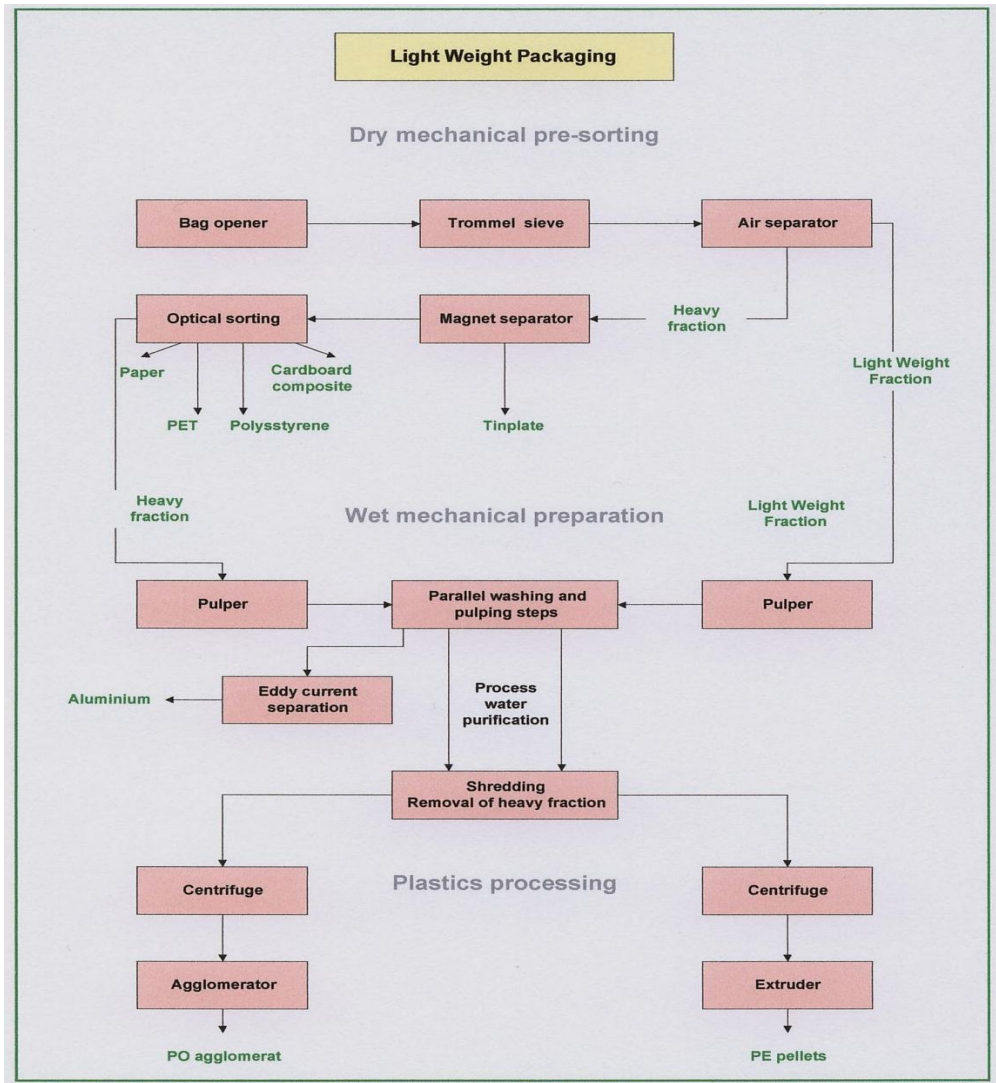
Figure 5-3. Example of a fully automated sorting line for mixed light weight packaging wastes (Kranert, 2004)
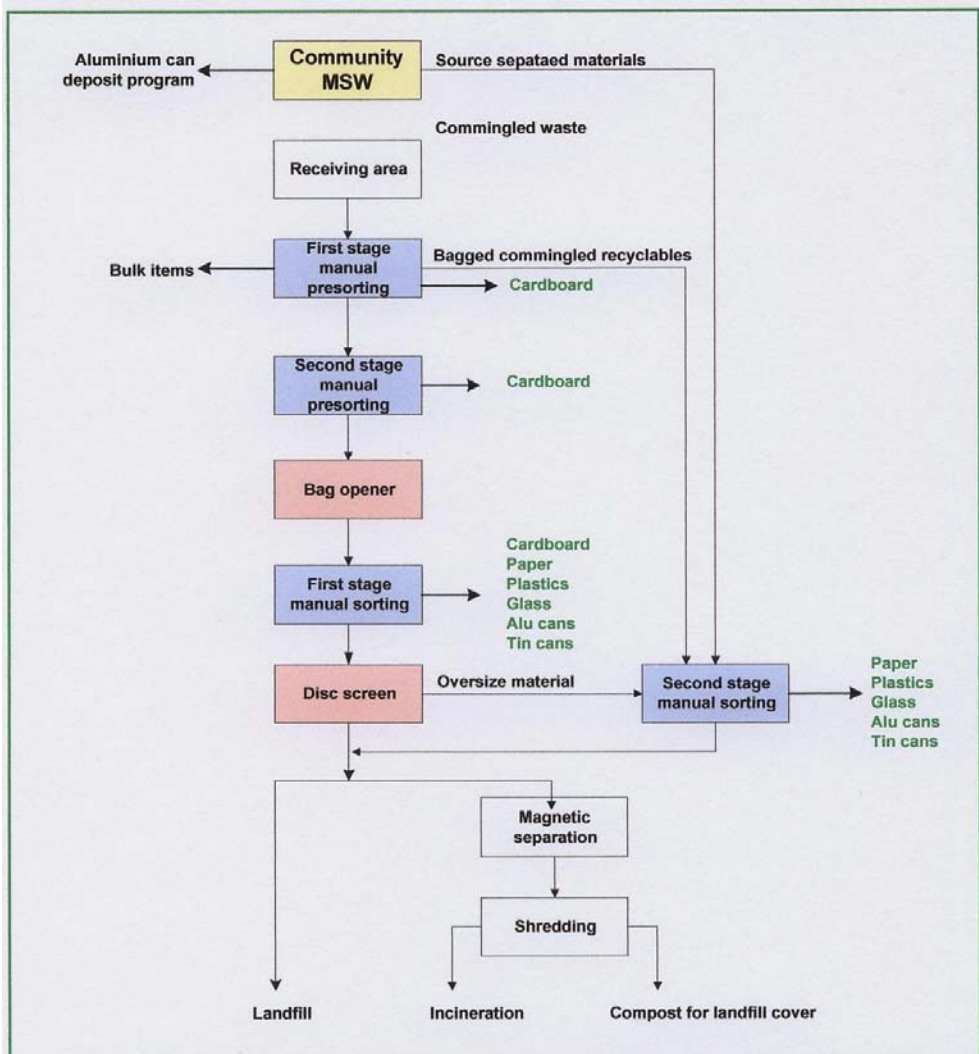
Figure 5-4. Sorting line for mixed municipal waste (Kranert, 2004)

5.2 Sorting of construction & demolition wastes

Typical solutions of C&D waste management comprises

1. Separate collection and temporary storage at the C&D location
2a. Delivery to the C&D waste recycling plant or
2.b. Sorting and treatment *in situ* by a mobile plant (installation)

C&D waste is a mixture of various materials (Figure 5-5).
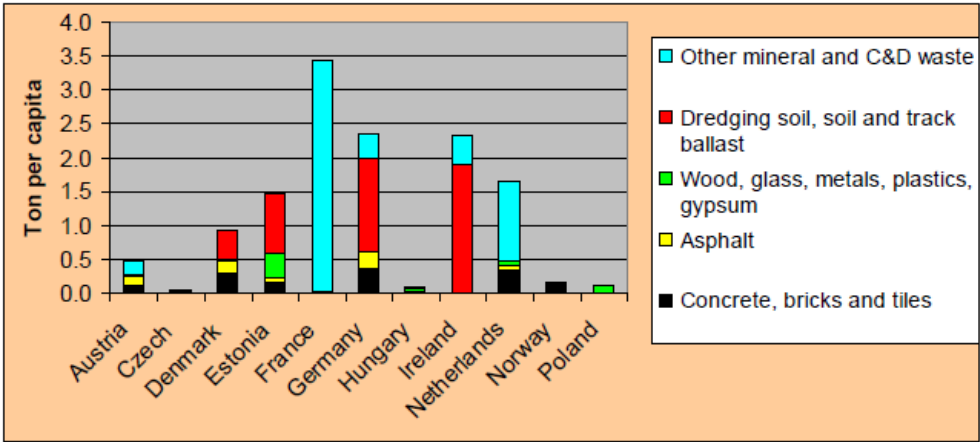
Figure 5-5. Composition of C&D waste in selected countries (Fisher, 2010)
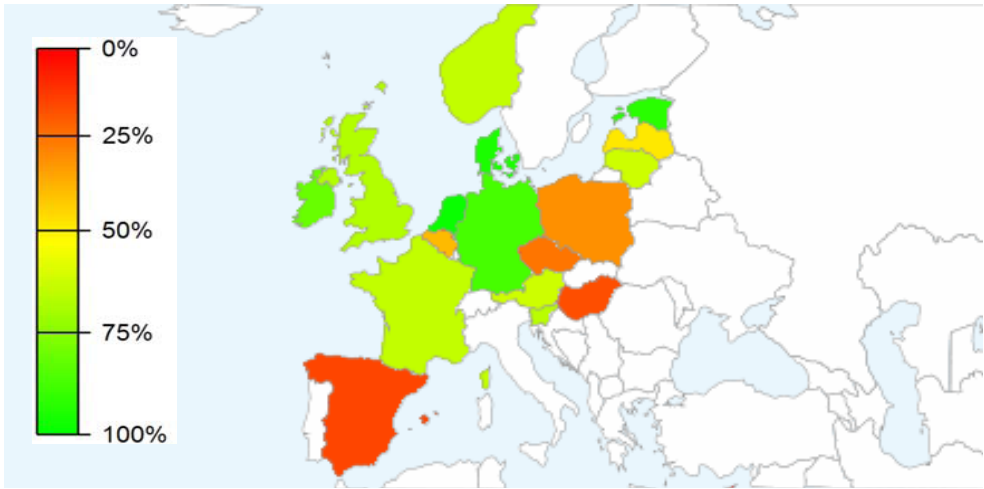
Figure 5-6. Efficiency of recycling C&D wastes in selected European countries (Fischer, 2010)

The present range of recycling C&D waste in Europe differs significantly among various countries (Figure 5-6). A typical sorting line for C&D waste includes

- primary manual sorting,
- magnetic separation,
- crushing of mineral materials and screening of defined granulometric fractions.

The main waste and product streams consist of

- separately collected and/or sorted out wood waste (for energy recovery)
- separately collected and/or sorted out metals (for recycling)
- aggregate produced by crushing and screening concrete waste (as a substitute material for recycling)
- other waste materials (fine fractions, stones, impurities etc.) for recovery or landfilling

6   Biological treatment of biowaste

This chapter is concerned with the biological treatment of separately collected biodegradable waste. The largest stream of biodegradable waste is "biowaste", which according to the definition of the Waste Framework Directive consists of garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants. Garden waste and park waste are often referred to as **garden waste** (alternatively - green waste), while the other biowastes are referred to as **kitchen waste**.

Biowaste is a putrescible, generally wet waste. Green waste includes usually 50-60% water and more wood (lignocellulosis), kitchen waste contains no wood, but up to 80% water. There are two biological technologies for treating separately collected biowaste: composting and anaerobic digestion. Both composting and anaerobic digestion may be classified as recycling when compost (or digestate) is used on land or for the production of growing media. If no such use is envisaged it should be classified as pre-treatment before landfilling or incineration. In addition, anaerobic digestion (producing biogas for energy purposes) should be seen as energy recovery (COM/2008/0811).

The main objectives of biological waste treatment include
- Reducing the mass and volume of waste and its moisture content
- Stabilization of organic matter destruction of biodegradable substances with
  - ✓ generation of simple oxidized or reduced products (as gases or liquids – CO_2, CH_4, H_2O)
  - ✓ evolution of heat energy
  - ✓ transformation of remaining organic substances in organic matter relatively stable in the environment (but undergoing further slow biodegradation without any nuisances for the people and environment),
- Higienization – pathogen destruction (partly or advanced), decreasing the population of pathogens in compost,
- Generation of usable products as biogas, compost, anaerobically digested organic matter.

6.1 Biowaste management in the EU

The total annual arising of biowaste in the EU is estimated at 76,5-102 million tones (Mt) kitchen and garden waste included in mixed municipal solid waste and up to 37 Mt from the food and drink industry. The overall potential for separately collected biowaste is estimated at up to 150 kg/inhabitant per year, including kitchen and garden waste from households, park and garden waste from public estates and waste from the food industry (80 Mt for EU27). Only about 30% of this potential (24 Mt) is currently collected separately and treated biologically. The total production of compost was 13,2 Mt in 2005. Most of it was produced from biowaste (4,8 Mt) and green waste (5,7 Mt) with the rest from sewage sludge (1,4 Mt) and mixed waste (1,4 Mt). The potential of compost production from most valuable inputs (biowaste and green waste) is estimated at 35 to 40 Mt. Given the low separately collected quantities, in the EU biowaste constitutes usually between 30% and 40% (but ranges from 18% up to 60%) of MSW, most of which is treated by options ranking low in the waste hierarchy. On average, 41% of MSW is landfilled, while in some Member States (e.g. PL, LT) this percentage exceeds 90%. However, as a result of national policies and the Landfill Directive which requires the diversion of biowaste from landfills, the EU average amount of landfilled MSW has dropped from 288 to 213 kg/capita per year (from 55 to 41%) since the year 2000 (COM/2008/0811).

Another biodegradable waste stream which can be treated together with the biowaste is paper and cardboard. However, according to the EU waste policy material recycling of paper and cardboard into the original material is preferred over its biological treatment.

Figure 6-1 shows the separate collection rates of biowaste in a few Member States.

Since 1995 the total amount per capita of recycled biowaste has increased in most of the old EU Member States and Norway, but Figure 6-2 and Figure 6-3 show huge differences between States in recycling levels (ETP/SCP 2009).

Spain recycles only 10 kilograms of biowaste per capita, while Denmark and Luxembourg each recycle about 130 kilograms of bio waste per capita. Figure 6-2 and Figure 6-3 also show for some countries the differentiation of recycled biowaste into kitchen waste and garden waste. This differentiation shows that the countries with the highest total amount of recycled biowaste, i.e. Denmark and Luxembourg, also have the highest amount of recycled garden waste per capita (80 to 120 kilograms). However, these two countries have a much lower level of recycled bio kitchen waste (18 to 42 kilograms).
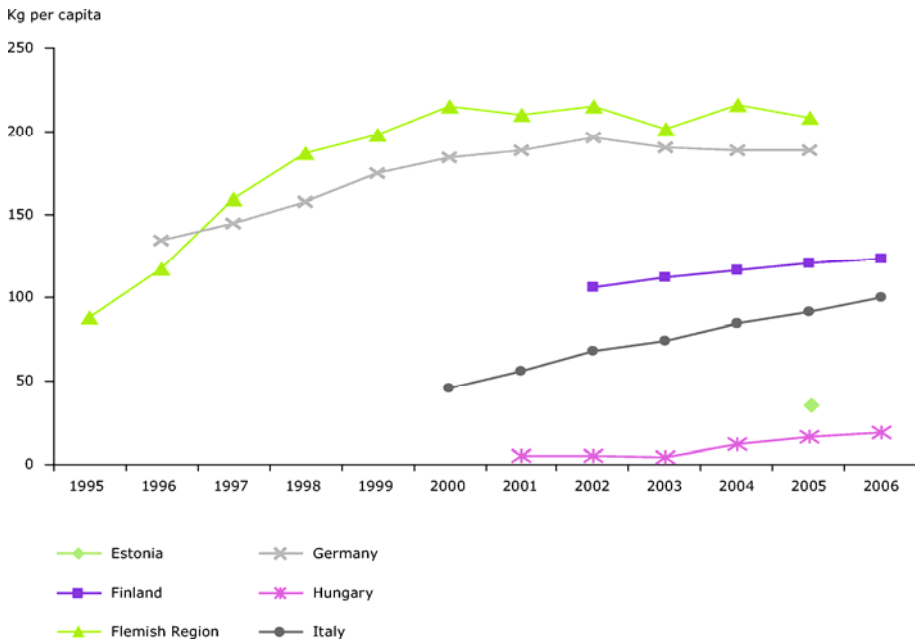
Figure 6-1. Separate collection of biodegradable municipal waste fractions (mainly paper and cardboard, packaging waste, and food and garden waste) [3] **(EEA 2009)**
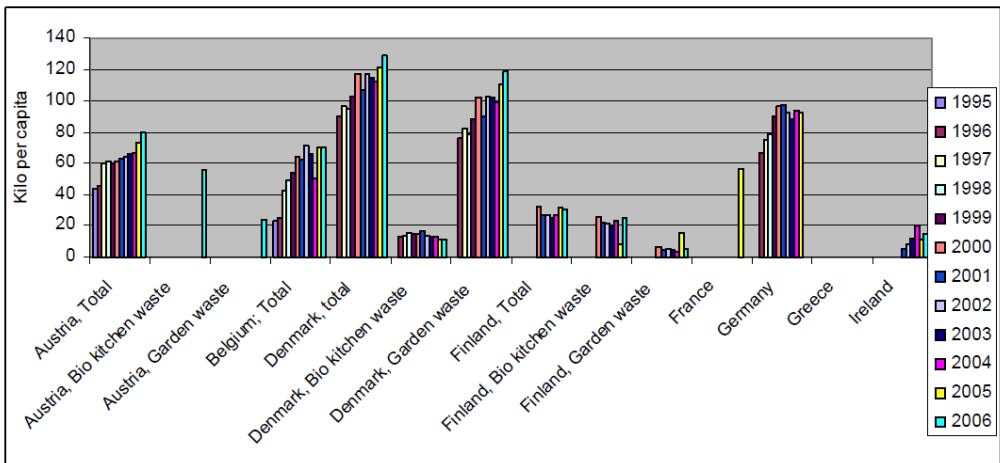
Figure 6-2 Development in the recycling of biowaste per capita in the old EU Member States and Norway. (Source: ETC/SCP 2009)

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[3] Figures for Italy 2002–2006 include food and garden waste, paper, textiles, wood packaging and bulky wooden waste.
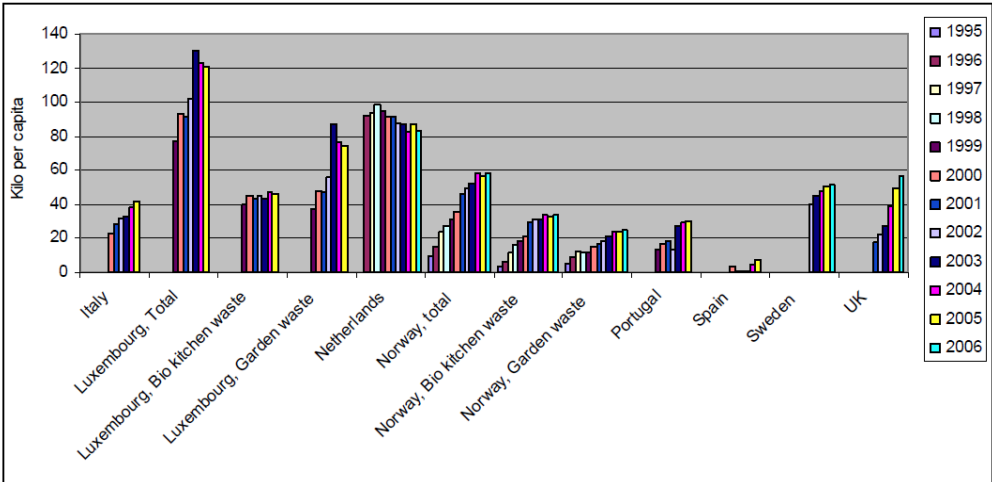
Figure 6-3 Development in the recycling of biowaste per capita in the old EU Member States and Norway. (Source: ETC/SCP 2009)

It is normally easier to recycle garden waste than kitchen waste since kitchen waste often can be polluted by non-biodegradable elements. On the other hand, kitchen waste includes the potential for making both bio gas and compost, whereas garden waste cannot be used for anaerobic digestion (bio gas) and is only able to be composted.

Figure 6-4 shows the development of biowaste recycling in the new EU Member States. In general, there has been little increase and apart from Estonia the level is quite low (under 10 kilograms). In the Czech Republic and Slovenia it is almost exclusively only garden waste that is recycled, whereas there is more recycling of bio kitchen waste in Estonia. Similarly, as in the old EU-Member States, there seems to be great potential in increasing the recycling of total biowaste in the new EU Member States (ETP/SCP 2009).

6.2 Composting

'Composting' means the autothermic and thermophilic biological decomposition of separately collected biowaste in the presence of oxygen and under controlled conditions by the action of micro- and macro-organisms in order to produce compost[4].

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[4] Definition according to the Working Document (2nd draft) on Biological Treatment of Biowaste (so called "Biowaste Directive"), by the European Commission Directorate-General Environment, Brussels 2001.
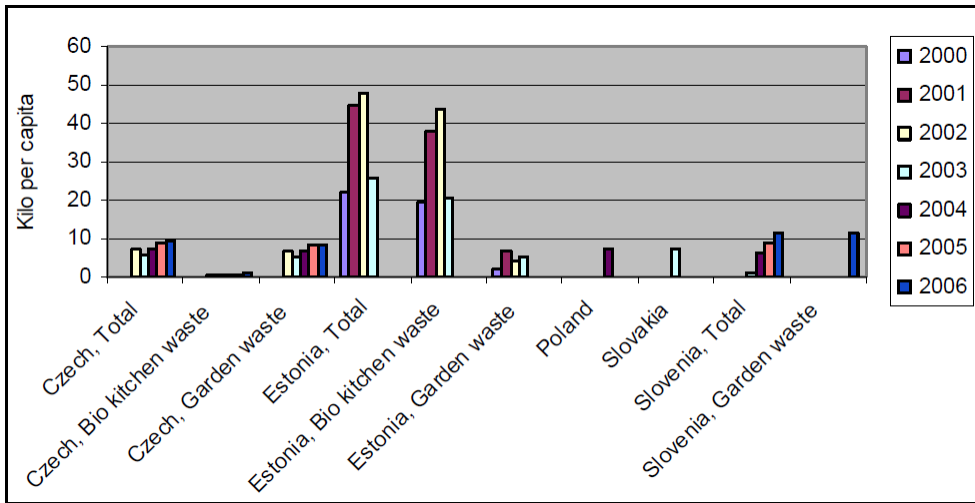
Figure 6-4 Development in the recycling of biowaste per capita in the new EU Member States (Source: ETC/SCP, 2009)

Thus, biowaste is composted with the objective of returning the waste to the plant production cycle as agricultural fertilizer and/or as a soil improver. This is done by converting putrescible organics to a stabilized form, at the same time assuring the destruction of organics pathogenic to humans. Composting is the most common biological treatment option (comprising some 95% of current biological treatment operations) (COM/2008/0811).

6.2.1 Characteristics of the input material

Biowaste is composed of a mixture of organic compounds, such as

- Carbohydrates, e.g. monosaccharides: glucose and fructose ($C_6H_{12}O_6$), disaccharides: sucrose and maltose ($C_{12}H_{22}O_{11}$), polysaccharides: cellulose ($C_6H_{10}O_5$)n, hemicelluloses (contains many different sugar monomers), pectin (contained in the cell walls of plants) and starch ($C_6H_{10}O_5$)$_n$
- Proteins,
- Fats, oils and wax,
- lignin.

Table 6-1 contains general chemical formulas of various organic compounds.

Table 6-1 General chemical composition of various organic compounds (Haug 1993)

Biowaste component		General formula
Protein		$C_{16}H_{24}O_5N_4$
Carbohydrates		$(C_6H_{10}O_5)_n$
Fat and oil		$C_{50}H_{90}O_6$
Wastewater sludge		
	Primary	$C_{22}H_{39}O_{10}N$
	Combined	$C_{10}H_{19}O_3N$
Biodegradable fraction of MSW		$C_{64}H_{104}O_{37}N$
		$C_{99}H_{148}O_{186}N$
Wood		$C_{295}H_{420}O_{186}N$
Grass		$C_{23}H_{38}O_{17}N$
Vegetables waste		$C_{16}H_{37}O_8N$
Bacteria		$C_5H_7O_2N$
Fungi		$C_{10}H_{17}O_6N$

Biodegradable materials, such as wastewater sludge, paper and partly MSW constitute a mixture of various organic and inorganic compounds. The average composition of a few examples of waste and their maximum decomposition rate in the composting process is presented in Table 6-2. The decomposition rate indicates the proportion of each substance which is transformed to inorganic substances with the remaining part forming humic matter.

Table 6-2 Characterization of waste input (Bidlingmeier 1998)

	MSW	Wastewater sludge	News-paper	Color news-paper	Decompo-sition rate
Ash	41	63	12,3	4,0	-
Nitrogen	1	3,4	0,3	0,1	-
TOC	36	26	49	45	50
Lipids	0,9	2,5	1,5	0,6	40
Wax	0,8	2,2	1,4	1,1	70
Sugars	2,3	2,5	1,9	0,2	70
Hemi-cellulose	13,0	3,2	13,0	11,8	70
Cellulose	29,0	4,5	64,4	31,6	50 - 90
Lignin, etc.	9,7	21	23,5	31,0	-
Proteins	2,3	1	-	-	50

6.2.2 Introduction to the composting process

Composting is an aerobic process in which organic matter is decomposed by microorganisms. Organic carbon is a predominant element of all organic substances contained in biowaste and is used by microorganisms as an energy source in the respiration process.

A major fraction of the organic carbon will be released as CO_2 to the atmosphere while the other part is assimilated by the biomass of the microorganism. The composting process is equivalent to the naturally occurring decomposition of biomass in soil (see Figure 6-5). The microorganisms responsible for the decomposition of organic matter are the so called primary consumers. These are organisms that take their nutrients directly from raw organic matter in compost. Primary consumers are mainly bacteria, actinomycetes and fungi which secrete enzymes to break down organic residues and absorb the nutrients released.
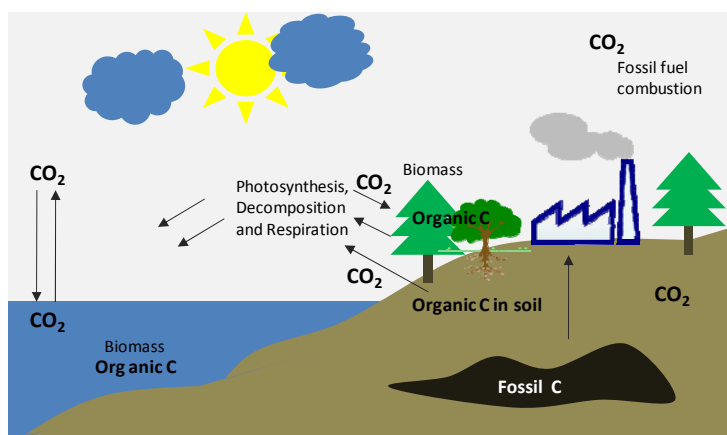
Figure 6-5. Carbon cycle in the ecosystem

Biowaste comprises several biodegradable organic and inorganic components (see Tables 6-1 and 6-2). Different fractions of the biodegradable organic components eventually mineralize to CO_2 and H_2O at different rates. A simplified metabolic pathway of carbon during aerobic degradation of lignocellulosic matter is suggested in Figure 6-6. Actual pathways can be rather complex and some, such as the humification pathway, are not yet well understood. Hydrolysis of the solid matrix is the first step towards the eventual substrate mineralization to carbon

dioxide and water. Cellulose decomposes down to glucose while other water soluble sugars (e.g. mannose, xylose and glucose) are the main hydrolysis products of hemicellulose. Additional water-soluble compounds can be amino acids, various sugars, alcohols, other types of acids which are either initially present or appear after the degradation of complex starting materials. Phenolic acids, produced during the partial degradation of lignin and humic matter (Stevenson 1994), are also water soluble compounds. The final carbon sinks during composting are carbon dioxide and humic matter. Humic matter is an almost stable organic polymer that mineralizes at minimal rates (Komilis 2006).
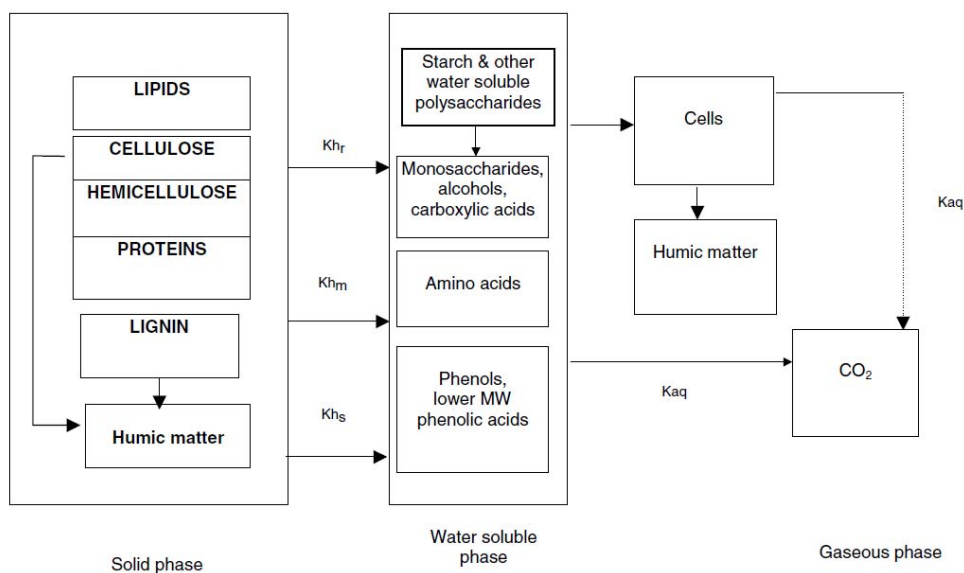
Figure 6-6. Simplified carbon degradation pathway during solid waste composting (Komilis 2006)

The general equation of the composting process can be summarized as follows:

$$\text{Organic substance} + O_2 + \text{biogenic substance} =$$
$$\text{biomass} + \text{undegraded organic substance} + CO_2 + H_2O + NH_3 + SO_4^{2-}$$
$$+ \text{heat energy}$$
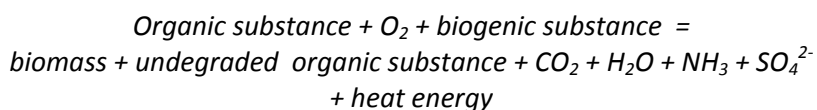
The biogenic substances are nitrogen, phosphorous, potassium and additional substances which are necessary for bacterial growth. Undegraded organic substances consist of humic matter (i.e. organic substances which are in a relatively stable, amorphous state and organic matter which is slowly

biodegradable) as well as intermediate products of the decomposition process, including various volatile organics responsible for odor nuisances in the process. In Figure 6-7 an exemplary mass balance of a composting process is presented. It can be seen that due to composting, a significant reduction of the mass of biowaste can be achieved (here approximately 63%). However, it should be noted that only 10% is due to the loss of the organic dry mass while the remaining 53% is due to the water evaporation. The main product of the process is compost with a water content of approximately 30%.
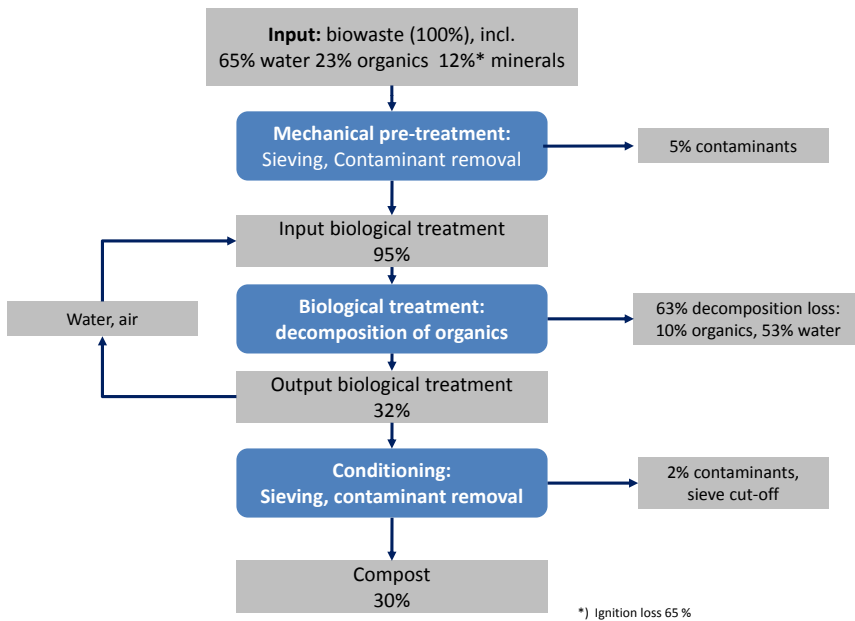
Figure 6-7 Material balance of the composting process (adapted from Bidlingmaier 2002)

6.2.3 Composting process parameters

The main factors influencing the kinetics and efficiency of the composting process are
- Composition of the composting mass:
  - biodegradable matter content (the higher the better),

- moisture content (40-70 %, optimum 50-60 %),
- availability of biogenic substances: C/N ratio (optimum 20-30), C/P ratio (optimum 100),
- absence of hazardous substances in input materials (inhibitors)
- pH (from 4,5 to 9,5 ; optimum 6,5),
- Presence of microorganisms (bacteria and fungi)
- Proper granulometry of waste (particle size 20 - 40 mm),
- Proper structure of composting mass:
  - free air porosity (> 30% of porosity) and
  - bulk density (less than 500-600 kg/m3),
- Effective mixing and aeration,
- Temperature:
  - optimum 55° C to maximize biodegradation rate and
  - 65° C to maximize hygienization – sanitization, pathogen destruction.

In the following, the influence of these factors is described in more detail.

Composition of the composting mass

The input to the composting plant strongly depends on the collection scheme for the biodegradable waste. Based on the collection model, the following composting inputs can be differentiated:

- **Biofraction (biowaste),** mainly kitchen waste, food waste (e.g. vegetables, tea and coffee residues from filters), polluted packaging paper – in most cases this fraction also contains garden waste added by residents of detached houses (grass, residues of vegetables, flowers etc.),
- **Green waste,** originating mainly from municipal green areas (parks, lawns), cemeteries – this waste consists not only of grass and flower residues, but also of leaves, branches and trunks of shrubs and trees,
- **Wet fraction**, separately collected waste materials not classified as the dry fraction in a two-bin collection system. This fraction consists of a mixture of kitchen waste and other waste not suitable for recycling.

Biodegradable matter content: Generally speaking, the higher the biodegradable matter content, the better for the composting process. The minimum content of biodegradable organic dry matter is 40% DM to achieve the process conditions assuring i) reaching the compost hygienization temperature, ii) minimum content of organic matter in product compost above 20% dm, and iii) effectiveness of final drying of compost to a moisture content below 35% of mass.

Separately collected biowaste and green waste are the most suitable fraction for composting due to their low contamination and high content of biologically degradable matter. In Table 6-3 the average characteristics of separately collected

kitchen waste and garden waste is provided. The total organic dry matter content in both types of waste is very high amounting to 87% and 84% of the dry matter for kitchen and garden waste, respectively, of which 100% is biodegradable. This is the amount of waste that undergoes biological transformation to either CO_2 and H_2O or to humus. The biologically degradable matter can be further subdivided into fast, medium and slowly degradable matter. Slowly degradable materials, such as cellulose and proteins, will only partly decompose to CO_2 and H_2O in a composting plant; thus, as presented in the mass balance of a composting plant in Figure 6-7, less than 50% of the organic matter is lost while the remaining fraction remains in the compost.

Table 6-3. Characteristics of separately collected biowaste (Vogt et al. 2002, Fricke et al. 2002c)

Parameter	Unit	Kitchen waste	Garden waste
Water content (WC)	%	55,0	57,0
Dry matter (DM)	%	45,0	43,0
Organic Dry Matter (ODM)	%DM	87,0	84,0
Biologically degradable ODM (bioODM)	%ODM	100	100
Organic Carbon	%ODM	51,9	49,8
Nitrogen (total)	%DM	1,7	1,2
Phosphorus (total)	%DM	0,4	0,5
Potassium (total)	%DM	0,9	1,5
Magnesium (total)	%DM	0,8	0,5
Calcium (total)	%DM	2,2	4,4
Cadmium	mg/kg DM	0,1	0,3
Chromium	mg/kg DM	1,8	4,6
Copper	mg/kg DM	9,2	0,1
Mercury	mg/kg DM	0,004	0,2
Nickel	mg/kg DM	1,3	3,7
Lead	mg/kg DM	2,6	4,8
Zinc	mg/kg DM	30,6	60,0

DM – dry mass, ODM – organic dry mass

Moisture content: The role of water in compost is to transport substrates and products of transformation processes. Only dissolved substances are available for microorganisms. The acceptable range for the initial water content in compost material is **40-70% with the optimum being 50-60%.** With a water content lower than 20%, biological processes are not maintained. With a water content lower than 40%, substrates are hardly available for microorganisms leading to process inhibition and slowing down the rate of transformations. In a very moist waste

with a water content above 70%, anaerobic zones are formed leading to the inhibition of the composting process. At the optimum moisture content the highest microbial activity has been observed.

The forced aeration and agitation of compost mass decreases its moisture content. Thus, additional watering of the composted mass is required to maintain the appropriate moisture level.

Availability of biogenic substances: The role of organic carbon in microbial growth is very important. It is the energy source for primary consumers and the basic building block of microbial cells (more than 50 %). However, the availability of further biogenic substances, such as nitrogen, phosphorous and potassium is equally important. These substances are complementary building blocks of microbial cells. For example, nitrogen is a crucial component of the proteins, nucleic acids, amino acids, enzymes and co-enzymes necessary for microbial cell growth and function. The optimum ratio between carbon and other biogenic substances can be derived from the relative abundance of those elements in a microbial cell. The optimum C/N ratio is ca. 20-30. At a C/N ratio lower than 20, nitrogen will be supplied in excess and will be lost in the form of ammonia gas causing undesirable odors. At a C/N ratio of over 30 there is not enough nitrogen for the optimal growth of microbial populations. In the latter case, compost will remain relatively cool and the degradation will proceed at a slow rate. The optimum C/P ratio is 100. The content of biogenic substances in separately collected biowaste is presented in Table 6-4.

Table 6-4. Biogenic substances in biowaste and green waste (Kern et al. 2010)

Input material	C/N ratio	N (%DM)	P_2O_5 (%DM)	K_2O (%DM)	MgO (%DM)	CaO (%DM)
Biowaste						
Bidlingmaier (1995)	14 – 36	0,6-2,1	0,4-1,4	0,6-2,1	0,3-1,2	2,2-6,8
Fricke et al. (2002)	10-25	0,6-2,7		0,5-1,6	0,5-2,0	0,5-5,5
Green waste						
Bidlingmaier (1995)	20-60	0,3-2,0	0,1-2,3	0,4-3,4	0,2-1,5	0,4-12
Fricke et al. (2002)	15-76	0,3-1,9		0,5-1,6	0,5-2,0	0,7-7,4
Siehler (1993)	15-30	1,3	0,9	1,6	0,6	2,9

If the C/N or C/P ratio is not in the optimal range, it can be corrected by adding other biodegradable waste. A too high C/N ratio can be corrected by the addition of e.g. wastewater sludges or animal excrements (C/N < 10) or grass, leaves and vegetables (C/N 10 – 30). A too low C/N can be corrected by the addition of e.g. wood or bark (C/N 50 – 500).

Absence of hazardous substances in input materials (inhibitors): A high level of the contaminants in the composted waste (oils, heavy metals, PAH, chlorinated organic compounds) can lead to process inhibition. The same is true for applying biological treatment in the purification of contaminated soils. Compost obtained by processing contaminated waste will not fulfill the quality criteria and will lead to their disqualification for use as a fertilizer.

pH value The optimal pH value for the composting process is determined by the toleration ranges of the microorganisms involved in the process. Optimal pH values for composting bacteria range from pH 6,0 to pH 7,5. Fungi tolerate higher pH variability, ranging from pH 5,5 to pH 8,0. Acidic conditions at pH values lower than 6 result in a decrease of the biodegradation rate. At a pH value of over 9,0 increased emissions of NH_3 can be observed leading to a higher odor nuisance.

In Table 6-5 the suitability of various biodegradable waste for composting is summarized.

Presence of microorganisms (bacteria and fungi)

The degradation of organic compounds in waste during composting is initiated predominately by a very diverse community of microorganisms: bacteria, actinomyctes and fungi.

Bacteria are the smallest known living organisms whose size is approximately 10^{-3} mm and they are the most numerous in compost. They make up 80 to 90% of the billions of microorganisms typically found in a gram of compost. Bacteria are responsible for most of the decomposition and heat generation in compost. In terms of decomposition abilities, they are the most diverse group of compost organisms using a broad range of enzymes to chemically break down a variety of organic materials. Bacteria are especially predominant in the early stages of compost breakdown where, if conditions are suitable, the temperature of the heap will rise very quickly, possibly up to 55°C and above. The types and distribution of bacteria in the heap will change as the temperature rises. Generally, due to different temperature tolerances, bacteria can be divided into three groups (see Table 6-6).

Table 6-5 Suitability of various input materials for the composting process (based on Amlinger et al. 2005)

Waste	Dominating organic compound	Nutrients content	C/N	Application as source of C	Application as source of N	Water content	Odor potential	Content of contaminants	Hygienization needed: weed	Hygienization needed: hum. pathogen	Recommendation
Kitchen waste	Cellulosis, proteins	++	15 - 20	-	++++	++	++++	+/++	+	+	Add structure material
Biowaste (households)	Cellulosis, lignins, proteins	++	20 - 40	++	+++	++	+++	+/++	+	+	Add structure material
Vegetables and fruit	Sugar, cellulos., proteins	+++	13 - 20	++	++/++++	+++	++++	+	+	-	Add structure material
Restaurant waste	Cellulosis, proteins	+++	12 - 20	+	++/++++	+++	++++	+/++	-	++	Add structure material
Fecalies of pets	Lignin, cellulosis, protein, clay	++	++	++	-/++	+	+++	+/++	-	++	Mix intensively with other materials, e.g. compost
Paper and cardboard	Lignin, cellulosis	+	>60	+++	-	-	-	+/+++ (print, chloride)	-	-	Add structure and nutrients, <10% input
Fresh grass	Cellulose, lignin	++++	12 - 25	+	+/+++	+++	+++	-/++ (streets)	+	+	Add structure material
Trimmings and branches	Cellulose, lignin	++	100 – 150	+++	-/+	+	+	-/+++ (polluted areas)	-	-	Add protein source
Timber chips	Lignin	+	100 - 500	+++	-	-	-	+/++ (pesticides)	-	-	Add protein source
Fermentation residues	Input dependant	++/+++	10 - 25	++	++	++++	++	+ (input dependant)	-	+	Add structure material
Sewage sludge	Organic P, organic N	++/+++	8 - 12	+	+++	++++	+/+++	+++ (organic contaminants)	+	+++	Hygienization, add structure materials > 50%

Table 6-6 Temperature requirements of different bacteria

	Range of tolerated temperatures, [°C]	Optimum temperature, [°C]
Psychrophilic	0 - 30	10 - 20
Mesophilic	10 - 55	25 - 40
Thermophilic	40 - 95	45 - 75

Psychrophilic bacteria are present in the initial composting phase. They are most active in temperatures of approximately 15 °C, but some are able to survive at temperatures below 0°C. As these bacteria decompose organic matter, heat is released creating an environment suitable for the next group - mesophilic bacteria. As the temperature rises above about 40°C, the mesophilic microorganisms become less competitive and are replaced by others that are thermophilic. At temperatures of 55°C and above, many microorganisms that are human or plant pathogens are destroyed. The thermal destruction of pathogens (sanitization) mainly depends on the level of temperature and length of its duration. Maintaining a temperature above 65°C (or in the case of enclosed composting facilities, 60 °C) for at least one week or above 55°C for at least two weeks in order to achieve sanitization of the biodegradable waste is prescribed in some national regulations (e.g. in Germany by the Ordinance on Biowastes - BioAbfV). Any input materials which have not yet undergone a sanitization process should be stored in such a way that contact is avoided with any heated-up composted materials. Because temperatures over about 65°C kill many forms of microbes and limit the rate of decomposition, compost managers use aeration and mixing to keep the temperature below this point. When conditions become unfavorable, some bacteria survive by forming endospores, thick-walled spores that are highly resistant to heat, cold, dryness, or lack of food (Trautmann and Olynciw 1996). Once the compost cools down, mesophilic bacteria again predominate.

Actinomycetes are a specific group as bacteria. Morphologically they resemble fungi because of their elongated cells. There are both thermophilic and thermo tolerant and mesophilic actinomycetes species. They play an important role in degrading complex organics such as cellulose, lignin, chitin, and proteins contained in woody stems, bark, or newspaper (Trautmann and Olynciw 1996).

Fungi form an extremely diverse group of lower plants that grow either on dead organic matter (saprophytic) or on living cells (parasitic) or in symbiosis with other organisms (lichens). Fungi include molds and yeasts, and collectively they are responsible for the decomposition of many complex plant polymers in soil and

compost. In compost, fungi are important because they break down resistant organic matter, such as cellulose and lignin, into smaller molecules which can be used by other composting organisms. They can attack organic residues that are too dry, acidic, or low in nitrogen for bacterial decomposition. Fungal species are numerous during both the mesophilic and thermophilic phases of composting. Most fungi live in the outer layer of compost when temperatures are high (Trautmann and Olynciw 1996).

Primary consumers also include **invertebrates** such as worms, snails and beetles which disintegrate, consume and digest the plant materials. Their excreta may contain shredded but undigested plant residues which provide increased surface area for further action by microorganisms.

Proper granulometry and structure of waste

Waste consists of three phases: solids, water and air. The latter ones are of antagonistic nature, i.e. the higher the water content the lower the free air porosity. For maintaining aerobic conditions in the waste mass as well as a relatively low pressure loss of aeration flux, a minimum of 30% free air porosity is required. This implies a maximum bulk density of waste at the level of 500 – 600 kg/m^3. A too low waste air porosity, e.g. in the case of composting wet kitchen waste or manures can be compensated by adding structure materials (wood and other green waste).

As for the solid particle size, in general, a smaller size implies faster material decomposition due to

- larger specific material surface,

- better access for microorganisms,

- better material homogeneity;

however, a smaller particle size worsens aeration conditions.

The optimum particle size of the composted solid phase is 20 - 40 mm in order to allow good process conditions. This means that larger constituents, e.g. stems and branches, require shredding before composting.

Effective mixing and aeration

The role of aeration in the composting process is twofold:
- Supplying oxygen as a substrate for degradation and removing gaseous biodegradation products,
- Final compost drying before screening to remove water as a vapor.

The optimal concentration of O_2 in the air of the compost mass amounts to ca. 14-16% by volume (equivalent to 2/3 of the natural oxygen concentration in atmospheric air). If the O_2 level falls below 5 % by volume, anaerobic conditions appear causing process inhibition and elevated odor emissions. Factors influencing oxygen demand include
- Process phase (higher oxygen consumption at the process beginning, decreasing with time)
- Type of composted material (high content of readily biodegradable material and of nitrogen increases oxygen consumption),
- Particle size of composted material (too small particles make compost aeration difficult),
- Moisture content (more wet waste consumes more oxygen – also for drying).

Optimal aeration rates for various waste types are presented in Table 6-7. As a general design recommendation an air-flow rate of 530 to 620 m^3/ton waste is given.

Table 6-7. Optimal aeration rates for various waste types (adapted from Bidlingmaier 1998)

Waste type	Aeration rate	Unit
Sewage sludge with structure material	5 - 12	g O_2/kg orgDM · h
Municipal waste	6 – 8	dm^3 air/kg DM · h
Organic waste	23 - 75	dm^3 air/kg DM · h
Municipal waste	0,3 – 0,8	g O_2/kg orgDM · h
Fresh compost	0,77 – 1,57	dm^3 O_2/kg DM · h
Cattle manure	2 – 4	g O_2/kg orgDM · h
Organic waste	1 - 5	g O_2/kg orgDM · h
Sewage sludge with structure material	31 - 61	dm^3 air/kg DM · h

Temperature of composted material

Aerobic decomposition of organic matter is an exothermic process, i.e. the energy balance is positive. With the decomposition of 1 mol of glucose, approximately 1/5 of the biodegraded organic matter is used for biomass growth while the rest is turned into energy. In total, 2.865 kJ of energy is released per 1 mol of glucosis, of which 40% is used by microorganisms (respiration) and 60% is released as heat.

In composting experiments the following heat release was measured per 1 g of decomposed matter:
- organic waste with wood chips or straw   15-19 kJ/g
- organic waste as above spiked with fat   22-25 kJ/g,
- average from all experiments   20,7±2,9 kJ/g, or 452±29 kJ/mol oxygen consumed (Sundberg 2003)

Due to the release of heat the temperature of the composted waste rises. A typical temperature curve during the composting process is presented in Figure 6-8. Under optimal conditions composting proceeds through three temperature phases:
- A mesophile or moderate temperature phase
- A thermophile or high temperature phase
- Cooling and maturation phases.

At the beginning of the composting process after a short lag, the temperature increases exponentially from ambient temperature to 70–75 °C. At 40 °C there is often a lag during the changeover from mesophilic to thermophilic microorganisms. Fast temperature growth at this phase is due to the degradation of easily degradable compounds such as sugar, starch, pectin and protein. At this phase oxygen consumption and thus the demand for aeration is the highest. Similarly, this phase is accompanied by the highest emissions of odors and leachates. Thus, this phase is often called intensive composting and normally takes a few weeks. After all easily degradable compounds have been oxidized, the degradation of difficult-to-decay degradable compounds begins such as hemicellulose, wax, fat, oil, cellulose and lignin. These processes are characteristic to the cooling and maturation phases, which can continue from a few weeks to up to one year. At this stage, mostly high molecular weight compounds, organisms and weed seeds are degraded. They are partly decomposed and partly condensated to form humus. The release of heat at this stage is low, resulting in a temperature decrease to ambient conditions. This phase is characterized by low

oxygen demand and low emissions of odors and leachates. Thus, this phase is often referred to as post-composting or the maturation phase.

In a composting plant the temperature can be controlled in order to achieve the pre-defined goals, e.g.

- ✓ 55° C to maximize the biodegradation rate (high temperatures inhibit the process)
- ✓ 65° C to maximize hygienization – sanitization, pathogen destruction.
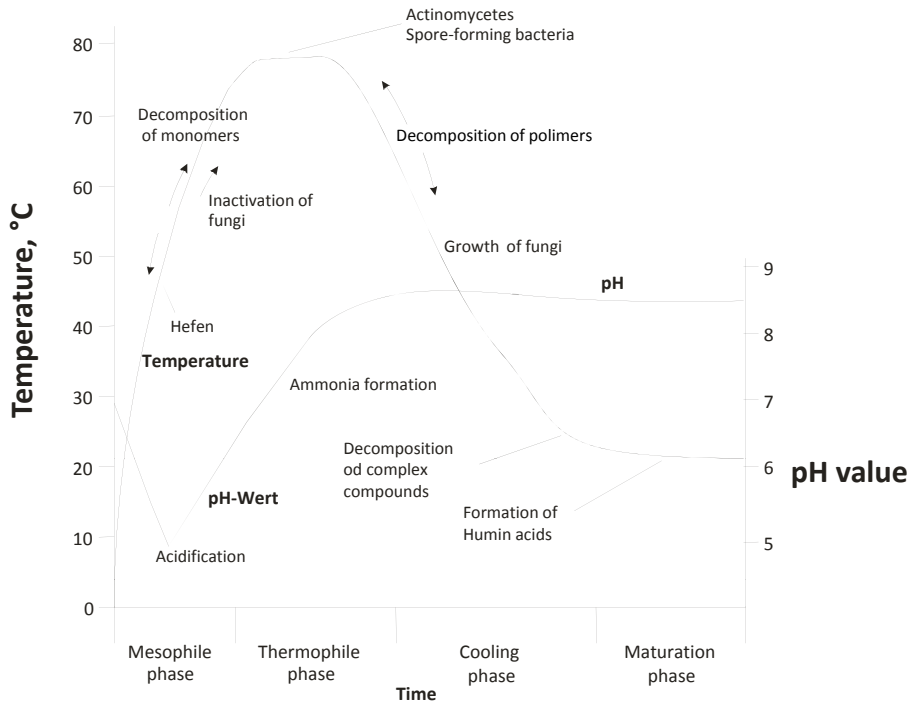

Figure 6-8. A typical temperature curve of a composting process (Flemming 1995)

The temperature can be controlled by modifying the aeration rates. An increase of aeration results in temperature reduction. In practice, the initial temperature rise is used to achieve the required compost hygienization. Afterwards, lower temperatures are preferred to maximize the biodegradation rate.

6.2.4 Classification of composting technologies

There is a large variety of composting technologies. The selection of the composting technology for a given application should be based on the following general criteria:

- Types and amounts of composted waste,
- Compost quality requirements or standards and places of their application,
- Local conditions depending on plant location, land availability etc.

With regard to the size and the range of service the following types of technology can be distinguished:

1) Home composting in individual (garden) composters – for composting by the residents of detached houses biowaste from kitchen and garden, etc.,
2) local composting plants, serving mainly small residential areas, parts of towns, selected structures (e.g. parks and other green areas in towns) designated primarily for separately selected biowastes, green waste etc. Capacity of <10 000 ton/year, and typically ca. 1000 ton/year,
3) regional (municipal) plants of mechanical-biological treatment of mixed municipal waste from various sources of generation.

Home composting is sometimes regarded as the environmentally most beneficial way of handling of domestic biodegradable waste as it saves on transport emissions and costs, assures careful input control and increases the environmental awareness of the users (COM/2008/0811).

With regard to the technology used, composting plants can be divided into open air composting and enclosed composting. Open composting is mostly realized in windrows and offers the cheapest method of biodegradable waste treatment. However, it has serious limitations such as odor emissions and a high area demand. Also the composting duration may be up to one year. Fully encapsulated plants allow for significant odor minimization. The flue gas in such plants is contained and fed through a purification system, normally a biofilter. Another advantage of such plants is automatic moisture and oxygen control allowing an acceleration of the composting process. Composting technologies may be also characterized as reactor systems and non reactor systems (see Figure 6-9).

Most common technologies applied in centralized composting plants include
Static pile (windrow) - traditional composting method in which waste is arranged in long heaps with a triangular or trapezoidal cross-section. Windrows remain in constant interaction with atmospheric conditions. Natural air flow is supported by occasional turning of the material. The advantage of this method is its simplicity and low-cost. The disadvantages include low control potential, dependence on

weather conditions and no possibility for odor control. Currently, static windrows are still applied for green waste composting and for the maturation process - a second step after enclosed intensive composting. Windrows are located on a hard insulated surface in order to prevent leaching to the ground.

Aerated static pile (aerated windrow) – a more advanced windrow system equipped with forced aeration.

composting systems
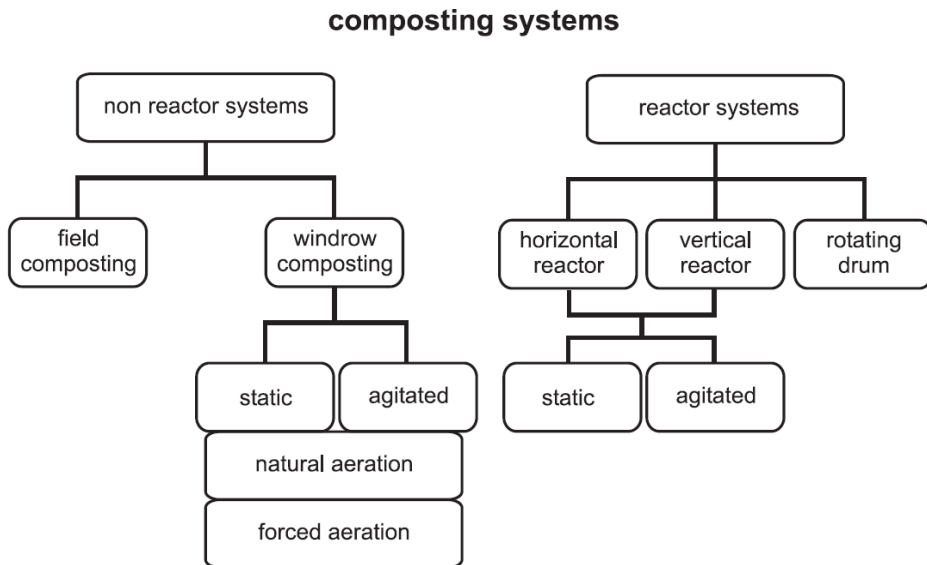
Figure 6-9. Classification of composting technologies (Schuchardt 2005).

Agitated bed (mechanical windrow) – an open kind of reactor, usually located in a composting hall, where the waste material can be turned mechanically and water can be added. The devices for mixing the wastes can be horizontally or vertically operating rotors or screws, scraper conveyors, or shovel wheels. Fully automated functioning of the whole process is possible.

Rotating drum – enclosed horizontal, slowly rotating reactors with forced aeration. The filling capacity is approximately 50%. The material is transported in the helical pathway from one end of the drum to the other and is mixed intensively. Self-heating starts after a short time. Water addition
is possible. Rotating-drum reactors can also be used as mixing equipment.

Box composting is an example of a state-of-the-art technology. It is normally realized as a first step in a two stage technology in which the first stage can be described as intensive composting and the second as maturation. Full automatization of intensive composting shortens the time of the fresh compost production process to 10 – 14 days. Also, this technology requires a limited amount of space. An additional advantage is the limited volume of waste air for purification. The system can be flexibly adjusted to the input waste quantity. The second stage, maturation is less odor intensive and can be performed both in open air windrows or in windrows in an enclosed building.

Currently, a trend towards fully encapsulated composting technologies can be observed. For example in Germany, the technology of enclosed intensive composting in composting boxes combined with maturation is frequently used as well as windrow 1-stage composting in enclosed buildings.

6.2.5 Process steps in a composting plant

The main process steps in a composting plant are presented in Figure 6-7. The process can be subdivided into initial mechanical treatment consisting of preparing and conditioning the raw material followed by the actual composting. To produce a marketable product, it is necessary to convert the compost to an end product, which involves additional mechanical treatment – conditioning of the composting product.

Mechanical pre-treatment and conditioning

The aim of raw material preparation is to optimize conditions for the subsequent composting process, to remove contaminants in order to protect the technical equipment and to meet quality requirements for the finished compost. The basic steps of raw material preparation are

- shredding (e.g. bulky wood scraps, trees, brush, long grass)
- dewatering of water-rich, structureless wastes (e.g. sludge, restaurant waste)
- addition of water if the wastes are too dry for the composting process,
- mixing of components (e.g. wet and dry wastes, N-rich and C-rich wastes, wastes
  with rough and fine structure)
- manual or automatic separation of impurities (glass, metals, plastics).

The actual composting

The actual composting is performed using one of the technologies described in the previous section. The current trend for municipal biodegradable waste

composting is two step technology – intensive composting in an enclosed reactor or in open reactors, but inside a building and the maturation phase in windrows outdoors on a paved surface covered with a roof.

Mechanical post-treatment and conditioning
The main product of the composting process is compost. Compost may require additional treatment before transport, storage, sale, and application. When post-preparation is needed, the basic steps can be

- sieving the compost to obtain different fractions for marketing or to remove impurities,
- manually or automatically removing contaminants
- drying wet compost to prevent formation of a clumpy, muddy product and drainage of water during storage
- disintegrating clumps in the compost by crushing or grinding to prevent problems that may occur when the fertilizer is packaged
- mixing the compost with additives (soil, mineral fertilizer) to produce potting mixes or gardening soils (Schuchardt 2005).

6.2.6 Compost application

The European Thematic Strategy for Soil Protection (COM(2006) 231) calls the use of compost one of the best sources of stable organic matter from which new humus can be formed in degraded soils. An estimated 45% of European soils have low organic matter content, principally in southern Europe but also in areas of France, the UK and Germany. The use of compost and digestate as soil improvers and fertilizers offers agronomic benefits such as improvement of soil structure, moisture infiltration, water-holding capacity, soil microorganisms and supply of nutrients (on average, compost from kitchen waste contains about 1% N, 0,7% P_2O_5 and 6,5% K_2O). In particular, the recycling of phosphorous can reduce the need to import mineral fertilizer while the replacement of peat shall reduce damage to wetland eco-systems. Increased water retention capacity improves workability of soils thereby reducing energy consumption when plowing. Better water retention (soil organic matter can absorb up to 20 times its weight in water) can help to counteract the desertification of European soils and prevent flooding. Finally, the use of compost contributes to counteracting the steady loss of soil organic matter across temperate regions (COM/2008/0811).

Composts can be used as fertilizer and/or a soil conditioner applied in
- Agriculture - food and non-food crops,

- Landscaping - properties and grounds maintenance
- Nurseries - potted plants, forest seedling crops
- Public agencies - highway landscaping, recreational areas, other public property
- Residences - home landscaping and gardening
- Other - land reclamation and landfill cover

Compost quality requirements

Compost quality requirements depend on its application, but in general the following aspects need to be addressed:
- optimal stability and maturity,
- favorable content of nutrients and organic matter,
- favorable C/N ratio,
- neutral or alkaline pH,
- low content of heavy metals and organic contaminants,
- no components that interfere with plant growth,
- low level of impurities,
- mostly free from germinable seeds and living plant parts,
- low content of stones,
- typical smell of forest soil, and
- dark brown to black color.

Stability can be defined as the bioavailability of organic matter, which relates to the rate of decomposition. Compost in which the level of fast degrading compounds is high has low stability.

Maturity is defined as the suitability of compost for plant growth and is associated with the degree of humification.

Several methods are used to determine maturity and stability, e.g. self-heating test, tests on plants, respiration activity, chemical analyses and nuclear magnetic resonance.

In the classic self-heating test according to Dewer, five compost classes are defined (see Table 6-8). The test is performed in a special vessel where a compost sample is stored at a constant ambient temperature of 20 ± 2°C for the whole test duration. The temperature inside the compost sample is recorded with a thermocouple (temperature sensor) located in the center of the flask. The highest temperature rise above ambient temperature is recorded and is used to classify the stability of the sample.

The compost will normally achieve the highest temperature within the first several days. Sometimes, a sample may require 5-10 days to heat. The temperature reading is continued until it declines for at least 2 days after the maximum is attained.

Table 6-8. Compost stability classes based on the original German Compost Association (BGK) Index from Dewar (1988)

Temp. rise above ambient in C	Official class of stability	Descriptors of class group	Major group
0 - 10°	V	Very stable, well-aged compost	Finished compost
10 - 20°	IV	Moderately stable, curing compost	
20 - 30°	III	Immature, young or very active compost	Active compost
30 - 40°	II	Immature, young or very active compost	
40 - 50° (or more)	I	Fresh, raw compost or just mixed ingredients	Fresh compost

More recently, a modified classification based on only three classes has been proposed (see Table 6-9). According to Brinton et al. (2009) composts are commonly recognized to have three states or potentialities: cool or cured, mesophilic, and thermophilic. A compost is of thermophilic potential at or above 45°C. Recent research shows that a small range of compost heating in the Dewar method (<5°C) corresponds to a significant amount of O_2 or CO_2 respiration; therefore, one level of "mature" compost with very little heating is recognized and one level for "mesophilic" which is the normal active and a transitional stage going into or coming out of the very active thermophilic state of composting.

Table 6-9 Modified compost stability classes (Brinton et al. 2009)

Temperature rise above ambient in C	Actual temperature in vessel in C	Class of stability	Interpretation
0 - 5°	20 - 25°	A	Mature, curing compost
5 - 25°	25 - 45°	B	Mesophilic, active
25 - 50°	45 - 70°	C	Thermophilic, very active

Compost stabilization can also be determined based on a respiration index (AT or RI). This method is based on the evaluation of oxygen uptake in mg O_2/g DM related to the compost sample (either within 4 days - AT4 or within 7 days - AT7) in standard conditions. Values of respiration indexes AT4 and AT7 after various composting durations ranging from 0 to 112 days are presented. The recommended AT4 value for finished compost is 5 mg O_2/g DM.
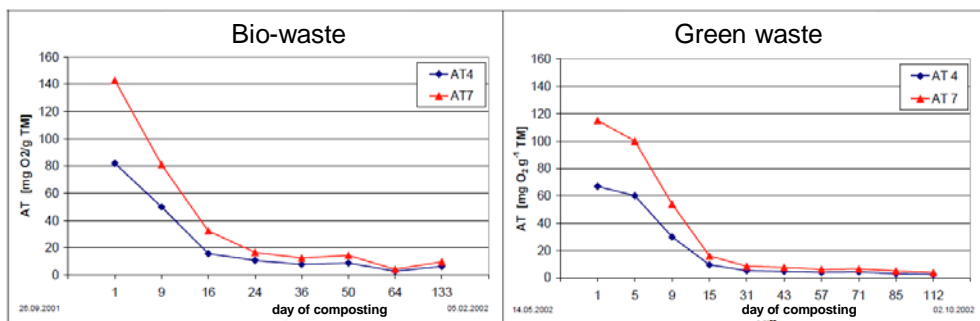
Figure 6-10. Respiration index of composts (Amlinger 2005)

Alternatively, the respiration-based method of evaluating compost maturation is the gas formation potential within 21 days – GB21. In this method, the potential of biogas formation in anaerobic conditions is evaluated. The GB21 of fresh biowaste ranges from 200 to 250 dm^3/kg DM The recommended value for finished compost is 20 dm^3/kg DM. Both respiration indexes – AT4 and GB21 are very well correlated with each other (Cossu 2008).

Environmental quality standards for compost

The agricultural benefits of compost use are evident, but there is debate about their proper quantification (e.g. by comparison to other sources of soil improvers), while the main risk is soil pollution from bad quality compost. As biowaste easily gets contaminated during mixed waste collection, its use on soil can lead to the accumulation of hazardous substances in soil and plants. Typical contaminants of compost include heavy metals and impurities (e.g. broken glass), but there is also a potential risk of contamination by persistent organic substances such as PCDD/F, PCB or PAHs (COM/2008/0811). Thus, proper control of input material coupled with the monitoring of compost quality is crucial. At the moment there is no valid European standard for the quality of compost. Some pieces of European requirements regarding compost applications can be found in

- The Organic Farming Regulation 834/2007/EC - conditions for the use of compost in organic farming.
- The eco-labels for soil improvers (2006/799/EC) and for growing media (2007/64/EC) - specify limits for contaminants and require that the compost be of waste origin only.
- draft Biowaste directive – not in force yet.

In addition, various Member States have issued their own national criteria. Only a few Member States allow compost production from mixed waste. Most require the separate collection of biowaste, often in the form of a positive list of waste which may be composted (COM/2008/0811). This approach limits the risk and reduces the cost of compliance testing by allowing less extensive monitoring of the production and use of compost. One of the very important criteria for the application of compost is its heavy metal content. The differences in the content of heavy metals in biodegradable waste collected in various collection schemes and in mixed waste are demonstrated in Table 6-10.

Table 6-10. Heavy metal content in biodegradable waste, mg/kg DM

Fraction of municipal waste	Cd	Cr	Cu	Pb	Zn
Biowaste	0,2 - 1,1	15 - 43	17 - 57	19 - 130	69 - 290
Wet fraction	1,6 - 2,5	60 - 95	130 - 270	400 - 680	460 - 880
Organics separated mecha-nically from mixed waste	1,7 - 1,9	20 - 59	70 - 710	600 - 670	540 - 710
Mixed waste	8,5	140	530	830	1600

It can clearly be seen that through separate collection a much better input quality for composting can be ensured. The limiting values for the heavy metal content in composts and stabilized biowaste according to the draft Biowaste directive are presented in Table 6-11. Considering that during the composting process metals get concentrated as they are related to the decreased mass of solids, only separately collected biowaste has the potential to fulfill the quality requirements for composts of class 1 and class 2.

The draft Biowaste directive also specifies the hygienization requirements for composts. Compost/digestate is deemed to be sanitized if it complies with the following:

– *Salmonella spp absent in 50 g of compost/digestate*
– Clostridium perfringens *absent in 1 g of compost/digestate*
Compost/digestate shall have less than three germinating weed seeds per liter.
With regard to the above requirements, the draft directive also specifies the
necessary treatment conditions and their duration (see Table 6-12).

Table 6-11. Proposed criteria for contaminant limits in composts and stabilized biowaste (draft Biowaste directive)

Parameter	Compost / digestate[1]		Stabilized waste
	Class 1	Class 2	
Cd, mg/kg DM	0,7	1,5	5
Cr, mg/kg DM	100	150	600
Cu, mg/kg DM	100	150	600
Hg, mg/kg DM	0,5	1	5
Ni, mg/kg DM	50	75	150
Pb, mg/kg DM	100	150	500
Zn, mg/kg DM	200	400	1500
PCBs, mg/kg DM[2]	-	-	0,4
PAHs, mg/kg DM[2]	-	-	3
Impurities >2 mm	<0,5%	<0,5%	<3%
Gravel and stones >5mm	<5%	<5%	-

[1] normalized to an organic matter content of 30%
[2] treshold values for these organic pollutants to be set in consistence of the Sewage Sludge Directive

Table 6-12. Composting conditions aimed at compost hygienization, specified in the draft Biowaste directive

	Temperature	Treatment time	Turnings
Windrow composting	≥55°	2 weeks	5
Windrow composting	≥65°	1 week	2
In-vessel composting	≥60°	1 week	N/A

6.3 Anaerobic digestion of biowaste

The anaerobic digestion process, often referred to as the fermentation process, can be defined as the microbial degradation of organic compounds in the absence of oxygen to biogas – a mixture of methane (50 – 70%), CO_2 (30 – 50%) and trace

amounts of H_2, NH_3 and H_2S. As opposed to composting, anaerobic digestion is most suited for wet waste with a weak structure, such as commercial kitchen waste, market waste and sewage sludge. The theoretical stoichiometric summary reaction of biowaste anaerobic digestion can be written as follows:

$C_aH_bO_cN_d$ + 1/4 (4a − b − 2c + 3d) H_2O = 1/8 (4a + b − 2c +3d) CH_4 + 1/8 (4a - b + 2c +3d) CO_2 + d NH_3
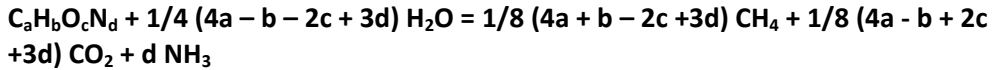
where the theoretic summary formula of
- a readily biodegradable substance: $C_{40}H_{65}O_{27}N$ and
- slowly biodegradable substance: $C_{20}H_{29}O_8N$

Water is a necessary substrate for the anaerobic digestion process. The stoichiometric water consumption of an anaerobic digestion process amounts to approximately 0,3 kg/m^3 biogas.

Anaerobic digestion as a waste treatment method is applied to a variety of substrates such as manure, slaughterhouse waste, waste water treatment sludge and biowaste. Also combinations of substrates, the so-called co-digestion or co-fermentation, is widely applied.

Anaerobic digestion is the result of the joint action of several populations of bacteria and occurs in a series of steps. The degradable solids that make up the waste are first biologically hydrolyzed to smaller soluble molecules. Then acid-forming bacteria use these soluble intermediates as substrates for energy and growth resulting in the formation of fermentation products such as Volatile Fatty Acids (VFAs). Finally, the methanogens or VFA-consuming bacteria produce methane and carbon dioxide.

The major steps of the anaerobic digestion process can be described as follows (see Figure 6-11):

1. The **hydrolysis** of high molecular often insoluble organic polymers such as proteins, carbohydrates and lipids. Enzymes convert them to soluble fragments (monomers) such as sugars, amino acids, long-chain fatty acids;

2. The **acidogenesis** or fermentation of amino acids and sugars: conversion of organic monomers to acetic, propionic and butyric acids (gathered under the 'VFA' label), hydrogen, carbon dioxide, and other organic products such as ethanol and lactic acid;

3. The **acetogenesis** – transformation of long chain fatty acids and alcohols to acetic acid, hydrogen and carbon dioxide by the obligate hydrogen producing acetogenic (OHPA) bacteria;

4. The **methanogenesis** - methane fermentation, i.e. conversion of acetate to methane and conversion of hydrogen to methane.
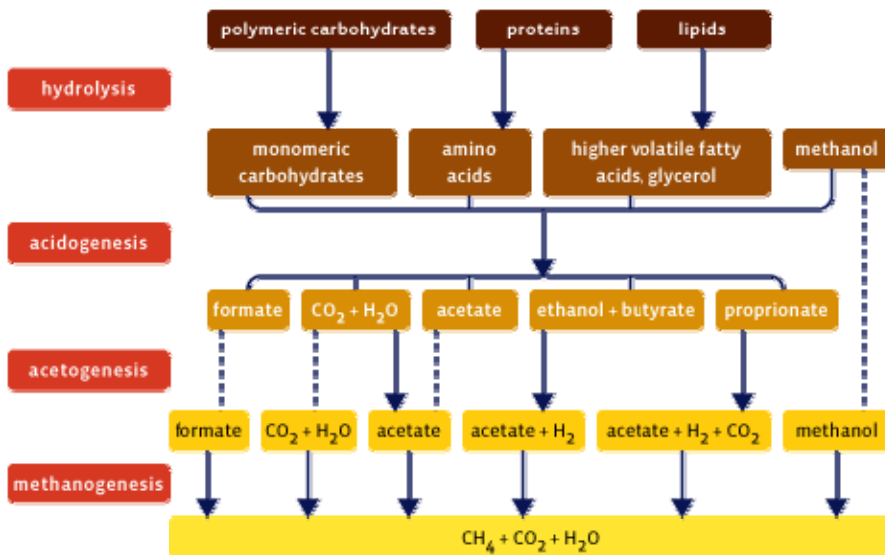
Figure 6-11. Anaerobic digestion process (Source: www.water-leau.com)

The energy balances of summarized aerobic and anaerobic biodegradation processes of glucose are presented below:

Aerobic decomposition: $C_6H_{12}O_6 \; + \quad 6\,O_2 \; \longrightarrow \quad 6\,CO_2 + 6\,H_2O$
heat = - 2875 kJ/mol

Anaerobic decomposition: $C_6H_{12}O_6 \qquad\qquad \longrightarrow \quad 3\,CO_2 + 3\,CH_4$
heat = - 132 kJ/mol

+ incineration of biogas: $\quad 3\,CH_4 \; + \qquad 6\,O_2 \; \longrightarrow \quad 3\,CO_2 + 6\,H_2O$
heat = - 2671 kJ/mol
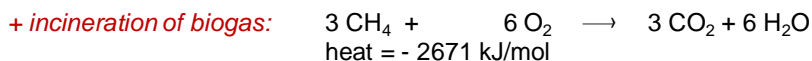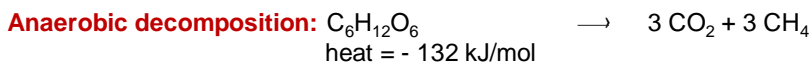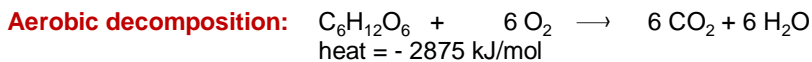
The quantity of energy released as heat in the composting process is much higher than the quantity of heat release during the anaerobic digestion process. Thus, in the anaerobic digestion process only a small temperature rise is observed. The

90

chemical energy which is preserved in biogas can be released in a subsequent process of energy recovery from biogas. This energy recovered from biogas combustion can be further converted to electricity or used in the form of steam or heat. This is a clear advantage of the anaerobic digestion process compared to composting. In the latter, a larger quantity of heat is released within the microbial decomposition, but it is technically not possible to recover this energy.

A second major difference between both technologies is the biomass production. In aerobic decomposition, the growth of microorganisms is significantly higher than in the anaerobic process. The biomass growth related to the decomposition of 100 g of dry organic mass, with a simplified average formula CH2O for both aerobic and anaerobic processes, is presented in Figure 6-12.
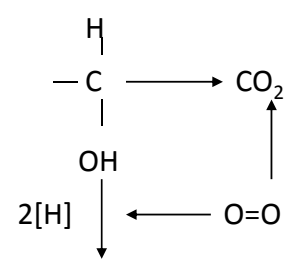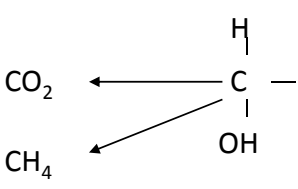
Composting process Oxidation of Corg	Fermentation process Reduction of Corg
aerobic, with oxygen	anaerobic, no oxygen
100 g dry mass (H_2O)+38,5 g O_2 → 63 g bio mass (micro-organisms) + 53 g CO_2 + 20 g H_2O	100 g dry mass (CH_2O) → 9 g bio mass (micro-organisms) + 66 g CO_2 + 24 g CH_4

Figure 6-12. Biomass production in aerobic versus anaerobic decomposition (Scherer 2002)

6.3.1 Anaerobic digestion process parameters

Anaerobic digestion processes are classified by critical operating parameters and reactor design such as continuity (batch versus continuous), operating temperature (psychrophilic, mesophilic and thermophilic), reactor design (plug-flow, complete-mix, and covered lagoons), solid content (wet versus dry) and the number of process steps: 1-stage or 2-stage processes. In Table 6-13, a five-year overview of the types of installed digesters in Europe is given. In the time span 2006-2010 the following trends can be observed:

- share of thermophilic installations increased to 41%

- share of one-phase digesters increased to 98%
- share of dry anaerobic digestion technology increased to 71%
- share of solid waste digesters increased to 95%
- share of biowaste digesters increased to 49%.

Table 6-13 Five-year development – digesters installed (de Baare 2008)

5 year development	1991 - 1995	1996 - 2000	2001 - 2005	2006 - 2010
Mesophilic installed/5 year	64%	64%	80%	59%
Thermophilic installed/5 year	36%	36%	20%	41%
One phase installed/5 year	85%	91%	92%	98%
Two phase installed/5 year	15%	9%	8%	2%
Wet installed/5 year	37%	38%	59%	29%
Dry installed/5 year	63%	62%	41%	71%
Solid waste installed/5 year	77%	90%	90%	95%
Codigest installed/5 year	23%	10%	10%	5%
Biowaste installed/5 year	92%	72%	41%	49%
Residual installed/5 year	8%	28%	59%	51%

The most important process parameters of the anaerobic digestion process include

- moisture
- the content of organic substances
- pH and redox potential
- content of biogenic substances
- presence of bacteria
- presence of harmful substances
- temperature and retention time - process duration.

Moisture

Water is essential for methane fermentation, as the nutrients for the micro-organisms must dissolve in water before they can be assimilated. Moisture facilitates microbe transport, the transport of nutrients in a high-solid bed and the balance between Volatile Fatty Acid production and the conversion of acids to methane. As the moisture content decreases, there is less water available for dilution and salt concentrations increase. Batch reaction rates were reported to

have been severely limited at an initial water content below 70%. Methane production decreases considerably at a water content <35%.

With regard to the water content, anaerobic digestion technologies can be divided into

- "Dry" anaerobic digestion – moisture content 60-80 % - waste at normal moisture content or additional water added (continuous of batch process)
- "Wet" anaerobic digestion – moisture > 85 % - waste moistened with water (continuous process, high digestion rate)

In terms of biological performance, dry systems can reach a high biomass concentration and have proved reliable. From a technical viewpoint, the dry systems appear more robust because frequent technical failures due to sand, stones, plastics and wood were reported in wet systems.

Content of organic substances

Typically, organic dry matter in a digested material is required to be higher than 60 % DM.

There is an optimum Organic Loading Rate (OLR) for a particular reactor which will produce maximum gas beyond which further increases in the quantity of substrate will not proportionately produce more gas. Depending on the biodegradability of the feedstock, the Organic Loading Rate may vary over a wide range.

The following OLR are applied:

1) Wastewater sludge:
   - low loading   0,8-2,2 kg dm/m^3d
   - high loading   2,5-5,0 kg dm/m^3d
2) Municipal and similar waste:
   - very high loading   10-13 kg dm/m^3d

pH and redox potential

A proper pH level is crucial for the optimal operation of a digester due to the high relevance of this parameter to microbial activity. In this context, the two-phase configuration of a anaerobic digestion process enables each to be optimized separately. Based on the simulation of anaerobic decomposition, it is known that a pH of 5,3 and 7,4, respectively, are optimal for the acidogenic and methanogenic phase. For the process design, it is normally assumed that hydrolysis proceeds well at a range of pH 6 – 6,5 and acidification at pH 5,8 – 6,2. In contrast, it is generally accepted that methane production from VFAs and lipids is optimal for a pH in the range 6,3 to 7,8 (Trzcinski 2009).

For single stage anaerobic digestion the optimum pH is 7,0 (range 6,0-8,0). Waste has a large buffering capacity due to the presence of acetic acid and acetates, phosphates and carbonates which slow down the pH reduction. Methanogens are most affected by changes in pH. A decrease in the rate of methane production at a pH below 6,3 may be related to the fact that methane formation proceeds at a slower rate than the production of organic acids. A decrease in methane production above pH 7,8 may be related to a shift of ammonium (NH^+) to the unionized form of ammonia (NH_3). The pH is known to influence enzymatic activity because each enzyme is active only within a specific and narrow pH range and displays maximum activity at an optimum pH (Trzcinski 2009).

The optimum redox potential amounts to -200 mV (range -50 to -500 mV).

Content of biogenic substances
The optimum nutrient content related to organic carbon is C/N ca. 30 and C/P ca. 100-120. The severe limitation of the nutrients N and P can have an adverse effect on the growth of microorganisms. In order to obtain optimum nutrient content, various types of waste have to be mixed with each other (see Table 6-5). A long anaerobic digestion process may need additional nutrient supplements.

Presence of microorganisms
Out of the microorganisms playing a role in the anaerobic digestion process, methanogenic bacteria are the most susceptible species. They are characterized by long time of growth, namely 4-6 days for the doubling of biomass, and are sensitive to changes in living conditions (pH, temperature).

The inoculation of fresh feedstock is required in an anaerobic digestion reactor to speed up the reaction processes. It is reported that the effective percentage of inoculation for acidogenic fermentation of organic municipal wastes in a plug-flow system is approximately 30% (w/w) (Yebo 2011). While reducing the amount of inoculum required in a digestion reactor will increase the reactor's utilization efficiency, it will also increase the retention time. The digestate or leachate of digestion can be recycled to inoculate the fresh feedstock.

Harmful substances

- High concentration of salts,
  High salt levels cause bacterial cells to dehydrate because of osmotic pressure. Some microorganisms are more susceptible to osmotic pressure than others. In anaerobic digestion experiments, inhibition and fifty percent inhibition was observed at 65 and 95 g/dm^3, respectively for adaptation to NaCl. Initial inhibition for the reactor adapting to NH_4Cl occurred at 30 g/dm^3 and a 50% inhibition was observed at 45 g/dm^3 NH_4Cl (FAO 1992).

- Heavy metals,
  Heavy metals are toxic to microorganisms and thus prevent biological decomposition processes.

- High concentration of ammonia (internal inhibition),
  Ammonia is the result of the degradation of proteins found in food waste and garden waste and this can lead to an accumulation in the anaerobic digestion reactor. It is generally accepted that ammonia concentrations of over 2-3 g/kg are inhibitory, if not toxic, to non-adapted systems, especially where the pH is high enough to enable a significant fraction to be in the unionized NH_3 form, which is the most toxic. The following equation describes the equilibrium:

$$NH_4^+ + OH^- \leftrightharpoons NH_3 + H_2O$$
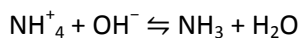
When the pH is sufficiently high (above 7,5), the equilibrium is shifted to the right and the free ammonia form predominates. There is a temperature and pH-dependent relationship between free ammonia and ammonia (Trzcinski 2009).

There are conflicting reports on the effect of total ammonia (unionized plus ionized form) concentrations. It was reported that concentrations between 1500 and 3000 mg/dm^3 were inhibitory at pH levels above 7,4 and those in excess of 3000 were toxic regardless of pH (Trzcinski 2009).

However, the lag phase time was dependent on the NH_3 level, but not on NH_4^+ and when NH_3-N was higher than 500 mg/dm^3, a notable shock was observed; this confirms the known fact that the NH_3 level is a more sensitive factor than the NH_4^+ level for an unacclimatized bacterial system.

Temperature and retention time
Anaerobic digestion can be run at both mesophilic and thermophilic temperatures. Mesophilic temperatures range from 10° to 45°C with the optimum at 35° to 38°C. The retention time is normally 15-21 days. Anaerobic digestion under mesophilic conditions exhibits a poor start-up performance and therefore takes more time to be completed than thermophilic anaerobic digestion.

Thermophilic temperatures range from 45° to 75°C with the optimum at 50° to 55°C. The retention time is ca. 14 days. The thermophilic operation of the anaerobic digestion process was developed later and it has been established as a reliable and accepted mode of operation. Operating digestion systems at thermophilic conditions (55 °C) can accelerate the digestion process. This also provides the added benefit of increased pathogen reduction during the anaerobic

phase. The added amount of heat required for thermophilic operation can be offset by the higher gas production yields and rates.

Thermophilic operations have been proved to be a reliable and acceptable option for the digestion of organic municipal waste. There is also considerable interest in applying digestion at thermophilic

conditions (55°C) to treat the biowaste fraction of municipal waste. The biogas yield of the anaerobic digestion of biowaste at thermophilic conditions is much higher than that in mesophilic conditions (Yebo 2011).

With regard to biowaste hygienization, the draft Biowaste directive sets the following minimum requirements for a digestion process:

- Minimum temperature of 55 °C for 24 hours without interruption and hydraulic dwell (retention) time in the reactor of at least 20 days.
- In case of lower operating temperature or shorter exposure:
  – the biowaste shall be pre-treated at 70 °C for 1 hour, or
  – the digestate shall be post-treated at 70 °C for 1 hour, or
  – the digestate shall be composted.

6.3.2 Anaerobic digestion technology

In single stage wet fermentation, the four anaerobic digestion steps take place in one reactor, i.e. they are not separated in time or in space. These types of plants have the advantage of being simple and easy to operate and they have a low investment cost. On the other hand, the biogas output is lower in comparison to multi stage fermentation. The retention time on this type of fermentation oscillates between the 14 and 28 days depending on the feed and operating temperature (Verma, 2002). Process parameters of single stage wet digestion technology are shown in Table 6-14.

Mechanical pre-treatment involves the following steps:

- feeding the system with wet biowaste and shredding it to obtain the optimum particle size of aerobic conditions. An anaerobic digestion rate increase with a greater surface area is possible for microorganisms. Ferrous metal removal can be introduced, but rather for mixed waste digestion.

Table 6-14. Process parameters of single stage digestion technology

Parameter	Unit	Range
Retention time	d	18 - 21
Degradation rate	% organic DM	47 - 60
AT4 after digestion	mgO_2/g DM	15 - 29
GB21 after digestion	dm^3/kg DM	13 - 52
TOC of eluate after digestion	mg/dm^3	200 - 390

- the mixed solution is sent to the pulper and mixed with process water where the light fraction (plastic contaminants) and the heavy fraction (glass, stones) are removed.
- additionally, a hydro-cyclone can be used to separate the solids (sand) from the liquids, producing a clean, homogenous and ready for digestion pulp.

The process scheme of a single stage plant is shown in Figure 6-13.
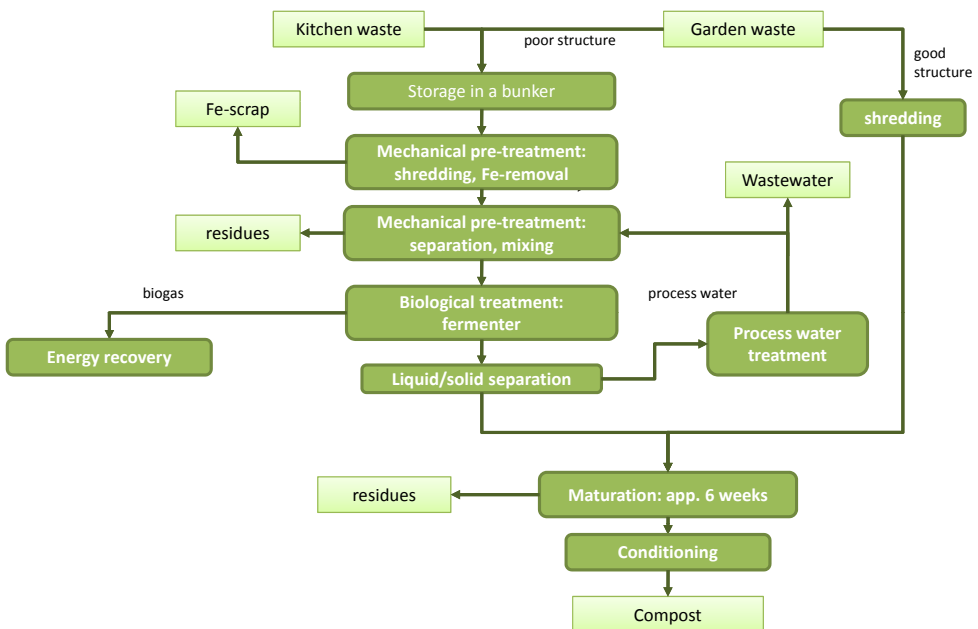
Figure 6-13. One-stage wet fermentation process (adapted from Bidlingmaier 2002)

The actual anaerobic digestion takes place in a reactor, often referred to as **fermenter or digester**. The pulp is heated and enters the reactor where

hydrolysis, acidogenesis, acetogenesis and methanogenesis take place. The digester contents are continuously mixed using either a mechanical device or compressed biogas. The biogas is stored in a separate tank and burned in a CHP to produce thermal and electrical energy while the rest of the substrate is mechanically dewatered and sent to post-composting (CCI BioEnergy, 2009).

The mass balance of a one-stage wet digestion process is presented in Figure 6-14.

*) Ignition loss 65 %
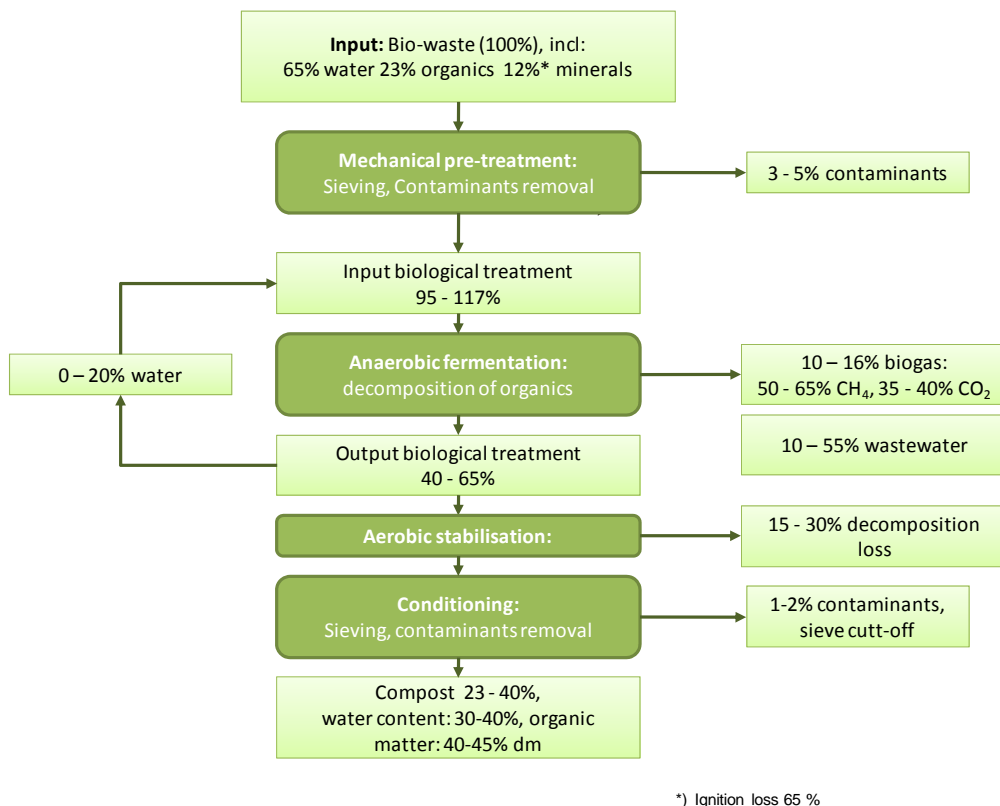
Figure 6-14. Mass balance of a one-stage digestion process (adapted from Bidlingmaier 2002)

In **multi-stage fermentation**, two or more reactors are applied to complete the anaerobic digestion. The idea of utilizing several reactors is to separate in space and time the hydrolysis and methanogenesis phases with the intention of increasing the biogas yield and making it safer to operate. The retention time in multi stage fermentation is approximately seven days, three days for the methanogenesis and between two and four days for the hydrolysis phase.

In Figure 6-15 a scheme of a multistage wet digestion process is presented. The mechanical pretreatment steps until clean homogenous pulp for digestion is provided are the same, as well as biogas-to-energy processes are similar to the single stage technology. The differences begin with pulp processing. In the multi stage process, the pulp is hygienized and centrifuged producing two fractions: one fraction is sent for hydrolysis and the other to the methane reactor. A fraction with "a high amount of already dissolved organic material is pumped directly into the methane reactor. The dewatered solids are mixed with process water and fed into the hydrolysis reactor to dissolve the remaining organic solids. After 2-4 days, the suspension is dewatered and the resulting liquid is also fed into the methane reactor" (BTA, 2007), while the solids are sent to post-composting. The wastewater resulting from the methane reactor is then treated by flocculation and denitrification.
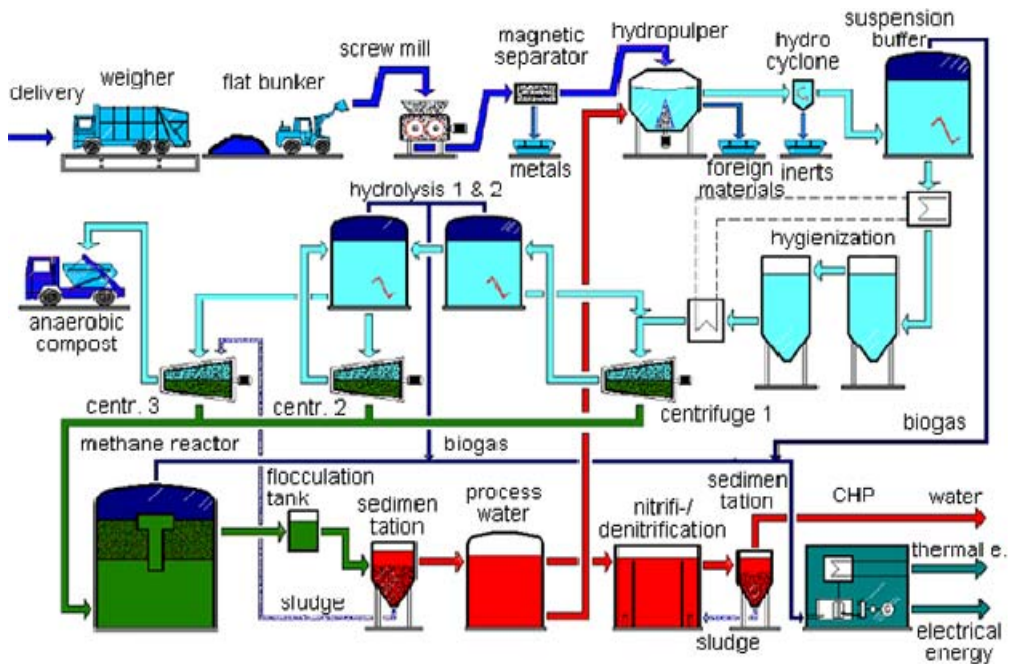
Figure 6-15. Multiple stage digestion process based on BTA technology (source: www.bta-international.de)

A comparison of single and multi-stage digestion is given in Table 6-15.

Table 6-15 Comparison of single and multi-stage wet anaerobic digestion, based on BTA technology (source: www.wtert.eu)

Operation parameter	BTA single stage	BTA multi stage	
	Digester	Hydrolysis	Methanization
Retention time [d]	14 - 16	2 - 4	3
T [°C] Mesophilic	37	37	37
T [°C] Thermophilic	55	-	-
Biogas [m^3/ton $_{biowaste}$]	80 - 90	100 - 130	
Methane content [%]	60 - 65	30 - 50	65 - 75
Heating value [MJ/m^3]	22 - 25	22 - 25	

6.3.3 Biogas generation and quality

The biogas potential of various organic compounds is presented in Table 6-16. It clearly shows that lipids have the highest gas generation potential followed by carbohydrates and proteins.

Table 6-16 Biogas potential of organic compounds

	Chemical composition	Gas generation, m^3/kg ODM	Methane content, %	Energy content of gas	
				MJ/ kg ODM	kWh/ kg ODM
Carbohydrates	$C_6H_{12}O_6$	0,83	50	15,1	4,2
Proteins	C_4H_6ON	0,72	71	18,4	5,1
Lipids	$C_{15}H_{31}COOH$	1,43	70	36	10

In Table 6-17 the potential biogas yield of various waste types is presented. It can be seen that municipal biowaste has an average gas biogas potential of 100 – 170 m^3/tone. The reported biogas yield per kg organic dry mass of biowaste varies from 0,20 to 0,50 m^3/kg DM (Hupe and Stegmann 1998). Out of all waste types, fats have the highest biogas potential.

Table 6-17 Biogas potential of various waste types (source: www.biogas-info.co.uk)

Feedstock	Dry Matter %	Biogas Yield m^3/ton
Cattle slurry	10	15-25
Pig slurry	8	15-25
Poultry	20	30-100
Biowaste (municipal)	32	100-170
Maize silage	33	200-220
Grass silage	28	160-200
Maize grain	80	560
Whole wheat crop	33	185
Wheat grain	85	610
Crude glycerin	80	580-1000
Rape meal	90	620
Fats	up to 100	up to 1200

The produced biogas, after cleaning (e.g. H_2S removal), is used for energy production. In a combined heat and power unit (CHP) the biogas is combusted producing both electricity and heat. The electrical efficiency of a CHP unit is approximately 30% and the thermal efficiency is approximately 56% (Vogt et al. 2002). Electricity is fed to a network and the produced heat is mostly utilized for the plant's own needs (heating of the buildings, heating up the feedstock). Surplus heat (especially in summer months) is often not utilized. A recent development is cleaning the biogas and feeding it into the gas network. In this way, a higher efficiency of energy recovery can be achieved.

6.3.4 Post-composting of the digestate

The dewatered digestate is typically processed further in an aerobic composting process. The minimum duration of the composting process is 4 weeks, of which a minimum of 1 week should take place in an enclosed reactor or in a building with process air control and cleaning in order to reduce emissions of odors. The remaining weeks can be performed in windrows with mechanical turning. Bulking materials are added to allow greater porosity and oxygenation during the aerobic finishing step.

In Table 6-18 the stability of biowaste after digestion and aerobic stabilization is presented. After digestion the waste is not stable enough. It requires at least a

further 10 weeks to achieve the recommended parameter values for finished compost respiration – GB21< 20 dm^3/kg DM and AT4 < 5 mgO$_2$/g DM

Table 6-18. Impact of digestion and aerobic stabilization on biowaste stability

Parameter	GB21 dm^3/kg DM	AT4 mgO$_2$/g DM	TOC eluate g/m^3	Celulose/Lignin
Raw waste before anaerobic digestion	151	55	2500	2,2
After anaerobic digestion	52	25	390	1
+ 3 weeks of aerobic stabilization	35	15	284	0,4-0,6
+ 8 weeks of aerobic stabilization	27	7	186	-
+ 12 weeks of aerobic stabilization	16	3	132	-

6.4 Composting versus anaerobic digestion

A summary of process parameters for both biowaste biological treatment options is provided in Table 6-19.

Table 6-19. Comparison of process parameters of composting and digestion

Parameter	Composting	Fermentation
Microorganisms	Bacteria, Fungi, Actinomycetes	Various bacteria
Biomass growth	50% of the decomposed organic carbon	10% of the decomposed organic carbon
Conditions		
Oxygen	5 - 15% in the pores	none
Water content	40 - 60%	60-90%
Nutrients	C/N = 20 - 35/1	C/N = 10-30/1
pH value	5,5 - 8	6,5 - 8
Receiving area	flat or vertical bunker	flat or vertical bunker
Pre-treatment	Dry	dry and/or wet
Inputs	organic waste, plus structure material, air	organic waste, plus water, heat

Parameter	Composting	Fermentation
Outputs	compost, exhaust air (treatment in biofilter), leachate, condensate	biologically stabilized, wet organic waste (dewatering though aerobic stabilization necessary), biogas (for energy recovery), wastewater (biological treatment necessary)
Decomposition rate	ca. 55% ODM	45-67%

6.5 Emissions from composting and digestion plants

As a result of the aerobic decomposition of the organic fraction, emissions to air and water occur. Decomposed organic carbon will be predominantly released from the process as air emissions. Based on literature data (Vogt et al. 2002), it can be concluded that throughout the whole composting process only approximately 1,3 % of the organic carbon content in the input waste is released to water.

6.5.1 Emissions to air

Air emissions typical for biological degradation processes include

- carbon dioxide and methane
- organic compounds that are metabolized or generated by biological reactions
- volatile substances that are stripped out from the original waste
- microbiological pollutants.

Heavy metals that are contained in waste mostly remain in the solid fractions and their presence in the waste gas stream is negligible.

Emissions of volatile organic compounds from the composting process and the aerobic treatment of digestate are presented in Table 6-20. It can be clearly seen that emissions from the aerobic treatment of fresh biowaste generates much higher emissions than aerobic post-treatment of digestate, which has been already stabilized in an anaerobic process.

Table 6-20. Volatile organic emissions from aerobic waste treatment (mg/m^3) (McDougall et al. 2001)

Compounds	Aerobic	Maturation after anaerobic	Ratio aerobic/anaerobic
Alcohols	283,6	0,033	8593,9
Ketones	150,4	0,466	322,7
Terpenes	82,4	2,2	37,5
Esters	52,7	0,003	17.566,7
Organic sulfides	9,3	0,202	46,0
Aldehydes	7,5	0,086	87,2
Ethers	2,6	0,027	96,3
Total Volatile Organic Compounds	588,5	3,017	195,1
NH$_3$	158,9	97,6	1,6
Total	747,4	100,617	7,4

Emissions of odors constitute a serious problem in biological treatment plants. Odors are related to many organic compounds which form during the partial decomposition of organic solids.

To measure the amount of odors, an odor is diluted to certain amounts to reach a detection threshold. The detection threshold is the concentration of an odor in air when 50% of a population can distinguish between the odorous sample and an odor free blank. This way of measuring odor concentration is the most widespread method to quantify odors. It is standardized in CEN EN 13725:2003 and is based on the dilution of an odor sample to the odor threshold (the point at which the odor is only just detectable to 50 % of the test panel). The numerical value of the odor concentration is equal to the dilution factor that is necessary to reach the odor threshold. Its unit is the European Odor Unit, OUE. Therefore, the odor concentration at the odor threshold is 1 OUE by definition. Threshold values for selected composting emissions are presented in Table 6-21.

The main factors influencing odor emissions at composting plants are the storage of raw materials – wet, high nitrogen-containing feedstock, release of ammonia and volatile organic compounds from the compost pile, anaerobic conditions in the compost pile and high temperatures.

Table 6-21. Detection thresholds for selected chemical compounds (adapted from Kalogerakis et al. 2010)

Chemical Compound	Detection Threshold (ppm$_v$)
Acetaldehyde	0,067
Acetic acid	0,008
Ammonia	0,370
Butyl mercaptan	0,0005
Diethyl sulfide	0,0008
Dimethyl amine	0,047
Dimethyl sulfide	0,001
Ethylamine	0,026
Ethyl Mercapton	0,0005
Hydrogen sulfide	0,00047
Methyl mercaptan	0,0011
α-Pinene	0,011
Butanone	0,29

ppm$_v$ – part per million (volume related)

Areas with a high emission potential in a composting plant are
- receiving area - leachates, odors
- mechanical pre-treatment: screening: dust, odors
- biological treatment - leachates, condensates from exhaust gas, odors
- post-treatment - odors, dust (screening, elimination of glass etc.)
- end product - heavy metals, organic pollutants (compost)

In Table 6-22 empirical data on odor emissions from various areas in a composting plant are shown. The highest odor emissions come from the biological treatment stage.

The exhaust air originating from the composting process itself is saturated with water and has a temperature range of about 20–40 °C in Central European climates with an odor concentration that is in the range of several thousand to several ten thousand OU/m3, but can be as high as 100.000 OU/m3 and above.
Emissions to air from a digestion plant are similar to a composting plant with regard to the waste receiving area, pretreatment, compost maturation, storage and final treatment. The largest difference is in the biological treatment step,

which in the digestion plant is encapsulated and thus causes no emissions. Instead some, but significantly lower emissions result from biogas combustion at a CHP unit.

Table 6-22. Odor emissions from a composting plant (adapted from Bidlingmaier 1993)

Composting stage	Air flow (m^3/h)	Odor emission (OUE/h x 10^4)
Receiving area	45 320	2 131
Pretreatment	14 825	156
Composting area:		
- sucking	13 450	15 380
- blowing	6 800	3 400
Storage	3 400	34
Final treatment	3 300	39

Waste air purification

Simple odor mitigation measures include maintaining aerobic conditions during the whole composting process and keeping the temperature lower than 60°C. Modern composting plants are equipped with air purification systems.

6.5.2 Biological waste gas treatment

Biological exhaust air treatment systems use state-of-the-art technologies for purifying emissions from the composting processes. The most relevant systems are bioscrubbers and biofilters. Which one of them is used or whether or not a combination of both is the best option depends on the specific situation and may be tested before implementation.

Bioscrubbers

Bioscrubbing is a process of biological waste gas treatment in which exhaust air is "washed" in an absorber with a scrubbing liquid. The scrubbing liquid is subsequently drawn off and transferred to an activation tank in which the constituents absorbed to the liquid are degraded by microorganisms. The liquid is continuously cycled through the process (VDI 3478, 1996). Bioscrubbers can be supplied with carrier materials to provide additional surface for microorganisms. The rapidly formed biofilm has a large effect on the degradation efficiency of the scrubber. To avoid clogging, the carrier material should have large pores and should be cleaned frequently. Nutrients may be added to the scrubbing liquid if

necessary to allow optimal conditions for the microorganisms (Schlegelmilch et al. 2005).

Biofilters

A biofilter is a fixed-bed reactor filled with biologically active packing material. Microorganisms settled on the media feed on the organic compounds that are contained in the waste gas. Biofilters usually are combined with wet scrubbers. The scrubbers are used to humidify the air passing into the filter in order to avoid drying of the filter material. Frequently used biofilter media are compost, peat, root wood, bark, wood chips (normally used as bulking agent) and various combinations. One important property of the media is its ability to store water. Biofilters may be open to the atmosphere or enclosed (VDI 3477, 2004).

In **open biofilters** the exhaust air flows through the biofilter bed in an up-flow direction. The biofilter bed is up to 2 m deep. A problem with open biofilters is that the biofilter media are directly exposed to climatic conditions which may influence the filter performance. A hot and dry climate may result in the drying of the filter media even though the waste air may be humidified. In these cases, a sprinkler system on top of the biofilter material may be installed. The opposite problem has been reported in places with very humid climates. In one case, heavy rainfall forced the operator to cover the filter (Accortt et al., 2001). Elements of an open biofilter are schematically presented in Figure 6-16. Process steps of biofiltration technology.

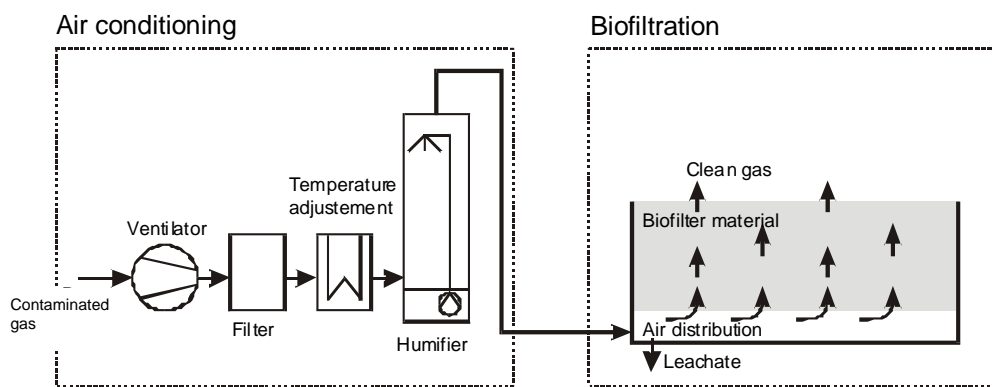**Figure 6-16. Process steps of biofiltration technology (adapted from Bockreis 2001)**

Enclosed biofilters are less affected by weather conditions than open filters. In addition, enclosed biofilters promise higher efficiencies than open biofilters (Heining, 1998). The air usually is distributed at the top of the biofilter and captured at the bottom. The water from the saturated air stream moisturizes the

biofilter material while trickling down by gravity. Additional sprinklers may also be installed, as experience has shown that drying of the filter media remains a problem (Schlegelmilch et al., 2005). In Table 6-23 efficiency of exhaust gas in biofilter is presented with regard to different pollutants.

Table 6-23. Biofilter efficiency (den Boer et al. 2005)

Parameter	Effect of purification, %
Methane	50%
Non-Methane VOC	83%
Total Organic Carbon (TOC)	82%
Ammonia	90%
N_2O	0%

In Table 6-24 the efficiency of a combined system: bioscrubber/biofilter is presented with regard to the removal of volatile organic compounds and odor degradation. Clearly, the combination of both processes shows an improvement in cleaning efficiency.

Table 6-24. Efficiency of bioscrubber-biofilter gas purification (adapted from Schlegelmilch et al., 2005).

Treatment device	VOC removal, %	Odor degradation, %
Bioscrubber	29	28
Bioscrubber/biofilter (coke–compost)	97	95
Bioscrubber/biofilter (screened compost)	99	99
Wet scrubber/open single bed biofilter (root wood)	96	95

6.5.3 Emissions to water

Wastewater from **composting plants** includes:
- leachate from waste stored on the plant surface,
- leachate and condensate from biological treatment amounting to 10-60 dm^3/t waste and 5 – 300 dm^3/t waste, respectively,
- condensate from the biofilter.

Leachates are heavily polluted, especially by organic compounds (high TOC); condensates show significantly lower pollution. Wastewater can be partly recycled within the plant.

Table 6-25 Parameters of wastewaters from static pile composting, mg/dm^3 (adapted from Haug 1993)

	Biological Oxygen Demand BOD	Chemical Oxygen Demand COD	Phosphate Phosphorus	Nitrate Nitrogen (NO_3-N)	Organic Nitrogen	Ammonia Nitrogen (NH_3-N)	Total Kjeldahl Nitrogen (TKN)	Alkalinity	pH
Conden-sate	2000	4050	1,87	0,73	139	1140	1279	4030	7,7
Leachate	2070	12400	3,13	0,46	655	905	1560	2930	7,7
Runoff	91	613	0,31	0,16	58	115	173	361	8,2

Water emissions in **anaerobic digestion plants** mainly occur in the fermentation phase. Table 6-26 shows selected parameters of wastewater from an anaerobic digestion plant- dry single stage and wet two stage. Generally, it can be seen that water effluents from anaerobic digestion process are less polluted than those from the composting process.

Table 6-26. Water emissions from the anaerobic digestion process (McDougall et al. 2001)

	Quantity dm^3/ton	BOD$_5$ mg/dm^3	COD mg/dm^3	Ammonium -Nitrogen (NH_4-N) mg/dm^3	Total Nitrogen mg/dm^3	pH
Dry single stage	540	740	1400	250	6	8
Wet two stage	500	60	200	100	na	na

7 Mechanical-biological (pre-)treatment of mixed residual municipal waste

The Landfill Directive prohibits the disposal of untreated waste and introduces targets for the diversion of biodegradable waste from landfills. Diversion can to some extent be accomplished through separate collection of recyclables and their recycling (e.g. paper) and through separate biowaste collection and related treatment in e.g. composting facilities, and by home composting in rural areas. However, recycling of separately collected waste fractions must generally be complemented with the treatment of waste remaining in the residual waste stream (Byström 2010).

7.1 Definitions

Residual waste is defined as the fraction of municipal waste remaining after the source separation of municipal waste fractions such as food and garden waste, packaging, paper and paperboard, metals, glass. It is unsuitable for the production of compost because it is mixed, combined or contaminated with potentially polluting products or materials (draft Biowaste directive).

Mechanical-biological pretreatment (MBP) is an alternative method to the incineration of pre-treatment of mixed or residual waste prior to landfilling. In the Biowaste directive mechanical-biological pretreatment is defined as the **treatment of residual municipal waste, unsorted waste or any other biowaste unfit for composting or anaerobic digestion in order to stabilize and reduce the volume of the waste.** The Green paper on waste defines MBP as techniques which combine biological treatment with mechanical treatment (sorting) with the objective to produce either a more stable input to landfills or a product with improved combustion properties or biogas (anaerobic MBP - an energy recovery process) (COM/2008/0811).

The MBP of municipal solid waste has been applied for over 15 years, initially in Germany, Austria and Switzerland. Over time, this treatment method became an established technology. The capacity for MBP has significantly increased in some European countries, with Italy having by far the largest treatment capacity at 240 kilograms per capita. The share of this treatment option in selected European countries is presented in Figure 7-1. The countries that use this treatment option all use or are planning to use dedicated incineration and co incineration of the refuse derived fuel (RDF) produced to generate energy (EEA 7/2009).

MBP technologies are commonly applied as regional waste treatment facilities for rural areas, small and medium size towns, whereas for large cities and

agglomerations incineration is the most suitable waste treatment option. Figure 7-2 depicts the position of MBP technology in a waste management system.
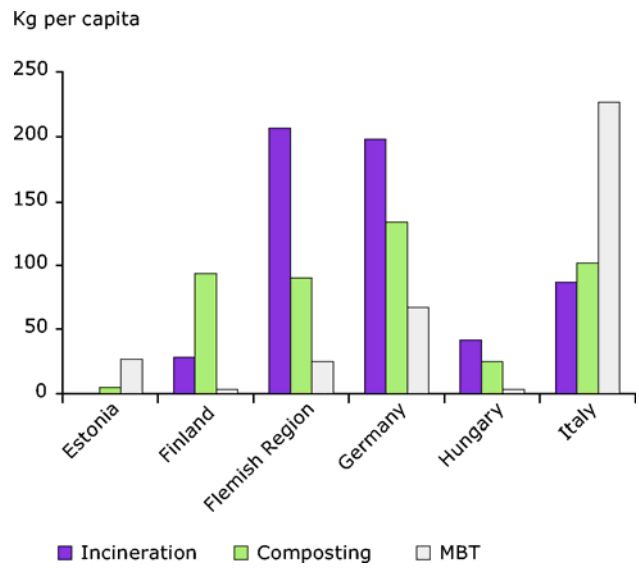

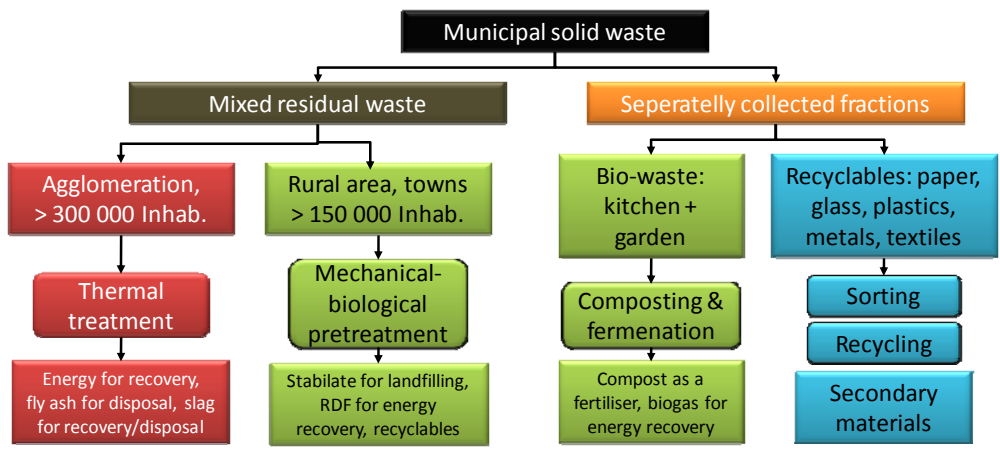**Figure 7-1 Shares of waste treatment technologies in selected European countries (EEA 7/2009)**

Figure 7-2 Position of the MBP technology in a waste management system

7.2 General objectives and related technological configurations

In the past 30 years the treatment of municipal waste has undergone an evolution which can be summarized in the following steps:

1. Landfilling of mixed waste in facilities complying with sanitary landfill standards
2. Separate collection of waste for recycling
3. Technologies for treatment of residual waste – incineration
4. Requirement of pretreatment of all waste prior to landfilling
   – **implementation of alternative methods to incineration waste pre-treatment, i.e. MBP technologies**
5. Energy recovery of the RDF from the MBP plants
6. Gradual elimination of the landfills that do not comply with the technical standards
7. Complete and environmentally sound recovery of residual waste (e.g. Germany plans to completely eliminate landfilling of waste by 2020).

Main objectives of MBP technology are

- Recovery of secondary materials: mechanical sorting enables recovery of the recyclable waste stream;
- Waste derived fuel production: mechanical sorting technologies enable recovery of combustible materials which can be used as an alternative fuel. In anaerobic processes biogas is produced which is an additional fuel for energy recovery;
- Reduction of the waste volume and organic matter through biological decomposition: due to sorting out some recyclables for recycling and combustibles for fuel, the total volume is reduced. Additional volume reduction is due to the decomposition of organics in the biological stage. Due to these measures, the total volume of waste needing to be landfilled can be reduced to 40% or less.
- Biostabilization of the organic matter: the decomposition of organic matter takes place in a controlled manner resulting in a stable residue for landfilling.
- Reduction of pollutants: the pollution arising at the plant can be easily controlled as opposed to pollutants emitted from a landfill. Effluent air is treated in a biofilter before entering the atmosphere and leachate is treated before being discharged.

Thus, with regard to landfill operation the benefits of MBP are

1) Extension of the operation time of a landfill through waste volume minimization
2) More reliable landfill operation through increasing the density of the input and hence increasing landfill stability and preventing water penetration
3) Reduction of emissions of landfill and leachate through the minimization of organic matter / moisture content in landfilled waste.

Depending on the purpose of MBP different types of technologies have been developed:

- **Biological stabilization (or biostabilization)** of waste prior to landfilling. In the initial, simplified applications the whole waste stream was stabilized and landfilled. State of the art plants apply the so called "splitting approach" in which during mechanical pretreatment the high caloric light fraction (containing mostly paper, plastics and textiles) for energy recovery is separated and the remaining heavy low-caloric fraction (containing biowaste, inerts and fine fraction) is biologically stabilized and landfilled. The objective of this concept is to split waste streams with different properties in order to ensure optimal recovery or disposal of theses waste streams.

- **Biological drying (or biodrying)** with the purpose of decreasing water content of the whole waste stream followed by energy recovery and material recycling. The purpose of this concept is the production of RDF, so called "dry stabilate" out of the whole mass of waste and its application in energy recovery (e.g. cement kilns). The RDF fraction is most often separated from the non-combustible waste fractions after the biodrying process. The ultimate goal of this technology is the recovery of all materials present in waste and the complete elimination of landfilling.

  Besides biological drying, physical drying is used in some plants having the same purpose. In this technology thermal energy is used to dry waste. This thermal energy can be obtained for example from landfill gas combustion.

In Figure 7-3, a general configuration of the process steps in both biostabilization and biodrying technology are presented. Installations involving the physical drying method have a similar configuration to biological drying. In 2007 in Germany, there were 46 plants with a total capacity of 5,6

million ton/year, of which 72% were applying stabilization (of which 80% used the aerobic process), 18% were applying biodrying and 10% physical drying.

7.3 Aerobic biological stabilization (biostabilization)

The general process steps of biostabilization technology are presented in Figure 7-4. MBP consists of mechanical and biological process stages and their combination.
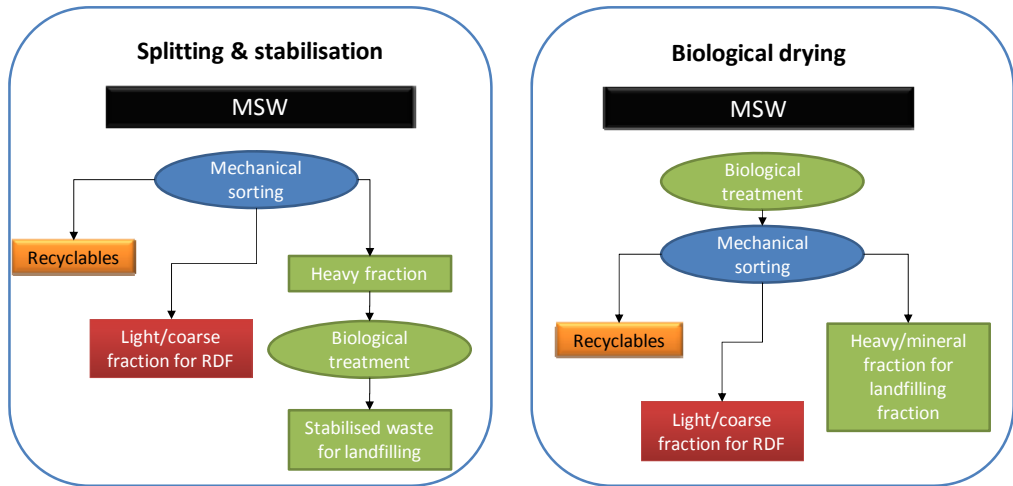

Figure 7-3. Configuration of main process steps in biostabilization technology (left) and biotreatment technology (right)

7.3.1 Mechanical treatment

The "mechanical" element is usually an automated mechanical sorting stage. This includes removal of contaminants and, depending on the plant option, separation of fractions for recycling (metals, plastics, paper, wood, composites) and energy recovery (high caloric waste). Mechanical treatment typically involves bag openers, conveyors, industrial magnets (separation of Fe-metals), eddy current separators (for non Fe-metals), drum screens, ballistic separators, air separators to separate heavy/light fraction, spectroscopic separators to separate pre-defined materials e.g. plastics, paper, glass, etc, shredders and other systems, or the sorting is done by hand.

7.3.2 Biological treatment

Aerobic biostabilization is based on the same principle as composting. The term biostabilization is used to differentiate between these two treatment options and to stress that the output is not compost but stabilate. The process parameters will not be described any further because they are analogical as in the composting process.
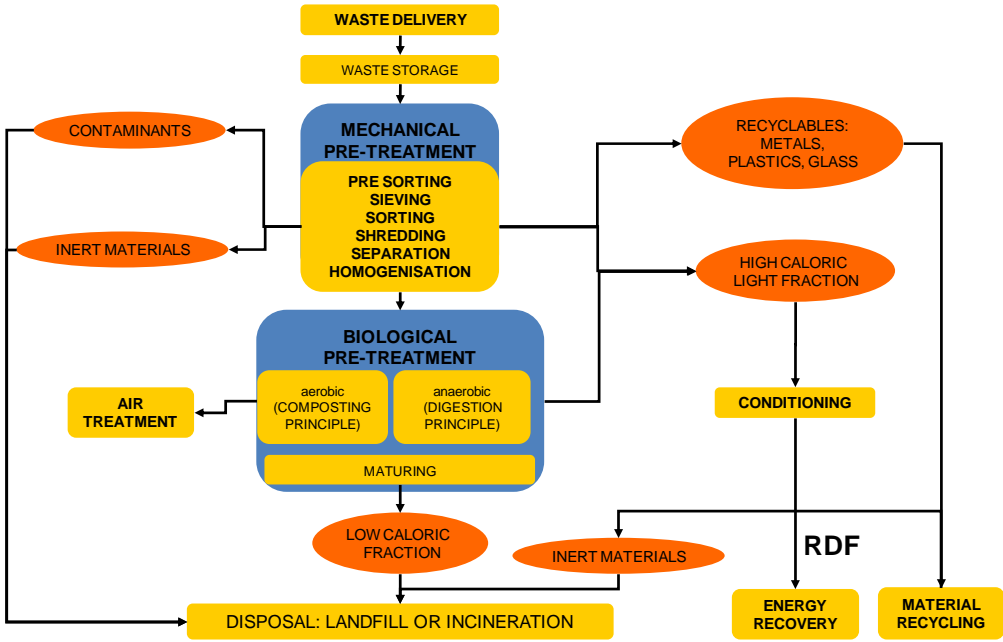

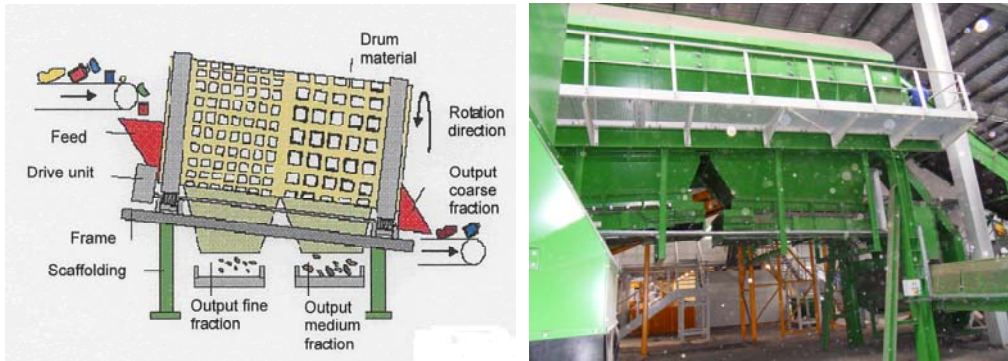**Figure 7-4 General process steps of biostabilization**

Figure 7-5 Drum screen (trommel) allowing the separation of three fractions: coarse, medium and fine
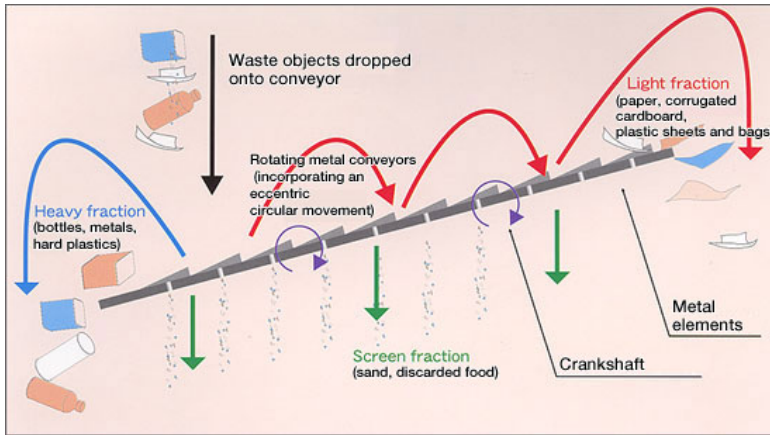
Figure 7-6 Principle of a ballistic separator (source: www.mrc.co.jp)
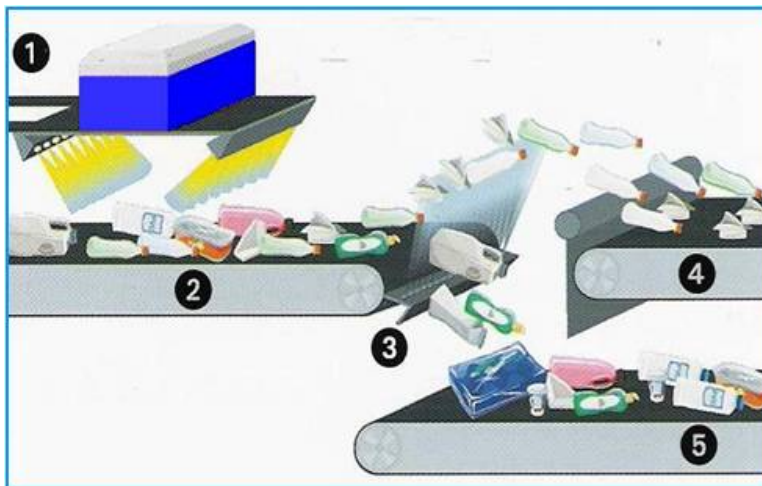
1.Scanner; 2.Conveyor with material in entry; 3. Air jets;
4. Conveyor with material selected, 5.Residual fraction

Figure 7-7 Principle of a spectroscopic separator, source: www.keyglobalitalia.it

The technologies applied for aerobic biostabilization are also analogical to those applied in the composting process and include windrows with or without aeration, containers, boxes, drums and tunnels. Figure 7-1 shows the MBP technology providers with the indication of the type of technology they supply.

Table 7-1. Overview of MBP technologies (Juniper 2005)

Technology supplier	Biostabilization				Biodrying	Percolation	Anaerobic digestion					
	Tunnel	In-hall	Continuously agitated	Simple covered			Wet, single-stage mesophilic	Wet, single-stage	Dry, single-stage mesophilic	Dry, single-stage thermophilic	Wet, multi-stage mesophilic	Wet, multi-stage thermophilic
ArrowBio											√	
Bedminster			√		√							
Biodegma				√	√							
BTA											√	
Civic			√									
Ecodeco					√							
GRL		√				√					√	
Grontmij								√				
Haase											√	
Herhof					√							
Hese												√
Horstmann	√				√							
ISKA						√					√	
Komptech												
Linde	√			√						√	√	
Nehlsen					√							
New Earth		√										
OWS										√		
Ros Roca	√						√					
Rumen												
SBI								√				
SRS	√											
Sutco		√										
Valorga									√	√		
VKW		√										
Wastec			√									
Wehrle					√	√					√	

Among the biostabilization technology providers Horstmann, Biodegma, Linde and Bedminster are the ones with the most plants constructed. Horstmann has supplied tunnel composting technology to most of the MBP plants. The majority of the plants produce stabilate for landfilling. Biodegma plants produce stabilate for landfilling and fuel for energy recovery (out of the coarse fraction). Linde supplies tunnel composting technology. The plants using Linde technology produce stabilate for landfilling or land recultivation. Plants with Bedminster technology co- stabilize municipal waste with sewage sludge (Juniper 2005). Thus, in the existing plants the main process output – stabilate is either landfilled or used for land reclamation. The more recent plants separate the light waste fraction for RDF production.

One of the differences between the different MBP technologies is the treatment duration. In Figure 7-8, the time applied for waste treatment in various technologies is provided. Biostabilization generally requires longer treatment times than anaerobic processes. The treatment duration is strongly dependent on the required stability of the process output.
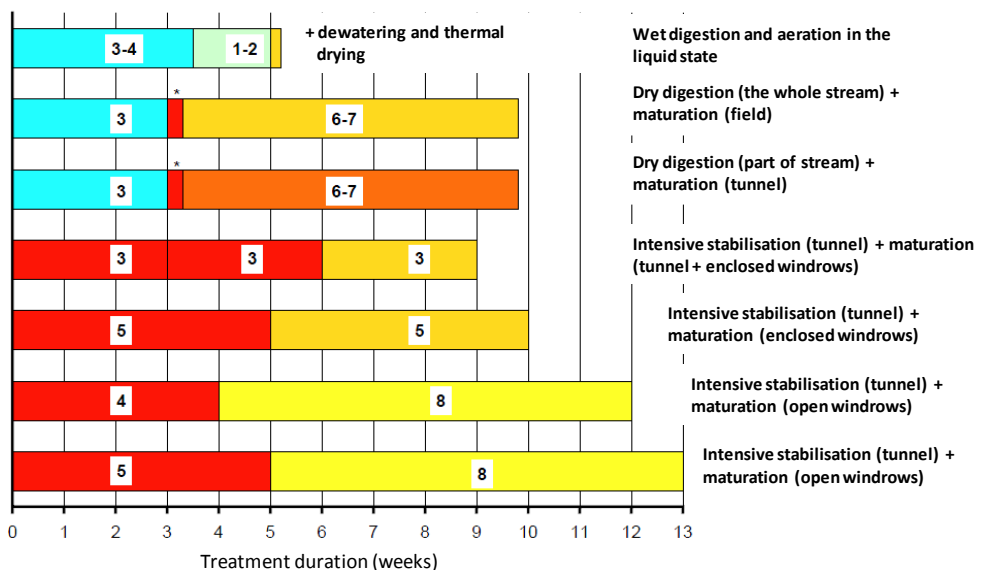

Figure 7-8 Treatment duration of various MBP technology configurations (Ketelsen et al. 2005)

The degree of stability of the stabilate can be determined in the same way as for compost, i.e. based on the respiration index within 4 days AT4 or gas formation

potential within 21 days. In Germany the Ordinance on Environmentally Compatible Storage of Waste from Human Settlements (Abfallablage-rungsverordnung – AbfAblV) of 20 February 2001 requires that stabilate in order to be accepted at class II landfills must fulfill the following requirements:

- Respiration index AT4 ≤ 5 mg O_2/g DM
- Gas formation potential GB21 ≤ 20 l/kg DM
- Gross calorific value ≤ 6.000 kJ/kg.

The last parameter implies that high caloric waste fractions, especially plastics, must be removed from the stabilate prior to landfilling.

7.3.3 Mass balance of aerobic biostabilization

In Figure 7-9, the composition of residual waste is given. This data is based on a waste analysis of Polish waste. The granulometric fractions of >80 mm, 20 – 80 mm and <20 mm account for 30%, 40% and 30%, respectively. Clearly, the fraction 20 – 80 mm contains most of the biodegradable waste. The coarse fraction contains burnable materials – paper and cardboard, composites and plastics. The fine fraction is dominated by minerals, street sweepings with some contribution of organics.
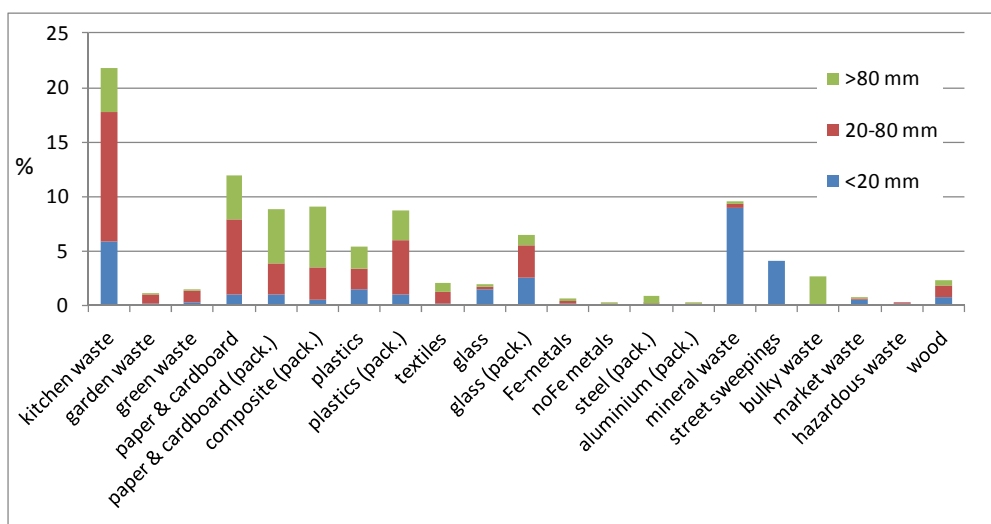
pack. – packaging

Figure 7-9. Composition of residual waste (den Boer et al. 2010)

In Table 7-2, the basic characteristics of waste fractions in the context of their mechanical-biological treatment is given. The fraction 20 – 80 mm shows the highest water content (52%) and relatively high organic content (50% DM). These are very good conditions for biological treatment. Therefore, this fraction is most

suitable for biostabilization. The fine fraction <20 mm is drier and contains less organics. However, it also needs pretreatment prior to landfilling in order to fulfill the landfill acceptance criteria. The coarse fraction has the lowest water content and the highest organic content. However, the organic matter present in the coarse fraction is to a great extent non-biodegradable organics, such as plastics and composites.

Table 7-2. Basic characteristics of waste fractions

Fraction	Share, %	Water content %	Organic dry mass content % DM
‹ 20 mm	30%	32%	40%
20 - 80 mm	40%	52%	50%
› 80 mm	30%	30%	77%
weighted average		39%	55%

In Figure 7-10, a mass balance for the treatment of the given residual waste according to biostabili-zation technology is given. The objective of the described technology is the separation of the light fraction for RDF and stabilization of the heavy fraction before landfilling. The main steps of the treatment are

Mechanical process steps:
- Manual sorting of bulky waste, hazardous waste, recyclables,
- Separation of Fe-metals in a magnetic separator
- Separation on a drum sieve to three fractions: <20 mm (fines), 20 – 80 mm (medium fraction) and >80 mm (coarse-light fraction for RDF)

Biological process steps:
- intensive stabilization in an enclosed reactor for 14 days,
- maturation in windrows for 10 weeks (medium fraction after intensive stabilization + raw fine fraction <20 mm)

Mechanical conditioning:
- sieving of stabilized waste at 40 mm in order to remove combustible fractions,
- cut-off (> 40 mm) added to the RDF fraction.

As a result of biostabilization, stabilate is produced accounting for approximately 42% of the total mass of the input residual waste and the RDF fraction accounting for nearly 26% of the input mass. The water loss accounts for nearly 13% of the input mass and the decomposition of organic dry mass approximately 7% of the input mass.
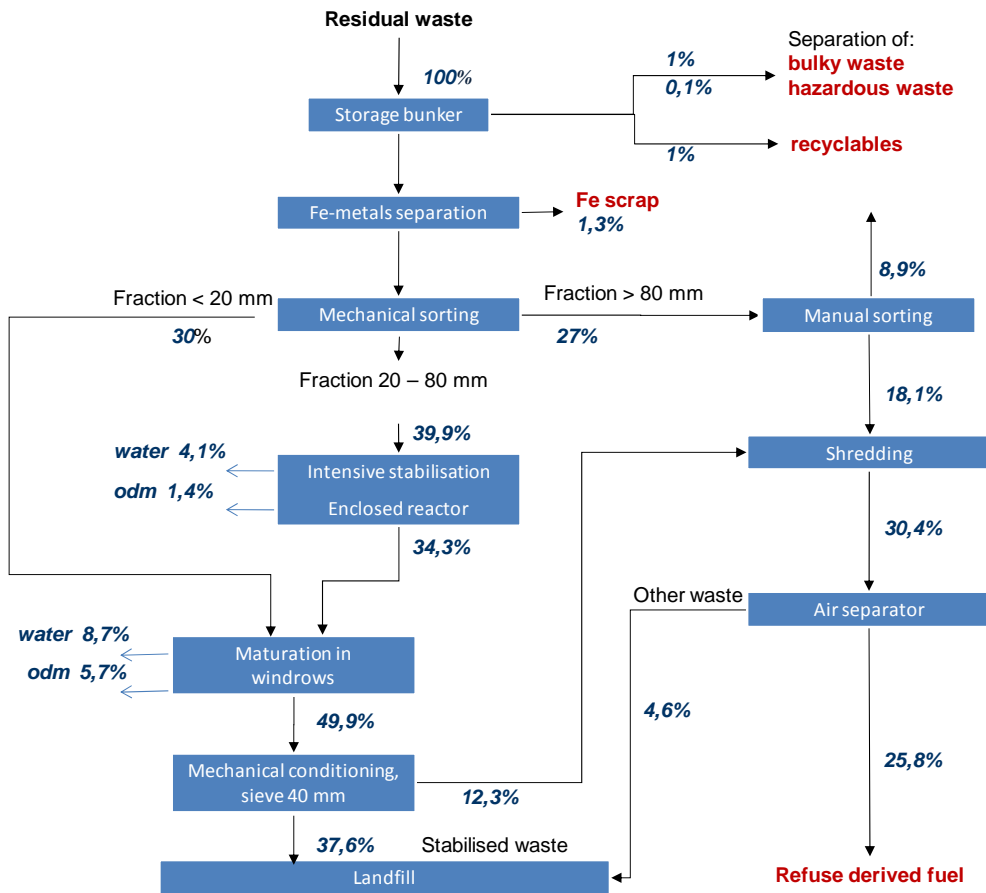
Figure 7-10. Mass balance of the biostabilization process of residual waste (den Boer et al. 2010)

7.4 Biological drying (biodrying) of waste

Biological drying is a variation of aerobic decomposition, used within MBP plants to dry and partially stabilize residual municipal waste. Biological drying MBP plants can produce a high quality solid recovered fuel (RDF), high in biomass content. In the reactors, waste is dried by air convection, while the heat is provided by the exothermic decomposition of the readily decomposable waste fraction. The main difference to biostabilization that here the main goal is to dry and preserve most of biomass content of the waste matrix rather than fully stabilize it. Commercial process cycles are completed within 7–15 days, with mostly H_2O and organic loses of ca. 25–30% w/w, leading to moisture contents of

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<20% w/w. The high airflow rate and moisture removal in the re-circulated process air provides for effective drying. Thus, the main objectives considered in the technological design are

- To achieve a good quality RDF with
  - increasing the energy content through moisture reduction (from above 40% to less than 20%), separation of noncombustible materials,
  - reducing the content of contaminants through mechanical sorting and the removal of metals, fines, PVC etc.
- To pretreat waste at the lowest possible residence time (5-15 days, preferably less than 10 days),
- To use the exothermic property of the biological decomposition process in order to dry the waste, but at a minimum organic matter loss in order to preserve the caloric value,
- To deliver stable RDF, suitable for storage and transport

The design of biological drying reactors includes a container coupled with an aeration system; containers can be either enclosed (Figure 7-11), open tunnel-halls, or rotating drums (Figure 7-12).

7.4.1 Operational parameters and process control

Biological drying technology originates from composting technology and differs mostly in process operation and control. Biodrying requires forced aeration in order to provide the air necessary for the decomposition process and to carry out the water vapor. In the following, the main process parameters are briefly described.

Mechanical pre-processing,
Mechanical pretreatment operations include shredding and/or mixing affecting the physical properties of the matrix, such as the resistance to airflow. Properties such as moisture content, air-filled porosity, permeability, mechanical strength, and compaction of matrix, have the potential to affect the resistance to the flow of air and, in turn, the level of the airflow rate necessary for effective biodrying. Currently, the pre-processing strategy in most biodrying MBP plants is limited to coarse shredding, e.g. at 300–150 mm maximum particle size (Velis et al. 2009).
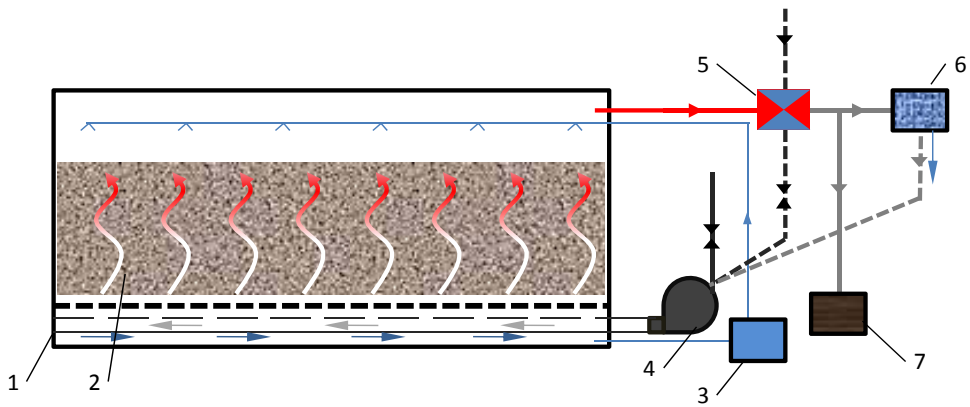
(1) enclosed box; (2) air flow through the waste body, heated by the exothermic aerobic biodegradation of readily decomposable organics; (3) leachate collection and circulation system; (4) forced aeration system with partial air recirculation, mixing ambient air and conditioned process air; (5) heat exchanger; (6) vapour condensator; (8) exhaust air treatment through biofilter or regenerative thermal oxidation (RTO).

Figure 7-11. Biological drying technology, based on Herhof boxes

Aeration system type

The forced aeration of waste is critical for biodrying. It provides the necessary airflow enabling (1) water content removal; (2) heat transfer involving the removal of excessive heat and adjusting the matrix temperature; and (3) supply of O_2 to meet the stochiometric demand for aerobic decomposition. Air management in biodrying varies according to reactor design and process complexity. In enclosed reactors, the usual configuration is positive pressure, forcing air through the matrix flooring and collecting off-gasses through openings located at the top. In static biodrying reactors problems with uneven air distribution occur leading to an uneven waste drying efficiency. To overcome this problem, the bottom of a commercial biodrying reactor (Herhof Rottebox) is divided into 12 parts enabling airflow to vary in each segment facilitating the control of temperature and moisture in the whole reactor (Nicosia et al. 2007). Moreover, some problems may occur in the lower parts of the reactors due to the downward leachate flow. Two solutions to mitigate this problem have been proposed: a) static reactors with alternating aeration systems blowing and pulling air through the matrix and b) rotating reactors enabling the constant mixing of waste and the even distribution of temperature and moisture (Bartha 2008). These solutions have been tested in pilot applications and especially the latter has been shown to improve the temperature distribution and homogeneity of dried waste (Velis et al. 2009).
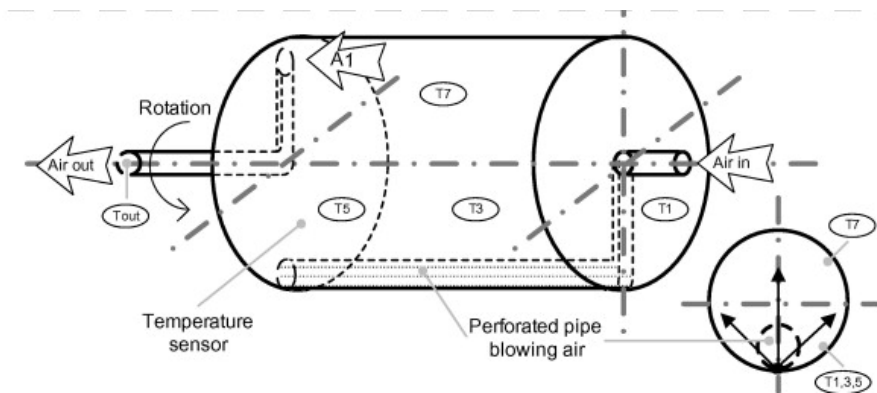
Monitoring points: T: temperature: 1–7 internal, out: exhaust air; P: pressure; rH: relative humidity; Q: airflow rate. BL: blower.

Figure 7-12. Biological drying technology, based on a rotation reactor (source: Velis et al. 2009)

Air partial recirculation systems are often used in biodrying to reduce the volume of exhaust air requiring treatment. Additionally, air recirculation during the winter can be used to pre-heat the cold ambient in-flow air to the reactor. This is done after vapor removal from the exhaust air by cooling in a heat exchanger and cooling tower

Moisture content
The moisture content of residual waste varies depending on the composition of generated waste, separate collection scheme (what kinds of materials are collected) and its rate, time of a year, etc. Generally, residual waste can be characterized by a water content above 40% to over 50%. In biodrying, the moisture content can be reduced to 10–20% (Velis et al. 2009). Water losses can be estimated using values of the airflow rate and the humidity of entering and exhaust air. The phenomenon used in any drying process is the exponential growth of the water carrying capacity of air related to temperature. Figure 7-13 demonstrates the air moisture holding capacity (kg water/kg air) and saturation quantity (g of water/m^3 of air). The air leaving a drying box is normally close to 100% saturated with air (100% relative humidity). Thus, the temperature difference of in-flowing and out-flowing air determines the extra water carrying capacity which can be used for waste drying. The mass balance of water content should include both metabolic water gains and evaporation–convection losses.
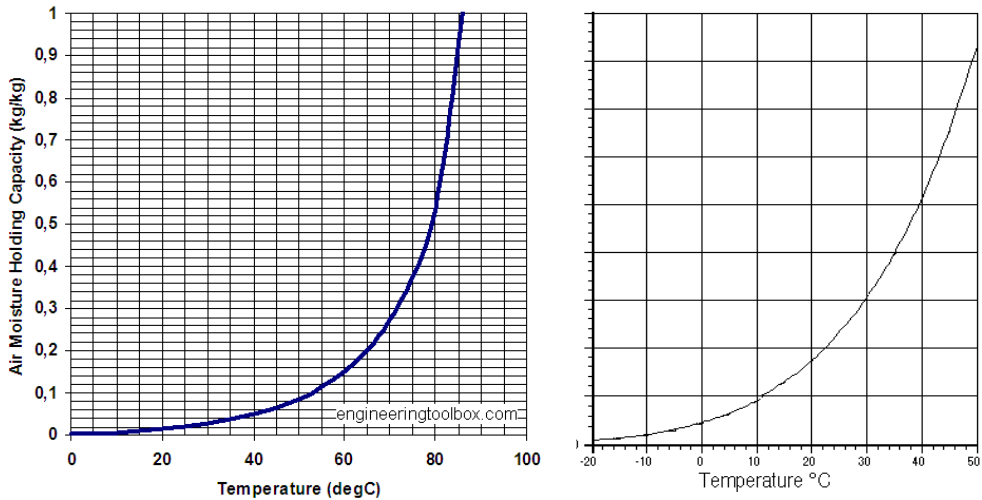
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Figure 7-13 Moisture holding capacity of air (left) and water saturation content of air (right), source: www.engineeringtoolbox.com

Typical water losses of 25% waste weight were reported by Rada et al. (2007) which constitute the sum of both water and organic matter loss. The ratio of weight losses between organic matter and condensed moisture is approximately 1:7 (Velis et al. 2009). The main goal of biological drying is to minimize the water content of the final product. On the other hand, moisture is one of the main parameters of microbial activity. For the composting process, water content of 20% was reported to prevent microbial decomposition (Haug 1993). In this case, the temperature in the reactor will drop to ambient temperature at which the efficiency of the drying process can be very low.

Aeration rate and temperature

Aeration rate is a crucial parameter for biological drying process control. Aeration rates in biological drying are much higher than in biological stabilization. The quantity of air needed for decomposition can be demonstrated by the aerobic decomposition of glucoses (see section 6.3). From the stoichiometric relations it follows that for the aerobic decomposition of 1 g of organic dry matter approximately 4 dm^3 of air is needed. Due to the decomposition of 1 g of organic DM approximately 0,55 g of water is formed (which increases the water content of waste). The amount of air to carry out 1 g of water vapor varies from approximately 3,5 dm^3 at 70°C to approximately 12 dm^3 at 50°C. Considering that the input waste has a water content of 40% and above and that it increases during the decomposition process, it becomes apparent that depending on the process

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temperature significantly higher air rates are required for waste drying than to simply maintain microbial activity.

As explained in the previous section, generally higher out-flow temperatures should improve the process performance due to a higher water/air carrying capacity. On the other hand, too high temperatures (of over 60°C) may become a limiting factor for microbial growth and thus slow down the decomposition rate and energy release.

Results of experiments indicate more effective heat removal at higher aeration rates resulting in lower waste temperatures (Skourides et al. 2006; VanderGheynst et al. 1997) with an optimal waste temperature of approximately 45 °C together with the corresponding airflow rates of approximately 0,023 m^3 /kg DM per hour. Contradicting results were reported in another trial where a temperature of 60°C maintained at the airflow rate of 0,0143 m^3/kg waste per hour (for 150 hours) yielded a 4% higher moisture reduction than the same experiment at 50°C with at specific airflow rate of 0,0164 m^3/kg waste per hour.

Most commercial biodrying processes operate in the temperature range of 40–70°C for outlet air for most of the residence time (Velis et al. 2009). Herhof Rottebox applies a staged outlet air control, consisting of four phases over one week: (1) start up and biomass acclimatization: 40 °C; (2) degradation: 40–50 °C; (3) sanitization and drying: 50–60 °C; (4) cooling to room temperature (60 °C to ambient T) (Nicosia et al., 2007).

Degree of organic matter decomposition
During the decomposition of organic matter energy is released which is used for heating up the waste matrix and for transferring water from the liquid state to water vapor. Thus, the amount of heat needed can be estimated based on the quantity of water to be carried out of the system. On the other hand, the objective is to preserve the maximum amount of biomass in the RDF. This enhances the energy balance of the process and yields a fuel with a higher biomass content that is beneficial in the context of biomass recovery (burning of biowaste is considered neutral in terms of greenhouse gas emissions). Biological drying completed in a short time (150 hours) at a high airflow rate was reported to yield product waste with a higher remaining energy content than the product from longer lasting trials (Adani 2002). Biological drying which took 250 hours to complete resulted in significantly higher biomass decomposition. Thus, the general objective is to shorten the decomposition time in order to limit the overall decomposition, but at the same time obtain a sufficient drying effect.

7.4.2 Mass balance

In Figure 7-14 a mass balance of the biodrying technology for residual waste characterized in Figure 7-9 and in Table 7-2 is provided.

The objective of the presented technology is RDF production out of the total mass of waste through

- Maximizing water evaporation with minimum decomposition of the organic matter in waste (minimum loss of the calorific value)
- More intensive waste aeration than in the case of waste stabilization

and recovery of all waste fractions.

Mechanical pretreatment: encompasses recyclables separation

Biological processes proceed in an enclosed reactor. Initial water content of waste is approximately 39%. In order to ensure a final water content of 15%, the decomposition of approximately 18% ODM is needed. The process duration is approximately 7 days.

Mechanical conditioning/RDF production: comprises separation of heavy/mineral fraction for landfilling and light fraction for RDF,

The total RDF mass accounts for 44% of the input mass. All materials suitable for recycling (Fe-scrap, non Fe-scrap and glass) are recovered. Minerals can be either landfilled or used, e.g. as aggregate in road construction.

7.4.3 Technology providers

In Table 7-3, the main process parameters and selected mass balance items of six commercial providers of biodrying technology are shortly described based on a review performed by Velis et al. (2009). The number of providers is higher than in Table 7-1 because this overview also includes providers which have not yet implemented their technology at an industrial scale. Out of the listed technology providers the Herhof technology has the largest installed capacity. Rotary shredders are used for mechanical pre-treatment (Rennerod: <150 mm; Dresden: < 200 mm) and a magnetic conveyor belt removes the ferrous material. The comminuted Fe-free output is biodried within air- and liquid-tight boxes (''Herhof-Rotteboxes®'') with a capacity of 600 m^3, receiving around 280 ton of waste each.
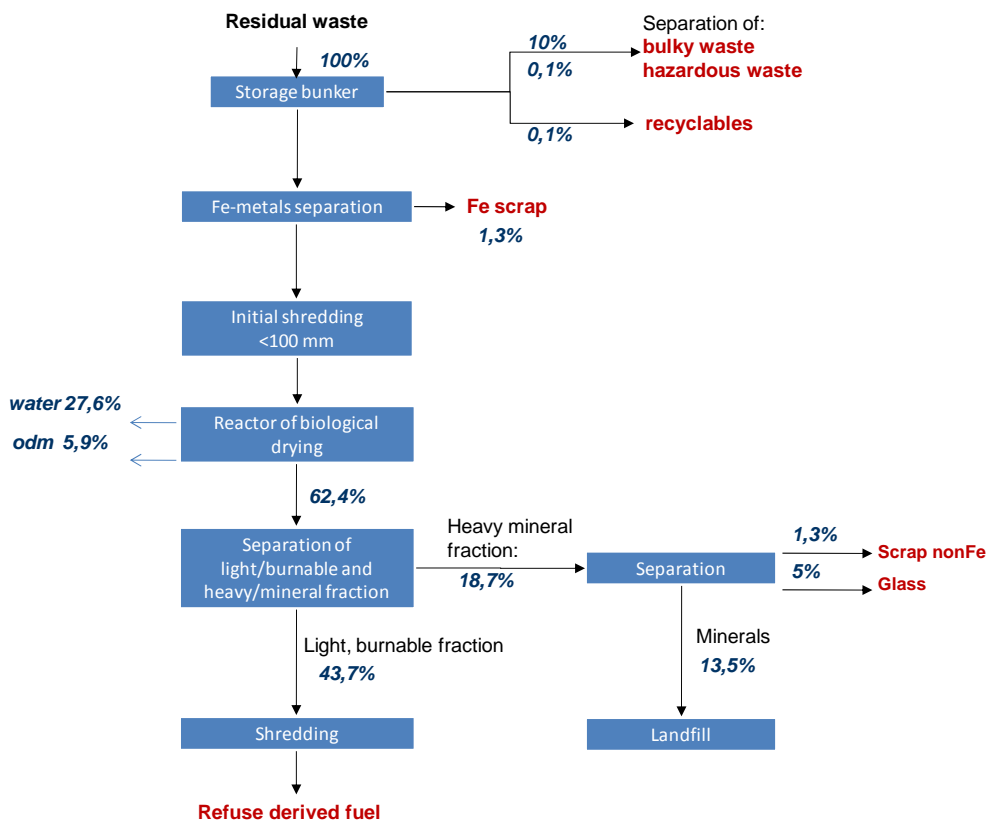
Figure 7-14. Mass balance of the biodrying process of residual waste (den Boer et al. 2010)

The biodrying reactor residence time ranges from 5 to 10 d, with 7 d the most common. The mass losses in the biodrying stage are around 30% w/w input. The initial water content of 42% is reduced to 12% after six days of biodrying in the Rennerod facility. The output RDF amounts to approximately 50% w/w input with a net calorific value of 15 - 18 MJ/kg and water content of approximately 15%. The content of the renewable energy sources (paper, textiles, wood, organic material, loss on ignition of the fine particles) is ca. 65 weight-%, plastics approximately 9% weight and other fossil energy sources (textiles, rubber, composites, etc.) ca. 25 weight-%. The remaining 1% constitutes non-combustibles (Öko Institut 2005). Except for BioCubi® technology by Eco-Deco, the other technologies have not been reported as being installed with significant capacities.

Table 7-3. Overview of mass balances and process parameters applied by biodrying technology providers (adapted from Velis et al. 2009)

Process feature	Eco-deco	Entsorga	Future Fuels	Herhof	Nehlsen	Wehrle Werk
Biodrying reactor type	BioCubi® Windrows in enclosed hall. Downward air suction through waste matrix	H.E.BIO.T.® Enclosed hall	Rotary bio-dryer (RDB), with internal lifters: circular cylindrical Length 25 m drum, Ø 4 m,	Herhof-Rotteboxes® Air and liquid-tight boxes. Upward blowing of circulated dehydrated air through waste matrix	Bio-cells, air and liquid-tight	Percotry® Enclosed tunnels with waste circulation
Installed capacity	10 plant plants in Italy, the UK, and Spain with an overall capacity of ca. 900.000 ton/a	One planned 60.000 ton/a at Westbury, UK	pilot-scale research	8 plants in Germany, Italy and Belgium, with overall operational capacity ca. 1.085.000 ton/a		
Input to the reactor	Residual unsorted MSW		Organic fraction of MWS, mechanically separated from residual unsorted MSW			Dry residuals of MSW percolation, dewatered to water content of 40%
RDF quantity, % w/w input	39 - 53	46 – 53,5		50 - 53	51 - 55	app. 35
Losses $(CO_2 + H_2O(g))$, % w/w input	20 – 28	29 – 31,5		30	16 – 30	15[a]
Solid rejected fraction, % w/w input	17	10 - 15		4 – 5 (dust from the bag filter)	15	22,7 - landfill 7,4 -waste incineration

[a] Biodrying reactor is fed with a fraction of plant input

Table 7-3 cont. Overview of balances and process parameters applied by biodrying technology providers (adapted from Velis et al. 2009)

Process feature	Eco-deco	Entsorga	Future Fuels	Herhof	Nehlsen	Wehrle Werk
Operational variables (manipulated)	Airflow rate		Airflow rate Drum rotation; pH of RDB input: 6,0–8,5; by recirculation of 10–20% w/w of biodried output; Heating cycle for T < 40°C: 30–35 m^3 h^{-1} ton^{-1}; Reactor static for 1–2 h; rotating for 10–15 min; Cooling cycle, for T >55°C: 120–150 m^3 h^{-1} ton^{-1}	Airflow rate 12 segments in bio-cell bottom		
Outlet air temperature T_{out} (°C)	50–70			ca. 50	up to 70	
Residence time	12–15 d	14 d	Aeration bay: 14–72 h, 3 d in Rotary biodryer: water content from 35–40% to 10–15%	5–10 d initial water content of 42% is reduced to 12% after six days biodrying	ca. 7 d	
Mechanical pre-treatment	Shredding 200–300 mm	Trommel	Primary shredding to 80–120 mm (Aeration bay) Trommel at 80 mm underflow fed to rotary biodryer; Metal separation of trommel overflow and sec. shredding at 80 mm, fed to rotary biodryer	Hammermill < 200/150 mm	Shredding <300 mm Single shaft cutting mills suitable for high plastic film contents	
Process air management	Negative pressure		Possible pre-heating of RBD inlet air by air-to-air heat exchanger using heat from aeration bays	Partial circulation for biodryer process air	Partial circulation of screening and refining process air after cleaning	
Air pollution control	Biofilter for biodrying	Biofilter	Biofilter	LARA® Regenerative Thermal Oxidation	Previously biofilter; upgraded to RTO	

7.5 *Anaerobic biostabilization*

The application of anaerobic technology has evolved with time. In Europe it started to be applied for the treatment of sewage sludge in the 1920's. Later, the technology was gradually adopted in the treatment of other organic waste: manures, organic industrial waste, co-fermentation of sewage sludge with municipal waste, digestion of biowaste and finally (since approximately 1995) the anaerobic digestion of residual municipal waste.

The anaerobic digestion technology applied for residual waste is very similar to that applied for the treatment of separately collected biowaste, as described in section 6.3. The major difference lies in the mechanical pretreatment for waste which is more complex for residual waste.

7.5.1 Mechanical treatment

Mechanical treatment in a anaerobic digestion plant for residual waste comprises the following process steps:

- manual and mechanical separation of non-biodegradable substances (light and heavy),
- crushing of the bio-fraction before digestion,
- wet treatment - pulping,
- moistening with process water, inoculation with methanogenic bacteria,
- thermo-alkaline treatment with soda lye and water vapor.

The first step in the mechanical pre-treatment of residual waste is the removal of bulky waste, hazardous waste and other waste which may interfere with the plant operation. Afterwards, shredding and screening to a particle size of <40 mm to <120 mm and the separation of metals take place.

The separation of contaminants such as sand, gravel, glass and plastics as a pre-treatment for anaerobic digestion is chiefly implemented to protect the plant equipment. This applies mainly to wet anaerobic digestion, but recently also to dry anaerobic digestion. Scum layers are a common problem in wet anaerobic digestion.

The contaminant removal method varies depending on the technology applied. Anaerobic digestion processes can be classified into wet and dry processes, where solids are digested, and the percolation process where the waste is washed (wash process) and the water is put through anaerobic wastewater treatment. Percolation, processes with hydrolysis and separation processes yielding wastewater enriched with easily biodegradable organic contents, are counted as a

wash process in this regard (Schu 2008). The contaminant removal technologies used before and after these three digestion methods also differ from each other.

Contaminant removal before and after dry anaerobic digestion.
The most recent trend in dry anaerobic digestion is to separate the fraction <60 mm from the residual waste. This fraction has proven to be the richest in biodegradable matter and still free from foils which disturb the process. The fraction over 60 mm may be used for RDF production and may require applying additional drying (e.g. biological drying to reduce its water content)

The mineral non-biodegradable fraction (stones, gravel, glass, hard plastics) is often removed in a ballistic separator (e.g. Dranco technology – see Figure 7-15) before entering the digester. However, some hard bio-degradable materials (e.g. round vegetables and fruit – potatoes, apples, etc.) are also removed at this stage and thus lost from further processing.

Sand and plastic foils are not removed before anaerobic digestion and are put into the fermenter together with the biomass <40 mm to <60 mm. Some plants were reported to have problems with sediments, e.g. in Hille (Dranco), Barcelona Ecoparque II (Valorga) and Rioja (Kompogas) in Spain (Schu 2007). For this reason, the capacity of the heavy fraction separators had to be refitted to achieve a higher separation rate. The Dranco plant in Kaiserslautern is equipped with a ball mill for the pretreatment of crushing glass and stones to avoid problems with sedimentation in the dry anaerobic digestion process (Schu 2007).

Within the anaerobic digestion stage recirculation of the fermentate is usually practiced in order to provide the fresh waste with necessary microorganisms. The excessive fermentation residue is dewatered and further biostabilized in the aerobic maturation process. The access wastewater is treated. The dewatering process is performed using various equipment, e.g. screw press. The dewatering equipment may suffer damage from the high load of contaminants and thus some existing plants were refitted with additional sand removing equipment.

Contaminant removal before and after wet anaerobic digestion.
In wet anaerobic digestion non-biodegradable contaminants strongly interfere by sedimentation and by building scum layers. Consequently, many different methods for the removal of contaminants have been developed. In the first mechanical step of wet anaerobic digestion technology, the organic fibers are disintegrated in the mixer. In some technologies a higher disintegration of organic fibers is achieved (the pulper technology, developed by BTA) and in other

Figure 7-15. Ballistic separator in a Dranco anaerobic digestion plant

technologies a lower disintegration of fibers takes place (e.g. WABIO-process, developed by Outokumpu Ecoenergy OY in the Vaasa demonstration plant, Finland) (Schu 2007). The water content of the waste is regulated by adding circulation water to the waste in a mixer until the contaminants can be separated by sink-and-float separation in the mixing tank. For reducing equipment size, several removal steps with declining dry matter content are installed.

In the demonstration plant of BTA technology in Garching, separation is carried out batchwise and the fermenter is equipped with a sand discharger.

Contaminant removal within the washing/percolation technology

The first process step in the wash technology is the screw press where the material is rinsed with hot water and then pressed. The liquid phase is digested in this technology. The percolation retention time is short and amounts to six hours in the Aguatherm-process. The IMK-process, implemented in Herten in 1996 for biowaste, is very similar to the Aquatherm-process. The material is heated using aerobic hydrolysis and washed during the three day retention time. The wash process consists of daily pressing, wetting and mixing. The process water is digested and recycled (Schu 2007).

A percolation process (developed by ZAK and implemented in Kahlenberg) is similar to the wash process. Coarse stones are removed from sieved residual waste by a ballistic separator. In a large paddle screw the material is mixed continuously and washed (percolated). The wash water is then treated in a three-stage wet mechanical separation unit before entering the digestion step.

After percolation, the solid material is dewatered by a screw press and then dried with biological drying to produce RDF.

To summarize, wash/percolation mechanical pretreatment consists of 1) solids treatment as in dry anaerobic digestion, and of 2) a sophisticated wastewater treatment as in wet anaerobic digestion and 3) an additional RDF production as in the biological drying process. The process is therefore rather complex. The ISKA-process implements similar technology as the ZAK-process and has been implemented in two large scale plants in Germany in Buchen and Heilbronn.

7.5.2 Biological process stage

Wet single stage digestion is the most commonly used technology for treating agricultural and municipal wastewater. However, solid waste behaves differently than wastewater sludge. Because of the heterogeneous nature of residual waste, a number of problems in wet digestion technology occur: formation of a scum layer which prevents the bacteria from degrading these organics, problems with clogging the pumps and pipes when it is removed from the reactors and many other operational difficulties. Hence, dry systems have become the most commonly applied in Europe to treat solid residual waste. Dry digesters treat waste streams with 20-40% total solids without adding dilution water. Most dry digesters operate as plug flow digesters, in which the digester contents are not completely mixed, but move as a plug through the reactor from the feed port to the exit, like stuffing a sausage casing (Rapport 2008, Yebo 2010). This prevents inoculation of the incoming waste. Therefore, most of the digester designs include an inoculation loop in which the incoming waste is mixed with some of the exiting digestate paste prior to loading.

In Table 7-4 a review of existing digestion technologies in large scale plants in operation (by the year 2008) is provided.

General operation parameters and configurations of anaerobic digestion were already described in section 6.3. The same rules apply for the anaerobic MBP plants. Therefore, in the following the emphasis is placed more on the providers of technologies and on the commercial technologies available on the market.

Table 7-4. Summary of commercial digestion technologies with large scale reference plants (based on Rapport 2008)

Technology name/ provider	No of plants[1]	Capacity range (thous. tons/year)	Number of stages		Total solids content		Operating temperature	
			single	multi	wet, <20%	dry, >20%	meso. 35°C	ther. 55°C
AAT	8	3 – 55	√		√		√	
ArrowBio	4	90 – 180		√	√		√	
BTA	23	1 – 150	√	√	√		√	√
Biocel	1	35	√			√	√	
Biopercolat	1	100		√		√	√	
Biostab	13	10 – 90	√		√			√
DBA-Wabio	4	6 – 60	√		√		√	
Dranco	17	3 – 120	√			√		√
Entec	2	40 – 150	√		√		√	
Haase	4	50 – 200		√	√		√	√
Kompogas	38	1 – 110	√			√		√
Linde-KC/BRV	8	15 – 150	√	√	√	√	√	√
Preseco	2	24 – 30						
Schwarding-Uhde	3	25 – 87,6		√	√			√
Valorga	22	10 – 270	√			√	√	√
Waasa	10	3 – 230	√		√		√	√

[1] Includes operational or planned plants that accept any of the following: residual waste, kitchen waste, food waste, yard waste, or green waste. Does not include food processing waste or wastewater. May include co-digestion with other organics such as biowaste or sewage sludge. Pilot and demonstration plants were excluded.

It can be concluded from the Table 7-4 that the most widely applied technology for the dry digestion of solid waste is Kompogas, Dranco and Valorga. In total, 77 facilities used one of these process technologies by the year 2008 and 24 of these have been in operation for 10 years or longer. All of these technologies are operated as a continuous single stage and at mesophilic or thermophilic conditions. The biogas yield of the systems ranges from 0,3 to 0,5 m^3/kg organic matter (Yebo et al. 2010).

The **Kompogas** process was developed in Switzerland in the 1980s. The Kompogas reactor is a horizontal steel tank with slowly rotating axial mixers which 1) moves the waste from the inlet to the outlet, 2) maintains heavy solids in suspension, and 3) allows the degassing of the thick digestate. The total solids in the reactor are held in the range of 23–28% to facilitate flow (Yebo 2010). Recycled digestate

is mixed with the fresh input in order to inoculate the material and process water may be added to reduce the solids content. The system is prefabricated in two sizes: 16.500 or 27.600 tons/year. Larger capacities can be acquired by combining the units in parallel. The system operates with a retention time of 15-20 days under thermophilic conditions. The biogas yield was reported at 0,11–0,13 m^3/kg wet weight (Rapport 2008).

The **Valorga** process uses vertical steel tanks with an inner wall extending to about ⅔ the diameter of the tank. Material enters at the bottom on one side of the inner wall and must flow around the wall before it can exit. Biogas is injected in the base of the reactor and the bubbles serve as a means for mixing and keeping solids suspended. These tanks can operate between 25% and 35% total solids (Yebo 2010). Biogas mixing systems are used to create local mixing in the tank. As a result, fresh input does not necessarily require inoculation with finished digestate or leachate before feeding to the reactor. Process water is recycled to reach a target of 30% solids inside the reactor (Yebo 2010). The problem of gas nozzle clogging (needed for the gas mixing system) has been reported with this technology. Biogas yields have been reported in the range of 0,22–0,27 m^3/kg organic matter which corresponds to 0,13–0,16 m^3/wet kg. The solid retention time is 18-23 days and post digestion solids composting takes about two weeks (Rapport 2008).

The **Dranco** process, marketed by Organic Waste Systems (OWS) of Belgium, uses a vertical silo design with a conical bottom discharge for its reactor. The tank has no internal mixing mechanism. Outside of the digester, in a separate chamber, up to 6 parts of recycled digestate are blended with 1 part of fresh input waste and steam prior to being delivered to the top of the tank. This system typically operates with 30% to 40% total solids in the reactor. It applies thermophilic temperatures. The retention time is 2-3 weeks and little or no liquid effluent is produced. The biogas yield amounts to 5-8 m^3 /day per m^3 of reactor volume (de Baere 2010). The remaining digestate is dewatered to a press cake which contains active bacteria, some ammonia and undigested solids and must be aerobically stabilized. Existing commercial Dranco systems are reported to have biogas yields in the range of 0,103 – 0,147 m^3/kg wet weight. The Dranco process produces heat or electricity from the biogas. The yield of electricity produced ranges from 0,15 – 0,32 MWh/ton feedstock (Rapport 2008).
According to the technology providers, Dranco plants have a number of advantages:
- waste is treated at a higher rate than in conventional wet digesters,
- operation at a high dry mass content prevents scum formation or settling of heavies,

- minimal heat requirements - injecting steam in the mixing part of the digester is kept to a minimum because only the incoming fresh feed needs to be heated up,
- thermophilic operation yields a higher gas production per ton of waste treated and can reach significantly higher loading (de Baera 2010).

Figure 7-16 shows a Dranco digestion plant with a vertical digester, round biogas gas container and digestate maturation in open windrows. Figure 7-17 depicts a conical outlet of a Dranco digester with an extraction screw.

Figure 7-16. A view of a Dranco digestion plant

7.5.3 Mass balance of a dry digestion process

In the following, a mass balance of a dry digestion process of residual waste is presented. The calculations were performed for the waste characterized in section 7.3.3. The main objectives of the presented technology are

- mechanical separation of the coarse fraction (>60 mm) for RDF production,
- anaerobic stabilization of the fraction <60 mm, based on dry fermentation technology followed by enclosed aerobic stabilization (including hygienization),
- energy recovery from biogas.

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Figure 7-17 Conical outlet of a Dranco digester with an extraction screw

Mechanical pre-treatment steps are analogical as for the aerobic stabilization technology (see section 7.3.3), except for the separation on a drum sieve to fractions >60 mm and <60 mm. In Table 7-5, the characteristics of waste input assumed for the mass balance calculations is summarized.

Table 7-5. Characteristics of residual waste input to the digestion plant

Fraction	Fraction share in total waste %	Water content, %	Content of organic dry mass, % DM
<60 mm	50%	48%	50%
>60 mm	50%	40%	70%
weighted average		44%	60%

The mass balance calculations were preformed under the following assumptions:

Anaerobic biological stabilization – dry digestion:
- mesophilic process,
- the feedstock water content of approximately 65% (in order to increase the initial water content in the residual waste from 48% to 65%, the recirculation of fermentate liquor and the input of additional process water was assumed),
- process duration - 21 days
- decomposition of approximately 30% organic dry mass - formation of biogas,
- mechanical dewatering of waste on a screw press: reduction of the water content from 69% to approximately 55%.

Aerobic biological stabilization of the digestate:
Objective: hygienization and further decomposition of the organic matter
- potential co-stabilization with the fine fraction separated from RDF
- duration: 2-4 weeks in an enclosed system with mechanical aeration.
- additional decomposition of approximately 25% organic dry mass
- conditioning of the stabilized waste on a 30 mm sieve (to remove combustible fractions).

RDF production
- drying of >60 mm fraction for RDF (option: biological drying).

In Figure 7-18 a mass balance of the anaerobic stabilization according to the above description is provided. This treatment technology results in a substantial diversion of waste from a landfill. The stabilate accounts for approximately 32% of waste input. This fraction either has to be landfilled or can be used for land recultivation. The other major output of the treatment is 28,5% RDF, accounting for approximately 28,5% weight related to the plant input.
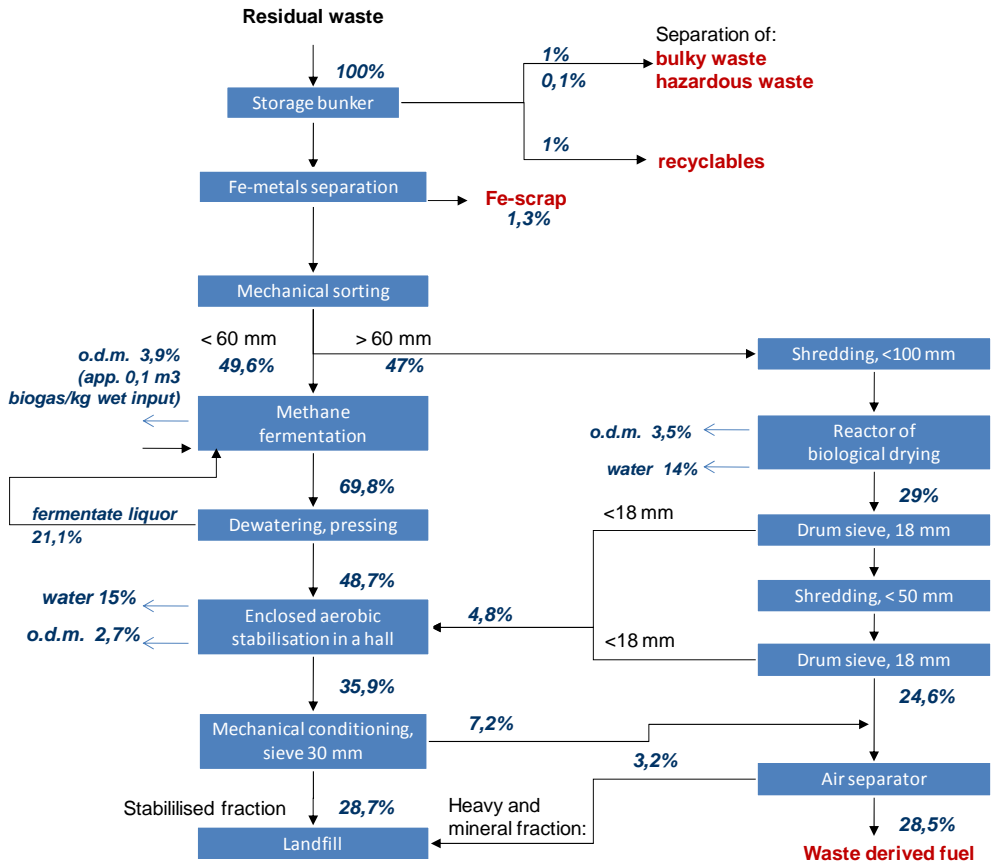
Figure 7-18. Mass balance of the dry digestion of residual waste (den Boer et al. 2010)

7.6 *Comparison of the MBP technologies*

In Figure 7-10, Figure 7-14 and Figure 7-18 mass balances are provided of the treatment of the residual waste (with the same composition) using three different MBP technologies: aerobic biostabilization, biodrying and anaerobic biostabilization. Below a very general comparison of these three technologies is provided.

Parameter \ Technology	I – aerobic mech.-biol. stabilization	II – biological drying	III – anaerobic mech.-biol. stabilization
Number of biological process steps	Two: intensive stabilization + maturation in windrows	One: biological drying	Two: fermentation + enclosed aerobic stabilization (+biological drying of RDF)
Treatment duration	2 weeks + 10 weeks	1 week	3 weeks + 4 weeks
Quantity of RDF, % input mass	26%	44%	28%
Quantity of waste for landfilling, % input mass	42%	13,5%	32%
Recovery of biogas	-	-	app. 0,1 m^3 biogas/kg wet input, (ca. 55% CH_4)
Complexity level	average	average	high
Emissions to air	high	medium	low

Comparison of the three MBP technologies:
- Duration: Biological drying – clearly the shortest treatment time,
- Fuel recovery: in all technologies recovery of RDF with a net calorific value approximately 15-18 MJ/kg,
- Quality of fuel: in biological drying the largest share of the recovered fuel, but of lowest quality,
- Biogas: only with anaerobic technology,
- Summary of energy recovery: lowest for aerobic biostabilization,
- Quantity of waste for landfilling: highest in aerobic biostabilization, lowest in biodrying,
- Complicity level: highest for anaerobic technology,
- Investment costs: highest for anaerobic technology,
- Operational costs: potentially lower for anaerobic (energy recovery),
- Emissions to air: highest emissions are expected from the biostabilization process, medium for biodrying (the process is shorter and fully encapsulated) and lowest for an anaerobic plant, (enclosed anaerobic

treatment does not cause air emissions), the second step is partly enclosed.

The final ranking of these technologies will strongly depend on local circumstances, e.g. whether there is market for RDF (e.g. demand at cement plants or power plants), whether stabilate can be used for land reclamation, where the plant is located - in a remote location or close to residential areas, etc. Thus, the selection of the technology for a given location should always be preceded by a thorough analysis of all these factors.

7.7 Emissions control from MBP plants

Generally, emissions from MBP plants are similar to those from the composting and anaerobic digestion of biowaste, except that the concentrations of some contaminants, e.g. heavy metals, Cl, persistent organic pollutants, etc. can be higher. Emission control measures applied at MBP plants are similar to those applied at biowaste treatment plants. Thus, biological air treatment methods such as biofilters are often used.

Emissions from MBP plants are not regulated at the European level. However, some countries, e.g. Germany and Austria have imposed very stringent targets with regard to air emissions from MBP plants (see Table 7-6).

Table 7-6. German limits for air emissions from MBP plants according to 30.BImSchV

Parameters	Target values clean gas		
Sample	½ h - mean	Daily mean	Monthly mean
TOC	40 mg/m³	20 mg/m³	55 g/Mg
Nitrous oxide (N_2O)	-	-	100 g/Mg
Ammonia (NH_3)	30 mg/m³ (TA air)		-
Dust	30 mg/m³	10 mg/m³	-
Dioxins / Furans	0.1 ng/m³		
Odor	500 OU/m³ (single measurement)		

OU: odour units as measured in a olfactometer

The German limits are very stringent, especially with regard to the total organic carbon. The required emission limits cannot be complied with using conventional biofiltration technology. Therefore, many MBP plants had to be equipped with more advanced air treatment technology, of which the most frequently used is regenerative thermal oxidation (RTO) technology.

RTO is designed to destroy air pollutants emitted from process exhaust streams at temperatures ranging from 800 to 1000°C. It utilizes ceramic media packed into vertical canisters as a high-efficiency heat exchanger. Oxidation is achieved as pollutants pass through the ceramic media. An auto thermal-operation form of RTO technology is possible at 1 g/Nm³ VOC loading in the exhaust gas. The clear disadvantage of the RTO method is its high investment and operational costs compared to biofiltration as well as its very high level of complexity compared to biological air treatment methods.

The positive impact of MBP on landfill operation is due to the reduction of uncontrolled emissions from landfilled waste. Table 7-7 provides the ranges of emissions from MBP waste (stabilate) as compared to raw waste. Waste pretreatment results in an approximately 90% reduction of emissions from the landfilled waste.

Table 7-7. Reduction of landfill emissions due to waste pretreatment (Stegmann 2005)

Emission	Raw waste	MBP-waste	Reduction
COD (mg/kg DM)	25.000 – 40.000	1.000 – 3.000	90%
Total Nitrogen (mg/kg DM)	1.500 – 3.000	150 - 300	90%
GB21 (dm^3/kg DM)	150 - 200	0 - 20	90%

8 Thermal treatment of waste

8.1 Introduction

8.1.1 History of thermal waste treatment

The thermal treatment of waste has a long tradition. The burning of waste materials for heating purposes, as a way of disposal or for hygienic reasons (e.g. the burning of cloths of the deceased in times of epidemics) is not a recent invention. Already in 1876 the first controlled incineration plant was operated in England. This technology was later transferred to the city of Hamburg because the city had nowhere to go with its cholera contaminated waste resulting in the operation starting in 1896 (Bilitewski et al. 1997). After WWII, with an increase in population and disposal space getting more and more sparse, incineration was often applied as a means of volume reduction (McDougall et al. 2001). Following the Seweso incident, incineration plants were increasingly considered as sources of dioxins and other dangerous substances which led to the development of flue gas cleaning technologies in incinerators. At the same time, waste incineration as a source of electrical and thermal energy is becoming increasingly relevant.

8.1.2 Disposal vs. recovery

According to the so-called waste management hierarchy also considered in European and national legislation, notably in the new Waste Framework Directive, the thermal treatment of waste can be considered as either recovery or disposal of waste. These are the last stages in the hierarchy:

a) prevention;
b) preparing for re-use;
c) recycling;
d) other recovery, e.g. energy recovery; and
e) disposal.

There are several ways to differentiate between thermal treatment that should be regarded as energy recovery and thermal treatment that is merely thermal disposal:

- the purpose of the installation: energy production or waste disposal. However, the road to hell is paved with good intentions and such a subjective differentiation would lead to hard to solve discussions.

- energy content of the waste: waste with higher energy content has a higher potential of recovery. This, however, leaves out the performance of the installation itself.

- efficiency of installations: this is what is introduced in the new Waste Framework Directive (to be implemented in national legislation by December 2010). This way of differentiating depends on both the installation performance and waste input.

Thus, the Waste Framework Directive indicates that incinerators to be qualified as energy recovery plants must have an efficiency of at least (EP&C 2008)

0,60 for installations in operation before 1 January 2009 and

0,65 for installations permitted after 31 December 2008.

To calculate the plant efficiency, the following formula should be used:

$$\eta_{nett} = \frac{E_p - (E_f + E_i)}{0,97(E_w + E_f)}$$

In which:

η_{eff} net energetic efficiency of the installation

E_p mean annual energy produced as heat or electricity. It is calculated with energy in the form of electricity being multiplied by 2,6 and heat produced for commercial use multiplied by 1,1 (GJ/year)

E_f mean annual energy input to the system from fuels contributing to the production of steam (GJ/year)

E_w mean annual energy contained in the treated waste calculated using the net calorific value of the waste (GJ/year)

E_i mean annual energy imported excluding E_w and E_f (GJ/year)

0,97   is a factor accounting for energy losses due to bottom ash and radiation.

Other ways of categorizing thermal processes can also be applied:

- single or multi-purpose processes: incineration or co-incineration. In a co-incineration process generally other fuels are the main energy input into the process. These are mostly primary fuels.

- type of input to the process: mass burning of mixed municipal waste, RDF (Refuse Derived Fuel), hazardous waste, medical waste

- produced outputs: nowadays most installations produce heat and/or electricity, which is sold. However, some older installations do not allow any form of energy recovery.

- purpose of the process: preliminary waste treatment or input in a production process (e.g. cement kiln, power plant)

8.1.3 Definition of incineration

According to the EU Incineration Directive, an incineration plant is defined by (EP&C 2000):

"any stationary or mobile technical unit and equipment dedicated to the thermal treatment of wastes with or without recovery of the combustion heat generated. This includes the incineration by oxidation of waste as well as other thermal treatment processes such as pyrolysis, gasification or plasma processes in so far as the substances resulting from the treatment are subsequently incinerated."

A co-incineration plant is

"any stationary or mobile plant whose main purpose is the generation of energy or production of material product and

- *which uses wastes as a regular or additional fuel; or*

- *in which waste is thermally treated for the purpose of disposal.*

If co-incineration takes place in such a way that the main purpose of the plant is not the generation of energy or production of material products but rather the thermal treatment of waste, the plant shall be regarded as an incineration plant."

Both definitions cover the site and the entire plant including all (co-)incineration lines, waste reception, storage, on site pretreatment facilities, waste-, fuel- and air-supply systems, boiler, facilities for the treatment of exhaust gases, on-site facilities for treatment or storage of residues and wastewater, stack devices and systems for controlling incineration operations, recording and monitoring incineration conditions.

8.1.4 Goals of thermal treatment

As mentioned before, the first incineration plants were constructed mainly for reasons of hygiene and to combat the spreading of epidemics. Later, the focus moved towards saving landfilling volume because of limited space for landfill sites

in Western Europe. Nowadays, the main goals of thermal waste treatment are manifold:

- Reduction of waste volume and the mass to be disposed. On average, the volume of waste after incineration is approximately 10% and the mass is app. 30% of the input material;

- Reduction of the environmental risks of the waste by
    - sterilization, mineralization and stabilization of the waste;
    - decomposition of organic compounds;
    - immobilization of heavy metals in mineral matrices (ash);
    - concentration of (volatile) hazardous substances in fractions for disposal;
- Production of energy and secondary materials. Energy can be produced in form of electricity, heat or process steam. The secondary materials obtained from thermal processes range from metal scraps to road foundation materials (bottom ash) to gypsum and acids.

8.2 Technologies

There are a number of technologies for thermal waste treatment available on the market. Some of these are more exotic and will be mentioned here only briefly. The standard and upcoming technologies are discussed in greater detail in the following chapters. Currently, the thermal treatment of waste encompasses five main technologies:

- Direct combustion: Grate fired mass burn installations. This is the standard technology for the incineration of municipal waste. The following chapters will focus on this technology.

- Direct combustion: Rotary drum. This technology is mostly applied for hazardous and medical waste. The waste is inserted in either solid, liquid or paste-like form into a rotating kiln. The main goal of the operation is clearly a reduction of the threat that the waste poses on human beings and the environment. The installations are generally smaller than grate fired processes.

- Gasification. In a gasification process organic material is decomposed with limited amounts of oxygen to form so-called syngas. Syngas consists of a mixture of carbon monoxide (CO) and hydrogen (H_2) and can be used either for energy production, e.g. in combined heat and power (CHP) units or for chemical production (methanol or synthetic fuels). The typical gasification reaction, here with celluloses as an example input is

(Heermann et al. 2001):

$$C_6H_{10}O_5 + O_2 \rightarrow 5CO + CO_2 + 5H_2$$
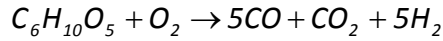
This process is best suited for homogeneous high caloric wastes such as wood or plastics. Residual or mixed waste can also serve as an input, e.g. in the Thermoselect process. In this process the ashes are vitrified. The plant in Karlsruhe (Germany) began operation in 1999, but had to close down only five years later.

- Pyrolysis. In a basic pyrolysis process organic material is decomposed at high temperatures under the exclusion of oxygen (which is also one of the initial process steps in incineration or gasification) producing tar, char and volatile substances. The typical gasification reaction here with celluloses as an example input is (Heermann et al. 2001)

$$C_6H_{10}O_5 \rightarrow 5CO + 5H_2 + C$$
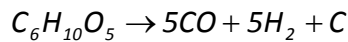
For processes involving waste as an input material mostly further steps are applied resulting in products similar to gasification plants. Also, pyrolysis is better suited for homogeneous, high caloric inputs. The Schwell-Brenn installation in Fürth, Germany was operated with municipal waste between 1997 and 1999 producing ash in the form of granulate.

- Plasma technologies. These technologies are special cases of gasification or pyrolysis processes using ionized gas. To create the plasma, a gas is passed between two electrodes, is ionized and turns into a plasma. The energy contained in this plasma (at temperatures of up to 4000 °C) breaks down the organic molecules contained in the waste. Plasma technologies are not yet established.

In Table 8-1 some characteristics of the various thermal treatment technologies are provided.

Table 8-1. Characteristics of thermal treatment technologies

Technology	Net calorific value (MJ/kg)	Energy recovery	Load range
Grate incinerator: air cooled water cooled	6-12 10-17	high high	60-100% 60-100%
Fluidized bed (gasification)	6-20	very high	40-100%
Pyrolysis	4-30	medium	30-100%

8.3 Incineration process

8.3.1 General overview

Grate fired incineration processes are highly complex and show a high variety in technological options. In Figure 8-1, a simplified overview of an incinerator is given.
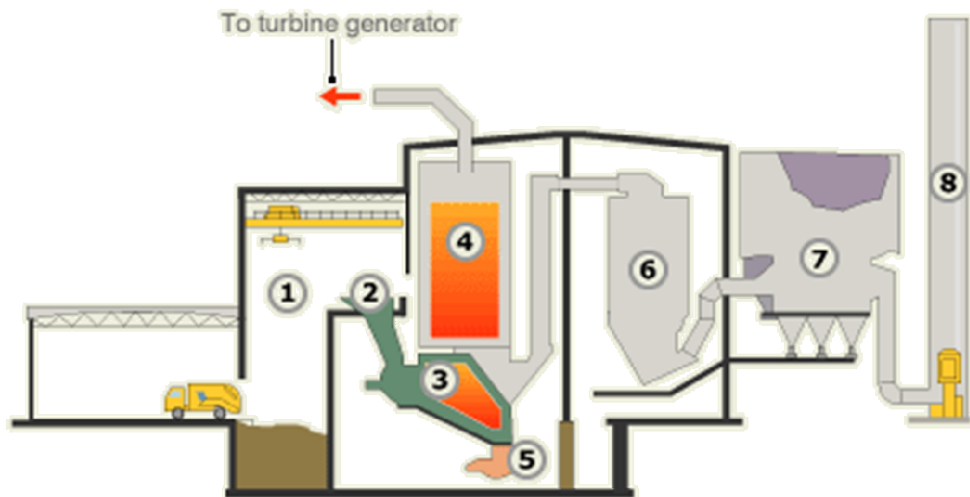

Figure 8-1. Simplified overview of a grate fired incinerator (BBC News 2006)

In the above picture the following components can be found:

1. Bunker. Here the waste is stored and homogenized by mixing.
2. Hopper. By a moving crane the waste is fed into the hopper.
3. Grate. Here the waste is actually combusted, it moves down, being fed with combustion air.
4. Boiler. The hot furnace gas is used to heat up steam which is used for energy production.
5. Bottom ash collection. Scrap is separated from the bottom ash.
6. Scrubber, for cleaning the flue gas.
7. Filter, to capture dust.
8. Stack, for emitting the flue gas to the air.

Later in this chapter the flue gas cleaning components will be discussed in more detail.

The basic incineration process consists of 6 separate steps which occur not strictly separated from each other on or above the firing bed.

1. Drying (100-200 °C). In this initial step most of the water contained in the waste is transferred to the gaseous phase
2. Degassing or pyrolysis. At a temperature of around 250°C volatile components are turned into a gas.
3. Ignition. The pyrolysis gas ignites in and over the waste body, thus providing energy for the following phases.
4. Gasification. At higher temperatures (400-600 °C) the organic compounds are decomposed into a carbon dioxide containing syngas.
5. Combustion. All produced burnable gases are combusted over the waste bed at temperatures of around 1000 °C. Left over carbon in the waste is burned out, leaving bottom ash of less than 0,5% ignition loss (Bilitewski et al. 1997).
6. Afterburning. Occurs in the boiler section of the plant. The flue gas is kept for minimal 2 sec. at 850 °C to assure complete combustion of volatile components, especially dioxins.

In modern incinerators the conditions for these separate steps are optimized by creating varying circumstances within and over the grate, e.g. by zones of high and low air inflow.

Taking the incineration process as a black-box, merging the single process steps into one overall reaction, the combustion reaction can be described by the following equation:

$$C_xH_y + (x + \tfrac{1}{4}y)O_2 \rightarrow xCO_2 + \tfrac{1}{2}yH_2O$$
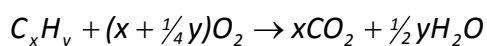
Here the incoming waste is simplified as C_xH_y. This is naturally more complex in reality. Apart from oxygen also several major elements are present in the municipal mixture. These elements either react during the combustion process or are freed from the waste body and emitted unchanged. Other contaminants in the flue gas are conglomerates with varying chemical structures and a variety of more complex substances. In the following Table 8-2 an overview of these major contaminants is provided.

Table 8-2. Overview of the main components in incinerator flue gas

Component	Origin
HCl	formed from Cl contained in waste
HF	formed from F contained in waste
SO_2	formed from S contained in waste
NO_x	formed from N contained in waste or air
heavy metals	emitted mainly Hg, Cd and Tl
organic pollutants	e.g. dioxins, contained in waste and de-novo synthesis
dust	particle matter containing many contaminants

Since municipal waste is a complex mixture of material, it is very difficult to describe its properties based on the joint contributions of all single components. Rather, the properties of the mixed material as a whole are considered in examinations and literature. The combustion properties of waste can be described in a simplified manner by providing the contents of combustible materials in relation to the mineral content and the water content. A convenient way of doing so is combining these values into a so-called Tanner's triangle. In Figure 8-2 an example for the average European municipal waste is shown. Bilitewski (1997) mentions a value for European waste of 35% water and combustibles combined with an ash value of 30%. This average value is therefore self-sustaining in combustion.

8.3.2 Flue gas cleaning

Flue gas cleaning takes up most of the space of a modern incinerator. This is also true for the investment costs; about two thirds stem from flue gas cleaning equipment. In the following, an overview of flue gas cleaning units is given, as these can be found in standard modern municipal waste incinerators. In practice, a large number of combinations of these and additional flue gas cleaning components are implemented. The Reference Document on the Best Available Techniques (BREF) for Waste Incineration mentions 408 different combinations (EIPPCB 2006).
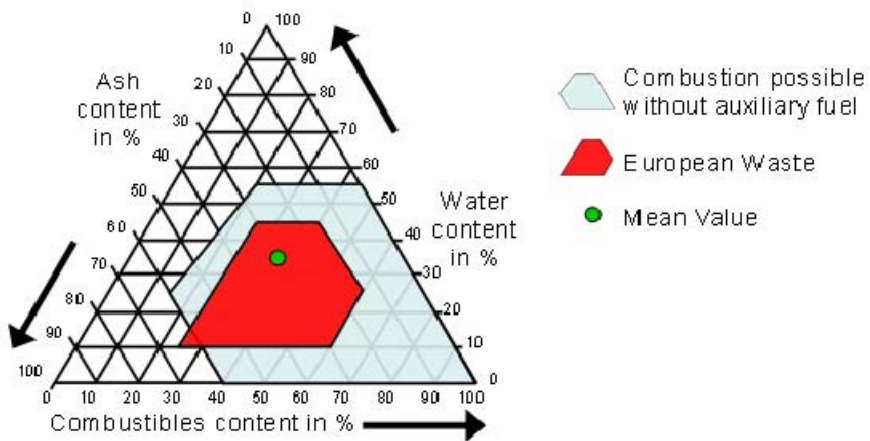
Figure 8-2. Tanner's triangle for average European municipal waste (Habeck-Tropfke 1985)

I.   Dioxins

Polychlorinated dibenzodioxins (PCDD, dioxins) and Polychlorinated dibenzo-furans (PCDF, furans) are groups of very toxic substances. Out of 210 different dioxins and furans, 17 are relevant in terms of toxicity. Within this group, the toxicity is described by the International Toxicity Equivalency Factor (I-TEF), which varies by a factor 1.000 between the single substances (the substance commonly known as dioxin, 2,3,7,8-tetrachlorodibenzo-para-dioxin is considered 1.000 times more toxic than e.g. 1,2,3,4,6,7,8,9-octachlorodibenzo-para-dioxin). The sum of the toxicities of the single substances in a certain sample is then provided in International Toxic Equivalents (I-TEQs).

The amount and ratio of the dioxins emitted in the raw gas of incinerators depends on the input material, but as well on the installation geometry and operation. In a first step of emission abatement, the Incineration Directive prescribes keeping the flue gas in the combustion chamber for at least 2 s at 850 °C which causes the dioxins to decompose (EP&C 2000). However, a de novo synthesis is possible between 200-600 °C. Smart plant operation (e.g. quenching of the flue gas) minimizes this process, but nevertheless as a last flue gas cleaning step **adsorption to coke or activated carbon** is needed to reduce the concentration to acceptable levels.

II.   Particulate matter

Dust in flue gas contains heavy metals and persistent organic pollutants (POPs) that adsorb to small particles. Part of the particulate matter is produced during the combustion process; another part stems from flue gas cleaning steps (scrubbers). The particulate matter separated from the flue gas stream is also known as fly ash. This fly ash is much more toxic than bottom ash and is generally disposed of at hazardous waste landfills, although there are processes for its recycling (Auber et al. 2004).

There are three main components that may be found in waste incinerators. A **cyclone** may be used as an initial step, but this is not sufficient to keep emission limits. **Electrostatic precipitators** (EPS) can keep these limits, but are thought to cause de novo synthesis of dioxins, especially at temperatures over 250°C (Duo and Leclerc 2007, EIPPCB 2006, Bilitewski et al. 1997). **Fabric filters** also are sufficient to clean the flue gas to meet the legal standards. Although modern incinerators mostly do keep the emission limits for dust, recently the emission of the smallest particles, PM2,5, by incinerators has been in critical discussions.

In Figure 8-3 an overview of the most widely applied dust removal equipment is provided showing their function.
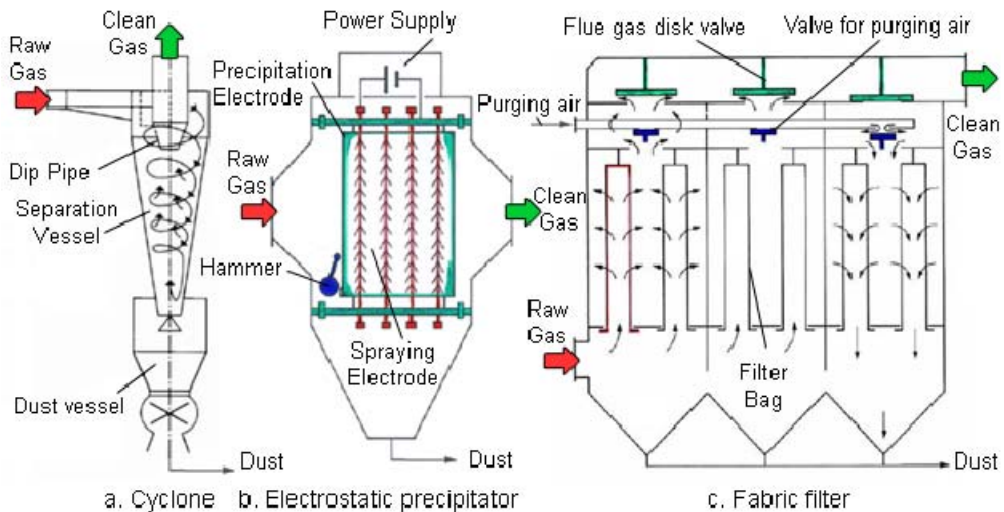

Figure 8-3. Overview and functioning of dust removal equipment in incineration plants (WtERT 2009)

III.   Acids

Acids present in the flue gas, mainly HCl and SO_2 and to a lesser extend HF, can be removed by scrubbers. There are three types of scrubbers: **wet scrubbers** (liquid absorptions agents in, liquid reaction products out), **dry scrubbers** and **semi-dry scrubbers** (liquid reactants in, solids out).

Wet scrubbers aim at removing HCl (acid scrubber) and SO_2 (alkaline scrubber) producing HCl (or NaCl) and gypsum (Na_2SO_4 or $CaSO_4$). In semi-dry scrubbers lime reacts with acids, whereas in dry scrubbers pulverized lime hydrate is used, which has a large specific surface. Dry and semi-dry scrubbers need additional filters.

In the following Figure 8-4 to Figure 8-6, these three principal scrubber types are shown.
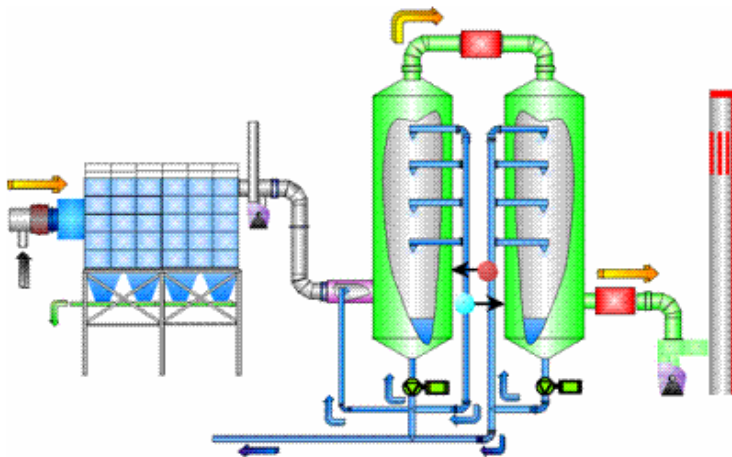

Figure 8-4. Wet scrubber with an upstream fabric filter (EIPPCB 2006)

I. NO_x

Nitrogen oxides are formed from both input material contained nitrogen and from nitrogen in the combustion air. The latter form of NO_x is strongly dependent on the temperature of combustion. By limiting the combustion temperature and the level of oxygen (e.g. by the use of secondary and tertiary air), the formation of NO_x can be partly prevented.

For the removal of left over nitrogen oxides there are two main technologies operated. Selective Non Catalytic Reduction (**SNCR**) uses a reagent that is inserted at the beginning of the flue gas cleaning system, directly into the combustion
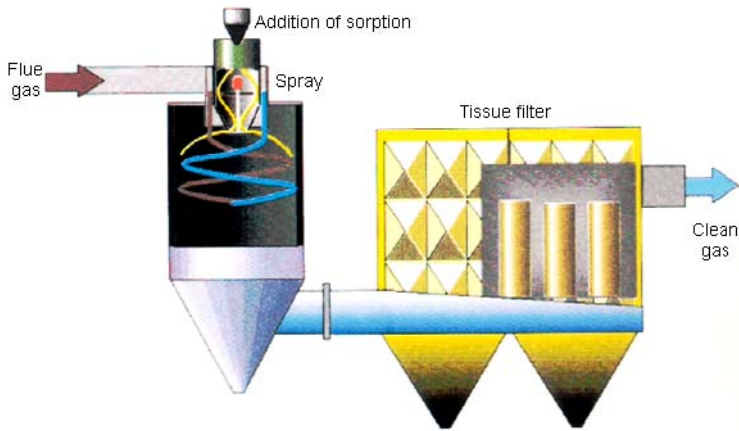
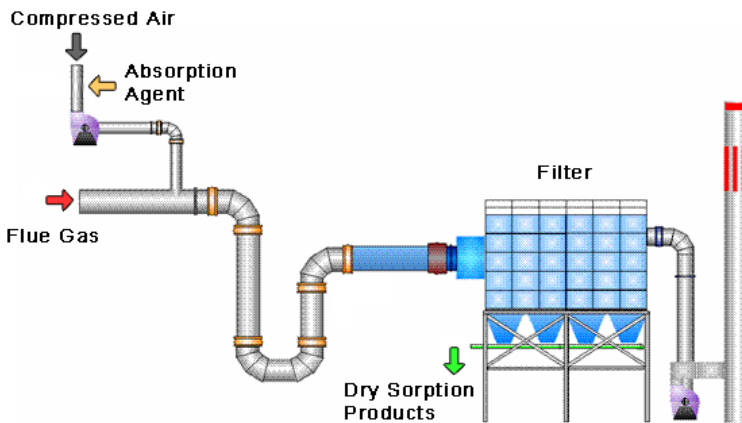**Figure 8-5. Semi-dry scrubber with downstream de-dusting (EIPPCB 2006)**

Figure 8-6. Dry scrubber with a downstream fabric filter (EIPPCB 2006)

chamber. Ammonia or urea act as reducing agents transforming the nitrogen oxides into elementary nitrogen and water vapor.

In a selective catalytic reduction (**SCR**), ammonia or urea are sprayed on the flue gas and led over a catalytic converter. This takes place later in the flue gas cleaning system after dust and acid removal. SCR can be operated almost stoichiometrically, whereas SNCR is a trade-off between excess ammonia (the so-called ammonia slip) or nitrogen oxide (formed from ammonia combustion) in the flue gas. SCR, however, may require reheating of the flue gas with a consequent negative effect on the energy efficiency of the incinerator (EIPPCB 2006).

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II.   Heavy metals

The main heavy metals contained in the flue gas are mercury, cadmium and thallium. Other metals are not volatile and therefore not problematic in the flue gas. The volatile metals may oxidize or be chlorinated (Hg), thus precipitating on dust particles or dissolving in wet scrubbers. Left over elementary volatile metals can be adsorbed to activated carbon or coke filters. These are usually the last flue gas cleaning step aimed at eliminating dioxins and other POPs.

The effects of flue gas cleaning technologies are substantial. In Table 8-3 an overview is provided of the raw gas values compared to the average clean flue gas leaving incinerators. It can be clearly observed that average incinerators comply with the European emission limits.

Table 8-3. Average emission values (mg/Nm^3) in clean and raw flue gas as well as emission limits of the Incineration Directive (EP&C 2000)

Substance	Raw flue gas	Limits clean gas	Modern plants
Dust	2.000-20.000	10	<1
TOC	10-400	10	<1
HCl	1.000-4.000	10	<3
HF	20-50	1	<0,3
SO_2	100-2.000	50	<5
NO_x	150-450	200	<30
Cd+Tl	<2,5	0,05	<0,01
Hg	0,4-0,7	0,05	<0,02
other heavy metals	10-60	0,5	<0,1
PCDD+PCDF (ng I-TEQ)	2-40	0,1	0,01
CO	80-800	50	<25

Municipal waste incinerators have often been criticized because of their dioxin emissions. Historically, these critiques were justified. Waste incineration was one

of the main contributors to the overall emission of dioxins in the Western world. However, during the 1990s the situation drastically improved. Whereas the total dioxin emission decreased, the contribution of incineration in both absolute and relative terms showed a sharp decline. In Figure 8-7, this process of improvement is shown for Germany and the USA.
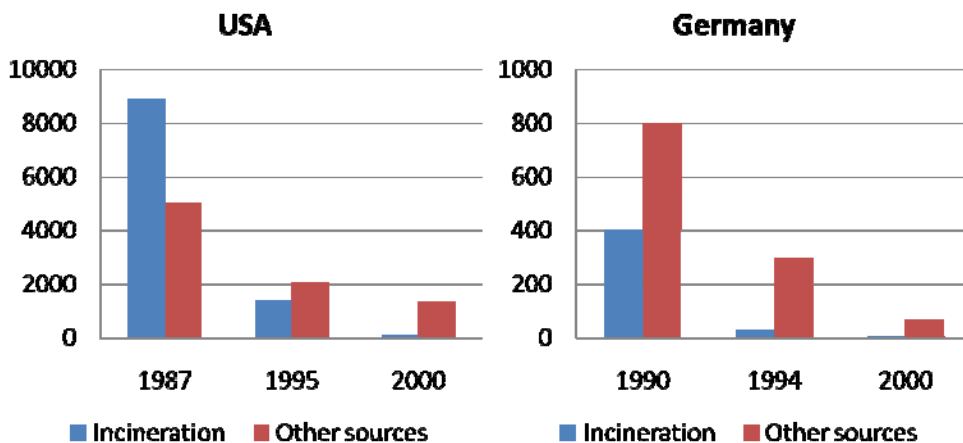

Figure 8-7. Development of the yearly dioxin air emissions (in g I-TEQ) in the USA and Germany (based on BMU 2005, US EPA 2006)

For other emissions the same trend can be observed. The mercury emissions from German incinerators decreased from approximately 1% in 1990 to 0,01% in 2001 of the total nationwide emissions in 1995. For lead, these values are 9% and 0,02%, respectively (BMU 2005).

8.3.3 Incineration mass balance

Apart from input waste a large amount of combustion air enters a waste incinerator. In addition to the flue gas containing emissions left over after the gas treatment installations, a number of solid products and materials for disposal are produced. In Figure 8-8 a mass balance for the average of 16 incinerators in North Rhine-Westphalia is provided. Since the data are based on average values from installations with varying technologies, such average values will not be found for single incinerators (because of alternative installation parts that are not constructed within one plant simultaneously).
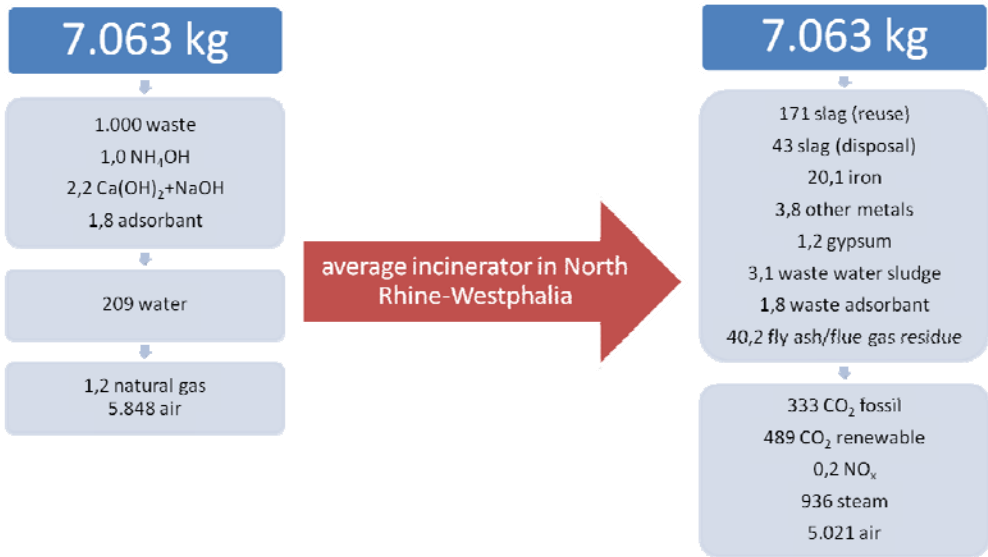
Figure 8-8. Mass balance (in kg, related to 1.000 kg of waste input) of an average incinerator in North Rhine-Westphalia (based on Fehrenbach et al. 2007)

Apart from the input of waste and air, water is the main auxiliary material on the input side. Most of the solid output materials can be recycled, although this is dependent on local markets and regulations, e.g. for bottom ash. After the major improvements in emission control in the 1990s, newly constructed incinerators show yet an additional level of improvement.

These incinerators, also known as 'of the fourth generation', show apart from further improvements in emission abatement, a significant increase in energy efficiency and a reduction in the amount of output material to be disposed of at (hazardous waste) landfills. To illustrate this newest development in Figure 8-9 a comparison between the aforementioned average incinerator in North Rhine-Westphalia and the new Waste Fired Power Plant in Amsterdam is provided.

The disposal rate of the average incinerator in North Rhine-Westphalia lies at approximately 10%, whereas the Waste Fired Power Plant in Amsterdam has to dispose approximately 1 % related to the input waste.
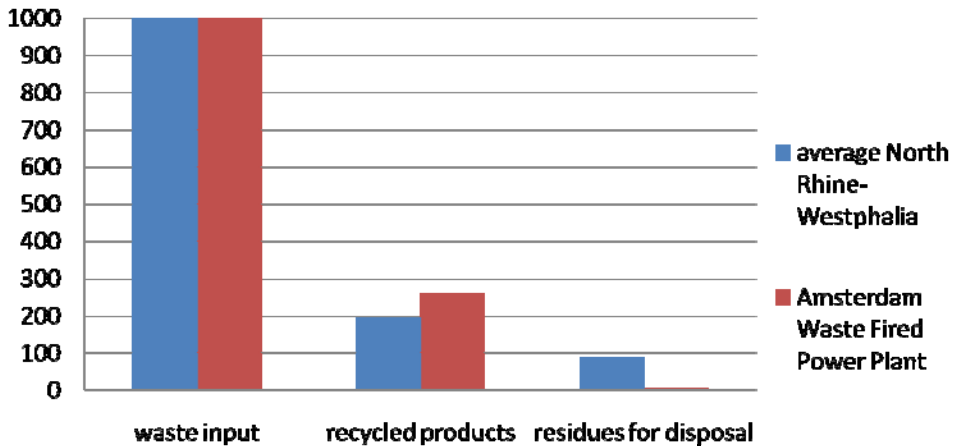
Figure 8-9. Comparison of output materials (in kg, related to 1.000 kg of waste input) between incinerators of the third and fourth generation (based on Fehrenbach et al. 2007, EAB 2008, 2010)

8.3.4 Incineration energy balance

Incinerators generally recover a part of the energy input (chemical energy of the input waste) by using it either for the sale of heat or steam. Alternatively, electricity can be produced. Also, combinations of these technologies are in operation. In comparison with power plants, relatively moderate figures are achieved for the net energy efficiency (production minus input needed for the operation of the incinerator) for incinerators. This is partly due to the more complex emission abatement technology installed, the more stringent emission norms and the relatively low calorific value of the input material (waste). Average values for the net energy efficiency of municipal waste incinerators are (EIPPCB 2006)

- electric efficiency (plants producing only electricity): 13% for German plants
- thermal efficiency (plants producing only heat/steam): 59%
- Combined Heat and Power plants (CHP): 49%. Here the electric efficiency is incorporated by multiplying it by a factor of 2,63.

This data is based on older sources (2001). ITAD (2010) mentions that in 69 German incinerators the amount of net produced electricity and heat increased by 39% and 10%, respectively between the years 2005-2008. In the same period the amount of incinerated waste increased by 20% in Germany (Eurostat 2010).

From this it is clear that the maximum energy recovery from municipal waste has not yet been reached in incinerators. In one of the newest incinerators, the Waste Fired Power Plant in Amsterdam, a net electric efficiency of 30% is reached (EAB 2010). This efficiency is already approaching that of normal coal fired power plants.

8.3.5 Incineration capacity diagram

The throughput of an incinerator is determined by two main parameters: the mass flow and the energy flow. Both set limitations to the maximum and minimum input flow which allow the installation to be operated in a feasible manner. In a capacity diagram this operational area is captured. Figure 8-10 gives an example of a capacity diagram of an imaginary incinerator for the city of Wrocław in the year 2020.
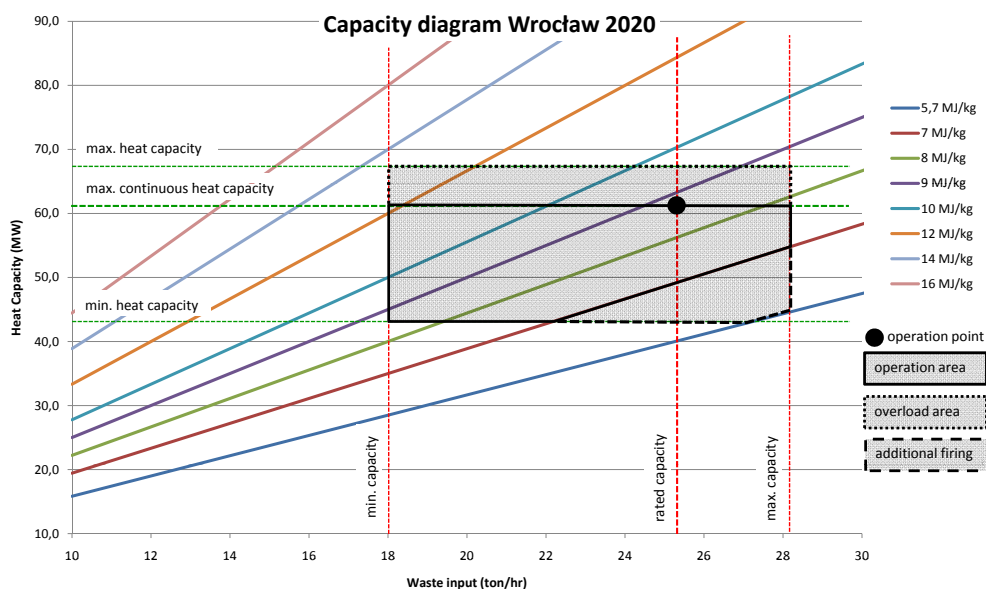

Figure 8-10. Capacity diagram for an imaginary incinerator for the city of Wrocław

In the capacity diagram it can be observed that the operation area of incinerators is limited by the mass flow and the heat capacity. The maximum continuous heat capacity can be exceeded, but this leads to higher maintenance costs. Generally, waste with a net calorific value of over 7 MJ/kg can be incinerated without the use of additional fuels.

The waste composition as it is predicted for Wrocław in the year 2020 (considering an increased level of separate collection of waste) leads to a net calorific value of 8,6 MJ/kg (Wameco 2010, Den Boer et al. 2005). This is considerably lower than the current value in Western Europe, e.g. in Austria a value of 10-11 MJ/kg is reported with an increasing tendency (BMLFUW 2010). This means that the same incinerator could treat a smaller amount of waste if it were built in Austria than if it were built in Wrocław.

Considering the average consumption of heat and electricity in Polish households, the incineration of the residual waste in Wrocław in 2020 (206.000 tons) in an incinerator similar to the Amsterdam Waste Fired Power Plant would lead to the production of (AEB 2010, BDR 2010, Dopke 2008, Wameco 2010):

- electricity: 148.200 MWh_{el} corresponding to the electricity consumption of 67.500 households (25% of Wrocław households)
- heat: 53.352 GJ_{th} corresponding to the heat consumption of 1.075 households (0,4% of households)

8.4 Incineration around the world

Incineration, being a high-tech and high-investment technology for waste management, can be predominantly found in the Western world. In the US incineration is less popular than it is in Europe. Tangri (2003) reports an increase of the number of incinerators since the late 1960s, reaching a maximum of 186 plants in 1990. In 2002 the number had decreased to 112.

As for many things, municipal waste incineration is rapidly developing in China. In 2008 approximately 70 incinerators were in operation with another 30 to be constructed by the year 2010 (Ni et al. 2009). Most of the Chinese incinerators were constructed after 2000, both with foreign and domestic technologies. However, emission levels of e.g. dioxins generally are, because of sub-optimal technologies and operation, much higher than in European plants (Cheng and Hu 2010).

In Japan incineration is the predominant waste treatment technology. Currently, its joint incineration capacity is even larger than the generation of municipal waste (ecoprog / Fraunhofer UMSICHT 2010).

In Europe a varied picture can be observed. In Figure 8-11 an overview of the number of incinerators and the amount of treated waste is provided.
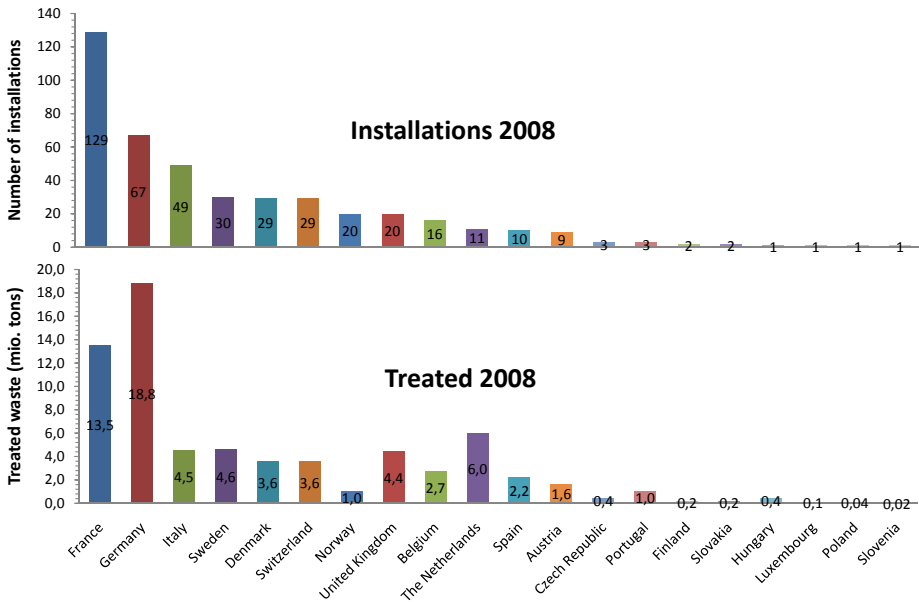
Figure 8-11. Number of operated incinerators and the amount of incinerated waste in European countries in the year 2008 (based on CEWEP 2010, Eurostat 2010)

Countries ahead in recycling at the same time tend to have a large share of their residual waste incinerated (e.g. Germany, The Netherlands, Switzerland, Denmark). Eastern European countries, as well as the (relative to its number of inhabitants) UK, Portugal and Finland have fewer incinerators, whereas e.g. in Greece and Ireland there are no incinerators at all. Between 1997 and 2008 the number of plants as well as the incinerated amount has more than doubled (+57% and +51% resp.) in Europe (CEWEP 2010, Eurostat 2010).

In total, 69 million tons of waste were incinerated in Europe in 2008. Most European incinerators recover part of the energy that enters the plant via the chemical energy content of the waste. With this the following amounts of energy recovery were achieved in 2008 (CEWEP 2010):

- electricity: 28 million MWh_{el}, corresponding to the electricity consumption of 13 million inhabitants;
- heat: 69 million MWh_{th}, corresponding to the heat consumption of 12 million inhabitants

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8.5 Refuse Derived Fuel

8.5.1 General

Instead of direct mass-burning of municipal solid waste in grate fired incinerators, the waste can also be pre-treated prior to thermal treatment. The high caloric fraction produced in such a pre-treatment is called Refuse Derived Fuel (RDF). This term is sometimes also used in a wider sense, e.g. including sorting residues from packaging wastes or processed industrial wastes. Also, other terms are in use such as Recovered Fuel (REF), Packaging Derived Fuel (PDF), Paper and Plastic Fraction (PPF) and Processed Engineered Fuel (PEF) (Gendbien et al. 2003).

RDF is similar to Solid Recovered Fuel or Specified Recovered Fuel (SRF). SRF, however, is produced according to the requirements of a quality label.

The produced RDF can be either burned in dedicated waste-to-energy plants producing electricity and or heat/steam or co-incinerated in industrial processes or power plants.

In Figure 8-12 a simplified overview is given for the effects of RDF production (e.g. in a mechanical-biological pre-treatment plant.
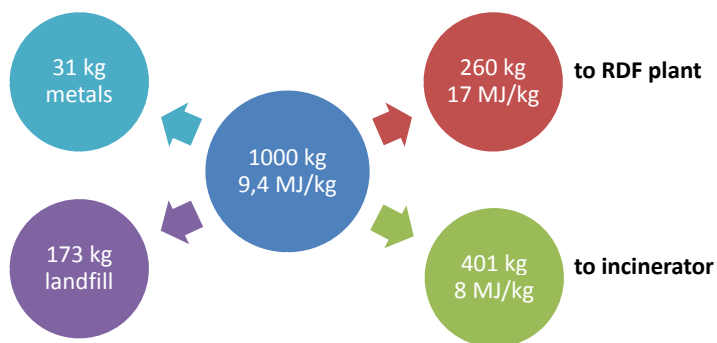

Figure 8-12. Exemplary effects of RDF production (based on Fehrenbach et al. 2007)

In this example it can be observed that splitting the input waste flow produces a high caloric RDF stream.

8.5.2 Reasons for RDF production

There are several reasons for the production of RDF instead of direct incineration:

- energetic efficiency of direct incineration is relatively low when compared to power plants or district heating plants, especially in the case of electricity production

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- regular fuels can be directly substituted (substitution of the heat content of the fuels, not of produced electricity or heat)
- RDF has less contamination than mixed waste, thus its thermal treatment needs less complex flue gas cleaning technologies
- RDF is more homogeneous and allows for treatment technologies that have difficulties treating mixed waste directly (e.g. fluidized bed reactors)
- incineration targets waste treatment (high caloric value limits the throughput (less t/hr) so a separation of RDF lowers the calorific value of the remaining waste enabling a higher input mass flow. This leads to more income from gate fees
- there is a demand for RDF for its high energy content in industrial plants
- the construction of a mass burn incinerator can be avoided, which may be interesting for reasons of high investments and strong opposition of citizens. Incineration is a technology that meets a high level of 'Not In My Backyard' (NIMBY)
- the emission limits for co-incineration are in certain cases higher and they are calculated based on the mixture of used fuels (partly primary, partly waste derived) (EC&P 2001, BMLFUW 2010). If the primary fuel causes emissions well below the limits, RDF could be treated in such amounts that the limits for the mixture are still kept. For some pollutants this would be the other way around. As a result, less complex flue gas cleaning is needed.

8.5.3 Quality criteria for RDF

Currently, there are no EU-wide quality requirements for RDF. The reason for this is the fact that various installations using RDF as a fuel have different requirements. Receiving installations have to comply with the emission limits for co-incineration and based on the technology and regular fuel they use this may lead to different demands for contaminant level and calorific value. As a result, individual installations do have requirements to their suppliers, but no EU-wide legal criteria have been set. For SRF, however, national quality criteria exist in some countries, e.g. in Finland, Germany and Italy (Van Tubergen et al. 2005).

The CEN/TC 343 Working Group 2 "Fuel Specifications and Classes" plan to come up with EU quality criteria for SRF in 2010. So far, a proposal was made based on three factors:

- Economic aspect: Net Caloric Value (the higher, the more regular fuel can be substituted)
- Technology aspect: Cl content (Chloride leads to corrosion in the installations)

- Environmental aspect: Hg (+Cd) content

Based on these factors five different classes of SRF can be distinguished, which are shown in Table 8-4. This classification is based on the specific needs of cement kilns, power plants and RDF plants.

Table 8-4. Proposed quality criteria for Solid Recovered Fuel (Van Tubergen et al. 2005)

Criterion	Unit	1	2	3	4	5
Net Calorific Value	MJ/kg ar[5]	≥25	≥20	≥15	≥10	≥3
Cl content	% DM[6]	≤0,2	≤0,6	≤1,0	≤1,5	≤3,0
Hg content	mg/MJ ar[7]	≤0,02	≤0,03	≤0,08	≤0,15	≤0,50

8.5.4 RDF treatment

RDF can be used as an additional fuel in industrial processes or be treated in dedicated RDF plants producing energy or syngas. Examples of installations using RDF as a main or additional energy input are

- cement kilns
- lime shafts
- power plants
- coke ovens
- brick furnaces
- district heating plants
- syngas production plants

In plants using RDF as an additional fuel, a wide variety in technologies is in use. In dedicated RDF plants the material is mostly gasified. Most common technologies for gasification are **fluidized bed** gasification or **fixed bed** gasification. In a fluidized bed reactor a carrier material (e.g. sand) is used and streamed through by a gas; thus it shows liquid-like behavior. This assures homogenous conditions within the reactor and good heat transfer possibilities. The produced syngas can either be directly combusted to produce heat or

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[5] ar = as received (related to wet material)

[6] DM = dry matter

[7] mg/MJ : related to energy content, not to mass. For Hg is related to median values of at least 10 measurements, Cl and NCV are mean values. For Hg alternatively an 80 percentile value exists, which is 2 times higher as the portrayed median values.

electricity. Another option is the use of the syngas as an input for the chemical industry. Also, combinations with existing installations are known, e.g. a RDF gasifier as an upstream fuel production unit of a power plant. Figure 8-13 shows the functional principle of a fluidized and of a fixed bed reactor.
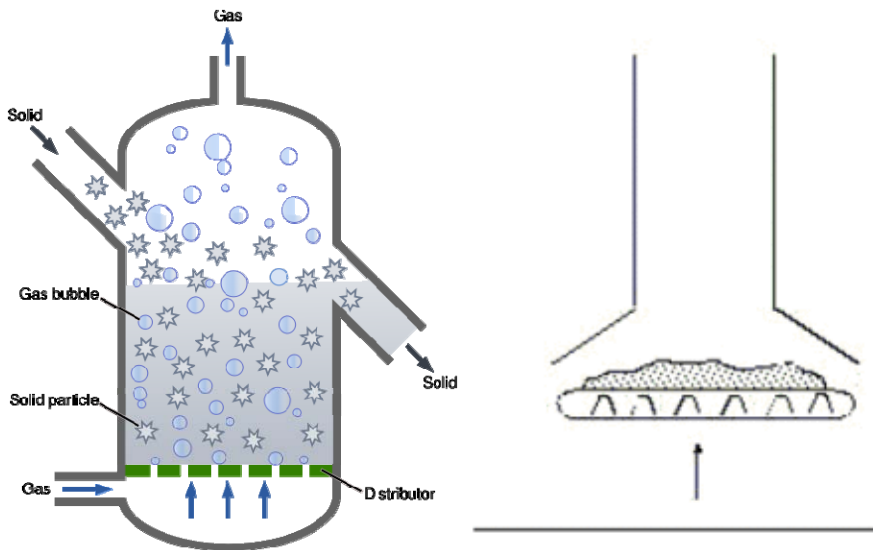

Figure 8-13. Fluidized bed (left) and fixed bed (right) gasification reactors (TUD 2010)

The production and use of RDF has been on the rise in the last decade. The reasons for this strong growth are the implementation of landfill bans for untreated municipal waste (e.g. in Germany, The Netherlands and Italy) and the high costs of landfilling (Gascoyne 2010). The production of SRF amounted to (Gascoyne 2010)

- 3 million tons in 2001
- 12 million tons in 2008 (of which 47% in Germany, followed by Sweden (8%), Italy (6%), Austria (5%) and the United Kingdom (4%).
- expected for 2015: 16,5 million tons

8.5.5 Emission limits for RDF treatment

The incineration directive sets emission limits for both incineration and co-incineration. For the co-incineration of pre-treated waste (which RDF is) the so-called mixing rule is applied (EP&C 2001). This means that the joint emission limit for the situation of co-incineration is based on the weighted average of the emission limits for the RDF and for the regular fuel. The weighing is based on the

flue gas volume resulting from the combustion from the secondary and regular fuel.

The emission limits for the regular fuel principally follow from the emission limits from the considered installation (national or other legislation). At all times the total emission limits for the most relevant heavy metals (Hg, Tl and Cd) and for dioxins from the joint co-incineration operation should be the same as from mono-incineration.

For combustion plants and cement kilns additional regulations are valid. For the emission limit share of the regular fuel in **combustion plants** (here only large plants are mentioned, >300MW$_{th}$)

- SO_2: 200 mg/Nm^3
- NO_x: 200 mg/Nm^3 for solid and liquid fuels, 300 mg/Nm3 for biomass
- Dust: 30 mg/Nm^3

For some emissions total limits for the joint co-incineration operation are given:

- other heavy metals: as for mono-incineration
- dioxins: as for mono-incineration

For **cement kilns** total emission limits for the joint co-incineration operation are set:

- SO_2: 50 mg/Nm^3, if the emission does not stem from the waste input, exemptions are possible.
- TOC: 10 mg/Nm^3, if the emission does not stem from the waste input, exemptions are possible.
- NO_x: 500 mg/Nm^3 for new plants
- Dust: 30 mg/Nm^3
- other emissions, apart from CO: as for mono-incineration

Incinerator operators do not like the situation in which co-incineration has emission limits deviating from those for mono-incineration. The Confederation of European Waste-to-Energy Plants (CEWEP) pleads for a 'level playing field' for all thermal waste treatment operations. On the national level the differences in emission limits may be more substantial (BMLFUW 2010)

9 Physical and chemical treatment

Physical and chemical treatment is a broad term defining the usage of various unit processes to achieve various goals:
- pretreatment waste before further recovery or disposal operations,
- direct recovery of waste,
- direct disposal of waste.

These processes are used mainly for the pretreatment of liquid and semi-liquid hazardous waste and according to the Waste Framework directive some of them may be classified as recovery or disposal processes as listed below:

R 2 Solvent reclamation/regeneration

R 3 Recycling/reclamation of organic substances which are not used as solvents

R 4 Recycling/reclamation of metals and metal compounds

R 5 Recycling/reclamation of other inorganic materials (including soil cleaning resulting in recovery of the soil and recycling of inorganic construction materials)

R 6 Regeneration of acids or bases

D 9 Physico-chemical treatment not specified elsewhere which results in final compounds or mixtures which are discarded by means of any of the operations numbered D 1 to D 12 (e.g. evaporation, drying, calcination, etc.)

9.1 Distillation and evaporation

Distillation is defined as the separation of two or more liquids of different volatility by vaporization and condensation. This process is used mainly in numerous industrial applications for the purification of chemicals and in solvent recycling.

Evaporation is defined as the conversion of a liquid into vapor. It is used for the separation of a liquid from a solid or slurry by vaporization. It is used for the treatment of radioactive liquids and sludges, metal-plating concentrated waste and in the treatment of landfill leachates.

Evaporation differs from distillation in that no attempt is made to fractionate the vapor (Freeman 1997). Both processes require the transfer of heat from a heating medium to the process fluid to vaporize the volatile solvent.

Two main types of distillation are distinguished:
- simple batch distillation,
- continuous distillation.
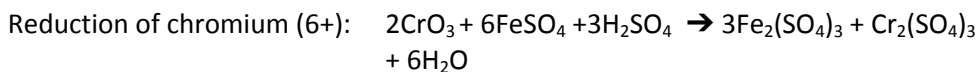
Six different types of evaporators are in use:
- rising-film evaporator,
- falling-film evaporator,
- forced-circulation evaporator,
- solar evaporators,
- batch-pan evaporator,
- natural-circulation evaporator.

The selection of the type of evaporator is based on the properties of the liquids to be treated such as heat capacity, heat of evaporation, density, thermal conductivity, boiling-point rise as well as heat-transfer coefficient, but other factors should also be taken into account (amount of waste to be treated, potential for corrosion, foaming, scaling, investment and operation costs).

9.2 Chemical oxidation and reduction

Chemical reactions in which the oxidation state of one reactant is raised while the oxidation state of the other reactant is lowered are defined as oxidation-reduction reactions (Freeman 1997).

These processes play an important role in the treatment of waste containing heavy metals (electroplating technologies) as well as inorganic and organic hazardous substances. The main oxidized or reduced compounds are sulfides, cyanides, chromium, phenols, pesticides and sulfur compounds. Typical reactions are presented below:

Oxidation of cyanides: $NaCN + H_2O_2 \rightarrow NaCNO + H_2O$

Reduction of chromium (6+): $2CrO_3 + 6FeSO_4 + 3H_2SO_4 \rightarrow 3Fe_2(SO_4)_3 + Cr_2(SO_4)_3 + 6H_2O$

Commonly used oxidation agents are listed below:
- Sodium hypochlorite $NaOCl$
- Calcium hypochlorite $Ca(OCl)_2$
- Hydrogen peroxide H_2O_2
- Chlorine Cl_2
- Ozone O_3
- Potassium permanganate $KMnO_4$

Typically used reduction agents:
- Sulfur dioxide SO_2
- Ferrous sulfate $FeSO_4$
- Bisulfite $NaHSO_3$
- Sodium borohydride $NaBH_4$

9.3 Filtration and separation

Filtration is defined as a method for the separation of solid particles from a fluid using a porous medium. Filtration needs a driving force which is a pressure gradient caused by gravity, centrifugal force and pressure lower or higher than atmospheric.

The main applications in waste management are
- dewatering of sludges after neutralization of acids and bases or precipitation of heavy metals as hydroxides, carbonates or sulfides,
- dewatering of oily sludges or floats before incineration or further treatment,
- dewatering of municipal wastewater sludges before further biological or thermal treatment.

Centrifuges, filter presses and belt filter presses are commonly used equipment for dewatering sludges.

9.4 Chemical precipitation

Chemical precipitation (ChP) – process by which a soluble substance is converted into an insoluble form either by chemical reaction or by changes in the composition of the solvent to diminish the solubility of the substance in it (Freeman 1997). The further removal of the precipitated solids is achieved by settlement and/or filtration. The main applications of ChP in waste management include the treatment and removal of heavy metals from aqueous hazardous waste such as Zn, Pb, Cd, Hg, Cu, Cr, As, Ni.

A typical scheme of an installation for the precipitation of heavy metals from aqueous solutions is presented in Fig. 9-1. A chemical agent is added to the solution in a precipitation tank equipped with stirrer.

The removal of the suspended reaction product from the water solution occurs in a clarifier and then the separated sludges are dewatered using filtration equipment.

Various chemicals are used as precipitation agents with lime being the most typical followed by sulfides and carbonates. Figure 9-2 shows the differences in solubilities of metal hydroxides and sulfides.
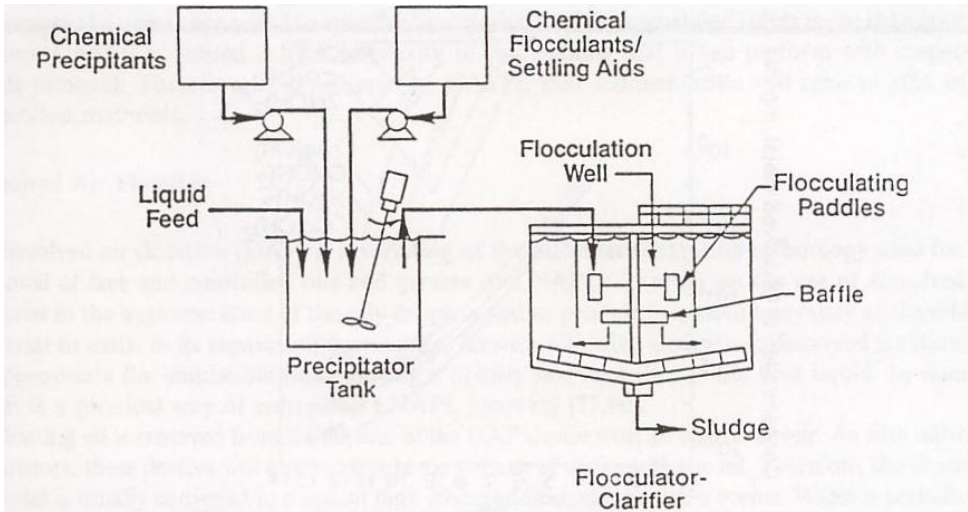
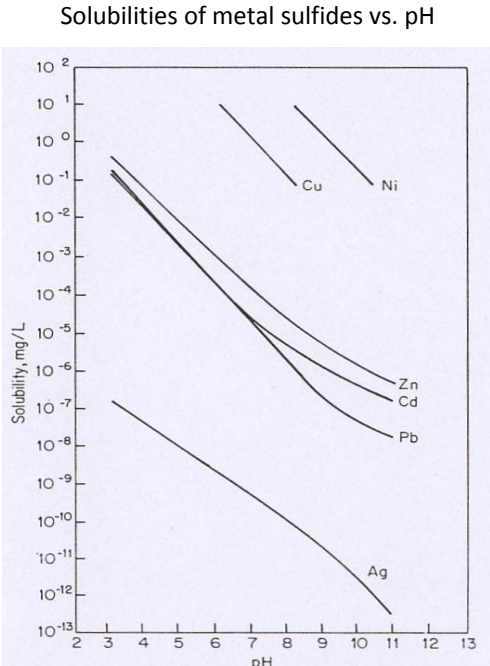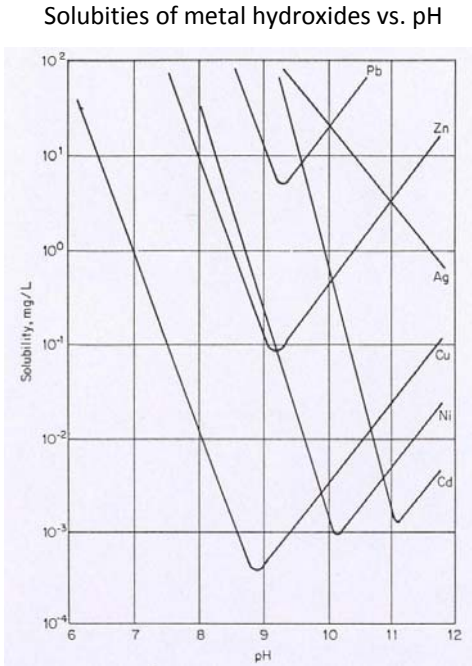**Figure 9-1. Typical flow chart of a precipitation installation (Freeman, 1997)**

Solubities of metal hydroxides vs. pH   Solubilities of metal sulfides vs. pH

Figure 9-2. Solubilities of metal hydroxides and sulfides as a function of pH (Freeman 1997)

Typical precipitation reactions are shown below.

Hydroxide precipitation $Me^{++} + Ca(OH)_2 \rightarrow Me(OH)_2 + Ca^{++}$

Sulfide precipitation $Me^{++} + FeS \rightarrow MeS + Fe^{++}$

Carbonate precipitation $Me^{++} + NaCO_3 \rightarrow MeCO_3 + 2Na^+$

9.5 Solidification and stabilization S/S

Solidification and stabilization are technologies use additives to decrease the mobility of pollutants and to produce a solid block of treated waste thereby enabling its landfilling under land disposal requirements. These processes improve waste-handling and the physical characteristics of waste, decrease the surface area exposed to the transfer or leaching of pollutants, decrease the solubility and in many cases detoxify the hazardous components. S/S technologies are designed for the treatment of liquid, semi-liquid, pasty and dusty waste. There are some definitions describing the processes that are part of S/S technologies (Freeman, 1997).

Solidification is defined as a process in which materials added to waste produce a solid, with or without any chemical reactions.

Stabilization is considered as a process by which a waste is converted to a more chemically stable form – this also includes solidification and chemical reactions to transform hazardous components to new non-hazardous substances or compounds.

Chemical fixation is defined as a chemical reaction causing the transformation of hazardous contaminants into new non-hazardous forms involving the bonding of the contaminate to the binder.

Encapsulation is a process involving the complete coating or closure of hazardous particles or waste agglomerates with a new substance, e.g. the S/S additive or binder.

Below are presented some types of mineral hazardous waste subject to S/S technologies (codes according to the Decision 2000/532/EC)

10 01 wastes from power stations and other combustion plants (except 19)
10 01 14*   bottom ash, slag and boiler dust from co-incineration containing dangerous substances
10 01 16*   fly ash from co-incineration containing dangerous substances

10 01 18*   wastes from gas cleaning containing dangerous substances

19 01 wastes from incineration or pyrolysis of waste
19 01 06*   aqueous liquid wastes from gas treatment and other aqueous
                liquid wastes
19 01 13*   fly ash containing dangerous substances
19 01 15*   boiler dust containing dangerous substances

The European List of Wastes (Decision 2000/532/EC) lists specific types of waste
        produced in S/S technologies:
19 03   stabilized/solidified wastes
19 03 04*   wastes marked as hazardous, partly stabilized
19 03 05   stabilized wastes other than those mentioned in 19 03 04
19 03 06*   wastes marked as hazardous, solidified
19 03 07   solidified wastes other than those mentioned in 19 03 06

According to definitions included in the Decision 2000/532/EC:
-    stabilization processes change the danger of the constituents in the waste
     and thus transform hazardous waste into non-hazardous waste.
     Solidification processes only change the physical state of the waste (e.g.
     liquid into solid) by using additives without changing the chemical
     properties of the waste.
-    waste is considered as partly stabilized if, after the stabilization process,
     dangerous constituents which have not been changed completely into
     non-dangerous constituents can be released into the environment in the
     short, middle or long term.

S/S technology may be characterized by the types of binders and types of
    processes. There are three main types of binders used:
•   inorganic binders – combinations of hydraulic cements, lime, pozzolans,
     gypsum and silicates
•   organic binders – epoxy, polyesters, asphalt, polyolefins, urea-
     formaldehyde
•   combinations of inorganic and organic binders, e.g. diatomaceous earth
     with cement and polystyrene; polyurethane and cement; polymer gels
     with silicate and lime cement.

Four main process types may be distinguished, as described below (Freeman,
1997):

- **in-drum processing (small scale)** – S/S binders are added to the waste in drum or container, then mixing, setting, disposal of waste with the drum,
- **in-plant processing** – stationary plant designed for S/S of bulk waste materials, from internal or external waste sources,
- **mobile-plant processing** – S/S equipment mobile or easy to transport and set up from site to site (e.g. for decontamination of polluted land),
- *in-situ* **processing** – addition of binders directly to a lagoon or the injection of S/S materials to the soil subsurface etc., applicable to remediation projects of contaminated land.

There are few factors affecting the selection and performance of S/S processes:

1) Treatment objective
   - change of waste consistency from liquid or dusty to solid – only solidification of waste is required,
   - making the waste acceptable for landfilling – solidification of waste has to fulfill acceptance criteria at different classes of landfills (min. solidification, leaching test - *Council decision 2003/33/EC, establishing criteria and procedures for the acceptance of waste at landfills*)
   - changing the waste properties and classification from hazardous to non-hazardous waste to be accepted for disposal in a non-hazardous waste disposal facility or for recovery (stabilization of waste, leaching test, EWC)

2) Waste composition and characteristics
   - Compatibility of waste and binder,
   - potential adverse effects of some organics and salts on cement and pozzolan-based S/S processes,
   - easier and more successful solidifying and stabilizing of inorganics than organics.
   - physical characteristics of waste and binders – particle size and shape in waste and binder, viscosity of mixture, proper water/binder ratio, intensity of mixing – positive or adverse effect on the strength of the final product.

3) Process type and processing requirements
   - process and installation type – mobile plant, stationary plant, in situ etc., easier stabilize waste in a plant than in situ,
   - specific process conditions
       ✓ modification of waste (e.g. dewatering),
       ✓ mixing modes,
       ✓ transportation,
       ✓ placement and storage.

4) S/S waste-management requirements - two main possibilities for product management:
  - landfilling depending on different degrees of stabilization and solidification and product leaching characteristics (three classes of landfills, different gate fees at landfills depending on waste type),
  - recovery of inert products in earth works.

There are two main criteria for evaluating the efficiency of S/S process: leachability and strength.

The leachability of solidified waste is to be measured by test methods according to Council Decision 2003/33/EC establishing criteria and procedures for the acceptance of waste at landfills. Threshold values for the strength of solidified waste were not established at the EU level.

5) Regulatory requirements concerning
  - methodology of leaching and limit values for waste designed for particular classes of landfills,
  - other waste acceptance criteria,
  - possibility for using solidified waste in a landfill operation.

6) Economics

Cost of S/S processing is rather low in comparison to other technologies (cheap reagents: cement, lime, fly ash, readily available rather simple equipment from the concrete industry). Cost depends on
  - characteristics of waste,
  - transport requirements of waste and reagents to the plants and finished products for disposal or recovery,
  - process type and type of reagents selected (cement cheaper than polyolefins),
  - increase of waste volume and mass after S/S processing,
  - other factors such as special health and safety requirements, special legal requirements, etc.

10 Landfilling

10.1 Legal framework

Landfilling of waste in the EU is regulated by

Council Directive 1999/31/EC of 26 April 1999 on the landfilling of waste

Council Decision 2003/33/EC of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 of and Annex II to Directive 1999/31/EC.

A landfill is defined as a waste disposal site for the depositing of waste onto or into land (i.e. underground), including

- internal waste disposal sites (i.e. a landfill where the producer of waste carries out waste disposal at the place of production), and

- a permanent site (i.e. more than one year) which is used for the temporary storage of waste

excluding

- facilities where waste is unloaded in order to permit its preparation for further transport for recovery, treatment or disposal elsewhere and

- storage of waste prior to recovery or treatment for a period less than three years as a general rule, or

- storage of waste prior to disposal for a period less than one year.

Three classes of landfills are distinguished:

- landfills for hazardous waste;

- landfills for non-hazardous waste;

- landfills for inert waste.

Landfills are regarded as necessary in all waste management systems (according to the German strategy – after 2020 there will be no more landfilling of municipal waste) and they shall not be noxious for the environment during operation and after landfill closure (30 years of aftercare monitoring). They are designed for raw, untreated waste (there is a ban on the landfilling of untreated waste with an exception of inert waste) and treated (stabilized) waste.

10.2 Landfill as a multibarrier system

The concept of controlling reactor landfills by barriers was developed in the 1980s is still valid for today's landfills. The landfills where untreated waste is deposited are defined as reactor landfills due to the biochemical processes occurring in the landfill body leading to emissions of gases and leachates. The five barriers are presented in Figure 10-1. The barriers 1, 2 and 5 should be regarded as physical chemical barriers, but barriers 3 and 4 express the quality of waste pre-treated before landfilling and the landfill techniques – operation of the landfill.

10.2.1   Barrier 1 – landfill location

This barrier is defined as the landfill location and comprises various elements, such as

- presence of a natural geological barrier,
- distance to residential areas (dust, odor, noise, microorganisms),
- access roads (noise, accident hazard, increased traffic, dust, exhaust gases),
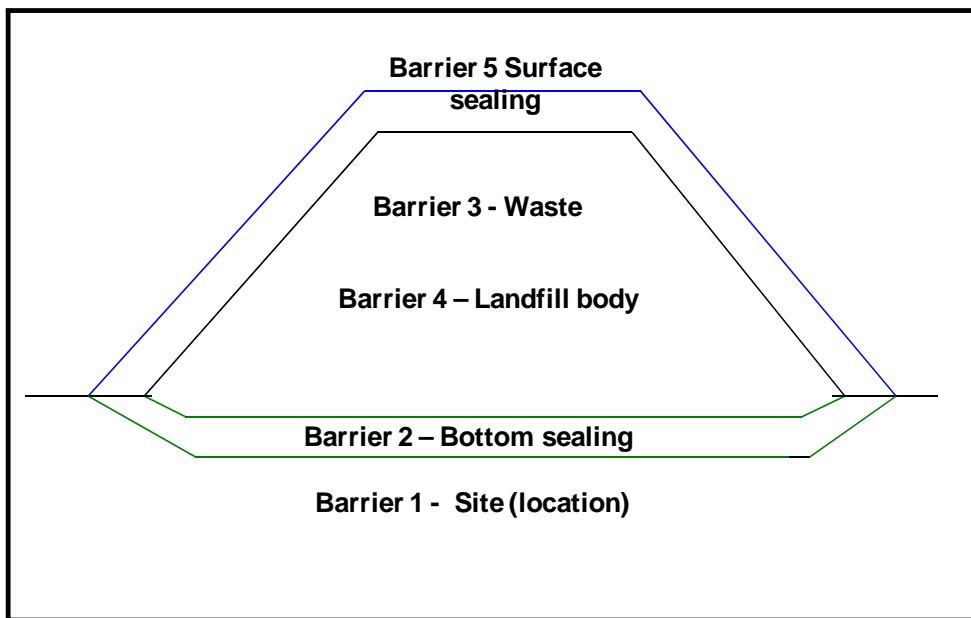- aesthetic aspects – landscape, greens.

Figure 10-1. Concept of the landfill as a multibarrier system

The location of a landfill must take into consideration requirements relating to
- the distances from the boundary of the site to residential and recreation areas, waterways, water bodies and other agricultural or urban sites,

- the existence of groundwater, coastal water or nature protection zones in the area,
- the geological and hydrogeological conditions in the area,
- the risk of flooding, subsistence, landslides or avalanches on the site,
- the protection of the nature or cultural patrimony in the area.

The landfill must be situated and designed so as to meet the necessary conditions for preventing pollution of the soil, groundwater and surface water and ensuring efficient collection of leachate.

The landfill base and sides shall consist of a mineral layer which satisfies permeability and thickness requirements with a combined effect in terms of protection of soil, groundwater and surface water at least equivalent to the following requirements:

- landfill for hazardous waste $k \leq 1,0 \times 10^{-9}$ m/s; thickness ≥ 5 m,

- landfill for non-hazardous waste $k \leq 1,0 \times 10^{-9}$ m/s; thickness ≥ 1 m,

- landfill for inert waste $k \leq 1,0 \times 10^{-7}$ m/s; thickness ≥ 1 m.

10.2.2 Barrier 2 – bottom sealing

According to the landfill directive, protection of soil, groundwater and surface water in the landfill vicinity is to be achieved by
- the combination of a geological barrier and a bottom liner during the operational/active phase and by
- the combination of a geological barrier and a top liner during the passive phase/post closure.

The geological barrier is determined by the hydrogeological conditions below and in the vicinity of the landfill site providing sufficient attenuation capacity to prevent the potential risk to soil and groundwater. Where the geological barrier does not naturally meet the permeability and thickness conditions it can be completed artificially and reinforced by other means giving equivalent protection. The thickness of an artificially established geological barrier should be no less than 0,5 meters. Additionally, in landfills for hazardous and non-hazardous waste, an artificial sealing system and a leachate drainage and collection system must be added to minimize leachate accumulation at the base of the landfill.

Two various types of artificial liners are used for landfill construction:
- geomembranes PEHD, thickness 1-2,5 mm, useful for sealing the bottom and top and welded to assure total tightness,
- bentonite liners (thickness 6-10 mm after swelling), allowed only for top sealing.

Conditions for the construction of the drainage layer, according to Polish regulations:
- thickness minimum 0,50 m
- permeability $> 10^{-4}$ m/s
- drainage pipes in the drainage layer
- drainage of internal landfill slopes – flow of leachates to the main drainage system.

10.2.3   Barrier 3 – deposited wastes

This barrier is defined as the quality and quality control of waste to be deposited in the landfill of particular classes. There is a general obligation to recover and treat the waste before landfilling as well as to minimize the amount and noxiousness of waste deposited.

According to the landfill directive:
- hazardous waste must be assigned to a hazardous waste landfill;
- landfills for non-hazardous waste may be used for municipal waste, for non-hazardous waste (fulfilling the acceptance criteria), for stable non-reactive hazardous waste (e.g. solidified, vitrified)
- landfill sites for inert waste must be used only for inert waste;
- criteria for the acceptance of waste at each landfill class must be fulfilled.

The criteria and procedures for the acceptance of waste at landfills were determined in the Council Decision 2003/33/EC.

The procedure for the acceptance of waste at landfills comprises three levels:
Level 1 Basic characterization- a thorough determination, according to standardized analysis and behavior-testing methods, of the short and long-term leaching behavior and/or characteristic properties of the waste
Level 2 Compliance testing: periodical testing by simpler standardized analysis and behavior-testing methods to determine whether a waste complies with

permit conditions and/or the specific reference criteria. The tests focus on key variables and behavior identified by basic characterization

Level 3 On-site verification – rapid check methods to confirm that a waste is the same as that has been subjected to compliance testing and that which is described in the accompanying documents. It may merely consist of a visual inspection of a load of waste before and after unloading at the landfill site.

Waste acceptance criteria are different for each class of landfill.

Criteria of landfills for inert waste include three basic elements:
- list of wastes acceptable at landfills for inert waste without testing (mineral waste like glass, concrete, bricks, soil and stones, tiles and ceramics and mixtures of these wastes),
- leaching limit values determined in standard leaching tests,
- limit values for the total content of organic parameters such as TOC, BTEX, PCBs, mineral oil and PAHs.

Criteria of landfills for non-hazardous waste include three basic components:
- list of wastes acceptable at landfills for non-hazardous waste without testing (non-hazardous municipal waste, separately collected non-hazardous fractions of household wastes and the same non-hazardous materials from other origins),
- leaching limit values determined in standard leaching tests,
- ban on landfilling gypsum waste together with biodegradable waste.

Criteria for hazardous waste acceptable at landfills for non-hazardous waste consists of three basic elements:
- leaching limit values determined in standard leaching tests,
- other criteria for granular waste (TOC, pH, ANC – acid neutralization capacity),
- special criteria for asbestos waste (limited requirements for landfilling of only construction material containing asbestos).

Criteria for waste acceptable at landfills for hazardous waste include two basic elements:
- leaching limit values determined in a standard leaching tests,
- other criteria for granular waste (TOC, LOI – loss on ignition, ANC – acid neutralization capacity).

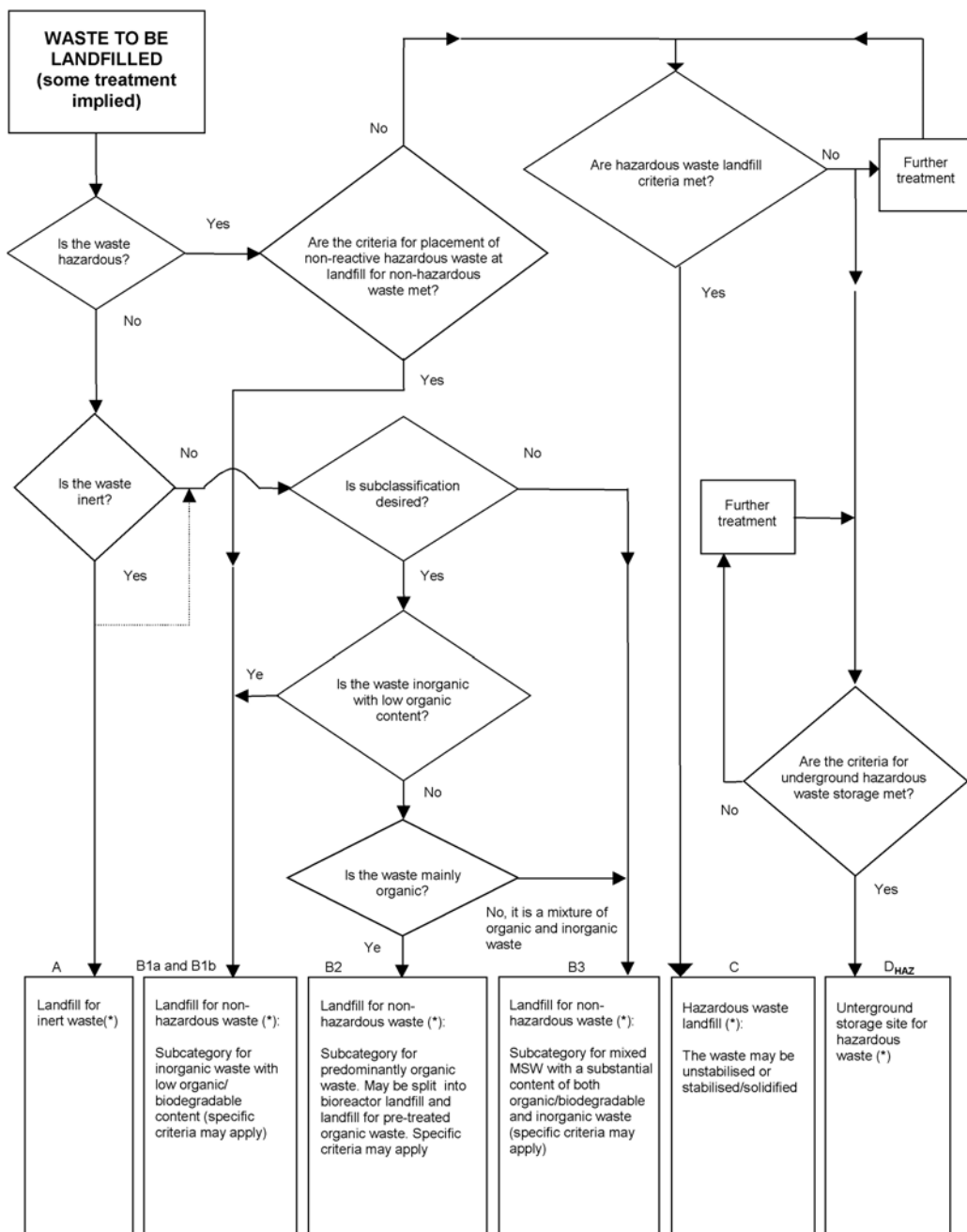A set of special criteria was established for underground storage sites.

Table 10-1 contains a comparison of leaching limit values calculated at a liquid/solid ratio 10 l/kg for the total release at various classes of landfills. There are also other limit values for other leaching tests.

Table 10-1. Leaching limit values, mg/kg dry substance

Parameter	Inert waste landfill	Non-hazardous waste landfill	Hazardous waste at a non-hazardous waste landfill	Hazardous waste landfill
As	0,5	2	2	25
Ba	20	100	100	300
Cd	0,04	1	1	5
Cr total	0,5	10	10	70
Cu	2	50	50	100
Hg	0,01	0,2	0,2	2
Mo	0,5	10	10	30
Ni	0,4	10	10	40
Pb	0,5	10	10	50
Sb	0,06	0,7	0,7	5
Se	0,1	0,5	0,5	7
Zn	4	50	50	200
Chloride	800	15000	15000	25000
Fluoride	10	150	150	500
Sulfate	1000	20000	20000	50000
Phenol index	1,0	-	-	-
DOC	500	800	800	1000
TDS	4000	60000	60000	100000

Figure 10-2 shows the landfilling options provided by the Landfill Directive

Some countries such as Austria and Germany introduced their own criteria for the acceptance of waste at landfills for non-hazardous waste and they are presented in Table 10-3. In Poland, special criteria for the acceptance of stabilized waste at landfills for non-hazardous waste are recommended.

WASTE TO BE LANDFILLED (some treatment implied)

Is the waste hazardous?

Yes — Are the criteria for placement of non-reactive hazardous waste at landfill for non-hazardous waste met?

No

Are hazardous waste landfill criteria met?

No — Further treatment

Yes

No

Yes

Is the waste inert?

No

Is subclassification desired?

No

Yes

Yes

Further treatment

Is the waste inorganic with low organic content?

Ye(s)

No

Are the criteria for underground hazardous waste storage met?

No

Is the waste mainly organic?

No, it is a mixture of organic and inorganic waste

Ye(s)

Yes

A
Landfill for inert waste(*)

B1a and B1b
Landfill for non-hazardous waste (*):
Subcategory for inorganic waste with low organic/ biodegradable content (specific criteria may apply)

B2
Landfill for non-hazardous waste (*):
Subcategory for predominantly organic waste. May be split into bioreactor landfill and landfill for pre-treated organic waste. Specific criteria may apply

B3
Landfill for non-hazardous waste (*):
Subcategory for mixed MSW with a substantial content of both organic/biodegradable and inorganic waste (specific criteria may apply)

C
Hazardous waste landfill (*):
The waste may be unstabilised or stabilised/solidified

D_{HAZ}
Unterground storage site for hazardous waste (*)

(*) In principle, underground storage is also possible for inert and non-hazardous waste.

Figure 10-2. Landfill options provided by the Landfill Directive (Council Decision 2003/33/EC)

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Table 10-2. Special criteria for the acceptance of waste at landfills

	Unit	Germany, after incineration	Germany, after MBP	Austria, after MBP	Poland , after MBP
TOC	% DM	5	18		
TOC of eluate	g/m^3		250		
Heat of combustion	kJ/g DM		6000	6000	
AT_4	mgO_2/g DM		5	7	10
GB_{21}	dm^3/kg DM		20	20	
Reduction of TOC or LOI during MBP	% of the initial content				40

According to the Landfill directive the acceptance procedure at the landfill site comprises the following 4 four steps:

1. before or at the time of delivery, or of the first in a series of deliveries of unchanged waste, the holder or the operator can show, by means of the appropriate documentation, that the waste in question can be accepted at the site according to the conditions set in the permit and it fulfills the acceptance criteria,

2. the following reception procedures are respected by the operator:
   - checking the waste documentation,
   - visual inspection of waste at the entrance and at the point of deposit and, as appropriate, verification of conformity with the description provided in the documentation submitted by the holder,
   - keeping the register of the quantities and characteristics of the waste deposited, indicating origin, date of delivery, identity of producer or collector in the case of municipal waste,

3. the operator of the landfill shall always provide written acknowledgement of receipt of each delivery accepted on the site

4. if waste is not accepted at the landfill the operator shall notify without delay the competent authority of the non-acceptance of the waste.

The following wastes are not accepted for landfilling:
- liquid waste;
- flammable waste;
- explosive or oxidizing waste;
- hospital and other clinical waste which is infectious;
- used tires, with certain exceptions;
- any other type of waste which does not meet the acceptance criteria.

10.2.4   Barrier 4 – landfill body

The landfill body as a barrier is regarded generally as a waste landfilling technology and landfill operation leading to the minimization of environmental impact, including
- minimization of working area (division of landfill in sectors),
- compaction, crushing and placing of waste in thin layers (0,2-0,5 m),
- using appropriate equipment for landfill operation (compactors, loaders etc.),
- moisture control of landfill body (recirculation of leachates),
- daily covers,
- current recultivation of slopes and top,
- control of dust emissions and windblown light materials,
- water and gas management.

Besides the general classification of landfills into three classes, there are also additional classifications of landfills receiving biodegradable wastes as "biological reactors" which can be operated as strongly compacted – anaerobic or weakly compacted – aerobic-anaerobic landfills.

According to Annex I of the Landfill Directive, appropriate measures shall be taken in order to
- control water from precipitation entering into the landfill body,
- prevent surface water and/or groundwater from entering into the landfill body,
- collect contaminated water and leachate,

- treat contaminated water and leachate collected from the landfill to the appropriate standard required for their discharge.

These provisions may not apply to landfills for inert waste.

The above mentioned Annex I also calls for appropriate measures to be taken in order to control the accumulation and migration of landfill gas:

- the collection of gas from all landfills receiving biodegradable waste and the landfill gas must be treated and used. If the gas collected cannot be used to produce energy, it must be flared.
- the collection, treatment and use of landfill gas shall be carried out in a manner which minimizes damage to or deterioration of the environment and risk to human health.

Leachate and gas management are described in more detail in other sections.

10.2.5 Barrier 5 – landfill top cover

The technical design of the landfill top cover must fulfill two main functions:

- ensure the long-term integrity and stability of the landfill by taking into account any possible emissions to the environment,
- support the development of vegetation growth or other final usage of the landfill site.

If the prevention of leachate formation is necessary, surface sealing may be prescribed.

Annex I of the Landfill Directive contains the following recommendations for the construction of the surface cover.

Elements of sealing	for non-hazardous waste	for hazardous waste
Gas drainage layer	required	not required
Artificial sealing liner	not required	required
Impermeable mineral layer	required	required
Drainage layer >0,5 m	required	required
Top soil cover >1,0 m	required	required

An example of a landfill cover design for biodegradable waste is presented in Figure 10-3.
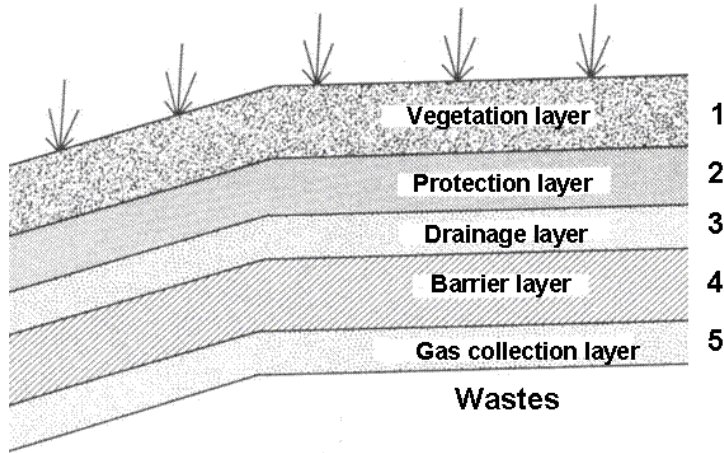
Typical materials: 1- topsoil, 2- soil, 3- gravel, sand, geonet, 4- compacted clay, geomembrane, geosynthetic clay liner, 5 – gravel, sand, geotextile, geonet

Figure 10-3. Elements of top cover systems – adopted from Sharma (1994)

10.2.6 Landfill biogas management

Biogas forms from the decomposition of biodegradable organic matter present in waste under anaerobic conditions. The process of anaerobic digestion is described in greater detail in chapter 6.3.

The main difference between the anaerobic processes in landfills and the anaerobic processes in the biological reactors of waste treatment plants is that there is only partial control of the reactions occurring in the landfill resulting in an unstable amount and quality of biogas generated there. A typical diagram of biogas generation in landfills is presented in Figure 10-4.

There are a variety of models describing biogas generation in landfills. The results of these calculations may differ significantly due to many assumptions introduced to the models and the lack of verification under various landfill conditions (Figure 10-5).
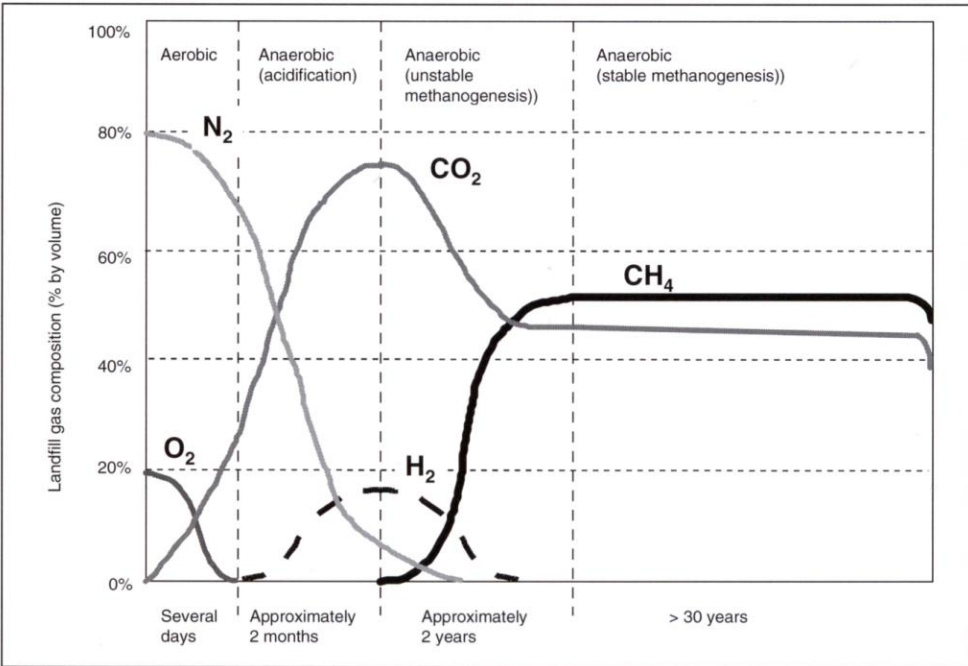
Figure 10-4. The composition of landfill gas over time (McDougall, 2001)
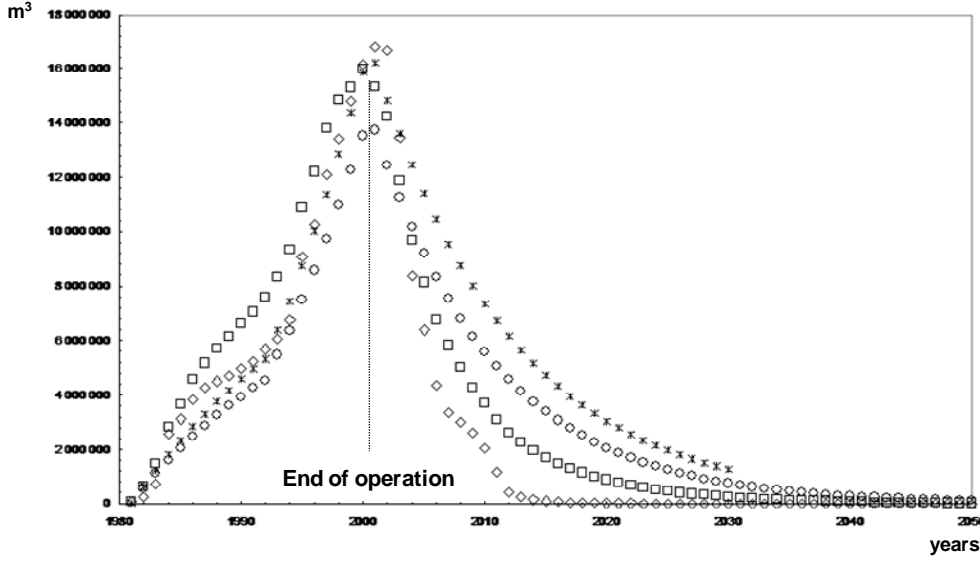
Figure 10-5. Results of gas production modeling using various models

The example of the WAMECO model for gas generation is presented below.

For $t_x < t_{1/2}$

$$dG \:/\: dT \:=\: k_1 \cdot G_0 \cdot N_{nx} \cdot \exp\left[-\:k_1 \cdot \left(t_{1/2} - t_x\right)\right]$$

For $t_x \geq t_{1/2}$

$$dG / dT = k_2 \cdot G_0 \cdot N_{nx} \cdot \exp\left[-k_2 \cdot \left(t_x - t_{1/2}\right)\right]$$

$k_1 = \ln(50) / t_{1/2}$
$k_2 = \ln(50) / (t_{99/100} - t_{1/2})$
$G_0 = 1,868\ Q\ C_{org}[0,014\ (T-273) + 0,28]$

where:

dG/dt	gas generation rate, m^3/a
G_0	total theoretical gas generation from waste, m^3
N_{nx}	participation of three fractions of waste with various rates of decomposition
k_1	kinetic constant of the first phase of anaerobic decomposition, a^{-1}
k_2	kinetic constant of the second phase of anaerobic decomposition, a^{-1}
$t_{1/2}$	time of decomposition of 50 % of mass of a given fraction of waste, a
$t_{99/100}$	time of decomposition of 99 % of mass of a given fraction of waste, a
t_x	calculation year
C_{org}	organic carbon content of waste, kg/ton
Q	amount of deposited waste, ton/a
T	temperature of decomposition, K

The typical range of biogas production in landfills receiving untreated municipal waste amounts to ca. 100-200 m^3/ton of waste. Gas production in landfills with stabilized waste after MBP is lower than 50 m^3/ton of waste and the gas production occurs rather slowly due to the lack of easily biodegradable fractions of waste which were decomposed in the MBP plant.

Biogas may cause adverse effects due to its hazardous properties, such as
- explosiveness – in mixture with air (4-14 % methane in air),
- toxicity – contains toxic components, such as carbon monoxide and hydrogen sulfide,

- greenhouse effect – due to the content of methane and carbon dioxide,
- pollution of the atmosphere,
- noxiousness due to odor emission,
- ability to migrate in soil, especially around a landfill without a sealed base,
- biogas emissions make landfill reclamation and restoration difficult.

However, the main component of biogas – methane (ca. 50-65% of the biogas volume) – has energy value and is regarded as a renewable energy source. CO_2 emissions from biogas burning for energy recovery purposes are considered as neutral. Typical energy values of biogas amount to ca. to 18-20 MJ/m^3 depending on the methane content. There is an obligation for the collection, treatment and energy use of biogas from all landfills receiving biodegradable waste. Cogeneration of heat and power (CHP) belongs to the most widely used biogas energy recovery methods. Other solutions include improving biogas to natural gas quality for supplying gas networks, using biogas outside the landfill for heating various objects such as greenhouses, industrial plants, etc. If the biogas collected cannot be used to produce energy it must be flared.

Biogas is collected in landfills using gas wells of various constructions under the natural pressure of the landfill body (passive collection) or under a vacuum produced by a suction system (active collection – figure 10-6).

Figure 10-6. Gas collection wells (active collection)

The example of gas energy recovery at the landfill Barycz (Figure 10-7):
- landfill surface 36 ha, closed 26 ha, under operation 10 ha
- operation since 1974, gas collection since 1992
- present gas collection on the surface 26 ha
- 4 energy modules – 1996, 1998, 2002
  - 2 x (300 kW el + 380 kW th)
  - 1 x (373 kW el + 553 kW th)
  - 1 x (460 kWel + 590 kWth)
- Theoretical gas generation – 3400 m^3/h
- Total gas consumption by modules – 750 m^3/h   (22 % of theoretical generation)
- Production of electricity – max. 7784 MWh, average 4400 MWh
- Sale of electrical energy 3800 MWh

Figure 10-7. CHP units at the landfill site Barycz

10.2.7   Landfill leachate management

The water balance of the landfill may be in part influenced by its controlled operation and proper methods of landfill closure and recultivation. Figure 10-8 shows the water balance of a landfill in operation. The main input is water from precipitation; water from chemical reactions may be omitted due to its small amount. For the operation phase, the most important element of the water balance is the retention of water in the landfill body because this may significantly reduce the amount of leachates taking into account the small share of evaporation from the waste surface in the water balance. Two different concepts of landfill closure are presented in Figures 10-9 and 10-10. The first concept does not provide any liner in the top cover and thus allows for limited infiltration of precipitation into the landfill body and the creation of leachates on the bottom liner. The second concept provides a top liner eliminating the infiltration of precipitation into the landfill body and preventing leachate from occurring on the landfill bottom.
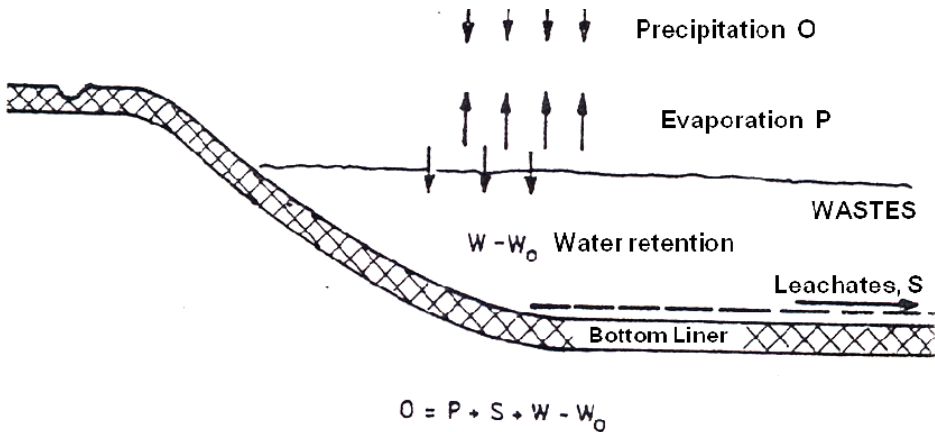
Figure 10-8. Water balance of a landfill under operation

$$O = P_g + P_r + T + A + B - B_o + W - W_o + S$$
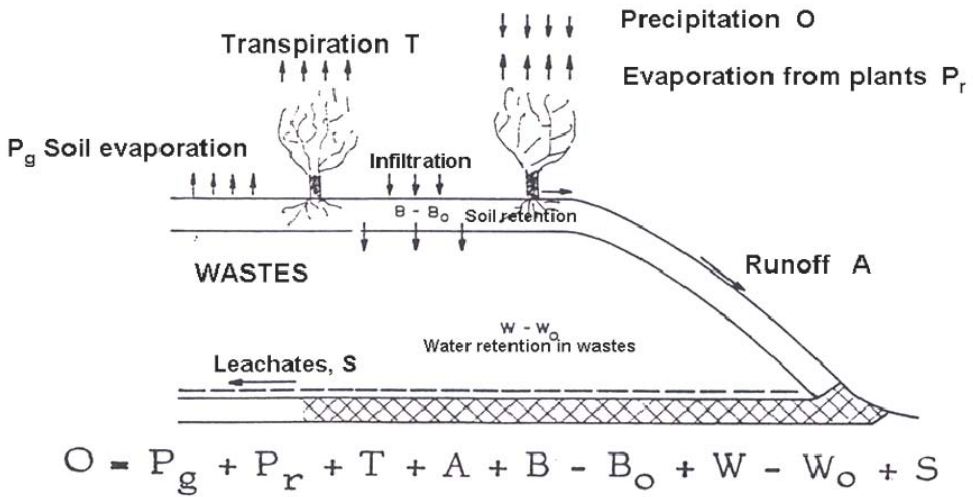
Figure 10-9. Water balance of a closed landfill without a top liner

$$O = P_g + P_r + T + A + A_p + B - B_o$$
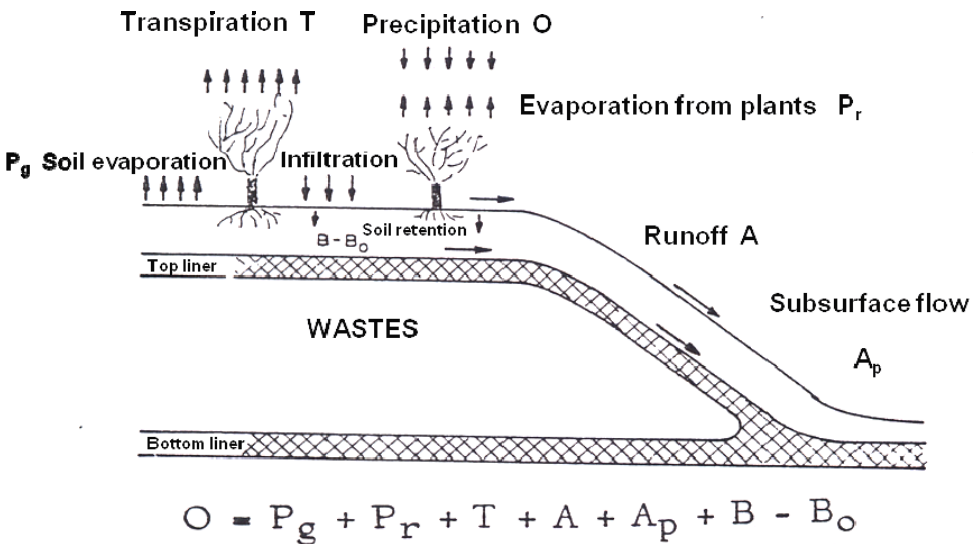
Figure 10-10. Water balance of a closed landfill with a top liner

The leachate flow during landfill operation depends on many factors such as
- landfill age
- waste compaction – density of waste in the landfill body

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- height of the landfill body
- amount of precipitation.

The amount of leachates grows with landfill age due to the decreasing water retention capacity with the degradation of organic matter, the increasing amount of precipitation over time as well as the decreasing waste density and height of the landfill body.

The average flow of leachates under Polish conditions amounts to ca. 25 % of annual precipitation. For precipitation of 700 mm/a leachate flow amounts on average to 1750 m^3/ha per year and ca. 5 m^3/ha per day. For precipitation of 500 mm/a leachate flow amounts to ca. 10 % of precipitation and for 800-900 mm/a it increases to ca. 30-40 % of precipitation. The factor of irregularity of leachate flow amounts to 3-5.

Leachates show a very high variability in composition depending on
- landfill age (decreasing biodegradability with age)
- landfilling technique / compaction of waste (higher concentrations of organic matter in leachates from landfills with strongly compacted wastes)
- pretreatment / stabilization of waste (lower biodegradability)
- precipitation – leachate dilution
- origin of waste – municipality / rural area (for example in Poland more mineral components of waste from rural areas, lower concentrations of organic matter in leachates)
- season – heating period / other periods (for example in Poland more mineral components of municipal waste in winter).

Typical pollutants and their concentrations characterizing the chemical composition of leachate are
- high BOD_5, COD, TOC,
- high concentrations of ammonia and organic nitrogen,
- high salinity – chlorides and sulfates,
- high hardness and alkalinity,
- acidic to alkaline pH,
- rather low concentrations of heavy metals,
- organic micropollutants.

Ranges of main pollutant concentrations in leachates from landfills of untreated municipal waste are presented in Table 10-3

Table 10-3. Characteristics of leachates from various landfills receiving untreated waste

Parameter	Unit	Landfill S1		S2	S3	Landfill S4	
Age	Years	1,0	12,3	2	0,5	1-3	16
pH		7,2-8,1	8,6-8,9	7,6	6,7	7,2	7,1
COD	g O_2/m^3	18300-58400	3380-4000	4500	1834	1148	58
BOD$_5$	g O_2/m^3	8400	300	2600			
BOD$_5$ / COD		0,46	0,09	0,58			
Ammonia nitrogen	g N/m^3	1302-2915	224	438	155	300	25
Organic nitrogen	g N/m^3	448-955	90-126		7	32	2
Chlorides	g Cl$^-$ /m^3	4500-7000	4460-5300	1270	810	1000	104
Sulfates	g SO_4/m^3	410-1053	74-132		103	64	346
Fe	g /m^3	2,5-150	5,0-6,25		150	8,1	1,42
Zn	g /m^3	9,5-24	6,25-12	3	0,17	0,38	0,18
Cu	g /m^3	0,5-3	0-1		0,01	0,06	0,06
Cr	g /m^3	3-9	1,25-1,5	0,9	0,02	0,20	0,06
Ni	g /m^3	0-5	0-0,4	3	0,19	0,24	
Cd	g /m^3		trace		0,03	0,04	

Leachates from aerobic landfills and from landfills with MBP pretreated waste show much lower concentrations of pollutants (Tables 10-5 and 10-6).

Table 10-4. Composition of leachates from aerobic landfills

Parameter	Unit	Landfill 1	Landfill 2
COD	g O_2/m^3	414-1064	600-1200
BOD$_5$	g O_2/m^3	<3 - 41	<20
Ammonia nitrogen	g N/m^3	<0,1 - 132	150-250
AOX	g/m^3	0,3-1,4	0,8-1,5

Leachate management includes the following steps:
- Minimization of leachate flow and concentration of pollutants
- Collection and retention in a basin
- Recirculation to the landfill body
- Pretreatment / treatment in local installations
- Discharge to municipal WWTP
- Treatment in a municipal WWTP

Table 10-5. Composition of runoff water and leachates from a MBP stabilized waste landfill (Ziehmann, 2002)

Parameter	Unit	Runoff		Leachates	
		at the beginning	after 7 months	at the beginning	after 7 months
pH	pH	7,0	7,4	7,1	7,5
Conductivity	mS/cm	7,0	2,94	16,6	3,71
TOC	g/m^3	429	122	1812	98
COD	$g\ O_2/m^3$	2000	352	4670	452
BOD$_5$	$g\ O_2/m^3$	66	19	244	15
Ammonia nitrogen	$g\ N/m^3$	89	<5	392	<5
Loss on ignition	g/m^3	5842	2280	14722	3009
TDS	g/m^3	1065	134	4015	167

The minimization of leachate flow is possible, but is strongly connected with waste landfilling technology – strong compaction and homogenization of waste by landfilling in thin waste layers, landfilling in small sectors to increase the height of yearly deposition and recirculation of leachates.

Due to a high variability of leachate flow and concentrations of pollutants, it is necessary to build leachate collection and equalization basins. They are used before leachate recirculation as well as before disposal to a municipal WWTP or local treatment plant. Aerated basins are recommended with mechanical aeration or compressed coarse air bubbles.

The required retention time of leachates in basins amounts to
- min. 7 days – before disposal to a municipal WWTP
- min. 2 months – before recirculation.

Recirculation to the landfill body is used for several purposes:
- increasing the moisture content in landfill reactors to intensify the
  decomposition of organic matter and the production of biogas,
- reducing leachate flow by using the water retention capacity of waste and
  increasing evaporation from the landfill surface,
- improving the compaction of waste and allowing better filling of landfill volume.

Pretreatment in local installations is implemented before leachate recirculation into the landfill body or before discharging to the municipal sewer and wastewater treatment systems. Full treatment in local installations must provide

the possibility for the direct discharge of treated leachates to surface water or the use for technological purposes at the landfill site.

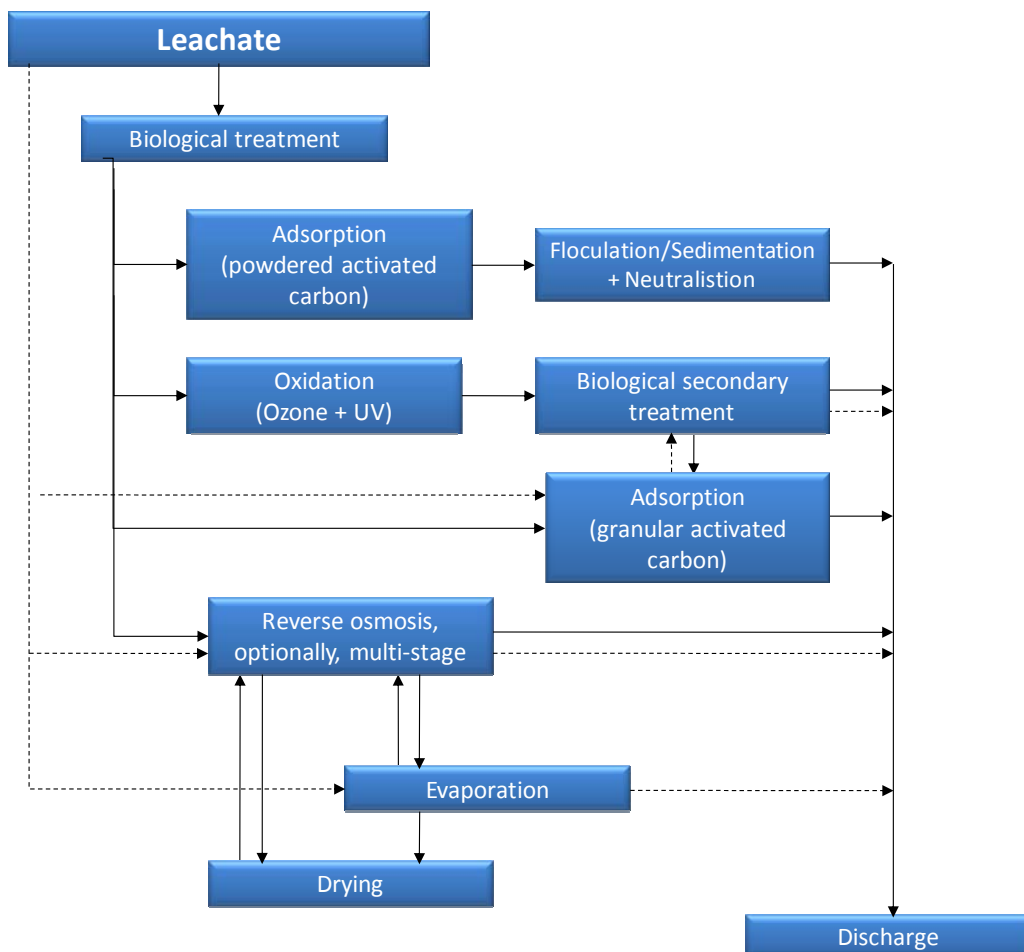Figure 10-11 presents possible technological schemes of leachate treatment.

Figure 10-11. Possible methods and their combinations for the treatment of leachates (Ehrig, 2002)

Recommended technological schemes of leachate treatment before discharge to surface water include

- **scheme I:** recirculation of leachate into the landfill body + biological treatment with nitrifying /denitrifying + adsorption on activated carbon + chemical precipitation + treatment of sludges,

- **scheme II:** recirculation of leachate into the landfill body + two stage reverse osmosis,

- **scheme III**: biological treatment with nitrifying /denitrifying + one or two stage reverse osmosis.

The discharge of raw untreated leachate to municipal sewer systems is prohibited in some countries while in other countries there is a special procedure for getting a permit.

Too high concentrations of ammonia, chlorides and sulfates as well as too high values of COD, BOD and AOX may be the main problems of accepting untreated leachate at municipal wastewater treatment plants.

Table 10-7. Comparison of leachate parameters with permissible values for discharge to sewer systems in Poland

Parameter	Unit	Leachate	Permissible values for discharge to the sewer
Ammonia nitrogen	g N/m^3	500-3000	100-200
Chlorides	g Cl$^-$ /m^3	500-5000	1000
Sulfates	g SO$_4$/m^3	200-1000	500
AOX	g /m^3		1,0

The removal of ammonia from leachate is possible using several combinations of typical wastewater treatment processes such as

- biological: nitrifying / denitrifying, using bioreactors SBR, membrane bio-reactors, combination of disc (rotary) biological filters and denitrifying reactors. Additional sources of biodegradable organic carbon include glucose, methanol and acetic acid,

- biological nitrifying + recirculation of leachates to the landfill body for denitrifying,

- biological nitrifying + usage of leachates for irrigation, e.g. plantation of willows (*Salix viminalis)* at the landfill site,

- physical-chemical involving: air stripping in an alkaline environment, neutralization of ammonia with sulfuric acid (production of ammonia sulfate) and re-acidification of leachates before discharge to a municipal WWTP.

There are also some attempts using natural or artificial wetland systems for the treatment of low concentrated leachates from aerobic landfills or landfills for MBP stabilized waste.

10.2.8   Landfill monitoring during operation and aftercare phases

The landfill operator is obliged to monitor landfill behavior and environmental impacts during the operation and aftercare phases of the landfill in order to identify any possible adverse environmental effects and to take appropriate corrective measures.

Annex III to the Landfill Directive provides the minimum procedures for landfill monitoring to check
- that waste has been accepted for disposal in accordance with the criteria for the category of landfill in question,
- that the processes in the landfill proceed as desired,
- that the environmental protection systems are functioning fully as intended,
- that the permit conditions for the landfill are fulfilled.

Additionally in Poland, the pre-operation monitoring phase involving sampling for reference data was established and is conducted before getting a permit for landfill operation.

The range and frequency of monitoring and sampling points have to be established in the decision approving the procedures for landfill operation given by the competent authority.

The following data should be monitored during both operation and aftercare phases (Table 10-8):
- meteorological data,
- amount and composition of surface water, groundwater, leachate and gas,
- level of groundwater in monitoring wells,
- settling behavior of the level of the landfill body,
and additionally the structure and composition of the landfill body during the operation phase.

Table 10-8. List of parameters and the frequency of their monitoring

Parameter	Operation phase	Aftercare phase
Volume of precipitation	daily	daily, added to monthly value
Temperature	daily	monthly average
Direction and force of prevailing wind	daily	not required
Evaporation	daily	daily, added to monthly value
Atmospheric humidity	daily	monthly average
Leachate volume	monthly	every six months
Leachate composition	quarterly	every six months
Volume and composition of surface water	quarterly	every six months
Potential gas emissions and atmospheric pressure (CH_4, CO_2, O_2, H_2S, H_2, etc.)	monthly	every six months
Level of groundwater	every six months	every six months
Groundwater composition	site specific frequency	site specific frequency
Structure and composition of landfill body	yearly	
Settling behavior of the level of the landfill body	yearly	yearly

The monitoring of groundwater must provide information on groundwater quality likely to be affected by the landfilling of waste. A minimum of three monitoring points are necessary – one in the groundwater inflow region and two in the outflow region. If there is more than one groundwater level potentially affected by the landfill, all levels are subject to monitoring (Figure 10-12).

Sampling must be carried out in at least three locations before the filling operations in order to establish reference values for future sampling. The parameters to be analyzed in the groundwater samples must be derived from the expected composition of the leachate and the groundwater quality in the area. In selecting the parameters for analysis, mobility in the groundwater zone should be taken into account. Indicator parameters should be selected in order to ensure early recognition of changes in water quality. For example, the recommended parameters in Poland for non-hazardous waste landfills: pH, conductivity, TOC, PAHs, Cu, Zn, Pb, Cd, Cr^{6+}, Hg.
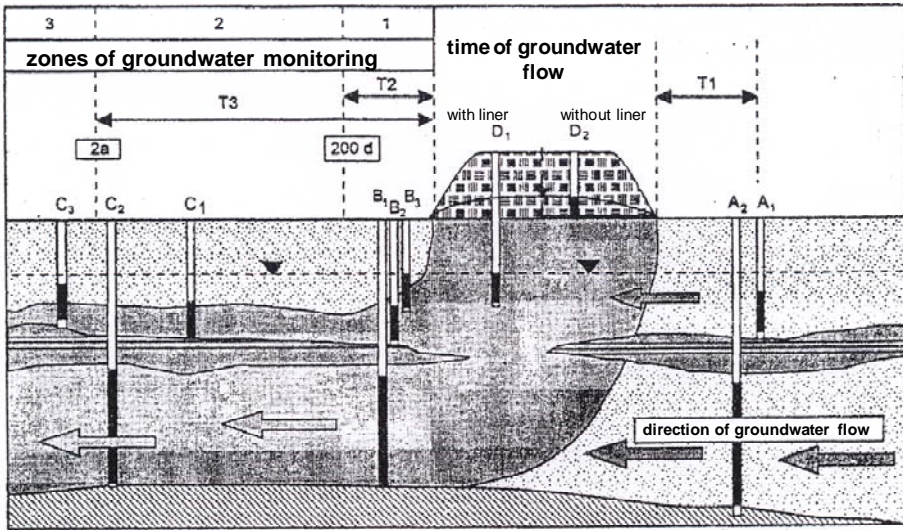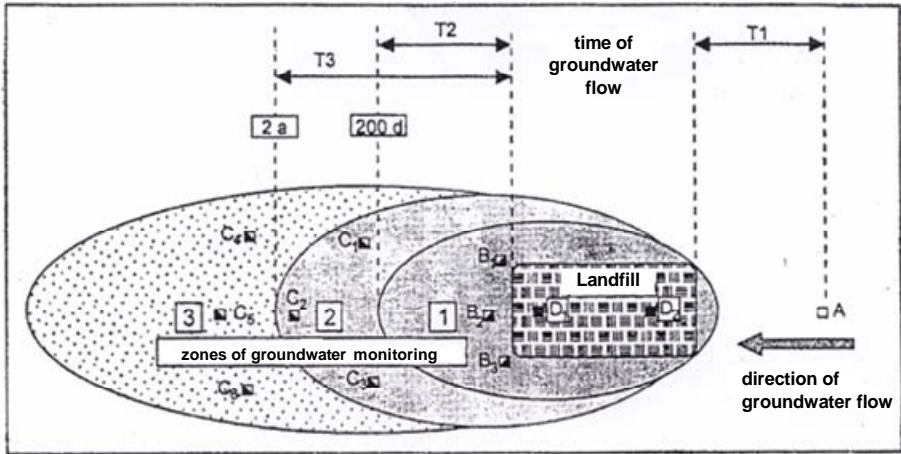
Figure 10-12. Example of groundwater monitoring well locations (adapted from Błaszyk i Górski, 1996)

The monitoring of surface water should be carried out at not less than two points upstream and downstream of the landfill. Surface water monitoring may not be required if there is no potential hazard of surface water pollution from the landfill operation.

The parameters analyzed in the samples must be derived from the expected composition of the leachate and the surface water quality in the area. They can be the same for leachate and groundwater.

Measures should be taken to minimize nuisances and hazards arising from the landfill through
- emissions of odors and dust,
- wind-blown materials
- noise and traffic,
- birds, vermin and insects,
- formation of aerosols
- fires.

10.2.9   Potential environmental impact of landfills

The potential environmental impact depends on the landfill class, types and quality of waste deposited, technical standard of the landfill, location and nearest land usage and development.
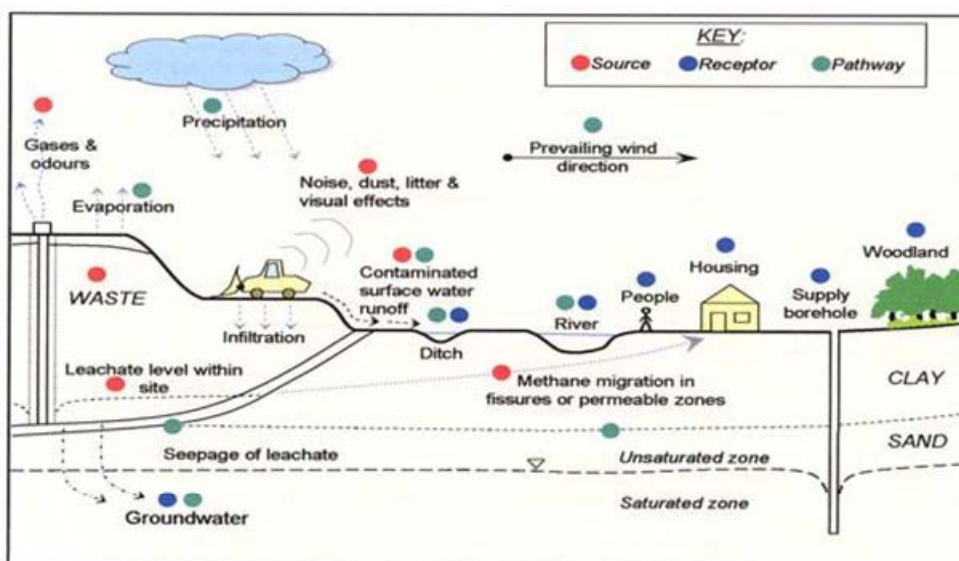
Figure 10-13. Illustration of Source-Receptor-Pathways (Wachulec 2004)

There is no direct environmental impact from landfills for inert waste, but some nuisances for the neighborhood may be caused by the emission of noise, gases and dust from vehicles delivering waste to landfills and from equipment used for landfill operation.

Emissions from non-hazardous and hazardous waste landfills may be noxious for the neighboring residential areas and for the environment depending mainly on the composition of waste (content of biodegradables, dusty fraction, volatile

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organic substances. Figure 10-14 shows the potential impacts of landfills to the environment as a function of distance from the landfill site.
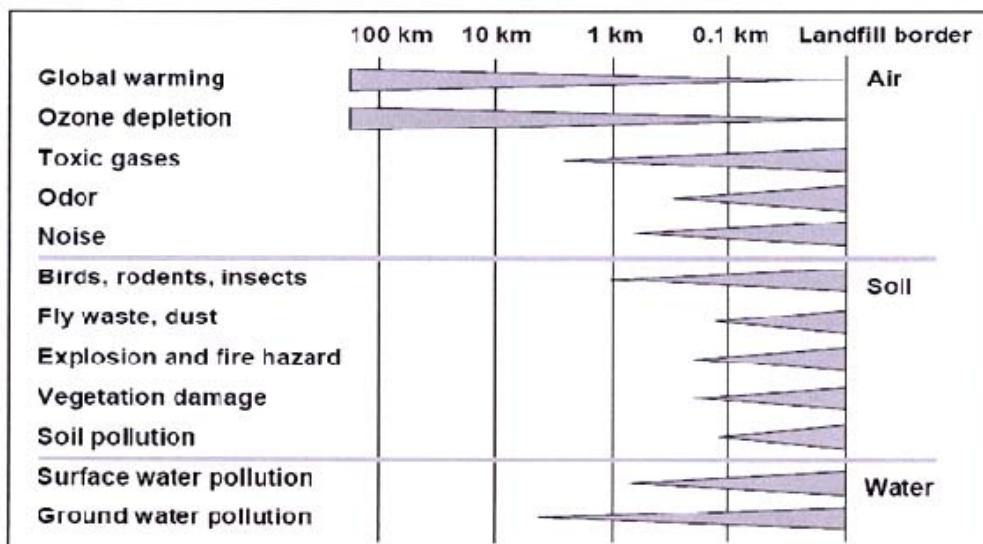
Figure 10-14. Potential environmental impacts of landfills to the air, soil and water as a function of distance from the landfill (Poulsen, 2003).

The proper location and construction of a landfill as well as its careful operation may significantly reduce or even eliminate some of the potential hazards. Most important may be the treatment of waste before landfilling by decomposition and the stabilizing of biodegradables, dewatering and solidifying sludges and semi-wet wastes, solidifying dusty wastes, etc.

11 Life cycle assessment in waste management

Life cycle assessment is a method to quantify potential environmental impacts over the whole life-cycle of a product or service.

11.1 Life cycle thinking in waste related legislation

According to the Waste Framework Directive (WFD) (2008/98/EC) the waste treatment hierarchy shall apply as a priority order in waste management. The waste hierarchy imposes the following order of actions:

(a) Prevention;
(b) preparing for re-use;
(c) recycling;
(d) other recovery, e.g. energy recovery; and
(e) disposal.

At the same time the current WFD allows more flexibility in applying the prescribed order, stating that: *"When applying the waste hierarchy (…) Member States shall take measures to encourage the options that deliver the best overall environmental outcome. This may require specific waste streams departing from the hierarchy where this is justified by **life-cycle thinking** on the overall impacts of the generation and management of such waste."*

Thus, it is directly stated that minimizing the overall impact of a given waste management system is the main objective. Life cycle thinking and related **life cycle assessment (LCA)** methodology and tools can be used to model, assess and compare the overall environmental impact of various waste treatment options.

Life cycle assessment has been already applied for the assessment of regional waste management plans in accordance with the requirements of the **Strategic Environmental Assessment (SEA)**[8]. The SEA shall evaluate the potential environmental impact of considered alternatives in a decision-making process.

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[8] In order to enhance environmental protection at an early planning stage the European Council adopted a Directive on the Assessment of the Effects of Certain Plans and Programmes on the Environment (2001/42/EC). The aim of this Directive is (2001/42/EC):
"…integration of environment considerations into the preparation and adoption of plans and programmes with a view of promoting sustainable development, by assuring that an environmental assessment is carried out of certain plans and programmes which are likely to have significant effects on the environment".
Such environmental assessment at a planning level is also known as Strategic Environmental Assessment (SEA).

According to Art. 3 of the SEA Directive, the SEA shall also be an element of waste management plans at the national, regional and/or local levels (if the latter are likely to have a significant environmental impact). Annex I of the Directive 2001/42/EC specifies required elements of the SEA. These elements include the assessment of the likely significant effects on the environment, including issues such as

"biodiversity, population, human health, fauna, flora, soil, water, air, climatic factors, material assets, cultural heritage, landscape and the interrelation between the above factors".

The guidelines for the SEA in waste management recommend the use of both qualitative assessment methods as well as quantitative environmental indicators enabling a comparison of alternatives. By now in most EU Member States SEAs have been performed. However, these were mostly for spatial planning and transport infrastructure and only in a few cases for waste management. Some examples of the latter are a SEA for the Dutch National Waste Management Plan, a SEA for the Waste Management Plan of the region Liguria in Italy (Brinkmann and Schelleman 2005), a SEA for the city of Vienna and for the province of Salzburg in Austria (Salhofer and Wassermann 2004). The Dutch SEA considered 12 environmental indicators and 40 waste streams. The SEA for the province of Salzburg considered environmental, economic and social effects both qualitatively and quantitatively (see Table 11-1).

Table 11-1. Indicators considered in the SEA for the province of Salzburg (Salhofer and Wassermann 2004)

Category	Quantitative indicators	Qualitative indicators
Environmental effects	Human toxicity, Terrestrial and aquatic ecotoxicity, Acidification, Eutrophication, Photochemical ozone creation, Global warming	Traffic flow, Hazardous incidents, Land use
Economic effects	Cost effects for waste producers	Regional added value, Synergy effects (treatment sites)
Social effects	Residues, Cost for waste producers	Appearance, Traffic flow, Regional jobs provided, Odor, Noise, Convenience, Autarky[9]

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[9] Self-sufficiency

In both the Dutch and the Austrian SEA the methodology of LCA was partly used to quantify the environmental aspects. An LCA approach allows the quantification of potential environmental impacts of products and processes and provides condensed results for various protection areas, such as human health and the natural environment. In the following sections the LCA methodology and its application to waste management planning is described in more detail.

11.2 Methodology of Life Cycle Assessment

The procedures of life cycle assessment (LCA) are part of the ISO 14000 environmental management standards: in ISO 14040:2006 and 14044:2006. (ISO 14044 replaced earlier versions of ISO 14041 to ISO 14043.) According to ISO 14044, life cycle assessment is defined as "the compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle".

The technique examines every stage of a product life cycle, from raw materials extraction, through manufacture, distribution, use, possible re-use/recycling and final disposal. Figure 11-1 shows a life cycle of a washing machine.
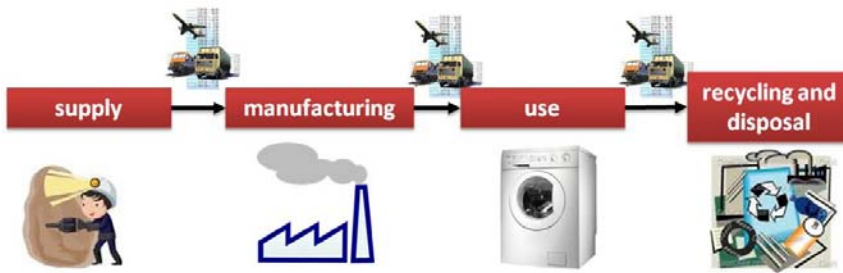
Figure 11-1. Life cycle of a washing machine

The phases of LCA, according to ISO 14044 (2006) are presented in Figure 11-2.

Goal and scope definition
A goal and scope definition is the phase in which the initial choices determine the working plan of the entire LCA. The goal of the study is formulated in terms of the exact question, target audience and intended application. The scope of the study is defined in terms of temporal, geographical and technological coverage and the level of sophistication of the study in relation to its goal (Guinée et al. 2001).
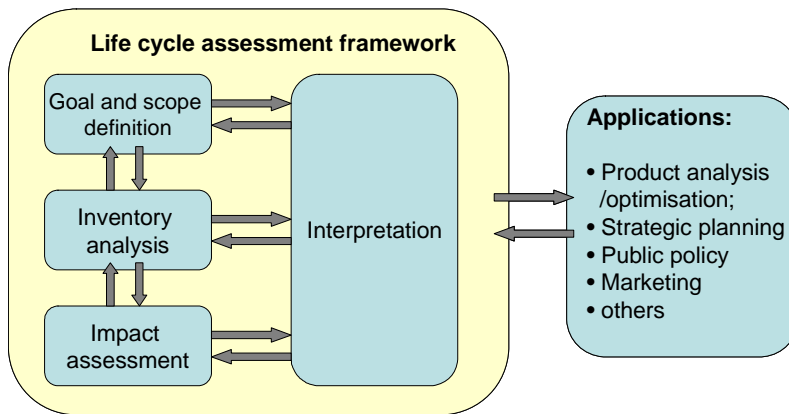
Figure 11-2. Phases of LCA according to ISO 14044

Within this stage the **functional unit** of the whole system is defined. The functional unit is a measure of the function of the studied system and it provides a reference to which the inputs and outputs can be related. This enables the comparison of two essential different systems. For example, single hand drying can be a functional unit to compare impact related to using paper towels, cloth towels and electrical hand dryers.

Another important step is the selection of the **system boundaries.** The system boundaries determine which unit processes are included in the LCA study

Life Cycle Inventory (LCI)

Within the phase of LCI for each "life stage": (i) the inputs (in terms of raw materials and energy) and (ii) outputs (in terms of emissions to air, water and solid waste) are calculated. The results of the inventory are aggregated over the entire life cycle. LCI is the most time-intensive step of an LCA study.

Life Cycle Impact Assessment (LCIA)

LCIA is the phase in which the output of the inventory analysis is further processed and interpreted in terms of potential environmental impacts. Within LCIA a list of relevant impact categories is defined and models for relating the environmental interventions to suitable category indicators for these impact categories are selected. Major environmental impact categories include:

- Global warming
- Ozone depletion
- Acidification
- Eutrophication
- Photo-Chemical Ozone Creation Potential
- Resource depletion

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- Ecotoxicity
- Human toxicity
- Odor, Noise
- Land use

The inputs and outputs (emissions of chemicals and consumption of resources) are then converted to their potential effects on the environment, i.e. potential environmental impacts. Figure 11-3 summarizes the overall framework of LCIA. It shows the relation between life cycle inventory results, impact categories, category indicators and category end point(s). Category end point is an attribute or aspect of the natural environment, human health or resources, identifying an environmental issue of concern, e.g. forest, coral reefs or crops (ISO 14044). The impact category "acidification" is used to illustrate this concept.

Results interpretation

A subsequent step of an LCA study is normalization. Normalization is defined as "calculation of the magnitude of indicator results relative to reference information". The reference information may relate to a given community (e.g. a country, Europe or the world), person (e.g. a Danish citizen) or the system, over a future target situation. The main aim of normalizing the category indicator results is to better understand the relative importance and magnitude of these results for each system under study (Guinée et al. 2001). Furthermore, impact categories can be aggregated into global, regional, and local scales. The results can be weighted according to their relative importance for the results interpreter. Weighting is necessarily based on value choices (monetary values, standards, expert panel). Finally, the results are interpreted. Life Cycle Interpretation is a phase in which the results of the analysis and all choices and assumptions made during the course of the analysis are evaluated in terms of soundness and robustness, and overall conclusions are drawn (Guinée et al. 2001).

Applications of LCA include optimization of products, processes, technologies, distribution methods (transport), waste management technologies as well as comparison of products. LCA was applied to assess

- Products: One-way vs. returnable packaging, glass vs. plastics, powder vs. liquid detergents, different isolation materials, etc.
- Services: different transport systems (truck, train, ship, etc.) waste management systems (collection, sorting, recovery, disposal)
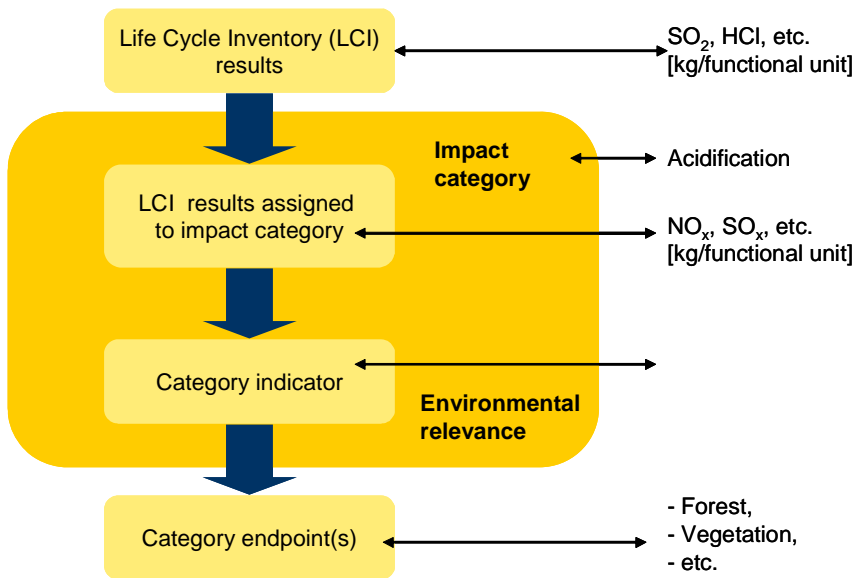
Figure 11-3. The conceptual framework for defining category indicators (ISO 14044)

11.3 Life Cycle Assessment in waste management

Applying LCA for the evaluation of the environmental performance of a waste management system (WMS) allows, in contrast to traditional environmental assessment methods, the borders of assessment to be extended to account for all relevant aspects resulting from a given WMS. In Figure 11-4 boundaries of a typical LCA study for a waste management system are presented. Thus, LCA allows the assessment of integrated waste strategies in a holistic way. LCA for municipal waste starts at the moment when solid waste is put to the waste bin and ends when the (mechanically, biologically or thermally pretreated) waste is disposed of in the landfill. LCA assesses the use of resources and the release of emissions to the air, water and land, and the generation of useful products from waste. The functional unit in LCA for a waste treatment system is most commonly one tone of waste treated by the system or waste treated by the system within the whole year. All impacts are then related to this functional unit and different system configurations can be modeled and assessed.

LCA has become an established tool to compare alternative waste management options in some European countries, e.g. in Germany, already for over 10 years LCA assessments have been used to inform policy and support decision making process in revealing solutions with the least environmental impacts (Giegrich et al. 1999).

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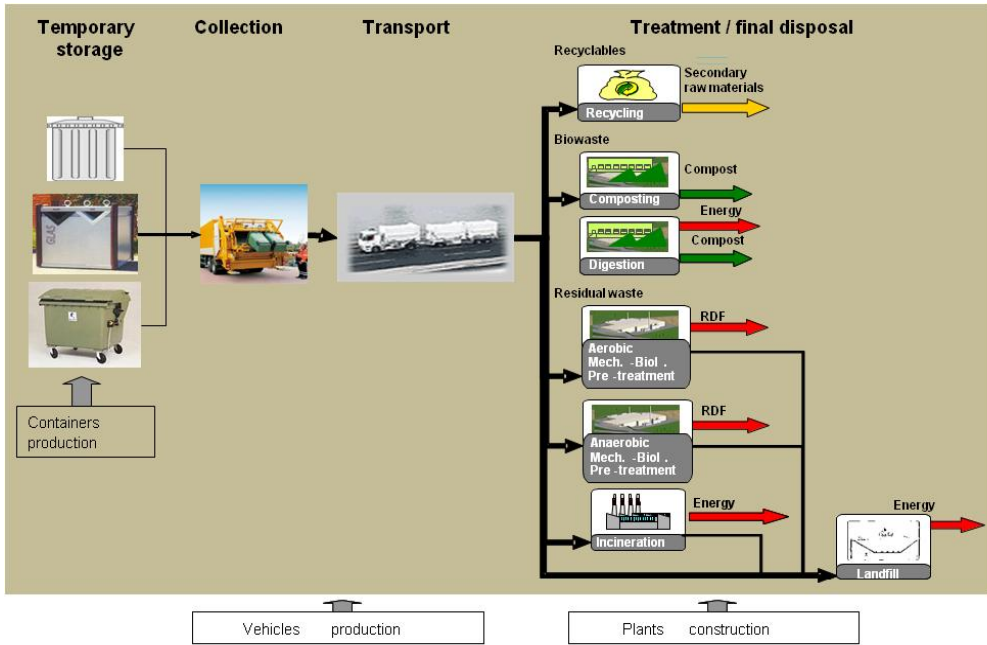
Figure 11-4. Boundaries of an LCA for a waste management system (den Boer et al. 2005)

Performing LCA studies for waste management is a complex task and requires careful selection of the scope of the study as well as the assessment method in order to obtain objective and reliable results. Reasons for variability of LCA results may be numerous: different inventory data, different borders of assessment, different technologies assessed, and different allocation and substitution procedures. Thus, before drawing any conclusion from an LCA study all the assumptions and related uncertainties have to be carefully evaluated in a sensitivity analysis (Koller 2001).

Existing LCA models for waste management and their application

In the past years a number of LCA software models have been developed. They allow the modeling of waste treatment systems and their assessment in a systematic way. Some of them are general LCA tools allowing modeling of various processes, i.e. products and processes LCA, these include

- GaBi developed by PE Consulting Group in cooperation with Stuttgart University and
- SimaPro provided by PRé Consultants from the Netherlands and

- Umberto developed by the Institute for Energy and Environmental Research (Institut für Energie- und Umweltforschung Heidelberg GmbH - ifeu) and the Institute of Environmental Computing (Institut für Umweltinformatik Hamburg GmbH - ifu) in Germany,

Some other tools were specifically developed for the modeling of waste management processes. Some well known examples of these are

- Integrated Waste Management (IWM2) of Procter & Gamble,
- ORWARE developed initially for modeling organic waste treatment by the Swedish Royal Institute of Technology in Stockholm,
- WISARD provided by the UK Environment Agency
- LCA-IWM developed within a fifth EU Framework Program in the Institute WAR, Darmstadt University of Technology.

Many of the tools apart from environmental assessment offer a possibility for the economic evaluation of a WMS (GaBi, Umberto, ORWARE and LCA-IWM). Most of the above tools have ready-made modules of standard waste management processes, such as waste collection and transport, waste incineration, landfilling and recycling, but only some of them incorporate newer technologies, e.g. mechanical-biological pretreatment of waste (e.g. Wisard, LCA-IWM).

11.4 Example of LCA application to waste management planning

Below the results of applying the LCA-IWM tool to waste management planning are shown (based on den Boer at al. 2005). These calculations were performed for the year 2015, assuming a population of 637,5 thousand inhabitants and municipal waste generation of nearly 500 kg/inhabitant per year. Waste composition was an input variable to the model. Four various scenarios were modeled and assessed for the city Wrocław, in Poland, with the first scenario being a baseline for assessment (current status).

Scenario 1: Separate collection and sorting of glass, plastics & metals and paper & cardboard for recycling, transport of residuals via transfer station to a landfill (Figure 11-5). In fact, although this assessment was performed in the year 2005, the current status of waste management in Wrocław in the year 2010 has only slightly changed. A composting plant for "public" green waste has been built, treating 6000 tons of waste annually. For residual waste a sorting plant has been build (capacity 210 thousand tones/year) which allows the separation of approximately 11% of material waste for recycling. The main disposal option remains landfilling.
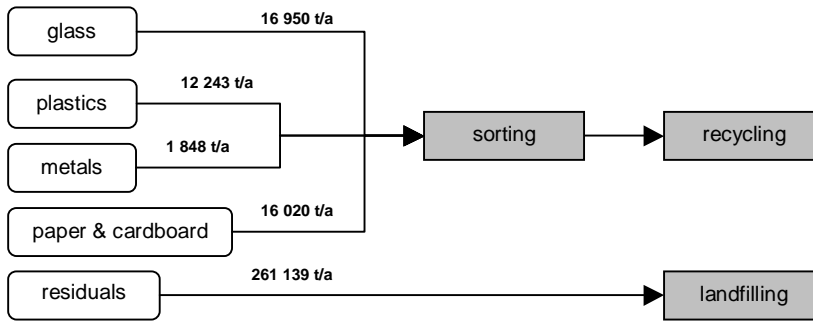
Figure 11-5. Current waste management in Wroclaw - Scenario 1 (den Boer et al. 2005)

Description of chosen future waste management scenarios for Wroclaw.

Scenario 2: Separate collection and sorting of glass, mixed dry recyclables (MDR) including plastics & metals, paper & cardboard, waste electric and electronic equipment (WEEE) and bulky waste for recycling, hazardous waste for treatment, green waste (from public green areas) for composting in windrows, aerobic mechanical-biological pre-treatment of residual waste before landfilling (Figure 11-6).
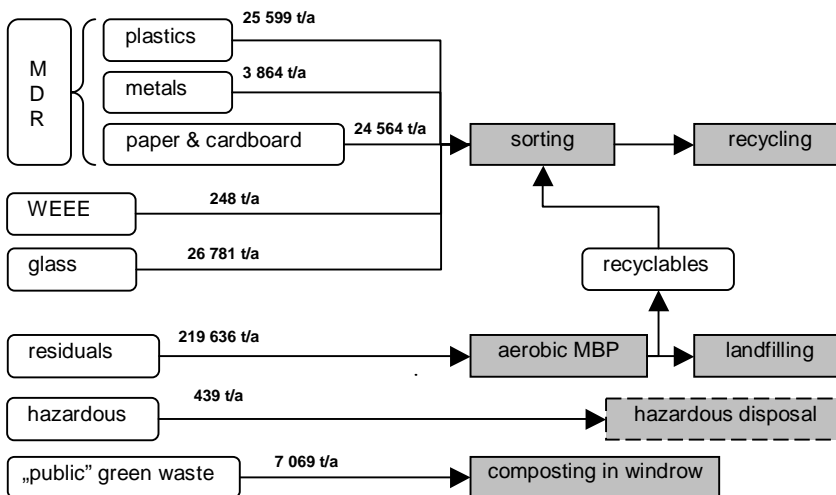
Figure 11-6. Wroclaw – Scenario 2 (den Boer et al. 2005)

Scenario 3: Separate collection and sorting of glass, plastics & metals, paper & cardboard, bulky waste and WEEE for recycling, hazardous waste for treatment, biowaste (kitchen and garden) and 'public' green waste for composting in a facility, aerobic mechanical-biological pre-treatment of residual waste before

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landfilling with separation of the high caloric fraction and usage as RDF in the cement industry (Figure 11-7).
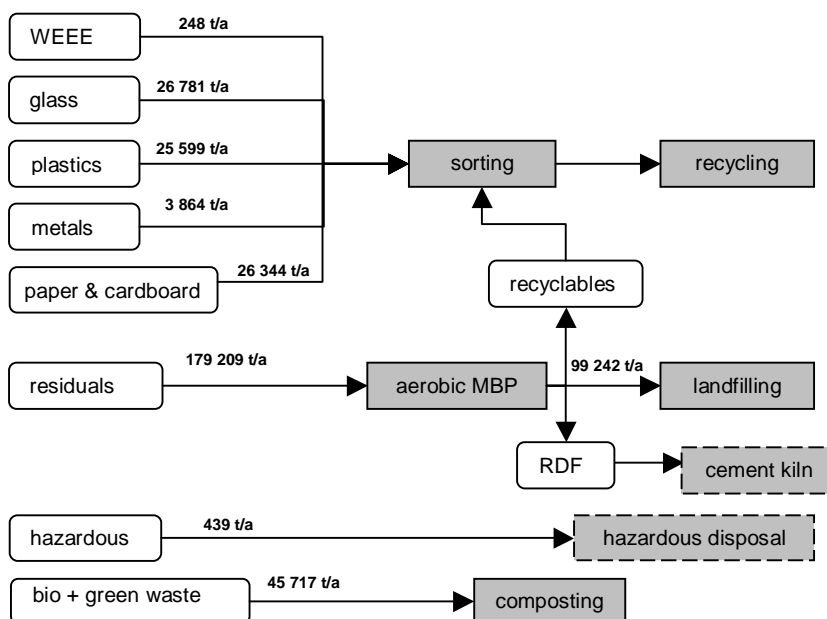
Figure 11-7. Wroclaw – Scenario 3 (den Boer et al. 2005)

Scenario 4: Separate collection and sorting of glass, MDR (plastics & metals, paper & cardboard), WEEE and bulky waste for recycling, hazardous waste for treatment, biowaste (kitchen and garden) and 'public' green waste for composting in facility, transfer station for residual waste, incineration of residual waste, landfilling of incineration residues.

LCA results for the proposed scenarios - results of modeling with LCA-IWM tool

In Figures 11-9 to 11-12 results of LCA for the four different scenarios are presented. In this tool six LCA impact categories have been selected as relevant for waste management assessment: abiotic depletion (use of resources), global warming, human toxicity, photo oxidation, acidification and eutrophication. The results are presented separately for each sub-system of the waste management system: temporary storage, collection and transport and treatment plant/disposal. The results have been normalized and expressed in Inhabitant Equivalents (IE).
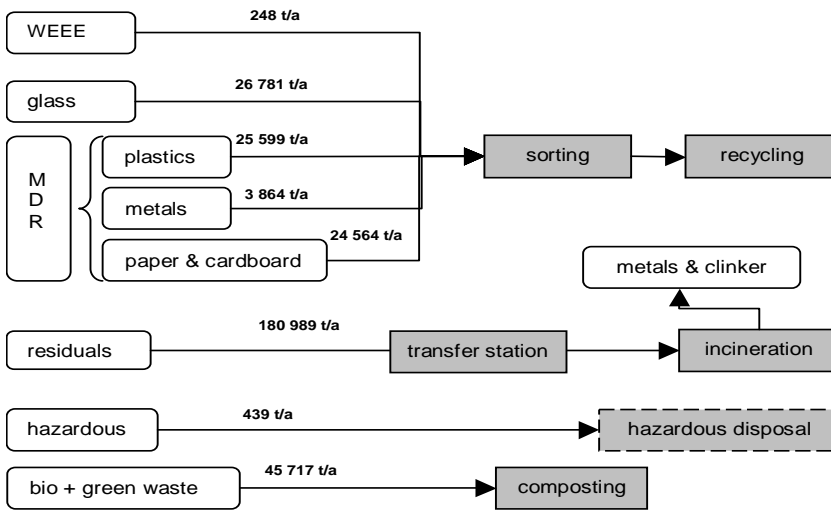
Figure 11-8. Wroclaw - Scenario 4 (den Boer et al. 2005)

For each category one IE represents the all the impacts generated in Europe (from industry, transport and human settlings) divided by total number of inhabitants in Europe. It can be seen that in all scenarios there are negative impacts. The negative impacts result from the so called substitutions of impacts of the conventional production processes of the recycled material and recovered energy.
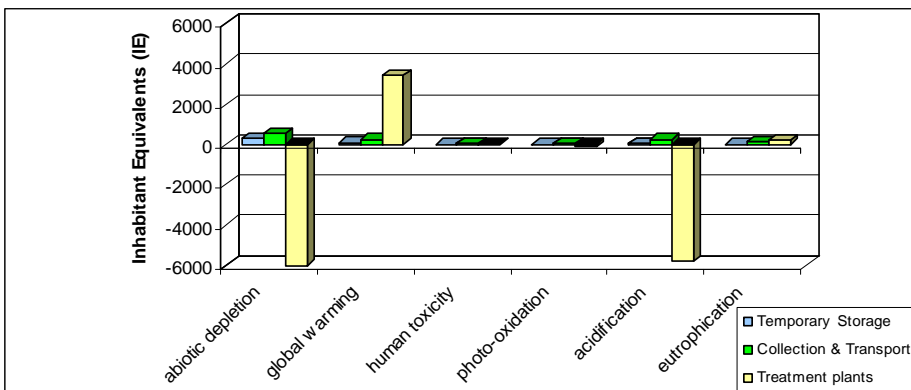

Figure 11-9. Environmental impacts of subsystems - Wroclaw Scenario 1 (den Boer et al. 2005)

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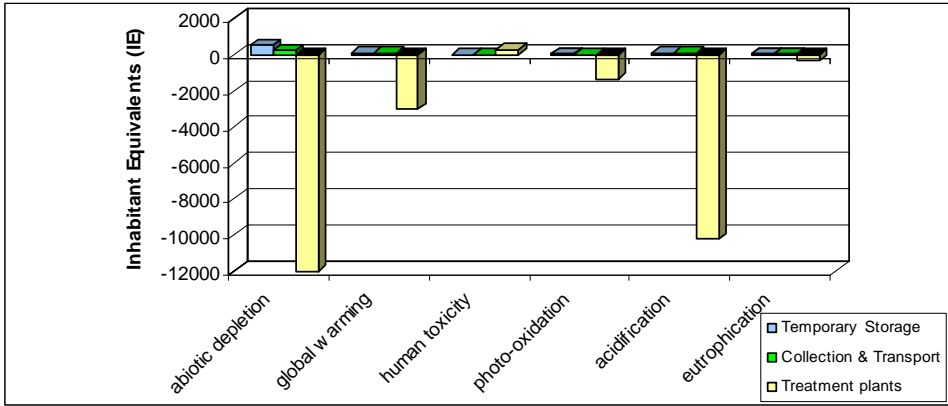
Figure 11-10. Environmental impacts of subsystems - Wroclaw Scenario 2 (den Boer et al. 2005)
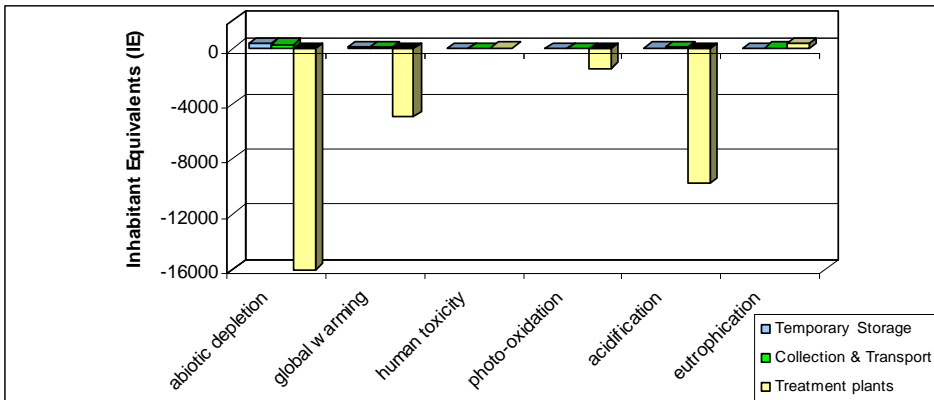
Figure 11-11. Environmental impacts of subsystems - Wroclaw Scenario 3 (den Boer et al. 2005)

For example if in the waste management system 10.000 tons of plastics are recovered for recycling, in the LCA the environmental impacts (emissions and resources consumption) related to the production of 10.000 tones of plastics will be deducted from the overall environment impact of the system. In the LCA of waste management we get waste "for free", which means we do not account for the emissions occurring at the production of all the waste materials, but start the assessment where waste is in the bin. Therefore, any recycling or recovery activity results in a high negative impact (environmental credit).
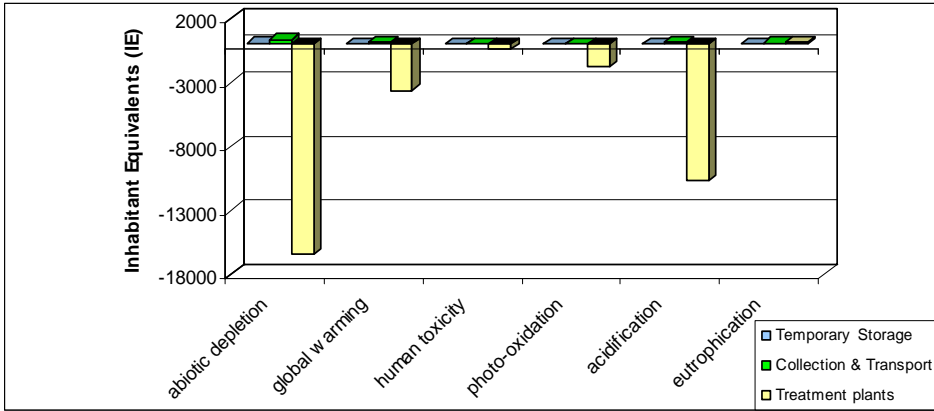
Figure 11-12. Environmental impacts of subsystems - Wroclaw Scenario 4 (den Boer et al. 2005)

In Figure 11-13 all the LCA results are put together. It can be seen that the current scenario has the most negative impact on the environment. The highest impact in the global potential category is related to emissions of greenhouse gases from the landfill – CO_2 and methane. In the category abiotic depletion all negative impacts are offset by the environmental credits of the recycling and energy recovery activities. The environmental credit for both abiotic depletion and global warming are highest here for scenario 3, which is related to the high recycling rate of this scenario and energy recovery of RDF in cement kilns. The impact of waste management systems in the categories human toxicity, photo oxidation and eutrophication is generally low. The general scenario 3 seems to show the best environmental performance.
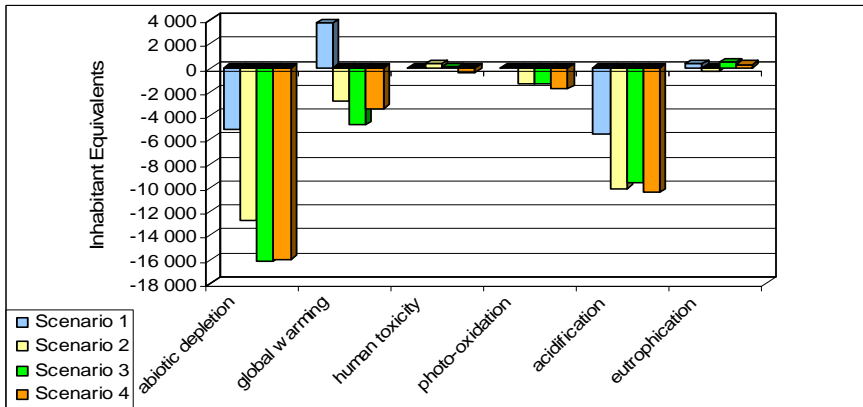

Figure 11-13. Environmental impacts of municipal waste management systems for Wroclaw (den Boer et al. 2005)

In parallel, the LCA-IMW tool performs an economic assessment of the systems according to a number of criteria. The main result of the economic assessment in terms of investment cost is presented in Figure 11-14. It can be clearly seen that a better environmental performance is unavoidably bound with a higher investment and thus also operational cost of the system. The scenario 4 with waste incineration is the most expensive option. Scenario 3 with a lower investment cost and good environmental performance seems to be the best solution, both in environmental and economic terms. However, the results of this assessment are based on the assumption that there is a market for recovered materials (recyclables and RDF).
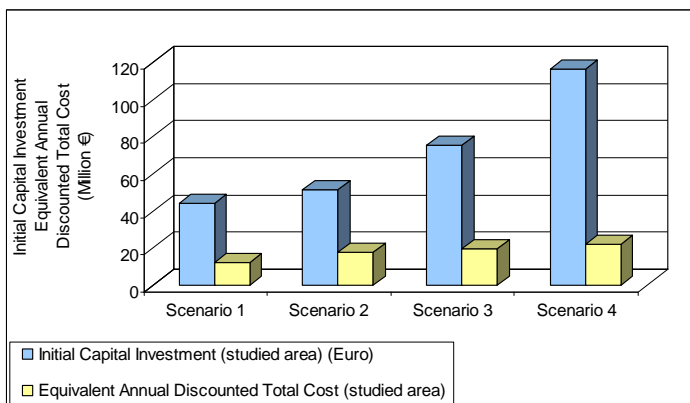

Figure 11-14. Economic impacts of waste management scenarios in Wroclaw (den Boer et al. 2005)

It should be kept in mind as well that LCA is a decision supporting tool for decision makers by providing a general trend for the potential environmental impacts of various alternatives. However, the final decision might depend on many other aspects that are out of the scope of an LCA study.

12 Industrial symbiosis

Industrial Ecology (IE) is the study of material and energy flows through industrial systems. Industrial ecology seeks to quantify the material flows and document the industrial processes that make modern society function. The primary goal of industrial ecology is to promote sustainable development at the global, regional, and local levels (Garner and Keoleian 1995).

Industrial ecology should promote the sustainable use of renewable resources and reduce the use of mineral non-renewable resources. Industrial activity is dependent on a steady supply of resources and thus should operate as efficiently as possible. Currently, due to growing consumption, growth of the world's population and replacing durable products by cheap short-living alternatives, the consumption of natural resources is steadily increasing. A number of elements, especially metals used in high-tech electronic products are threatened by depletion. According to scientific reports, without more recycling, antimony, which is used to make flame retardant materials, will run out in 15 years, silver in 10 and indium in under five. In a more sophisticated analysis, taking into account the effects of new technologies, projections were made as to how many years are left for some key metals. The author estimates that zinc could be used up by 2037, both indium and hafnium - which are increasingly important in computer chips - could be completely depleted by 2017, and terbium - used to make the green phosphors in fluorescent light bulbs - could run out before 2012 (Cohen 2007).

In the past, alternatives were found to replace the depleting raw materials; however, in the long term it will not be possible to find substitutes. Therefore, it is a major challenge of modern industry to increase reuse, recycling and recovery of resources in order to become sustainable in the long term (Garner and Keoleian 1995).

Industrial symbiosis is an approach developed within industrial ecology which can significantly contribute to global resource savings. According to Chertow (2004):

"industrial symbiosis engages traditionally separate industries in a collective approach to competitive advantage involving physical exchanges of materials, energy, water and/or by-products. The keys to industrial symbiosis are collaboration and the synergistic possibilities offered by geographic proximity".

12.1 By-products instead of waste

The concept of by-products has been introduced by the updated European Waste Framework Directive (WFD) (Directive 2008/98/EC), as follows:

"A substance or object, resulting from a production process, the primary aim of which is not the production of that item, may be regarded as not being waste (…) but as being a by-product only if the following conditions are met:
a. further use of the substance or object is certain;
b. the substance or object can be used directly without any further processing other than normal industrial practice;
c. the substance or object is produced as an integral part of a production process; and
d. further use is lawful, i.e. the substance or object fulfills all relevant product, environmental and health protection requirements for the specific use and will not lead to overall adverse environmental or human health impacts."

Moreover, the directive also defines conditions which may render end-of-waste status to waste substances which have undergone a recovery operation, including recycling, when it complies with specific criteria.

These new regulations enable the implementation of the industrial symbioses concept in a much broader context than ever before. Until now production residues were classified directly as waste and therefore were subject to strictly regulated handling options which hindered their exchange between industrial entities.

Figure 12-1 schematically presents how by-products can be circulated in the economy in accordance with the concept of industrial symbiosis.
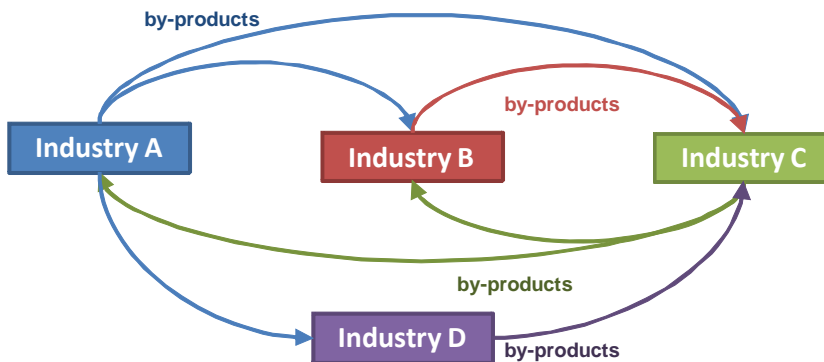

Figure 12-1. The concept of industrial symbiosis

Figure 12-2 and **Figure 12-3** present the desired transformation of industrial activities towards the implementation of the industrial symbiosis concept. **Figure 12-2** presents the average current production model. On the input side, the majority of feedstock materials and energy needed for industrial processes are primary natural materials and only a small proportion originates from recycled secondary materials. On the output side, the following streams can be distinguished:

- emissions from processing of materials and energy supply
- products that after the use phase often are disposed (partly recovered/recycled)
- production of waste that is disposed often through landfilling and partly recovered/recycled.

Thus, despite much environmental progress that has been achieved in the past years, many resources are still wasted.

Figure 12-3 presents the improvement potential from the current situation when the concept of industrial symbiosis is implemented. On the input side many of the feedstock materials, operating materials and energy, can be derived from other industries, partly directly (if they have the required quality) and partly after conditioning at a recycling plant. On the output side the following improvements could be realized:

- reduced emissions to the environment due to less energy use and less pollution from by-products processing compared to raw resources extraction,
- products can be partly reused (especially if supporting actions are implemented such as: design for reuse, design for reparability)
- landfilling of production waste is reduced; instead by-products are used as input materials for other industries either directly (if the quality is sufficient) or after necessary re-preprocessing in a recycling plant.

Thus, the introduction of industrial symbiosis concepts, especially supported by other measures, targeted at waste reduction can bring a substantial improvement in environmental performance through a reduction in waste disposal and the minimization of raw materials extraction.

However, industrial symbiosis can only be successfully implemented if it is viable economically. Ideally, all participants should benefit from such symbiotic relations – the company which delivers by-products does not have to bear the costs of waste treatment and disposal and the company which utilizes by-products has lower production costs due to cheaper raw materials.
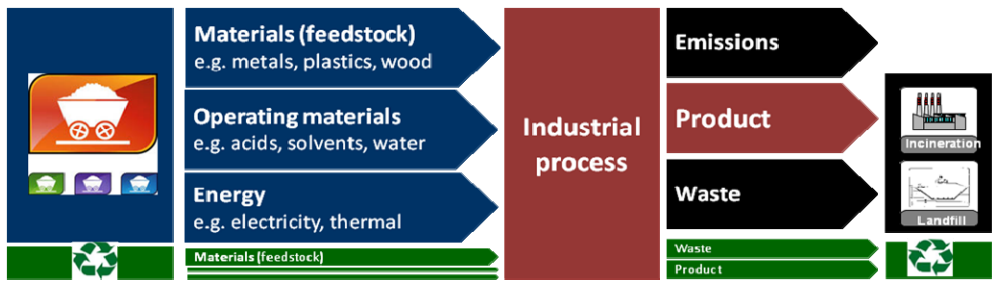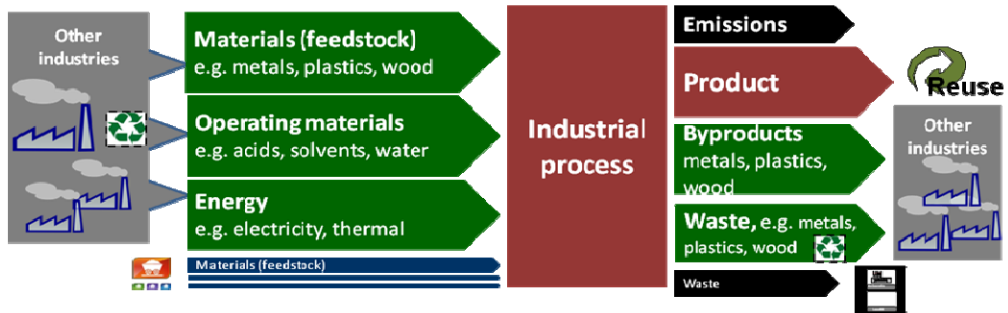
Figure 12-2. Current production model

Figure 12-3. Production model based on the industrial symbiosis concept

12.2 Existing industrial symbioses

Kalundborg, Denmark

The Industrial Symbiosis of Kalundborg is built as a network co-operation between the seven companies and the Municipality of Kalundborg technical departments. The philosophy behind the Symbiosis is that the seven companies: DONG Energy Asnæs Power Station, the plasterboard factory Gyproc A/S, the pharmaceutical plant Novo Nordisk A/S, the enzyme producer Novozymes A/S, the oil refinery Statoil A/S, the soil remediation company RGS 90 A/S as well as the waste company Kara/Noveren I/S and the Kalundborg Municipality exploit each other's residual or by-products on a commercial basis.

The general principle is that **"One company's by-product becomes an important resource to one or several of the other companies".** The outcome is reduced consumption of resources and a significant reduction in environmental strain. The success of the Kalundborg symbiosis lies in its economic profitability over a long time.

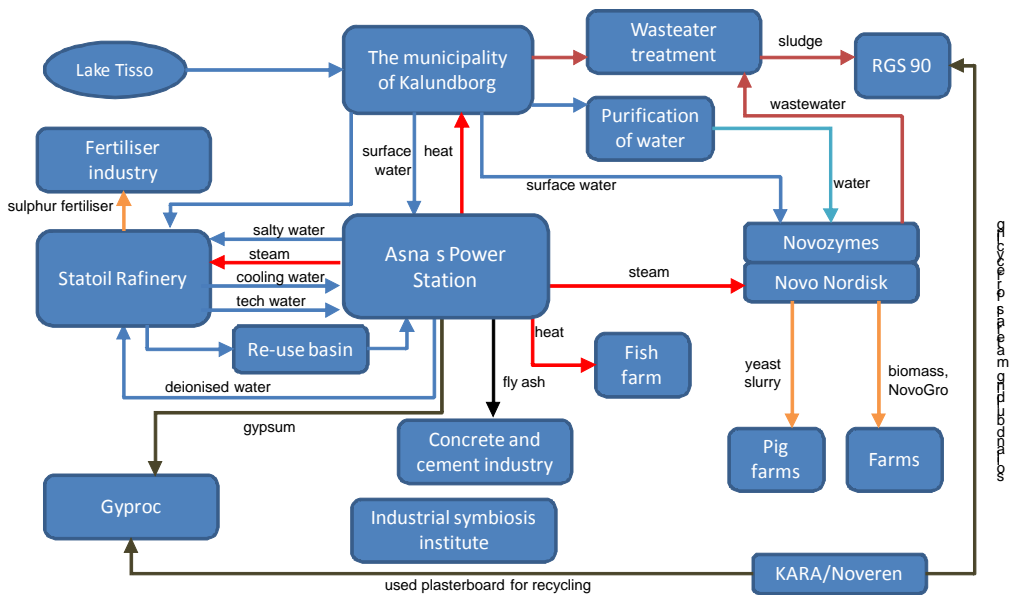Figure 12-4 depicts symbiotic relations between the companies participating in the Kalundborg industrial symbiosis.

Figure 12-4. Overview of the symbiotic relations between companies in Kalundborg (Industrial Symbiosis Institute 2008)

Material and Energy Exchanges at Kalundborg have evolved over a long time. The industrial symbiosis is regarded to have started when Gyproc located its facility in Kalundborg to take advantage of the fuel gas available from Statoil. The energy and material exchanges include the following (based on the report of Industrial Symbiosis Institute, 2008):

Energy related symbioses:

- Since 1981 the Asnæs power plant supplies households with steam and the surrounding companies with process steam. In 1992, the power plant began substituting fuels, using surplus refinery gas in place of some coal. This only became possible after Statoil built a sulfur recovery unit to comply with regulations on sulfur emissions; the gas was then clean enough to permit its use at the power plant. Currently, Statoil Refinery, Novo Nordisk and Novozymes receive approximately 1,5 mio. GJ/year as process steam preventing about 240.000 tons/year in CO_2 emissions.

- A 2nd generation bio-ethanol plant is being built next to the power plant. This new facility will utilize straw- by-product from agriculture.

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- The waste company Kara/Noveren operates an incineration plant in which nearly 1,5 mio. tons of household waste is combusted, generating electricity and providing central heating to the surrounding houses.

Water related symbioses:

- Purified wastewater from the treatment plant is used in the power station flue gas treatment. Water is reused 3 to 4 times between the Statoil Refinery and the power plant (reuse of cooling water as process water, delivery of deionized water and steam). Wastewater is sterilized using excess heat from the Asnæs Power Plant in order to produce clean process water for industrial processes.
  The recycling and reuse of water between the enterprises saves 3 mio. m3 of groundwater and surface water from Lake Tissø

Exchange of by-products:

- **Industrial gypsum** is a by-product of the Asnæs power plant produced in the desulfurization of flue gas by adding calcium and recycled treated waste water. Industrial gypsum is utilized by the plasterboard manufacturer Gyproc replacing imported natural gypsum.

- **Used plasterboards** are collected at Kara/Noverens collection sites and diverted for reuse to Gyproc. In total the secondary gypsum is equivalent to over 15 million m^2 of plasterboards.

- **Yeast slurry** is a by-product of insulin production at Novo Nordisk. Yeast slurry is used as a feed for pigs, replacing approximately 70% of the traditional soy proteins in traditional feed mixes. In Novo Nordisk sugar, water and lactic acid bacteria are added to the yeast making it more attractive to the pigs.

- **Access activated sludge** from the industrial waste water treatment plant at Novozymes is hygienized to the fertilizer NovoGro® (app. 150.000 tons/year) which is delivered to more than 600 farmers replacing up to 60% of the fertilizer needs depending on the crops produced.

- **Sludge from the municipal waste water treatment plant** is used by RGS 90 in the bio-remediation process of soil contaminated with oil and chemicals. Approximately 250.000 tons/year of soil is treated. After treatment, the clean soil is utilized as filling material at various construction sites.

The principal message from the Kalundborg experience is that all contracts have been negotiated on a bilateral basis. Each party must view the contract as economically attractive, otherwise no matter how environmentally attractive, it will not be pursued.

Landskrona, Sweden
This industrial symbiosis project was initiated in 2002 by the International Institute for Industrial Environmental Economics (IIIEE) and involved the creation of a network between the large numbers of industrial producers from a variety of sectors located in the region. The formal version of the initiative, the Landskrona industrial symbiosis program (LISP) financed by the Swedish Business Development Agency (NUTEK), was initiated in the spring of 2003 and involved over 20 companies and three public organizations. Strict and long-standing legislation on environmental standards and renewable energy in Sweden encouraged the development of the initiative. In Figure 12-5 the existing and planned (dotted lines) relations between the participating parties are presented. Similarly like in the case of Kalundborg, industrial waste heat used for heating is one of the main flows. Also, wastewater from car glass manufacturing was substituted for quality drinking water used in the wet scrubber for volatile organic compounds removal in the printing company. Due to the improved efficiency of this scrubber the content of ethanol in the wastewater increased. This ethanol is utilized by the wastewater treatment plant for its operation and reduces the amount of purchased ethanol required.

A number of other examples of industrial symbioses exist worldwide, e.g. Recycling Network Styria in Austria, petrochemical companies in Alberta (Canada), Brownsville / Matamoros Eco-Industrial Parks, Texas (USA), Burnside Eco-Industrial Park, (Canada), Gladstone Industrial Area Network (Australia), Golden Horseshoe (Canada), Kawasaki Zero Emission Industrial Park (Japan), Kwinana Industrial Area (Australia), Polish Copper Industrial Symbiosis in Głogów (Poland), Map Ta Phut Industrial Estate (Thailand), Tampico (Mexico) and others (Onita 2006). In the United Kingdom, a National Industrial Symbiosis (NISP) Program has been initiated to promote the development of symbiotic cooperation.

The major material exchanges in the existing industrial symbioses are concerned with
- gypsum recovery from the desulphurization stage of air treatment in power plants,
- use of ashes and slags from thermal processes in the construction industry,
- steam and heat recovery from various thermal processes,
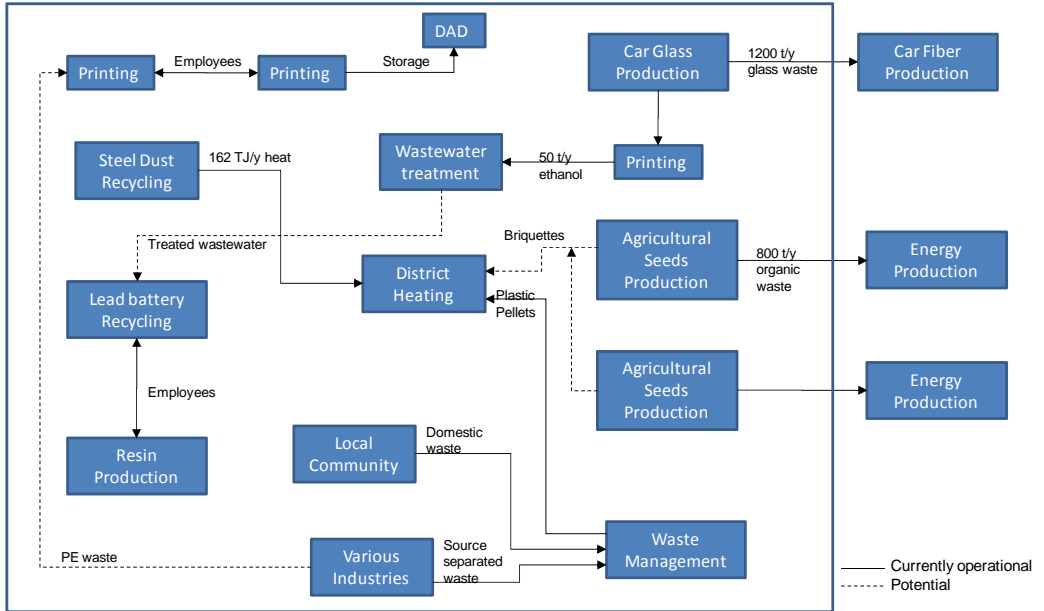- energy recovery from organic, waste in the form of waste derived fuels,

Figure 12-5. Industrial symbiosis In Landskrona, Sweden (adapted from Mirata and Emtairah 2005)

- recovery of plastics and their reprocessing to other products,
- reuse of packaging.

The majority of companies involved in industrial symbioses are large companies able to ensure a constant (in both quantity and quality terms) supply and/or demand of/for by-products. The European research project ZeroWIN has been initiated targeted at the development of industrial symbioses between the information and technology sector, photovoltaic sector, automotive sector and the construction industry (www.zerowin.eu). The challenge for these sectors is that they often involve small and medium companies which rely on a large number of suppliers scattered all over the world for individual materials and components.

12.3 Methodology for developing industrial symbiosis

Industrial symbiosis offers an approach to achieving resource efficiency by utilizing the waste material generated by one company's process as a resource for another company. By doing so, the net emissions of solid waste, liquid waste and

gaseous waste to the environment surrounding the industries involved is diminished. As a result, a company can not only reduce their costs related to waste disposal, treatment, storage, regulatory issues, and legal and liability issues, but can also achieve savings in resource costs. Thus, some industrial symbioses have evolved spontaneously, as the companies revealed a business opportunity with a savings potential. More recently, because of the growing concern about the depletion of basic resources, targeted programs have been initiated to support the development of industrial symbioses (e.g. Landskrona/Sweden, NISP/UK). The methodology followed in the development of the industrial symbiosis in Landskrona is presented in Figure 12-6.
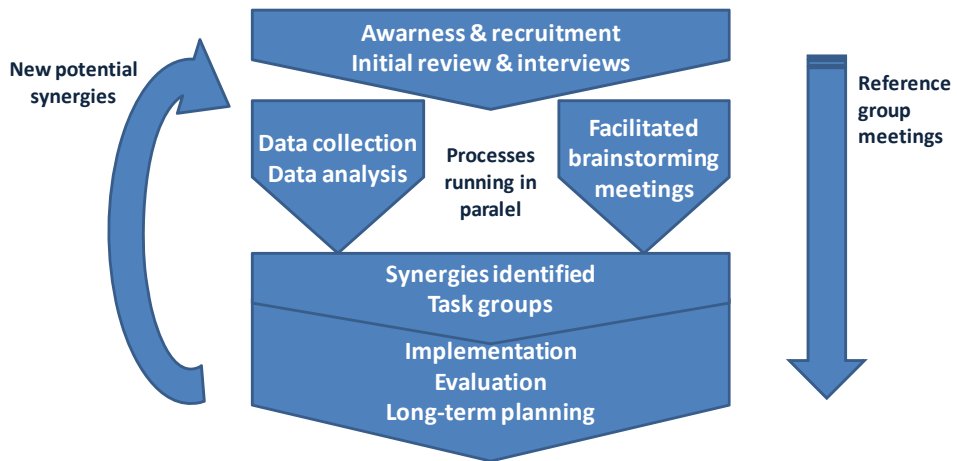

Figure 12-6. Methodology of the development of industrial symbiosis (adapted from Starlander 2003)

The development of an industrial symbiosis is a long-term process requiring extensive data collection, data analysis, facilitating contacts between various non-related industries and overcoming various burdens. Obstacles to developing industrial symbioses include
- lack of information on waste/by-products arising in other industries,
- insufficient specification of waste quantities/quality,
- globalization of the economy (extensive supply chains, limited control),
- lack of motivation (unnecessarily high quality specifications for raw materials, disqualifying secondary materials),
- confidentiality of data,
- complicated procedures due to the waste status of by-products (changed by the new Waste Framework Directive)
- need for a new business model.

13 Transboundary shipment of waste

13.1 Legal framework

There are few international regulations concerning the transboundary shipment of waste:

- Basel Convention of 22 March1989 on the Control of Transboundary Movement of Hazardous Wastes and their Disposal [Basel Convention],

- OECD Council Decision C(2001)107/Final of the OECD Council concerning the revision of Decision C(92)39/Final on the Control of Transboundary Movement of Wastes Destined for Recovery Operations [OECD Council Decision],

- Regulation (EC) No. 1013/2006 on shipments of waste [EC Waste Shipment Regulation],

- Commission Regulation (EC) No. 1418/2007 concerning the export for recovery of certain waste listed in Annex III or IIIA to Regulation (EC) No. 1013/2006 of the European Parliament and of the Council to certain countries to which the OECD Decision on the control of transboundary movements of wastes does not apply [Commission Regulation (EC) No. 1418/2007],

The Basel Convention is a global control system for the transboundary shipment of hazardous waste. It introduces an export ban on hazardous wastes from OECD member countries to OECD non-member economies. However, due to the lack of ratification by a sufficient number of Parties to the Convention, this export prohibition has not yet entered into force internationally. It has been transposed by the EU regulation on the shipment of waste.

The OECD Council Decision is a system for the control of the import and export of wastes destined for recycling in order to allow their tradability within the OECD member countries.

Two annexes (Annex VIII and Annex IX) to the Basel Convention contain lists of wastes. The hazardous wastes are listed in Annex VIII.

The wastes regulated in the OECD Decision are grouped into risk categories as Green and Amber Lists. Wastes from the Green List are not subject to any waste-law control and are regarded as goods. Amber List wastes are subject to a control procedure consisting of notification (application) and the consent of the applicable authorities. Also required are legally binding contracts from the parties involved in the shipment of wastes who have legal control of the wastes and the recycling plants.

The two waste lists of the Basel Convention have been integrated into the OECD lists in such a way that the "Green Procedure" applies to Annex IX and the "Amber Procedure" applies to the wastes listed in Annex VIII of the Basel Convention.

Regulation (EC) No. 1013/2006 on shipments of waste is the European transposition of the international regulations on shipments of waste. This Regulation implements the international obligations of the Basel Convention and the OECD Decision while also containing the internationally acceded objective that wastes shall be disposed of in an environmentally sound manner. It also prohibits the shipment of hazardous wastes in particular (Annex V) from the EU to non-OECD countries.

The regulation contains the following lists of wastes:
- Annex III with Green List of wastes,
- Annex III A for mixtures of Green Listed wastes,
- Annex IV with Amber List of wastes,
- Annex V, covering wastes subject to the export prohibition.

Aims of the regulation are
- strengthening, simplifying and specifying the procedures for control waste shipments to improve environmental protection,
- reducing the risk of waste shipments not being controlled,
- seeking to include into Community legislation the provisions of the Basel Convention as well as the revision of the Decision on the control of transboundary movements of wastes destined for recovery operations, adopted by the OECD in 2001 (Sander, 2010).

The Regulation applies to shipments of waste:
- between Member States, within the Community or with transit through third countries;
- imported into the Community from third countries;
- exported from the Community to third countries;
- in transit through the Community, on the way from and to third countries.

The Regulation concerns almost all types of waste shipped. Only radioactive waste and a few other types of waste do not fall within its application insofar as they are subject to separate control regimes. Derogations concern, for example, shipments of waste generated on board vehicles, trains, airplanes and ships, until such waste is offloaded for recovery or disposal, etc.

Figure 13-1 presents the general overview of international regulations on the shipment of waste. On the basis of the shipment of WEEE some differences in waste nomenclature among those regulations have been shown. *Hazardous WEEE according to the European List of Wastes should, for the purposes of Regulation No 1013/2006, be classified as hazardous WEEE by using the Basel entry A1180, unless another entry contained in Annex IV applies, and that hazardous WEEE cannot be classified appropriately as either GC010 or GC020.*
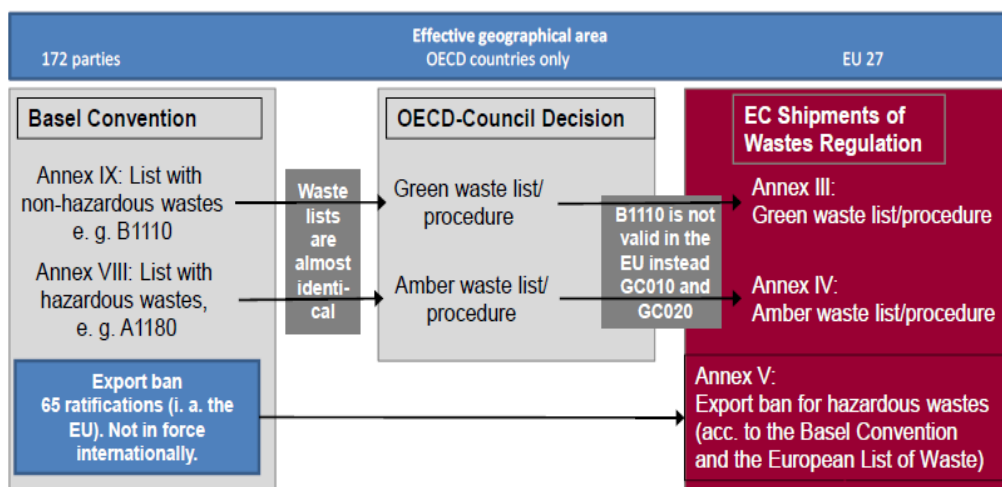

Figure 13-1. Overview of regulations on the shipment of waste (Sander, 2010)

The Regulation 1013/2006/EC reduces (in comparison to the previous regulation 259/93/EC) the number of lists of wastes whose shipment is authorized from three to two. Wastes subject to notification are set out in the "Amber List" (Annex IV) while wastes subject only to information requirements are set out in the "Green List" (Annex III). Wastes for which export is prohibited are listed separately (Annex V).

The Regulation also reduces the number of waste shipment control procedures from three to two:

- the "green listed" procedure applies to non-hazardous waste intended for recovery;

- the notification procedure applies to shipments of all waste intended for disposal and hazardous waste intended for recovery;

There are the following main rules of notification:

- whatever the procedure, all persons involved in the shipment must ensure that they take all necessary measures in order that waste is managed in an environmentally sound manner throughout the shipment process and when it is recovered or disposed of,

- the notification procedure requires that the competent authorities of the countries concerned by the shipment (country of dispatch, country of transit and country of destination) give their consent prior to any shipment,

- waste shipments must be the subject of a contract between the person responsible for shipping the waste, or having it shipped, and the consignee of such waste. Where the waste in question is subject to a notification requirement, the contract must include financial guarantees.

Under the notification procedure, the notification must be submitted by the notifier only to the competent authority of dispatch which, in turn, will be responsible for passing it on to the competent authorities of destination and transit. The competent authorities must give their consent (with or without conditions) or express their objections within 30 days. Any changes involving the main aspects of the shipment (quantity, itinerary, etc.) must be the subject of a new notification, save in cases where all the competent authorities grant the notifier an exemption from this obligation.

Furthermore, interim recovery and disposal facilities are bound by the same obligations as final recovery and disposal facilities. The authorization of a shipment involving interim operations can only be sanctioned if the shipment of the waste in question has also been authorized.

If a shipment cannot be completed (including the recovery or disposal of waste), the notifier must take the waste back, normally at his own expense.

The take-back obligation does not apply:

- if the competent authorities of dispatch, of transit or of destination concerned by the recovery or disposal of the waste consider that the notifier or, if that is impracticable, the competent authority of dispatch or a physical or legal person acting on their behalf, can recover or dispose of the waste in another way in the country of destination or elsewhere;

- if the waste has been irreversibly mixed with other types of waste before a competent authority concerned has become aware of the fact that the notified shipment cannot be completed.

Other general provisions of the Regulation:

- a ban on the mixing of waste during shipment,

- the making available to the general public of appropriate information, and

- the obligation on the part of the notifier, the competent authority, the consignee and the facilities concerned to keep documents and information.

Exports to third countries of waste intended for disposal are prohibited, except to European Free Trade Association (EFTA) countries which are party to the Basel Convention. Exports of hazardous waste intended for recovery are prohibited, except those directed to countries to which the OECD decision applies.

Imports from third countries of waste intended for disposal or recovery are prohibited, with the exception of imports:

- from countries to which the OECD Decision applies,

- third countries which are party to the Basel Convention,

- countries which have concluded a bilateral agreement with the Community or Member States, or

- other areas during situations of crisis.

Member States must make provisions for the organization of checks throughout the entire waste shipment and waste recovery/waste disposal process.

13.2 Development of shipment of waste in EU countries

EU Member States must report shipments of notified waste (such as hazardous waste) to the European Commission and to the Basel Convention Secretariat. According to data of the European Commission, the shipment of notified waste from EU countries is increasing - Figure 13-2 (Waste without borders, 2009). The notified waste comprises mainly hazardous and problematic waste.
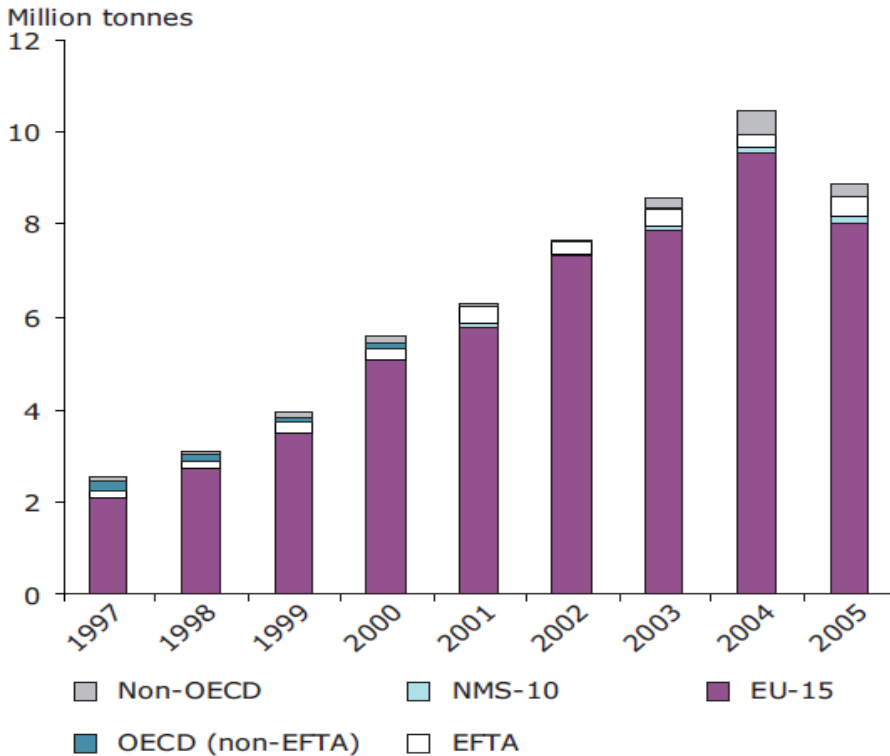
Figure 13-2. Shipments of notified waste from EU Member States to other EU and non-EU countries, 1997–2005 (EEA 1/2009)

The levels of export and import of notified waste differ significantly among the EU countries. Figure 13-3 shows the export and import per capita in the EU countries and Norway. The most significant exporters are the Netherlands, Ireland, Luxembourg, Belgium, Denmark and Lithuania, while Germany, Norway, Netherlands and Sweden import the largest amount of wastes.

There are the following main drivers of the increasing transboundary shipment of hazardous and problematic waste:

- differences in prices for treatment and disposal.

- differences in taxes for waste disposal (landfilling),

- insufficient capacities of waste treatment installations,

- lack of installations for special treatment of individual wastes,

- generally better development of treatment and disposal facilities in large countries.
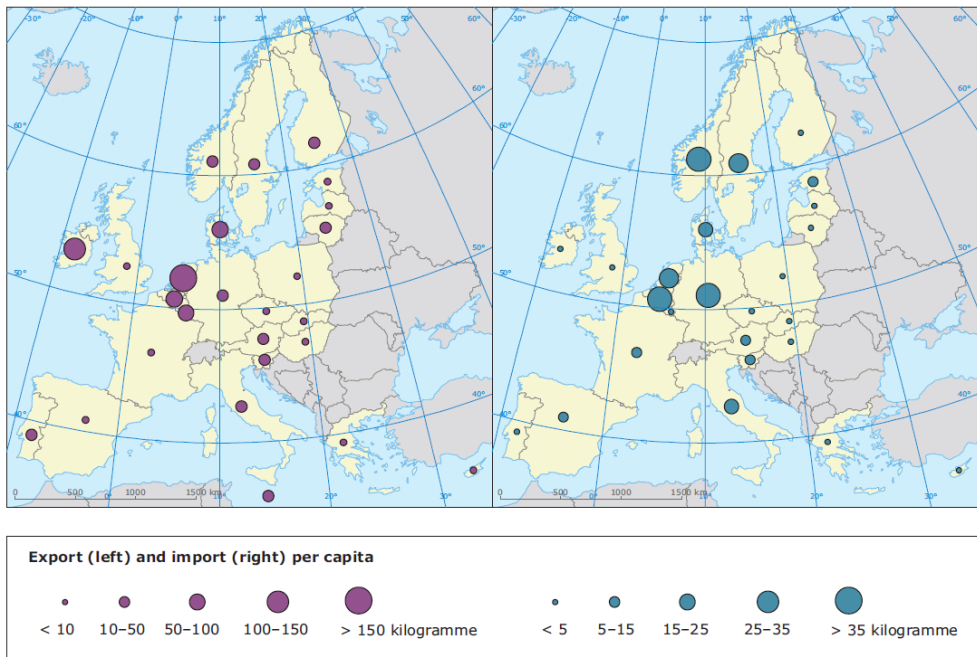
Figure 13-3. Export and import per capita of notified waste in 2005, kg/capita (EEA 1/2009)

In 2005, nearly 20 % of the shipped waste was waste for disposal (mainly incineration) and the remaining 80 % was destined for recovery operations (mainly recycling and incineration with energy recovery) – figure 13-4. The increase in exports was connected to a move towards incineration and recycling of organic and metal waste.

The political ambition of the EU to be self-sufficient in handling its landfill and other waste disposal activities has almost been achieved and only a limited amount of waste is shipped for disposal in other OECD countries.

In 2007, the amounts of waste notified for transboundary shipment were about 4,7 million tons for hazardous ones while 6,4 million tons for non-hazardous wastes (Fischer, Davidsen, 2009).

Ca. 1,08 million tons or 23% of the total amount of shipped hazardous waste were construction and demolition waste and the main part was polluted soil and asbestos. There was also a high amount of transboundary shipped waste from waste treatment facilities (1,3 million tons).
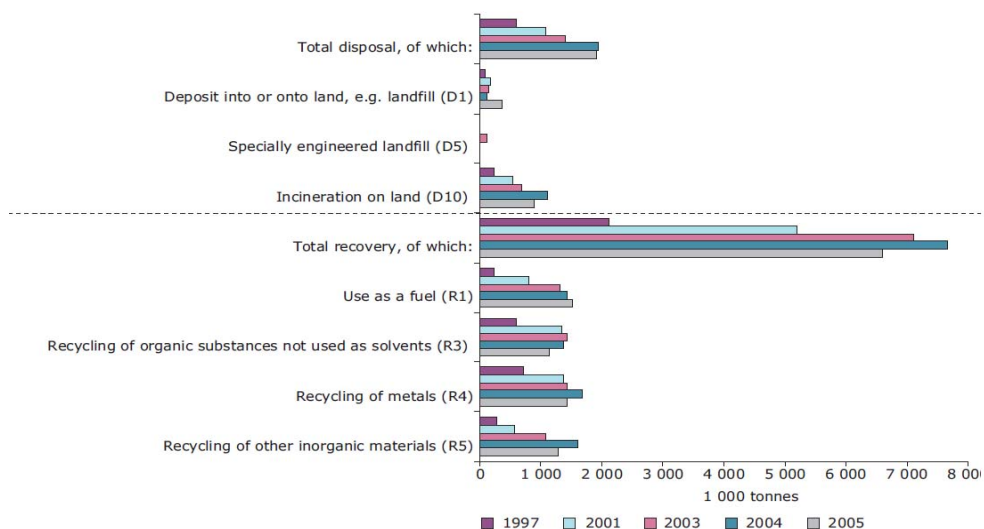
Figure 13-4 Treatment of notified waste shipped from EU Member States to other EU and non-EU countries (EEA 1/2009)

Three main waste treatment operations produce hazardous waste that is shipped:

- waste from flue gas cleaning from incineration of waste,

- waste from mechanical sorting of waste, especially contaminated wood,

- waste from physical and chemical treatment of waste.

In addition to the growing quantities of notified hazardous and problematic waste, the shipments of non-hazardous or 'green-listed', waste such as paper, plastics and metals have also increased considerably in the last ten years with a rapid increase of shipment to the Far East, particularly China. These wastes have value as resources and the development of the Asian economy was the main driver for the increase of such shipments. After 2008, due to the world's economic crisis, these tendencies became reversed.

13.3 Illegal shipments of waste

Illegal shipment is not only a simply case of transporting a certain type of hazardous waste to a country where it is prohibited, but encompasses also the unintentional breach of law caused by, for example, an administrative error or a carefully planned illegal shipment of waste. Examples of illegal shipment may include (Waste without borders, 2009):

- transporting any waste subject to the Basel Export Ban out of the EU or the OECD;
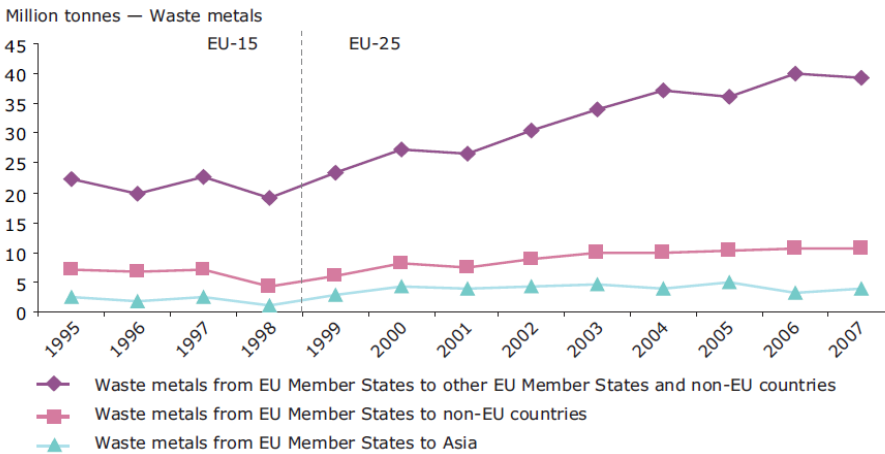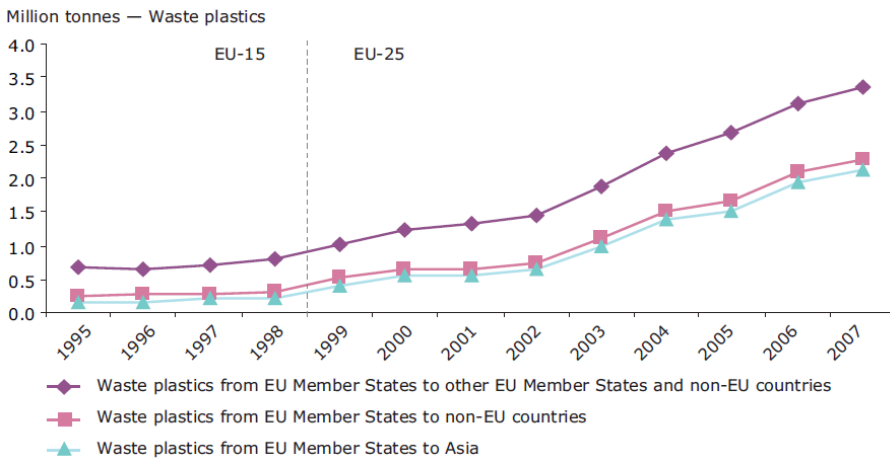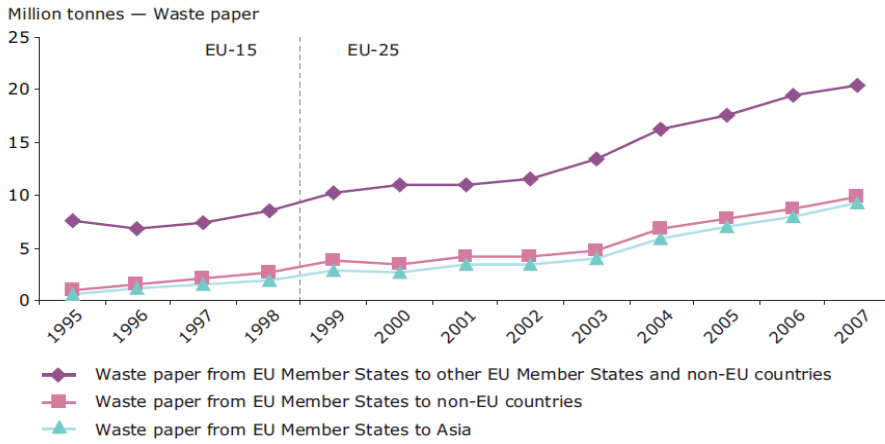
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Figure 13-5. Developments in shipments of waste paper, waste plastics and waste metals out of and within the EU from 1995 to 2007 (EEA 1/2009)

- transporting waste without notifying the authorities of source and destination when such a notification is necessary;

- falsifying any documentation regarding waste loads or not declaring waste on documentation;

- mixing certain types of waste;

- classifying hazardous waste as non-hazardous ('green-listed');

- shipping waste whilst falsely claiming that it comprises second-hand goods and is therefore not subject to waste regulations.

Reported annual illegal shipments in the years 2001-2005 varied between 6 000 and 47.000 tons with an average of about 22.000 tons, equivalent to 0,2 % of the notified waste (Figure 13-6). These are probably minimum figures, as many of the reports from the EU countries did not contain information on the amounts shipped. The number of reported illegal shipments has increased during the period 2001 to 2005. It is expected that reported cases represent only a fraction of the actual number and that the number of illegal shipments is considerable.
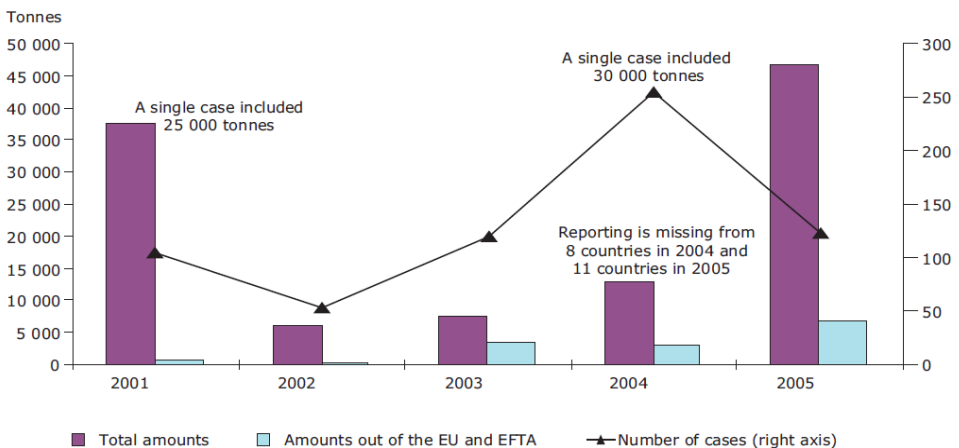

Figure 13-6. Reported illegal waste shipments in the EU from 2001 to 2005 (EEA 1/2009)

14 Economic aspects

14.1.1 Waste prognosis

In order to properly plan a waste management system, both current and future waste amounts and composition are needed. In most countries and municipalities some sort of waste statistics are available. However, these are not always reliable[10] and rarely give a complete overview of the actually collected amounts of all waste fractions. In situations where waste management is privatized, as currently still is the case in Poland, waste statistics tend to be less reliable because the data is provided by companies not having financial obligations to the governmental institutions. Moreover, collected waste is not always weighed, although nowadays most treatment and disposal sites have a weigh-bridge. Estimations based on numbers of trucks and their volumes are naturally sources of fault.

Data mostly available are the amounts collected of residual/mixed waste and of recyclables. To assess which treatment option for the residual fraction is advantageous, not only knowledge about the amounts is needed, but also the quality or physical and chemical properties of the waste are required. For initiating or optimizing separate collection the knowledge of amounts is not enough. If for instance 15 kg/inh. of paper waste is collected from a total of 300 kg/inh. of municipal waste (5%), it is hard to say how far this situation can still be optimized. The composition of the residual waste fraction should be known to determine the rate of separate collection. For this purpose sorting analyses should be undertaken.

A **traditional waste prognosis** is based on the extrapolation of historic data. In Figure 14-1 an overview of waste prognosis in five steps is given.
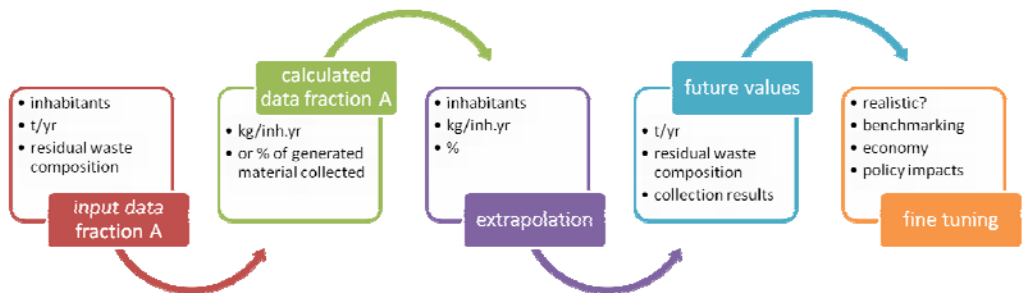

Figure 14-1. Traditional waste prognosis in five steps

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[10]) "There are three kinds of lies: lies, damned lies, and statistics", attributed to Benjamin Disraeli, British Prime Minister
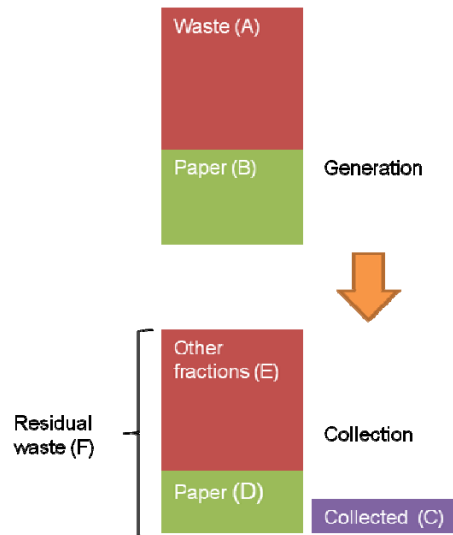
In the above overview of a traditional waste prognosis, a prognosis of actually two different indicators is undertaken: based on the collected amounts only and based on both the collected amounts and the composition of the residual waste. The latter gives a better insight about realistic maximums of the prognosis and is more closely related to the central decisive factor in waste generation: the inhabitants.

In the second step the historical data is transformed to the indicators considered in the actual prognosis: kg/inh.yr and % of generated waste that is collected separately. The third step consists of the extrapolation of both the waste related indicators and the number of inhabitants. The latter can be obtained from official prognosis from the municipality, which may also include future plans for new residential areas. In the fourth step the future values for the total amounts of waste and the separate collection rates are determined. The last step consists of a control to see whether the obtained values are realistic. Here external factors, e.g. economic development, impacts of new policies on waste generation and collection and a comparison with similar municipalities may used to adjusted values for the prognosis.

The traditional approach for waste prognosis is a black-box method. It does not evaluate the driving factors behind the changes in waste generation. Newer or alternative methods and tools do attempt to include these driving factors. The main focus lies on

- consumption data. To a large extent, waste generation is a (timely delayed) consequence of consumption. This is true for both the consumed goods themselves and their packaging material. And/or
- socio-economic parameters. Countries with a higher GDP tend to show higher waste generation figures. The factual parameters used for waste prognosis do not always show a direct connection. Beigl et al. (2008) found for example that the infant mortality rate and life expectancy are also of significant influence (Den Boer et al. 2005).

The more comprehensive prognostic methodologies require as well a critical view on the outcomes. In Poland, historic data on waste amounts and composition is often not reliable. Therefore, the indicators often provided in the national waste management plan are used on a city level. This, however, leads to predicted amounts of generated waste that may deviate significantly from reality.

14.1.2 Investment calculations

A significant factor in the overall costs of a waste treatment or disposal site and of a waste management system as a whole are the investment costs. These costs occur at the beginning of an investment, but have effects over the economic lifetime of the invested good or installation. In the following, these aspects are clarified.

Amortisation is the accounting of an investment over a certain period of time. In this way, the yearly costs of the investment, generally paid for by a loan, can be calculated. This includes both a pay-back factor of the investment sum and a factor that accounts for the costs of the use of the capital sum (e.g. the interest). There are three general methods of amortisation used:

- linear depreciation
- fixed depreciation rate
- annuities

When **linear depreciation** is applied, the yearly depreciated amount is kept constant. Investment goods are put on the balance sheet of a company/municipality. The value they represent is often decreased over the years at a constant rate (as related to the total investment sum). The depreciation rate with **linear depreciation** is calculated as follows:

$$D_t = \frac{I - R}{n} = d_I * I$$

With:
- D_t = Depreciation in period t
- R = Rest value after n years
- I = Investment
- d_I = yearly depreciation (% of investment)
- n = life time

Applying a **fixed depreciation rate** is similar to a linear depreciation. Here the depreciated rate of the current balance value of the investment is kept constant. This form of depreciation can be used to keep total costs equal in case in later years higher costs are expected (e.g. maintenance). The depreciation rate at a **fixed depreciation rate** is calculated as follows:

$$R = I(1 - d_B)^n \rightarrow d_B = 1 - \sqrt[n]{\frac{R}{I}}$$

$$\Rightarrow D_t = B_{t-1} - B_t = d_B(1 - d_B)^{t-1} * I$$

With:
- B_t = Balance sheet value in period t
- d_B = yearly depreciation (% of balance sheet value)

Using **annuities**, the sum of depreciation and interest is constant. This form is applied to determine equal yearly costs for capital investments. It is also often used for mortgages on houses. The **annuities** are calculated as follows:

$$Ann. = (I - R) * \frac{i(1 + i)^n}{(1 + i)^n - 1}$$

$$\Rightarrow CCR = Ann. + i * R = (I - R) * \frac{i(1 + i)^n}{(1 + i)^n - 1} + i * R$$

With:
- i = interest rate
- Ann. = Annuity
- CCR = Capital Cost Recovery

Hint: in the Polish version of Microsoft Excel the Capital Cost Recovery is calculated as follows:

CCR = PMT(stopa; liczba_rat;wa;wp;typ) = PMT(i;n;-I;R;0)

To illustrate the use of different forms of amortization, as an example the investment in a compost plant is used. In the tables below the yearly costs are calculated based on the following data:

i	=	10%
n	=	4 years
I	=	10.000.000 Euro
R	=	2.000.000 Euro

This leads to the following results:

Table 14-1. Calculations of the capital investment costs, Linear depreciation

Year	Depreciation D_t	Balance Sheet value B_t	Interest I	Total cost $(D_t + I)$
1	2.000.000	8.000.000	1.000.000	3.000.000
2	2.000.000	6.000.000	800.000	2.800.000
3	2.000.000	4.000.000	600.000	2.600.000
4	2.000.000	2.000.000	400.000	2.400.000
Total	8.000.000		2.800.000	10.800.000

Table 14-2. Calculations of the capital investment costs, Fixed rate depreciation

Year	Depreciation D_t	Balance Sheet value B_t	Interest I	Total cost $(D_t + I)$
1	3.312.597	6.687.403	1.000.000	4.312.597
2	2.215.267	4.472.136	668.740	2.884.007
3	1.481.438	2.990.698	447.214	1.928.652
4	990.698	2.000.000	299.070	1.289.767
Total	8.000.000		2.415.024	10.415.024

Table 14-3. Calculations of the capital investment costs, Annuities

Year	Depreciation D_t	Balance Sheet value B_t	Interest I	Total cost $(D_t + I)$
1	1.723.766	8.276.234	1.000.000	2.723.766
2	1.896.143	6.380.090	827.623	2.723.766
3	2.085.757	4.294.333	638.009	2.723.766
4	2.294.333	2.000.000	429.433	2.723.766
Total	**8.000.000**		**2.895.066**	**10.895.066**

14.1.3   Costs of waste treatment and disposal facilities

The costs of facilities for waste treatment and disposal consist of three main contributors. The **capital costs** are the yearly costs of investment which are the sum of interest and depreciation. Capital costs are also known as fixed costs since their sum is not dependent on the operation (i.e. the number of tons treated) of the plant. **Operating costs** are all expenses related to the operation of a plant. These consist of costs for personnel, maintenance, overhead, energy, consumables, insurances etc. They are also known as variable costs because their sums by and large vary with the number of tons treated. **End-of-Life** (EoL) costs are costs for dismantling facilities at the end of their lifetime. This is especially relevant for landfills with their recultivation costs (often mandatory by law), but also for other large constructions such as incinerators. These costs only occur after the end of the lifetime of the facility when there is already no cash-flow to finance it. Therefore, the EoL costs should be 'earned' during the facility operation. Often, a fixed rate of the investment sum is assumed for EoL costs. These costs can be calculated yearly by accounting for them reciprocally. Alternatively, a fixed sum of the yearly capital costs or operational costs can be taken.

Although the costs for both the investment and operation of facilities depend strongly on local and international circumstances (e.g. level of the average wage, level of the economy, legal requirements, market situation, interest rate, bulk commodities price level, energy price level, etc.), in the following figures examples of investment costs and operation costs are provided. Displayed are extrapolated curves based on a variety of literature and real data for actually constructed and

operated facilities. The presented curves are of illustrational character only. In the case of planned investments more accurate and actual estimates should be provided.
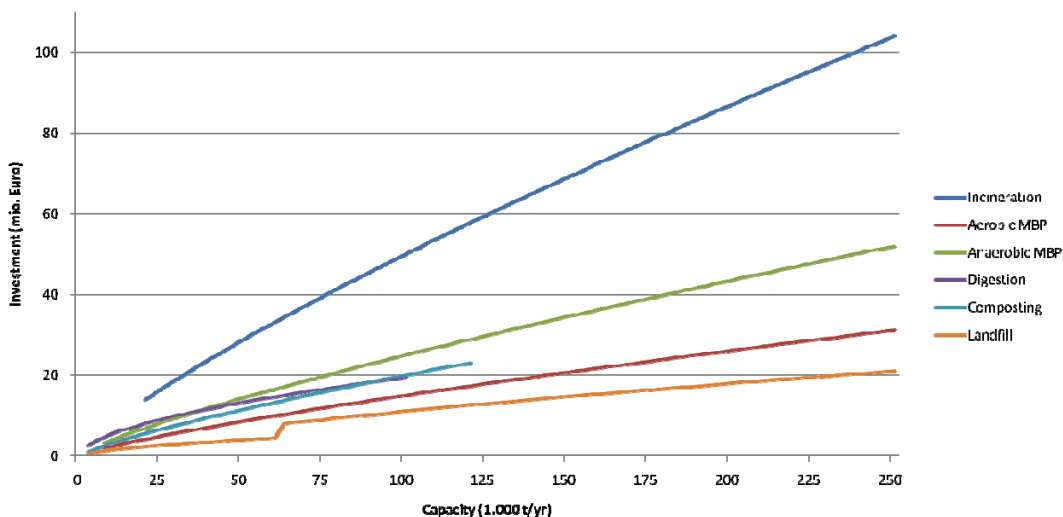
Figure 14-2. Average investment costs for various waste treatment and disposal facilities (Den Boer et al. 2005)

From Figure 14-2 it clearly follows that facilities with larger planned capacities required higher investment costs. At the same time, this dependency is not linear: the investment cost per ton of capacity decreases at an increasing facility size.

Figure 14-3 shows that the operation costs per ton of waste input decrease with increasing facility capacity. This is especially true for the treatment facilities in the range up to 50.000 t/yr.

Taking an Aerobic MBP with a capacity of 100.000 t/yr as an example, this leads to the following costs of the facility (lifetime: 20 yrs, interest: 8%, rest value: 0 Euro, EoL: 10% of investment; for EoL reciprocal discounting with the same interest rate):

1. investment capital costs (annuities) 15,0 mio. € → 1,53 mio. €/yr
2. operation costs: 40,0 €/ton→ 4,00 mio. €/yr
3. EoL costs: 1,5 mio. € → 33 th. €/yr
4. **Total costs of the Aerobic MBP**: 5,56 mio. €/yr → 56 €/ton

In these costs the revenues of sales of recovered materials and the costs for treatment and disposal of residuals are not included.

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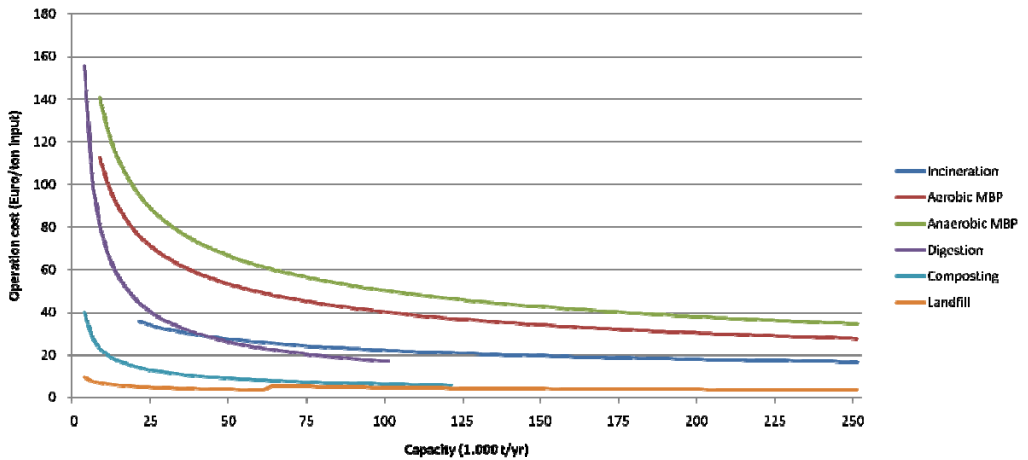
Figure 14-3. Average operation costs for various waste treatment and disposal facilities[11]
(Den Boer et al. 2005)

14.1.4   Costs of waste management systems

In most European countries, the local government organizes the municipal waste management system from both a planning and a financial perspective. Either the local government itself or an executive body that is controlled by it takes the decisions related to waste management. This does not necessarily mean that the local government itself executes all municipal waste management tasks. Often the collection, treatment or disposal of waste is undertaken by commercial parties selected by public tender procedures. The costs for individual stages of a waste management system occur therefore in form of

- direct costs (investment capital costs, operation costs, End of Life costs) of publicly owned treatment or disposal installations, collection services, temporary storage;
- gate fees or bills in case of external collection & treatment costs;
- overhead costs of the waste department of the local government (e.g. for public relations, management, awareness raising campaigns).

For Poland (2010), the situation is different with fully privatized waste management. Here the local governments are not the owners of the generated

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[11] excluding residue treatment & disposal costs, excluding revenues. Landfill not for the treatment of residues, price level 2004

waste and the collection companies have direct contracts with the citizens. This situation, however, is due to change soon.

As a rule of thumb, the costs of a municipal waste management system are about half caused by the waste logistics varying from a 30%-70% to a 80%-20% division (Bilitewski et al. 1997). Considering municipal waste management systems, the following general statements considering the costs of the waste logistics can be made:

- costs for bag collection are smaller than for container collection. This depends on the legal restrictions in force. Bags are a one-way system only, whereas containers have to be returned, at least to the curbside;
- costs for smaller containers are similar to larger containers. The emptying of the large containers costs less time per ton (but more per container). Generally, driving times between large containers are longer. The outcome of this trade of depends on the district type;
- the costs for mixed waste are lower than for separately collected fractions. For the treatment of the wastes, however, it is the other way around.

To illustrate how the costs of municipal waste management look in practice, here an example is provided of a city of 30.000 inhabitants. The particular city has a year round collection system for garden waste through depots on a district level leading to very high collection results for garden waste. In Figure 14-4, an overview of the city's waste management system is shown; the data provided is from the year 2002.
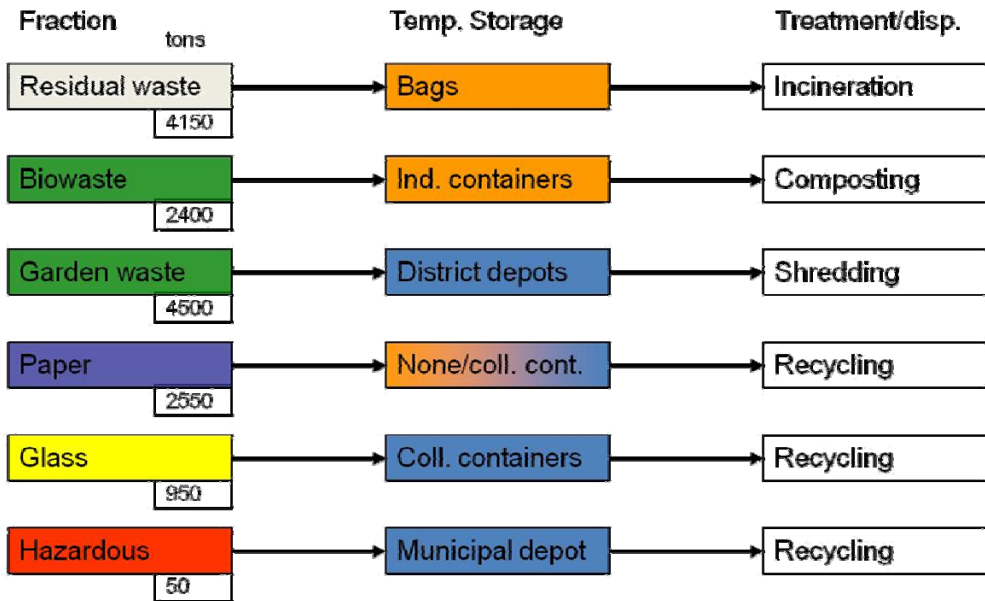
Fraction	tons		Temp. Storage		Treatment/disp.
Residual waste	4150	→	Bags	→	Incineration
Biowaste	2400	→	Ind. containers	→	Composting
Garden waste	4500	→	District depots	→	Shredding
Paper	2550	→	None/coll. cont.	→	Recycling
Glass	950	→	Coll. containers	→	Recycling
Hazardous	50	→	Municipal depot	→	Recycling

Figure 14-4. Overview of the waste management system of an example city of 30.000 inhabitants

The waste management system as shown in the figure above leads to a variety of costs for the municipality. These are shown in Table 14-4, in Euro per ton.

In terms of waste management there is no standard city. For the planning of any waste management system, the local situation should be considered. In the case of the exemplary city considered here, there are some peculiarities that are uncommon in European waste management. Yet, looking at the cost data provided in Table 14-4, it can be concluded that

- waste separation pays of financially. The costs for the separately collected fractions are significantly lower than for residual waste.
- for biowaste this is not the case. This follows from the fact that residual waste is collected in bags in the low rise areas. This leads to low costs for temporary storage and collection. In the case of containers for residual waste collection, the total costs would be similar for residual waste and biowaste. In many cases, the treatment costs for biowaste (composting) will be lower than for the incineration of residual waste.
- garden waste is collected separately in an intensive manner. For the considered city this pays off because of the high treatment costs for biowaste. On the other hand, inhabitants of the neighboring municipalities use this disposal possibility (waste tourism).

Table 14-4. Costs of the waste management system of an example city of 30.000 inhabitants (Euro/ton)

WM Stage	Residual	Biowaste	Garden waste	Paper[12]	Glass	Hazar-dous[13]
Temp. storage	7 €	22 €	8 €	- €	45 €	- €
Collection	39 €	62 €	16 €	39 €	30 €	7 €
Transport[14]	16 €	25 €	5 €	4 €	- €	23 €
Treatment/ disposal	85 €	70 €	30 €	- 9 €	- 9 €	644 €
Total	**147 €**	**179 €**	**58 €**	**35 €**	**66 €**	**674 €**

- the treatment costs per ton of hazardous waste are very high. For curbside collection also the logistic costs are at similar levels (does not apply for the considered city). For the total waste management system, the contribution of hazardous waste, however, is limited.

14.1.5   Pay as You Throw

The total costs of waste management systems are to be paid by the citizens. Mostly households pay a kind of waste tax. This can be a fixed-rate or it can be based on the number of persons in the household. In the ideal case, the municipal waste management expenses are covered fully by the taxation, but in practice it happens that a part is paid for by the general municipal budget. Contrary to fixed-rate taxation, a system of differentiated fees or Pay-as-You-Throw (PAYT) considers the quantity of waste that is generated. Those who generate more waste have to pay a larger contribution. The reasons for introducing PAYT are several (AOO 2004, Wille 2002):

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[12]Kerb-side collection of bundled paper: no temporary storage. Some paper is collected through local social and charity groups

[13] Bring system (to bulky waste station only): costs for battery collection only

[14] Costs of the bulky waste/transfer station are not included. For glass and part of the paper the transport costs are included in the collection costs.

- introduction of the polluter pays principle;
- waste prevention. If generating additional waste costs additionally, less waste will be generated;
- increased separate collection rates. As mainly residual (and to a lesser extent biowaste) has to be paid for when PAYT is introduced, it pays off to better segregate waste;
- enlarged environmental awareness of citizens. The introduction of PAYT is accompanied by awareness raising campaigns;
- lower overall costs for waste management. In The Netherlands, municipalities with PAYT have lower overall waste taxations than municipalities without PAYT (227 and 274 Euro/household resp.); however, this applies mainly to rural municipalities which already before introduction showed lower waste management costs (Agentschap NL 2010);
- exclusion of free riders from the waste management system. PAYT enables limiting the accessibility to the collection system for only those who are registered. In the case of open, non-PAYT systems, a considerable share of the small commercial waste can be disposed of through the household waste management system. With PAYT these can access the system only when they pay for it.

PAYT can be operated in different ways with varying levels of accuracy in the relation between the amount generated and the price paid. The fees can be based on

- **volume**. The size of the container determines the fee to pay.
- **volume and frequency**. Payment, depending on the container volume, for every time the container is offered for emptying. Collection vehicles and containers need to have hard- and software (transponders and reading units).
- **weight**. The actual amount of generated waste is weighed. Collection vehicles need additional weighing scales for the full and empty containers.
- **expensive bag**. Citizens can only offer waste for collection in special bags that have to be purchased at a price that includes the PAYT fee.

PAYT fees exist partly as a fixed basic fee and a flexible fee depending on the amount of waste generated (as is often the case for electricity, gas, water, telephone contracts). The fixed fee can include already a standard amount, e.g. 26 collections per year. Since the cost driving factors depend both on the frequency of collection and on the amount collected, also mixed forms of the above mentioned fee systems occur (e.g. weight and frequency).

To give an impression of the varying effectiveness and efficiency, in Table 14-5 an overview of various factors concerning PAYT fee systems is provided.

Table 14-5. Effects of PAYT systems in The Netherlands (AOO 2004, Agentschap NL 2010)

WM Stage	House-holds (%) 2010	Total fee (€) 2010	Residual waste Amount (kg/inh.) 2001	Residual waste Effect (%) 1995-2001	Recyclables Amount (kg/inh.) 2001	Recyclables Effect (%) 1995-2001
Volume	7,5	254	230	+3	214	+13
Volume & frequency	12,0	219	164	-17	184	-13
Weight	3,3	220	127	-39	162	-15
Expensive bag	3,5	208	121	- 43	229	+17
Other	0,1	-	-	-	-	-
Total PAYT	*26,4*	*227*	-	-	-	-
Number of persons	57,6	276	-	-	-	-
Fixed rate	15,9	270	-	-	-	-
Total non-PAYT	*73,5*	*274*	*245*	*+17*	*210*	*+13*
Total	**100**	**251**	-	-	-	-

In the above table it is clearly shown that the **average fees** for municipalities with PAYT systems in the year 2010 is significantly lower than for those without PAYT (-17%). The systems in which citizen can continuously save costs by offering less waste for collection show the strongest effect. The 'expensive bag' system is the cheapest because it has the lowest costs for the PAYT system itself. It should be kept in mind that the municipalities having PAYT systems are less urban (smaller inhabitant density) than those that do not have it.

The amount of **generated residual waste** in 2001 was significantly lower for municipalities that introduced PAYT with again the 'volume' system the showing

the least apparent effect. To consider the effect of the introduction of PAYT within one and the same municipality, the changes in the residual waste generation in the years 1995-2001 are shown. The considered municipalities introduced the PAYT system between the years 1996 and 2000 so the depicted changes reflect the actual effect of the introduction of PAYT. The non-PAYT municipalities are a selection of such municipalities. Municipalities registered a reduction in residual waste generation of up to 40%, while in the non-PAYT municipalities the amounts rose 17% in the same period.

The **recyclables** (glass, paper and biowaste) show a less clear picture. It would be expected that waste generators show better waste separation results after the introduction of PAYT resulting in higher amounts of recyclables. This is only the case for the 'volume' and 'expensive bag' systems, which show similar figures as for the non-PAYT municipalities (in absolute numbers; as a % of the total generated waste, the PAYT systems shown higher results). The other PAYT systems show a decrease in recyclables collection results. A reason for this development may be the fact that (in 2010) for the latter systems also a fee, in comparable height to the residual waste fee, was put on biowaste (Agentschap NL 2010). Biowaste comprises the largest fraction of the recyclables. For glass and especially paper, all systems showed increased collection results in the period 1995-2001. Also, later studies show a varied picture: Reichenbach found a clear link between reduced residual waste generation in PAYT municipalities in Germany (2008), whereas Dahlen and Lagerkvist did not in Sweden (2010).

Apart from the above mentioned reduction of residual waste generation and waste fees, PAYT also brings along a number of **adverse effects**:

- waste tourism. If one pays extra for every kg of waste generated, it pays off to take it along to work, family, friends, neighboring municipalities, where the waste management system is freely accessible (no PAYT introduced yet). Introduction of PAYT works best when done in cooperation with surrounding municipalities. Estimates of this effect are difficult to make, Linderhof et al. (2003) found a value of 5% of the generated waste disappearing to neighboring municipalities or employers;
- waste dumping. A more extreme form of waste tourism is the dumping of waste in forests and fields. AOO (2004) suggests that this effect is minimal, referring to experiences in Belgium, Switzerland and The Netherlands. In the case of tourism, the waste still ends up in containers and is collected in the ordinary collection system. Wild dumping, on the other hand, is clearly felt as being illegal and as directly endangering the environment. Nevertheless, if already a culture of wild dumping exists

(e.g. of construction waste), the introduction of PAYT likely leads to a significant increase of this practice;

- pollution of recyclables. In a PAYT system, it pays off to discard part of the residual waste through the bins for recyclables. This effect will be the largest for commonly used containers. Afvalsturing Friesland (2003) found an increased contamination of the separately collected paper waste in PAYT systems in comparison to non-PAYT municipalities;
- increased amounts of bulky waste. As a result of the introduction of PAYT, residual waste collection is restricted to containers (with often size-limited insert openings) or expensive bags. Thus, a part of the residual waste does not fit in or is not allowed next to the temporary storage units which leads to increased amounts of bulky waste. AOO (2004) reports varied on the effects on bulky waste generation;
- increased amounts of street litter. A danger of limiting the accessibility of commonly used containers is the additional occurrence of waste placed next to the temporary storage utility;
- problems of public acceptance. These may have a number of underlying causes. E.g. some citizens find it undesirable that the largest generators pay the most and suggest that fees should be equal for all (so that e.g. a family with small children does not have to pay extra for the generated diapers. For other utilities and services this is also the case. Post delivery, telecom services and electricity supply do not have differentiated rates, although costs are lower in cities than in the countryside). Others simply do not want to pay more, being the larger generators. Some inhabitants of high rise areas with commonly used containers may be required to have an individual accounting system (and not pay the average costs of the concerned container). Supplying such systems is more complex and expensive;
- PAYT systems come with high costs in investment and operation.

Considering the above mentioned threats to PAYT systems, it becomes clear that the introduction and maintenance of a PAYT system should be done very carefully. A strong focus should be put on prevention measures against wild dumping and waste tourism. Public awareness campaigns are essential for a successful introduction. It is generally considered as counter-effective, or at least very difficult, to introduce PAYT systems in larger cities.

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