

*Chapter 7*

## **SUSTAINABLE LANDFILL AND BIOLOGICAL STABILITY**

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### **ABSTRACT**

The landfill represents the most common way of municipal solid waste (MSW) disposal. On the other hand the anaerobic degradation of the biodegradable fraction of the landfilled MSW, causes several environmental problems such as the production of methane, VOC odors and leachate, the presence of vectors more – insects, rodents, and birds – public health hazard, explosions and plants toxicity. All these negative impacts and the long time required to stabilize the materials (after care period) are the major issues that make landfills unsustainable. On the contrary, a landfill is defined sustainable if there is a safe disposal of waste, and its subsequent degradation to the inert state in the shortest possible time-span.

The strength of environmental impacts of MSW depends fundamentally on both the quantity and the characteristics of the organic fraction of MSW (OFMSW).

The reduction of OFMSW in landfills can be obtained by three different approaches: i) separated collection of OFMW to produce compost; ii) waste burning to produce energy and iii) mechanical-biological treatment (composting-like process) to produce a stabilized material.

Irrespective of the process, all these methods achieve high levels of biological stability of the residual to be landfilled. Biological stability indicates the extent to which biodegradable organic matter has decomposed. The importance of knowledge concerning the biological stability of a waste is recognized because it affects the potential for MSW impacts: high biological stability mean a more sustainable landfill.

In this chapter, we discuss the MSW impacts in landfill and processes, i.e. mechanical biological treatment, able to reduce waste impacts when it is landfilled. Reduced impacts can be obtained by getting high level of biological stability. Methods to determine the biological stability and full-scale application will also be discussed.

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## 1. INTRODUCTION

### 1.1. Not Sustainable Versus Sustainable Landfill

A landfill can be defined as “the engineered deposit of waste onto and into land in such a way that pollution and harm to the environment is prevented and through restoration, land provided which may be used for another purpose” (ISWA, 1992).

Municipal solid wastes (MSW) are mainly disposed into a landfill (Table 1), because a landfill is the simplest, cheapest and low cost-effective method of waste disposal (Allen, 2001).

Municipal Solid Wastes are defined as the waste fraction collected by a municipality. It concerns waste from households, small business, office buildings and institutions such as schools, hospitals, government buildings, waste from parks and street cleaning (EUROSTAT, 2006).

Generally the MSW composition takes in: (1) food waste, (2) garden (yard) and park wastes, (3) paper and cardboard, (4) wood, (5) textiles, (6) nappies (disposable diapers), (7) rubber and leather, (8) plastics, (9) metal, (10) glass (and pottery and china), (11) other (e.g. ash, dirt, dust, soil, electronic waste).

The composition and the volume of disposed wastes vary nationally and regionally in relation to human activity and to the quantity and type of products that communities consume (Taylor and Allen, 2001) (Table 2).

**Table 1. MSW production in the world and fraction of MSW disposed in landfill (IPCC, 2006 modified)**

	MSW generation rate (Mg/cap/year) <sup>†</sup>	MSW disposed in landfill (% total MSW)
Africa	0.29	69
Asia	0.28±0.15	63±32
America	0.40±0.20	61±15
Europe	0.52±0.10	67±23
Oceania	0.69	75

<sup>†</sup>data referred to 2000.

**Table 2. Composition of MSW in the world**

Fraction	UE <sup>†</sup>	African cities <sup>‡</sup>	USA <sup>§</sup>	Asia (urban zone) <sup>¶</sup>
	Content (% by weight)			
Paper	25-35	0-12.9	35.2	4.2-19
Plastic	7-10	0-6.3	11.3	4-19
Metals	3.5-7	0-2.6	8	0.1-6
Glass	5-10	0-1.9	5.3	0-3
Ceramic	1-2	-	-	-
Yard Waste	10-15	-	12.1	-
Hazardous Waste	10-15	-	-	-

Fraction	UE†	African cities‡	USA§	Asia (urban zone)¶
Rubber, Leather And Textiles	-	0-3	7.4	0-9
Wood	-	-	5.8	-
Food, Vegetable And Fruits	25-35	-	11.7	-
Other	-	-	3.4	2-37.7
Biodegradable Fraction	-	55.8-94	-	39-74
Organic fraction#	60-90	55.8-100	72.1	43.2-100

† Data from: Bidlingmaier et al., 2004

‡ Data from: Asomani-Boateng and Haight, 1999

§ Data from: USEPA, 2003

¶ Data from: Zurbrügg, 2002

# Organic fraction=paper+rubber+leather+textiles+wood+food+vegetable+fruits+biodegradable fraction+yard waste

In all the cases, the organic fraction constituted the major component of MSW (Table 2).

In the landfill the biodegradation of the organic fractions is the major mechanism that governs biological pollution (odours, biogas and leachate) (Derham, 1995; Driessen et al., 1995; Westlake, 1997; Read et al., 2001; Visvanathan, 2005). The uncontrolled productions of landfill gas (CH<sub>4</sub>, CO<sub>2</sub> and trace of non-methane volatile organic carbons) lead to ozone depletion and contribute to global warming effect (Bogner and Matthews, 2003). The unregulated dispersion of leachate results in soil, surface and groundwater contaminations (El-Fadel et al., 1997; Taylor and Allen, 2001; Kylefors, 2002). In order to reduce or prevent the environmental pollution three different landfill approaches have been adopted (Read et al., 2001).

1. *Dilute and attenuate landfill.* It is the principle of landfill disposal for unconfined sites (open dump). They are most common in *rural*, remote and developing areas. This approach takes into account the presence of environmental auto-depuration processes. Nevertheless the risk of contamination is high (Westlake, 1997).

## 2. *Entombment landfill*

The main concept of dry tomb is to isolate waste from the environment in a compacted soil and plastic sheeting tomb. Plastic sheeting is a thin layer of high-density polyethylene (HDPE). It is combined with a compacted soil-clay layer to form composite liner (Read et al., 2001).

3. *Containment landfill.* The containment landfill is defined as: “the landfill site where the rate of release of leachate into the environment is extremely low. Polluting components in waste are retained within such landfills for sufficient time to allow biodegradation and attenuation processes to occur, thus preventing the escape of pollution species at an unacceptable concentration” (ISWA, 1992). Although this definition referred to leachate production it can be extended to all impacts (e.g. biogas production) (Allen, 2001). When MSW and other organic materials are contained, the landfill is called a sanitary landfill. In some countries (e.g. UE), all landfills are sanitary landfills (Allen, 2001).

These landfills usually have physical barriers such as leachate and biogas collection systems (Visvanathan, 2005) and procedures to protect the public from exposure to the disposed wastes.

The three different approaches to landfill construction lead to different impact levels and a different time-scale (Figure 1). While the dump landfills have the maximum impact during its management, the dry tomb and the containment landfills inhibit waste degradation and considerably prolongs the activity of waste, slowing down its degradation into an inert state (Allen, 2001; Visvanathan, 2005). However, due to containment, the potential pollution of the landfill remains in place. Therefore the containment strategy is effective as long as the impermeable shell remains intact (Allen, 2001). Given the uncertainty regarding the durability of artificial lining systems over a long time-span, it also increases the potential for environmental pollution in the long term (Allen, 2001).

As a consequence of that, the dry tomb and the containment landfills can show a high level of pollution after the closure time. Similar approaches to the landfill management do not allow a sustainability of the landfill (Visvanathan, 2005).

The concept of sustainability with respect to landfills as been developed by Allen (2001): “a landfill is defined sustainable if there is a safe disposal of waste, and the subsequent degradation of organic matter proceeds in the shortest possible time-span...and with minimal damage”. Westlake (1997) defined the sustainable landfill as “a landfill designed and operated in such a way that minimizes both short and long-term environmental risks to an acceptable level”. More correctly a sustainable landfill should consider, also, the entire chain of the waste management as it influences the quantity and quality of the MSW disposed in a landfill (Derham, 1995; Driessen et al., 1995; Allen, 2001).

An approach to a sustainable waste management is starting in both the European Union (EU) and the United States (USA) and the demand for a such system is also increasing in other countries (e.g., Japan, Australia and Taiwan) (Visvanathan, 2005).

Sustainable MSW management is known as "waste management hierarchy" consisting in the three Rs (3Rs): reduce, re-use and recycle wastes that are the options preferred to other waste management such as landfill or incineration (Figure 2).

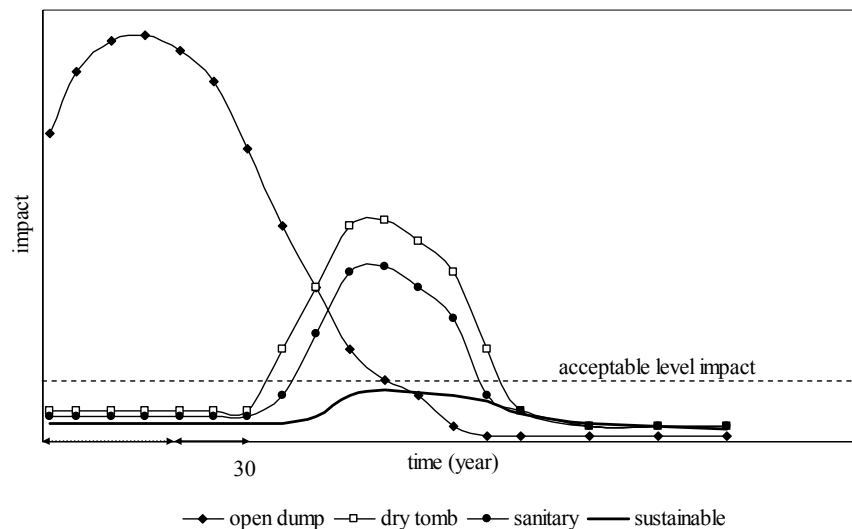


Figure 1. Impact level as consequence of different approaches in landfill construction (Visvanathan, 2005, modified).

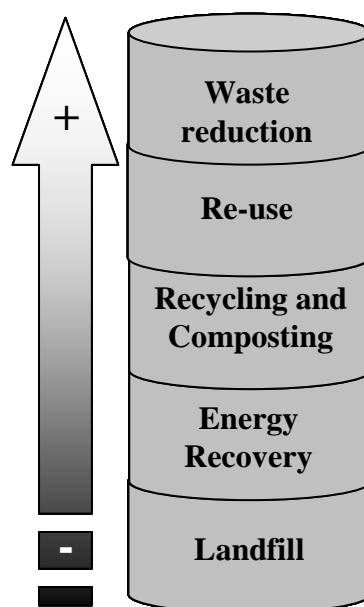


Figure 2. A representation of the 3Rs waste management hierarchy

changes that make these methods not so attractive for the society. As a consequence of that, both incineration and, above all, landfilling are nowadays the most common ways used to manage MSW (Table 1) (IPCC, 2006).

The reduction of organic fraction of MSW to be landfilled can be obtained by three different approaches: i) source separated collection of OFMSW to produce compost; ii) MSW burning to produce energy and, iii) mechanical-biological treatment of MSW to produce a stabilized or a composting-like material to be landfilled (Table 3).

**Table 3. Positive (+) and negative (-) effects of different MSW pre-treatments on landfill management (Visvanathan, 2005, modified)**

	Source separated collection	Mechanical pre-treatment†	Biological pre-treatment	Thermal pre-treatment‡
Organic fraction content	+	--	+	++
Leachate production	+	+	++	+++
Biogas production	+	--	++	+++
Reactions in landfill	+	-	++	+++
Waste mass reduction	-	++	--	-
After care period	+	--	++	+++

†Solid Refuse Fuel production and the residual into the landfill.

‡Residual waste into the landfill.

Biological pretreatment of MSW prior to landfilling helps to reduce the organic matter and above all the putrescible fractions contained in the waste, supporting control and

minimizing landfill emissions (Raninger et al., 1999; Lechner et al., 2001; El Fadel et al., 2002) (Table 3).

## 1.2. Environmental and Health Impacts

The major concern regarding landfilling sites is the release of gases and leachate that can contaminate the terrestrial and aquatic systems (El-Fadel et al., 1997; Tsiliyannis, 1999) (Figure 3).

The biodegradation of the organic fraction of MSW, the major factor that determines the landfill impacts, follows five different separated phases (Barlaz et al., 1990; El-Fadel et al., 1997; Tsiliyannis, 1999; Allen, 2001) (Figure 4).

*Phase I:* Aerobic phase. Phase of short duration (months) that is sometimes divided into an oxygen-consuming and a nitrate-consuming phase. The organic components (long molecular chains of complex carbohydrates, proteins and lipids) are hydrolyzed and then oxidize aerobically into  $\text{CO}_2$  and  $\text{H}_2\text{O}$  by the bacteria.

*Phase II:* Anaerobic acidic phase. The depletion of oxygen during Phase I, creates the anaerobic conditions in the landfill. The anaerobic bacteria convert hydrolyzed compounds into organic acids and alcohols. A decrease of pH occurs.

*Phase III:* Acetogenic phase. The organic acids are transformed into acetate.

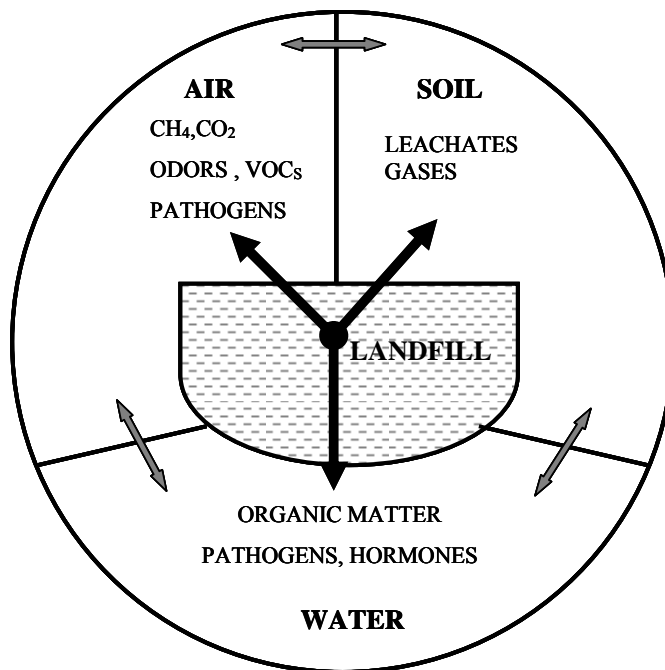


Figure 3. The effects of landfill contamination into environmental compartments (Dubois et al., 2002, modified).

*Phase IV:* Methanogenic phase. The degradation products of the acidogenic phase are further degraded into gaseous carbon as methane and carbon dioxide.

*Phase V.* Aerobic final phase. During this phase degraded OM creates voids in the landfill and O<sub>2</sub> penetrates in these voids re-starting aerobic processes (e.g. methane oxidation) (e.g. Bozkurt et al. 1999; Revans et al. 1999, Kjeldsen et al. 2002). Some authors used to divide this phase into different sub-phases (Bockers and Steinberg, 2005).

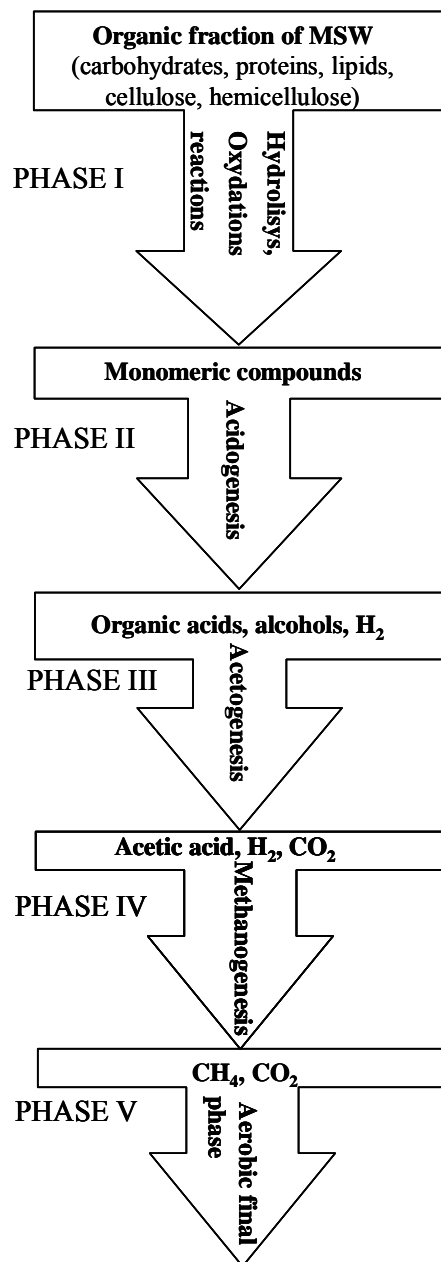


Figure 4. Landfill phases scheme of the organic matter degradation (El-Fadel et al., 1997, modified).

### 1.2.1. Leachate

#### 1.2.1.1. Leachate Production

Leachate is formed when the refuse moisture content in the waste exceeds its field capacity which is defined as the maximum moisture that is retained in a porous medium without producing downward percolation (El-Fadel et al; 1997; Driessen et al., 2001). Moisture retention depends on the holding forces of surface tension and capillary pressure. Percolation occurs when the magnitude of the gravitational forces exceed the holding forces. Percolating water provides a medium in which waste can undergo a change into simpler substances through a range of chemical reactions (dissolution, hydrolysis and redox reactions) and microbial metabolism (El-Fadel et al., 1997).

Many factors contribute directly to landfill moisture and affect leachate or moisture distribution within the landfill (Table 4).

Mechanisms regulating mass transfer from wastes to leaching water can be divided into three groups: (i) biological processes (e.g. hydrolysis and biodegradation reactions), (ii) solubilisation of soluble salts contained in the waste and, (iii) suspension of particulate matter (Kjeldsen et al., 2002).

During the Phase I, the leachate produced is low but it is hardly polluted. The hydrolysis and oxidation reactions determined the formation of water and CO<sub>2</sub> that is partly dissolved in the water (HCO<sub>3</sub><sup>-</sup> formation). The leachate pH is approximately 8 due to high NH<sub>3</sub> concentration (Kylefors, 2002).

During the following acidogenic phase, the pH decreases (4.5<pH<7.8) (Table 5) and the volatile fatty acids (VFA) and the ammonia increase.

The successive oxygen consumption causes a depletion of volatile fatty acids with an increase of the pH and alkalinity values (acetogenic phase). Leachate from this phase is characterized by high values of biochemical oxygen demand (BOD<sub>5</sub>) and high BOD<sub>5</sub>/COD ratios (>0.5) (Table 6).

At the metanogenic phase the composition of leachate is characterized by a pH of 6-8. At this time the composition of leachate is characterized by a low concentration of volatile acids and total dissolved solids (TDS) and by a moderate organic content that is dominated by fulvic acids (Christensen, 1992; Kylefors, 2002). The degradation processes convert nitrogen into ammonium. Leachate produced at this stage is characterized by relatively low BOD and low BOD<sub>5</sub>/COD ratio (Table 6) (El-Fadel et al., 1997; Kjeldsen et al. 2002).

**Table 4. Factors influencing leachate production (from El- Fadel et al., 1997, modified)**

Landfill leachate	Factors
Composition	Site operation and management
	Refuse characteristics
	Internal processes
Formation	Climatic and hydrogeological
	Site operation and management
	Refuse characterization
	Internal processes



**Table 5. Leachate characterization during the different landfill phases (Kylefors, 2002, modified)**

	Landfill phases			
	Acidogenic†	Acetogenic ‡	Methanogenic†	Methanogenic ‡
	Interval		Interval	
pH	4.5-7.8	6.73	6.4-9	7.52
BOD <sub>5</sub> (mg/L)	4,000- 68,000	18,632	<0.5-1,770	374
COD (mg/L)	400-152,000	36,817	<1-8,000	2,307
VFA (mg/L)	963-22,414		<5-146	
DOC (mg/L)	-	12,217	-	733
NH <sub>4</sub> <sup>+</sup> (mg/L)	8.5-3,610	922	<1-2,040	889
NO <sub>3</sub> <sup>-</sup> (mg/L)	<0.2-18	1.8	<0.1-64	0.9
PO <sub>4</sub> -P (mg/L)	<0.05-22.6	-	<0.01-18.4	-
P (mg/L)	-	5	-	4

† Data from: Ehrig, 1983; Robinson and Gronow, 1993; Kylefors, 2002.

‡ Data from: DOE, 1995. Average of 35 (acetogenic phase) and 25 (methanogenic phase) landfills.

**Table 6. BOD<sub>5</sub>/COD ratio during the different landfill phases (El-Fadel et al., 1997, modified)**

Landfill phase	Significance	BOD <sub>5</sub> <sup>†</sup> (mg L <sup>-1</sup> )	COD <sup>†</sup> (mg L <sup>-1</sup> )	BOD <sub>5</sub> /COD ratio
Phase II	Young	13,000‡	22,000‡	0.58†
	unstable landfill	4,000-40,000 §	6,000-60,000 §	>0.5#
Phase IV	Old stable	180 ‡	3,000 ‡	0.06†
	landfill	20-550 §	500-4,500 §	<0.1#

† Data from: Kjeldsen et al., 2002.

‡average value

§range value

#Data from: El-Fadel et al., 1997.

### 1.2.1.2. Leachate Environmental Impacts

The dispersion of leachate around the landfill site causes the pollution phenomena in freshwater, groundwater and soil. The high presence of organic matter in the leachate causes the depletion of oxygen (water anoxia phenomena) (Dubois et al., 2002). Moreover the high presence of nutrient (nitrogen and phosphorous) may lead to the water eutrophication (Schwarzbauer et al. 2002). The leachate can also contain hormones, that affect the physiological characteristics of the fauna of the aquatic systems it pollutes (feminization process), and pathogen microorganisms (Donnelly et al., 1981; Pahren, 1987). Nevertheless pathogenic organisms seemed to be of minor importance in landfill leachate (Christensen et al., 1994).

### 1.2.2. Biogas Production

The biogas produced in a landfill is mainly composed of methane and carbon dioxide with trace amounts of non-methane organic compounds (NMOC) that include air pollutants and volatile organic compounds (Themelis and Ulloa, 2007) (Table 7).

*The greenhouse gas production.* The biogas (CO<sub>2</sub> and CH<sub>4</sub>) is one of the products of the biodegradation of waste in landfills. Landfills release large amounts of CO<sub>2</sub> and CH<sub>4</sub> which are greenhouse gases (GHGs). The total anthropogenic methane emissions on Earth were calculated for the year 2000 to be of 282.6 x 10<sup>6</sup> Mg (USEPA, 2003) (Table 8). The 13% of this amount (36.2 x 10<sup>6</sup> Mg) was due to landfill emissions (Table 8, 9).

The composition of biogas depends on the landfill phases (Bockers and Steinberg, 2005). In the Phase I (aerobic phase), the gases are mainly nitrogen and oxygen. The microorganism metabolism takes up the amount of CO<sub>2</sub> concentration. During the Phase II (acidogenic phase) the oxygen is consumed completely; the emitted gases are hydrogen and CO<sub>2</sub> while nitrogen remains present. Phase II is instable and the process can reverse back if the oxygen contacts the waste.

During Phase III, the chemical condition for the growth of methanogen community is created. As a consequence, the CH<sub>4</sub> production increases to reach a level of 50-60% v/v (Bockers and Steinberg, 2005). At the same time, the concentration of CO<sub>2</sub> decreases from 70% v/v to 40% v/v. In this phase the production of biogas is large but lower than Phase III.

**Table 7. Landfill biogas composition**

Compound	(Themelis and Ulloa, 2007)	(Daskalopoulos et al., 1998)
	% (v/v)	
CH <sub>4</sub>	50	63.8
CO <sub>2</sub>	45	33.6
Nitrogen	5	2.4
Oxygen		0.16
Hydrogen		0.05
Long chain alkane		0.05
Ethene		0.018
Ethane		0.005
Acetaldehyde		0.005
Butane		0.003
Propane		0.002
CO		0.001
Helium		0.00005
Other		0.00005
Halogen compounds		0.00002
H <sub>2</sub> S	<1	0.00002
Sulphide organic compounds		0.00001
non-methane organic compounds (NMOC)	2700 ppmv	
Alcohol		0.00001

The production during this phase remains constant as amount and as composition ( $\text{CH}_4:\text{CO}_2$  ratio is 1.25) (Bockers and Steinberg, 2005).

Phase V is the long-term phase in which most of the gas emission occurs with pressure inside the landfill higher than atmospheric pressure. The air infiltration in the landfill determines the re-start of aerobic microorganism metabolism and the oxygen oxidizes the methane inside the landfill body with a progressive reduction to reach 0% v/v (Bozkurt et al. 1999; Revans et al. 1999; Kjeldsen et al. 2002).

The landfill methane estimation is determined considering theoretical or real conditions. The theoretical approach supposes that the organic fraction decomposed at an optimized biodegradation rate (Bogner and Matthews, 2003).

**Table 8. Global anthropogenic methane emissions (USEPA, 2003)**

Font	Anthropogenic methane emission %
Enteric fermentation	28
Natural gas	15
Solid waste	13
Rice	11
Waste water	10
Coal	8
Biomass burning	5
Bio fuel combustion	4
Manure	4
Oil	1
Fuel stat and mobile	1

**Table 9. MSW landfills biogas emissions**

Country	kg $\text{CO}_2$ eq. /person†	Total emissions† (%)
USA	397.72	17.47
China	31.82	6.15
Mexico	281.82	4.45
Canada	695.00	3.38
Russian federation	217.73	4.57
Saudi Arabia	652.73	2.59
India	13.18	2.12
Brazil	80.45	2.22
Ukraine	260.91	1.79
Poland	403.18	3.60
South Africa	345.45	2.24
Turkey	134.09	1.39
Israel	1422.72	1.29
Australia	390.45	1.16
Dem. Rep. of Congo	107.27	0.99
Rest of the world	104.09	44.59

†data from EPA 2006

Data referred to 2005

In this case the CH<sub>4</sub> amount depends on the biodegradable organic carbon content (Bogner and Matthews, 2003). The literature reports several theoretical CH<sub>4</sub> productions for MSW and for different countries (0.18 kg CH<sub>4</sub> kg<sup>-1</sup> MSW -wet basis- for U.S. and 0.25-0.47 kg CH<sub>4</sub> kg<sup>-1</sup> MSW -wet basis- for Germany and Italy) (Halvadakis et al., 1983; Bogner and Spokas, 1993).

However the real landfill condition does not allow the highest methane production obtaining lower CH<sub>4</sub> emission (0.036 kg CH<sub>4</sub> kg<sup>-1</sup> dry solid waste) (Richards, 1989; Bogner and Matthews, 2003).

Data referring to landfills shows different production ranging from 0.1 to 1.0 Mg CH<sub>4</sub> ha<sup>-1</sup> d<sup>-1</sup> (Nozhevnikova et al., 1993; Hovde et al., 1995; Borjesson, 1996; Czepiel et al., 1996; Mosher et al., 1999; Galle et al., 2001; Morris et al., 2001).

### 1.2.3. Odors

The concentration of odorigen molecules in the biogas causes the landfill odor emissions.

The presence of odors in the landfill depends on MSW composition, MSW putrescibility and biogas production. The anaerobic biodegradation of MSW determines the formation of odorigen molecules (mercaptan, thiophenol, thioalcohol, thioacids, and aliphatic amine) that during the biogas production are emitted (Table 10) (APAT, 2003). The concentration of odors depends on the landfill's age and by microbial populations (Young and Parker, 1984).

Although some molecules contained in the biogas can be toxic, the main impact concerns the environmental nuisance (El-Fadel et al. 1997).

**Table 10. Odorigen molecules detected near MSW landfills (APAT, 2003)**

molecules	Perception level (mg/m <sup>3</sup> )	MSW landfills		
		A	B	C
limonene	0.057	0.4	4	1.9
xylene	0.4	0.1	0.2	0.1
ethyl benzene	0.2	0.1	0.1	0.1
Propyl benzene	0.04	1.7	2.4	3.0
Butyl benzene	0.1	1.4	0.6	0.3
Methane thiole	0.00004	5	1	2000
Dimethyl sulphur	0.02	0.1	0.4	0.1
Buthane-2-holo	0.1	0.1	0.2	2.1
Methyl buthanate	0.005	0.1	0.1	0.8
Ethyl propilate	0.1	0.1	0.1	0.1
propylpropionate	0.1	0.1	0.1	0.1
Butyl acetate	0.003	0.1	0.1	0.1
propylbutaneato	0.1	0.1	0.1	0.1
dipropylethers	0.07	0.1	0.1	0.1

#### **1.2.4. Pathogens**

Among the landfill microorganisms, human pathogens can be found. Their presence has three different sources:

- 1) initial contamination of MSW. MSW have a high variable composition on microorganisms including pathogens. Household waste can contain nappies that can be soiled with faeces and animal wastes (dogs and cats faeces). In addition, MSW can contain bloodstained materials (pads, tampons, etc...) with possible sources of infectious viruses disposed in landfills though viral survival away from the host is considered rare (Pahren, 1987).
- 2) Pathogens may be transported to landfill sites by vermins (rats) and other scavengers in particular seagulls (Taylor and Allen, 2001).
- 3) Growth of pathogen microorganisms in the landfill body. During the Phases III-IV (long landfill phases) the landfill is similar to an anaerobic reactor. Most of the organisms that carry out these processes are harmless saprophytes but a small percentage of this population is composed of opportunistic pathogenic microorganisms (Taylor and Allen, 2001).

The fate of pathogens in landfill is not understood although it is generally assumed that most are rapidly inactivated by the condition that prevails in landfills (initial temperature rise and persistent low pH, high interspecies competition) (Surflita et al., 1992).

## **2. DISCUSSION**

### **2.1. Organic Fraction of MSW and Waste Impacts**

The complete characterization of the organic fraction of MSW is fundamental to measure the potential impacts of waste. Doing so, both a quantitative and qualitative approach should be considered (Figure 5). In the following paragraph, the methods to quantify and qualify the organic fraction of MSW are described.

#### **2.1.1. Biogenic/Biomass Fraction of the Municipal Solid Waste**

Municipal solid waste contains organic matter of biogenic origin: the biomass. This fraction is biodegradable and it is responsible for the biological waste impacts in landfills.

Recent definitions of biomass, biogenic and biodegradability can be given as in the following:

*Biomass*: material of biological origin excluding material embedded in geological formation or transformed to fossil (European Committee For Standardization, 2006).

*Biogenic*: produced in natural processes by living organisms but not fossilised or derived from fossil resources (European Committee For Standardization, 2006).

*Biodegradable*: material capable of undergoing biological anaerobic or aerobic decomposition under conditions naturally occurring in the biosphere (Landfill Directive 1999/31/EC).

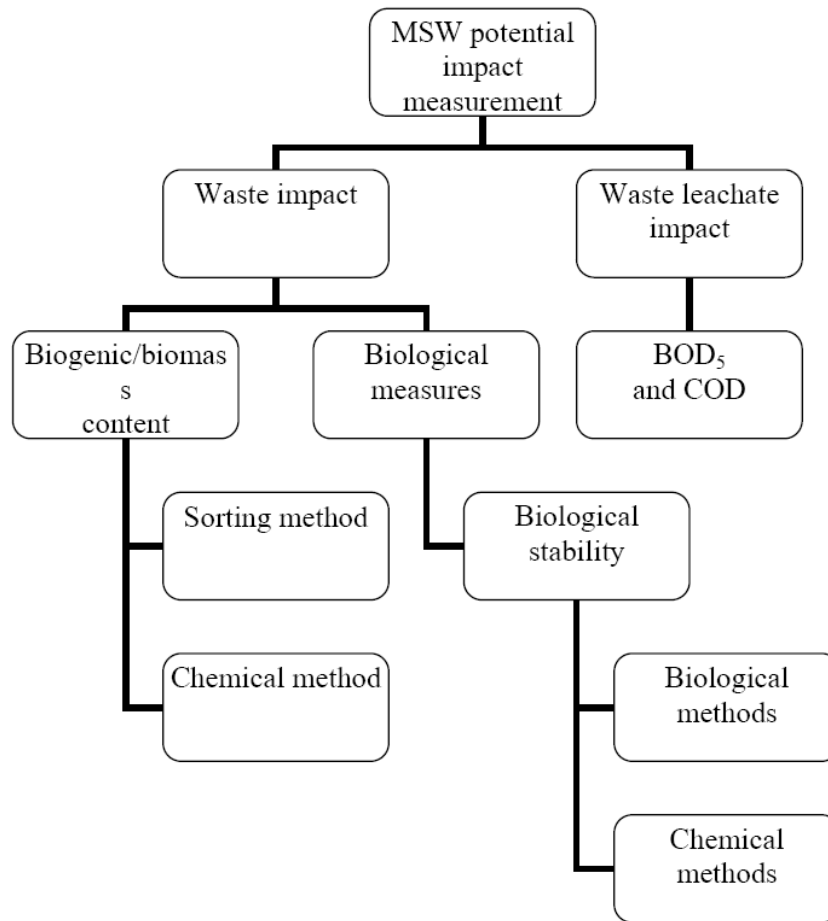


Figure 5. Potential waste impact measurement: logic flow chart.

Waste impacts depend on the amount of biomass present in the waste and on its degradability (Bogner and Matthews, 2003). Therefore a correct approach to the determination of quantity/quality of biomass is an important argument in waste management.

Operatively the biogenic fraction of MSW and derived products can be determined by using the manual sorting method and the chemical dissolution method (European Committee For Standardization, 2006).

*Manual sorting method.* The various components of MSW are sorted out by a combination of hand picking and mechanical sorting. Prerequisite for the manual sorting is the aspect of MSW and its size ( $\varnothing > 10$  mm) (European Committee For Standardization, 2006). The manual sorting is applied only to fresh municipal solid waste because for treated MSW the different fractions cannot be identified. MSW is selected by the operator in the following fractions: glass, metals, plastics, non-combustibles and biomass/biogenic fraction (Table 11). The biomass/biogenic fraction is composed of kitchen, garden, wood, leather, paper, cardboard, nappies and textiles collected separately (European Committee For Standardization, 2006).

**Table 11. Manual sorting fractions (European Committee For Standardization, 2006)**

MSW fraction	Biogenic/not biogenic/inert classification
Biological waste (leaves, grass, food residues, vegetables, etc...)	Biomass/biogenic
Paper-paperboard (Packaging paper, cardboard, newspaper, milk carton, etc...)	Biomass/biogenic
Wood (Waste wood, sawdust, waste wood from garden, etc..)	Biomass/biogenic
Tissue (diaper, tissues, tampons, etc...)	Biomass/biogenic
Fabric	Biomass/biogenic
Leather-rubber	Biomass/biogenic
Glass	Inert
Stone	Inert
Fines	Inert
Plastics (soft-rigid)	Not biogenic
Carpet/mats	Not biogenic
Iron	Inert
Non ferrous materials	Inert

**Table 12. Manual sorting results of residual Italian MSW collected with different collection systems (APAT, 2003)**

	Collection System 1 †	Collection System 2 ‡	Collection System 3 §
% MSW wet weight (w.w.)			
Fraction <20 mm#	3.42±0.86 †† a	9.15±1.47 b	11.11±0.53 b
Organic Fraction	12.18± 1.74a	15.83±6.13 a	27.15±3.34 b
Paper	17.10±5.33 a	13.98±4.83 a	18.82±6.18 a
Paperboard	12.56± 3.37a	13.88±4.49 a	10.43±1.01 a
Textile	5.60±3.45 a	3.43±1.00 a	3.43±2.30 a
Leather, Rubber	0.33± 0.32a	3.27±1.27 b	1.17±1.17 a
Wood	1.05±1.21 a	5.35±3.05 b	1.98±1.36 ab
Nappies	13.55±1.3 b	5.35±1.52 a	2.72±1.21 a
Total Biogenic Content	65.79	70.24	76.81

† Separated recyclables and organics (door to door collection).

‡ Two bins collection for recyclables and for general household waste.

§ Single bin collection.

# The fraction  $\varnothing < 20$  mm were determined differing by the European Committee For Standardization, 2006 indication ( $\varnothing < 10$  mm),

†† Within lines, means followed by the same letter are not significantly different according to Tukey Test ( $p < 0.05$ )

The fines fraction of the MSW (fraction <10mm) is also considered as part of the organic fraction. The weight of each fraction is recorded and the composition of the MSW calculated in terms of percentage by wet weight (Table 12).

*Chemical dissolution method.* When the size of the waste is too small to make the manual sorting (e.g. after biological treatments), the chemical method is carried out. This method is based on the selective dissolution of biogenic and not-biogenic organic fractions with an oxidant agent. The biogenic fraction is defined as the portion of the total organic matter that is solubilized by two-step treatments with H<sub>2</sub>SO<sub>4</sub> (78% w/w) and H<sub>2</sub>O<sub>2</sub> (35% v/v). More details are reported in European Committee For Standardization (2006).

This method showed a good reliability when it is applied to pure materials contained in MSW (Table 13). During the dissolving test, biodegradable plastics will behave like biomass, and plastics synthesized with a base of vegetable products will sometimes behave like non-biomass. In addition, there are also various uncommon biomass materials that are not entirely converted during the dissolving test (Laine-Ylijoki et al., 2004) (Table 13).

**Table 13. Amount of biogenic fraction contained in different pure organic material composing MSW, detected by dissolution chemical method**

Materials		Biogenic fraction (% dm)	
		Experimental data	Difference between experimental and theoretical data
Bio degradable plastic	Based on starch	99.3 †	0.7 †
	Based on corn	-0.2 †	n.d.
Fabric	Cotton	99.4 † 77 ‡	0.6 †
	Leather	92.7 §	8.3 †
	Linen	99.5 †	0.5 †
	Nylon	97.3 †	-97.3 †
	Viscose	61.4 †	38.6 †
Paper	Copy paper	99.1 † 86.69±0.04§	0.9 †
	Glossy paper	99.1 †	0.9 †
	Paper sludge	28 ‡	n.d.
	Carton	82 ‡	n.d.
	Methyl cellulose	81 ‡	n.d.
	Newspaper	96.7 †	3.3 †
Plastic	HDPE	0.9 † 0.42±0.11 §	-0.9 †
	LDPE	0 † 0.45±0.11 §	0 †
	PC	-0.6 †	-0.6 †
	PET	0 † 0.74±0.55 §	0 †
	PP	0 †	0 †



Materials	Biogenic fraction (% dm)	
	Experimental data	Difference between experimental and theoretical data
PS	-1.5 †	1.5 †
Plastic bag	0 †	0 †
PUR	94.8 †	94.6 †
PUR	98.3 †	-98.3 †
PVC	1.7 †	-1.7 †
Cellophane	0.38±0.08 §	n.d.
Primary Fuel		
Hard coal	46 †	54 †
Lignite	93 †	93 †
Peat	96.8 † 53 ‡	96.8 †
Rubber		
Natural	84.4 †	15.6 †
Butyl rubber	0 †	0.2 †
Chloroprene rubber	9.7 †	9.7 †
Silicon rubber	85.7 †	85.7 †
Wood	99.06 † 81 ‡ 86.69±0.04 §	0.94 †
Composite materials		
Tetrapak with aluminium	86 †	0.7 †
Tetrapak without aluminium	76 †	46 †

† Data from: European Committee For Standardization, 2006

‡ Data from: Laine-Ylijoki et al., 2004

§ Data from: DiProVe

HDPE =high density polyethylene

LDPE=low density polyethylene

PC =poly carbonate

PET=poly ethylene terephthalate

PP=polypropylene

PS=polystyrene

PUR dense=polyurethane

PUR foam =polyurethane

PVC =Poly Vinyl Chloride

n.d. not detected

The application of the dissolution method to different MSW and their derived products indicated the presence of high content of biogenic for Italian MSW, compost and for the SFR fraction (Table 14). The Finnish and German MSW biogenic content were lower such as the undersize fraction from SRF production.

**Table 14. Biogenic fractions content of different MSW and treated municipal solid waste**

Samples	Experimental Data (% dm)
Unsorted Italian MSW	55.95± 3.5†
Finnish waste (grain size < 80 mm) from pilot MBT-plant	24 ‡
Finnish waste pre-sieved (grain size < 80 mm) from MBT-waste pilot plant	23 ‡
German waste (grain size< 50 mm) from full scale MBT-plant.	23 ‡
SRF from mechanical separation	64.11±4.06†
SRF from mechanical biological treatment	50.09±10.07 †
Compost	61‡
Undersize fraction from SRF production	15.86±5.33 †

† Data from: DiProVe

‡ Data from: Laine-Ylijoki et al., 2004

MBT= mechanical biological treatment

SRF= solid refuse fuel

### **2.1.2. The Biological Stability**

The determination of the total content of biogenic/biomass in MSW is not sufficient to describe the potential biological impact of waste in landfill as no indication of the quality of this fraction is determined. Organic matter (OM) transformation in landfill is a subject not completely understood. Easily (e.g. sugar, aminoacid) and moderately degradable organic fractions (e.g. cellulose, hemicelluloses) are extensively degraded in a landfill (Barlaz et al., 1990; El-Fadel et al., 1997; Kylefors, 2002). On the other hand, more recalcitrant molecules (e.g. lignin, cutin, suberin) are preserved and transformed in humic-like matter (Filip and Küster, 1979; Barlaz and Ham, 1993; Christensen et al., 1994; Lechner et al., 2002; Ryan et al., 2006). Therefore the OM quality is a key concept in determining waste impact in landfills and it can be defined as the capacity of OM to be utilized by microbes as a source of energy, i.e. biodegradability.

Theoretically all biomass is biodegradable (European Committee For Standardization, 2006). The time-scale and the environmental conditions considered during biomass degradation influence the rate of degradation (Bogner and Matthews, 2003). Therefore, the biodegradability degree of a biomass is of difficult definition. One approach is to define both time-scale and conditions during which degradation occurs. For example in the case of a landfill, biomass is no longer degradable when it does not cause biological impacts or at least they are strongly reduced. When these conditions are assured, the biomass got the full biological stability.

The biological stability determines the extent to which readily biodegradable organic matter has decomposed (Lasaridi and Stentiford, 1998). It identifies the actual point reached in the decomposition process and represents a gradation on a recognized scale of values, which thus enable comparison of the process of decomposition (Lasaridi and Stentiford, 1998). Therefore knowing the degree of biological stability possessed by the organic matter

contained in wastes is important as it can be related to the potential impacts (e.g. odour generation, biomass re-heating, residual biogas production, re-growth of pathogens, phytotoxicity) (Müller et al., 1998; Science and engineering of composting, 1993; Adani et al., 2004a).

Many indicators and indexes have been proposed in the past to measure the biological stability of wastes (Komilis et al., 1999; El Fadel et al., 2002; Fricke et al., 2005; Robinson et al., 2005; Ryan et al., 2006). These methods can be classified in biological (e.g. anaerobic biogasification potential, respiration index and biological oxygen demand) (Adani et al., 2001b), and chemical (e.g. C/N, lignin/N and cellulose/lignin ratios, volatile solids) (Adani et al., 1998; Laine-Ylijoki et al., 2004; Ryan et al., 2006).

Biological methods allow a direct measurement of the biological stability of waste under standardized conditions (Barrena Gómez et al., 2006), giving more reliable results. On the other hand these methods can fail in the presence of conditions that inhibit the microbial activity (e.g. low pH, high content of volatile fatty acids and/or the presence of xenobiotic molecules or heavy metals) (ASTM, 1996; US Composting Council, 1997; UNI, 2006).

Chemical methods are always applicable but they are far from reproducing a biological process.

#### **2.1.2.1. Biological Stability: Chemical Methods**

The OM composition with regard to the macromolecular components appears to be important in determining OM degradability. For example, lignin, phenols and tannin, i.e. the recalcitrant fraction in the OM, are very important in determining the OM decomposition.

The ratio of chemical-labile *versus* chemical-recalcitrant compounds is generally used to describe OM quality (Six et al., 2002; Mikutta et al., 2006). Soil chemistry (Rovira and Vallejo, 2000) described OM degradability in soil by dividing the OM in a labile and a recalcitrant pool, depending on the solubility of the OM in strong acids. A similar approach can be used to describe the biodegradability and so the potential impacts of waste in landfills (Adani et al., 2001a). Anyway, chemical methods did not find wide application in waste characterization.

#### **2.1.2.2. Biological Stability: Biological Methods**

*Respirometric approach.* The most used and accepted methods to measure the biological stability are the respirometric methods (Iannotti et al., 1992; U.S. Composting Council, 1997). A detailed review of the different respirometric methods can be found in Barrena Gómez et al. (2006).

A simple classification of the respirometric methods should take into consideration:

- 1) respirometric activity measured as CO<sub>2</sub> produced during the biological activity (e.g. Naganawa et al., 1990);
- 2) Respirometric activity measured as O<sub>2</sub> consumed during the biological activity (Iannotti et al., 1992; Palestky and Young, 1995; Lasaridi and Stentiford, 1998; Adani et al., 2006); For respirometric purposes oxygen uptake is preferred (Lasaridi and Stentiford, 1996) and it has been proposed for adoption as the standard method (ASTM, 1992; U.S. Composting Council, 1997; CEN, 2006; UNI, 2006). The main disadvantage of this approach is that it needs more specific instrumentation and more skilled labour. Besides, equipment needs constant maintenance and frequent

calibration (Barrena Gómez et al., 2006). The main disadvantage of methods that measure CO<sub>2</sub> is that they are unable to distinguish between CO<sub>2</sub> produced aerobically from that produced anaerobically (Barrena Gómez et al., 2006). Moreover, correct measurement occurs only if the respiration quotient CO<sub>2</sub>/O<sub>2</sub> (mol/mol) is 1. This happens only for stabilized material (Adani et al., 2001b). Therefore, an underestimation of the respirometric activity occurred for material that was not stabilized. As a consequence, some authors argued that they cannot be used to estimate correctly the biological stability of a material (Lasaridi and Stentiford, 1998).

- 3) The absence of continuous aeration able to force O<sub>2</sub> into the biomass free air space, i.e. static methods (e.g. Binner and Zach, 1999, U.S. Composting Council methods, 1997).
- 4) The presence of continuous aeration able to force O<sub>2</sub> in the biomass free air space, i.e. dynamic methods (Palestky and Young, 1995; Adani et al., 2004a; Berthe et al., 2007). The absence of continuous aeration limited oxygen transfer through the biomass layers and into the bacterial cell wall, that is typically considered to be the rate-limiting step in fixed-film biological reactions (Palestky and Young, 1995). Therefore, when static methods are used, underestimation of oxygen up-take is possible, especially when fresh organic matter is present. These problems can be solved by continuous aeration of biomass (Adani et al., 2001b), i.e. dynamic method.
- 5) Pre-determined-temperature.
- 6) Self-heating temperature. Respirometric methods can be performed at pre-determined temperature. Different temperatures give different biological stability degrees, so that respirometric data referred to different temperatures cannot be compared. Typically mesophilic conditions (e.g. 20-25 °C) are considered (e.g. Sapromat test) (Govind et al., 1997). Iannotti et al. (1993) suggested 37°C to be a right compromise between mesophile and thermophile conditions suggested. ASTM (1996) 56°C to be optimal temperature.
- 7) Self-heating respirometric tests (e.g. Adani et al., 2001b; Berthe et al., 2007) are preferred by using apparatus designed in order to allow heat produced during aerobic reaction to be kept in the mass, miming the “core” of a biomass during an aerobic full scale process. Advantage in the self-heating consists in the fact that microorganisms decide temperature on the basis of the intensity of the microbial activity miming a real bio-oxidation process.
- 8) Solid state methods. Liquid state methods. Solid state assures conditions close to reality so that the data obtained reflect well what biomass is in terms of biological stability. Nevertheless, sometimes, the presence of toxic compounds as volatile fatty acids coming from temporary anaerobic conditions could inhibit microbial activity (Adani et al., 2006).

Liquid state methods (e.g. SOUR method) (Lasaridi and Stentiford, 1998), as it allows sample to be dilute in aqueous media and pH buffered, eliminates this problem. Moreover continuous biomass-stirring, combined with intermittent aeration allows O<sub>2</sub> to be dispersed in the suspension.

*Biogas production test.* Tests for the measurement of the potential biogas productions are performed in the laboratory to estimate the maximum amount of biogas producible by a waste

(Binner and Zach, 1999). The laboratory tests are classified in fermentative and incubation tests (Adani et al., 2001a). The fermentative tests are carrying out in solid condition in order to simulate the landfill state. These tests have the disadvantage of being subjected to toxicity phenomena above all when samples have low biological stability underestimating the potential biogas production.

Incubation tests are carried out under conditions that avoid toxicity phenomena, i.e. aquatic milieu at 35°C and inoculum adding (Adani et al. 2001a).

Biogas tests take a long time to be performed ranging from 21 to 100 days (Binner and Zach 1999; Adani et al., 2001a; Hansen et al., 2004).

*Leachate characterization.* The BOD (biological oxygen demand) and the COD (chemical oxygen demand) indirectly assess the amount of organic compounds in leachates and so their potential impact. The chemical oxygen demand is the measure of all oxydable compounds present in the leachate. The BOD indicates the amount of biodegradable organic matter present in the leachate. The BOD/COD ratio is a good indicator of the leachate quality.

### ***2.1.3. The Dipartimento Di Produzione Vegetale (DiProVe) Approach to Potential Impact Measurements***

In the last 10 years as a consequence of the DiProVe research activity, new methods have been developed to be applied to waste characterization with particular attention to the potential waste impact measurements. DiProVe approach consisted in the use of a Dynamic Respiration Index to measure the biological stability degree of wastes. Method set up is nowadays both a national (UNI, 2006) and a European Standard method (CEN, 2006).

More recently both anaerobic biogasification potential (ABP) and leachate characterization have been introduced to better describe the potentiality of impact of wastes.

In the following paragraphs these methods are presented and used to discuss some results about the effect of mechanical biological treatment (MBT) on the reduction of potential waste impacts.

#### **1. Dynamic Respiration Index (DRI)**

The Dynamic Respiration Index is measured using the method suggested by Adani et al., (2001b and 2004a), which involves a 20 l capacity adiabatic respirometric reactor (Costech International, Cernusco S.N., Italy; DiProVe, Milan, Italy). The respirometer is composed of an insulated container (the reactor), a control cabinet, an air supply system, a PC unit and a biofilter. A Clark-type temperature compensation electrode and differential-pressure electronic transmitter ensured both oxygen and airflow measurements every 10 s. The instantaneous data are then input into the software that calculated the DRI. Adani et al. (2001b) previously reported an extensive description of the scientific apparatus.

About 10 to 13 kg of wet sample are used for the test. The sample is optimized for moisture content (750 g kg<sup>-1</sup> of the maximum water retention) (Adani et al., 2001b). The oxygen uptake rate obtained under dynamic conditions, (i.e., DRI) is determined by measuring the difference in oxygen concentration (ml l<sup>-1</sup>) between the inlet and outlet air flow, the air having passed through the biomass, as well as by using knowledge of the absolute content of starting volatile solids (VS) (kg) in the biomass, the air-flow rate (l h<sup>-1</sup>), and the time (h) during which oxygen consumption was measured.

The hourly Dynamic Respiration Index ( $DRI_h$ ) is determined by measuring the difference in oxygen concentration ( $\text{ml l}^{-1}$ ) between the respirometer inlet and outlet air flow and calculated as reported by Adani et al. (2006):

$$DRI_h (\text{mg O}_2 \text{ kg}^{-1} \text{ VS}(\text{dm}) \text{ h}^{-1}) = Q \times \Delta O_2 \times Vg^{-1} \times 31.98 \times \text{VS}(\text{dm})^{-1} \quad (1)$$

where  $DRI_h$  is the hourly DRI,  $Q$  ( $\text{l h}^{-1}$ ) the airflow,  $\Delta O_2$  ( $\text{ml l}^{-1}$ ) the difference in oxygen concentration in the inlet and outlet air flow of the reactor,  $Vg$  ( $\text{l mole}^{-1}$ ) the volume occupied by one mole of gas at inlet air temperature,  $31.98$  ( $\text{g mole}^{-1}$ ) is the molecular weight of  $O_2$ , and  $VS$  or  $dm$  ( $\text{kg}$ ) the initial total volatile solids and dry matter content, respectively.

The Dynamic Respiration Index (DRI) is calculated as the average value of 24  $DRI_h$  taken over the 24-hour period characterized by the most intense biological activity (mobile mean). Time testing is a maximum of 4 days (Adani et al., 2004) and test-end was decreased when the 12 highest  $DRI_i$  values are registered (Adani et al., 2004a). Therefore, effective test length depended on waste stability degree and lag-phase.

Test is performed by setting an  $O_2$  concentration in the outlet airflow of  $140 \text{ ml l}^{-1}$  (Adani et al., 2001b). This value was maintained by a feedback control that automatically adapted airflow rate as a function of the oxygen concentration in the outlet airflow.

Respirometric activity and related biological stability degrees are resumed in Table 15.

## 2. Anaerobic Biogasification Potential

The anaerobic bigasification potential (ABP) method is determined according to the method of Adani et al. (2001b) with few modifications developed, according to Hansen et al. (2004). In 100-500 ml serum bottles, 0.62-3.1 g of dried sample was added to 37.5-187.5 ml of inoculum and 22-110 ml of de-ionized water. The batch tests were carried out with 60-300 ml samples (about 3.5%  $dm$ ) and 40-200 ml of headspace. The fresh feedstock and inoculum percentages of total solids were respectively 35% and 65%. Control blanks were prepared using 60-300 ml of inoculum.

All batches were sealed with Teflon hermetic caps, flushed with an  $N_2$  atmosphere and incubated at  $37 \pm 1^\circ\text{C}$ , until no further biogas production was detected (normally around 60 days). Assay bottles were periodically analyzed for both quantitative and qualitative determination of biogas production. Quantitative biogas production was estimated by withdrawing extra-pressure gas with a 60-300 ml syringe. Biogas production of blank control batches was subtracted from biogas production of every sample. Qualitative characterization of biogas was performed by a Gas-chromatograph (Carlo Erba Megaserie 5300, capillary column 25 m x 0.32 mm diameter and flam ionization detector FID) to determine  $CH_4:CO_2$  ratio in the biogas.

**Table 15. Biological stability identification by DRI, COD, BOD<sub>5</sub> and ABP values**

Biological stability level	DRI ( $\text{mg O}_2 \text{ kg dm}^{-1} \text{ h}^{-1}$ )	COD ( $\text{mg O}_2 \text{ L}^{-1}$ )	BOD <sub>5</sub> ( $\text{mg O}_2 \text{ L}^{-1}$ )	ABP ( $\text{L kg dm}^{-1}$ )
High	<200-300	<1500	<500	<50
Medium	500-700	2000	1000	100
Low	>1000	>3000	>1200-1500	>100

ABP and related biological stability degrees are reported in Table 15.

### **3. Leachate Characterization: BOD<sub>5</sub> and COD**

Leachate characterization refers to the determination of both chemical (COD) and biological oxygen (BOD<sub>5</sub>) demands of water-extract using standardized procedure (Daskalopoulos et al., 1998; El Fadel et al., 2002; Robinson et al., 2005).

The BOD<sub>5</sub> and COD (APHA, 1992) are determined using waste elutes obtained by water extraction of waste (1:10 ratio solid liquid) that is shaken for 24 hours in distilled water (EN, 2002). BOD<sub>5</sub> and COD values and corresponding biological stability degrees are reported in Table 15.

## **2.2. Mechanical Biological Treatment and Sustainable Landfill**

### ***2.2.1. Mechanical-Biological Treatment: The Definition***

The Mechanical Biological Treatment (MBT) refers to a process consisting in the mechanical fractionation of MSW into a wet (mostly organic) and a dry fraction (mostly plastic and paper) and in the successive biological treatment of the wet fraction (Wiemer and Kern, 1995; Adani et al., 2000; Calcaterra et al., 2000). In the last 15 years these kinds of processes found a wide diffusion in Europe (e.g. Austria, Germany and Italy).

The biologically treated wet fraction is known as biostabilized organic or compost-like fraction (Wiemer and Kern, 1995) and its fate is the landfill or its usage as organic amendment for particular uses (e.g. landfill and land restoration). On the other hand the dry fraction is landfilled or used to produce solid refuse fuel (SRF) (Leikam and Stegman, 2004). More recently mechanical biological processes assumed different forms and have been classified on the basis of the presence or absence of physical separation of waste into different streams before the biological treatment. Therefore it was defined the mechanical-biological treatment with separate fluxes (MBTSF) (Adani et al., 2000) and the mechanical-biological treatment with unique flux (MBTUF) (Calcaterra et al., 2000).

The MBTSF (Figure 6) consists in a mechanical pre-treatment of the waste (sieving) giving a wet ( $\emptyset < 50$ -80 mm; OM  $> 80$  % w/w; moisture  $> 50$  % v/v) and a dry ( $\emptyset > 50$ -80 mm; OM  $< 25$  %; moisture  $< 25$ -28 %) fraction, and the successive biological treatment of the wet fraction. This treatment went on for several weeks using an aerobic biological treatment in order to reduce the content of putrescible organic fraction leading to a high degree of biological stability (biostabilized product). The length of the biological treatment can vary, so that a biostabilized product shows different degree of biological stability. For example, in Austria and Germany the biostabilization process takes place for 2-6 months. On the other hand, in Italy, the biostabilization process takes place, generally, only for 2-4 weeks (Adani et al., 2001c). Italian choice came from the fact that the degradation process is described by a first order kinetic and processes longer than 4 weeks do not a significant increase in the degree of biological stability (Adani et al, 2000). In any case, an optional curing phase can be considered as an extra time in order to produce a compost-like material.

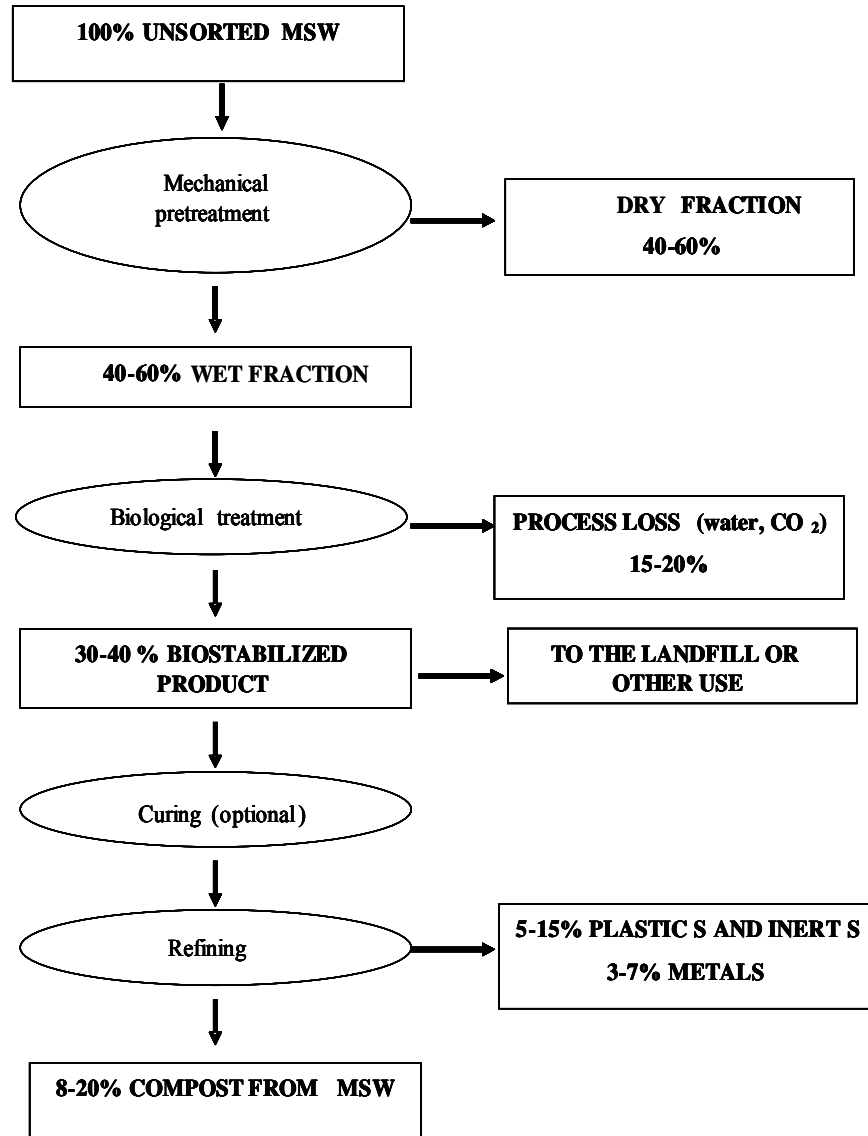


Figure 6. MBTSF mass balance scheme.

In the MBTUF process (Figure 7) the entire waste after a bland mechanical treatment used to reduce waste size, is processed by a biological process. Depending on the process management (air, temperature and moisture management) fast and long biological treatments can be indicated (Adani et al., 2002). A fast biological process gives a well-dried material with a high calorific value and a low degree of biological stability (Adani et al., 2002). In this case, the fate of the biodried material is the production, after refining, of SRF. Long biological process gives a product characterized by low calorific value (depending on the process) and high biological stability (Adani et al., 2002; Forte et al., 2003).



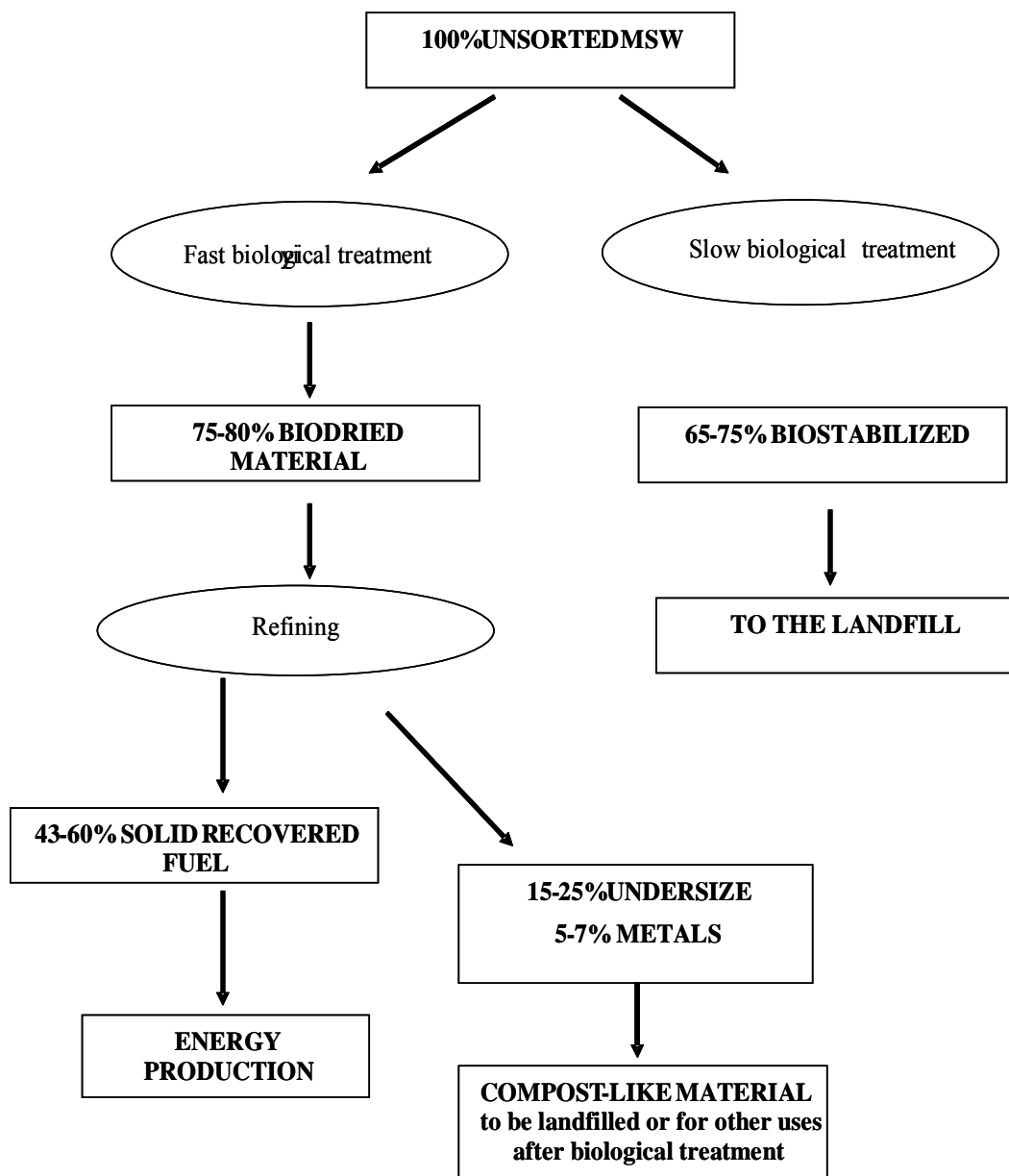


Figure 7. MBTUF mass balance scheme.

A particular application of the MBTU is to perform the process directly in landfill sites allowing a low-cost biostabilization technology. This option of pre-treatment leads to reducing the plant cost and preserving high process performance (Adani et al., 2000; Forte et al., 2003) (Figure 8). Moreover, the modularity of system and the low cost favoured their application in developing and emerging countries.

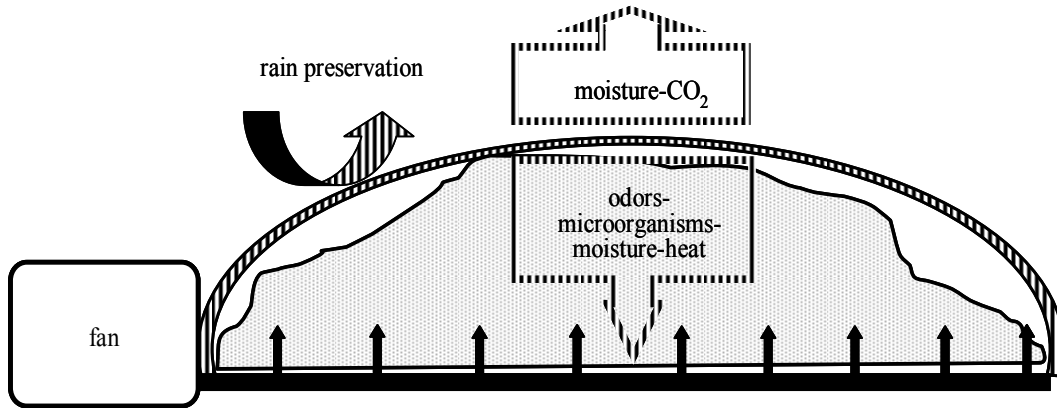


Figure 8. Particular of the MBTUF technology performed in landfill.

**Table 16. Differences observed for the biological section of MBTUF and MBTUF processes**

	MSW treated (initial amount %)	Area required (m <sup>-2</sup> Mg <sup>-1</sup> <sub>MSW</sub> )	Process Time (weeks)	Biostabilized products (% initial MSW)	Biostabilized products: fate
MBTSF	40-60	0.76	4	30-40	Landfill or particular employments
MBTUF	100	0.5†	2†	65-75†	Landfill and/or SFR

† referred to low biological process.

### 2.2.2. Differences between Biostabilization Processes

The main differences between MBTSF and MBTUF refer to the biological treatment section (Table 16) (Figure 6 and 7). The MBTSF should be preferred to produce from the wet fraction, a compost-like material to be used for landfill capping or for agricultural-like purpose (e.g. landfill or degraded land restoration).

On the other hand, MBTUF should be preferred to produce a biostabilized material whose fate is the landfill (low biological process) (Forte et al., 2003) or a biodried material (fast biological process) for SRF production (Calcaterra et al., 2000) (Figure 7).

### 2.2.3. Output Fraction of MBT and Possible Usage

As stated in previous chapters, a new view of MBT should consider a variety of products with different characteristics and usages, such as the following:

- biostabilized/compost-like material from MBTSF processes to be landfilled or used as compost-like material;
- SRF from MBTSF process, to be used to produce energy;
- biostabilized material from MBTUF to be landfilled;

- biodried SRF from MBTUF to be used to produce energy;
- compost-like material from MBTUF to be landfilled or used as compost.

Bio-products, such as biostabilized/compost-like material can be used for landfill capping and/or agriculture-like use because of their characteristics. The relative high content of heavy metals suggests the use of these products *una tantum* for particular purpose such as the remediation of a contaminated area or capping of an old landfill (landfill closure). In this case these materials act as organic amendments to improve the soil fertility. In any case, the use of these products should be discouraged preferring high quality composts obtained from the organic fraction from separate collection.

Alternatively, biostabilized material can be disposed in a landfill such as, also, biostabilized from MBTUF. In this case, product quality refers to the waste potential impact, i.e. biological stability. In this way, an oxygen uptake rate, measured by the Dynamic Respiration Index (DRI) (Adani et al., 2004b) of 300-500 mg O<sub>2</sub> kg dm<sup>-1</sup> h<sup>-1</sup> indicates a good biological stability allowing the reduction of more than 60-80 % of the residual biogas production, COD, BOD<sub>5</sub> and odour impacts (Adani et al., 2003) (Table 17-18).

Solid recovered fuels are other important products of MBT.

As previously reported MBTUF gives SRF by mechanical separation. As a consequence of that this fraction does not undergo the stabilization process. In this case, SRF could be biologically unstable and, if stocked for a long time, it could produce odours and leachate or self-burning because its moisture content (22-28 % wet weight- w.w.) is enough to assure microbial activity. On the other hand, SRF from MBTUF can be easily stored as it typically presents lower moisture content (10-15 %) than MBTUF and being biologically dried (Figure 4). In both cases, primary SRFs can be refined in order to remove inorganic fraction (metals and inert material) to increase calorific power and products homogeneity. This operation gives a higher quality SRF, that should assure high standard quality (e.g. Italian law) (EU TC/343: Solid Recovered Fuel standardization) (Table 16).

#### ***2.2.4. The Effect of MBT Treatment on the Reduction of Landfill Impacts Determined by Biostabilization Process***

The biostabilization processes are effectiveness in the waste management to reduce the landfill potential impacts. The organic matter degradation that occurs during biological treatment determines an elevated reduction of MSW putrescibility getting a high biological stability (Adani et al., 2002; Forte et al., 2003) (Table 17).

The Dipartimento di Produzione Vegetale of the University of Milan (Italy) (DiProVe) has produced in the last 10 years a large amount of data on this subject. In Table 18 some unpublished data regarding the measurement of the potential waste impact reduction during full-scale biological processes, measured by DRI and, BOD<sub>5</sub> and COD determined on water extracts, are reported.

Processes studied refer to both MBTUF and MBTUF processes performed in full-scale plants for 2-4 weeks. Data reported indicated an effective reduction of the potential impacts measured by parameters indicated. This reduction was of 66.70% for DRI, 17.54 % for COD, 50.19% for BOD<sub>5</sub> as average for MBTUF and of 73.91% for DRI, 54.99% for COD and 64.18% for BOD<sub>5</sub> as average for MBTUF (Table 18).

**Table 17. Potential waste impact measured of untreated and treated waste: literature data**

Treatment period	Respiration activity <sup>†</sup> (mg O <sub>2</sub> g dm <sup>-1</sup> )	BOD <sub>5</sub> (mg l <sup>-1</sup> )	COD	Biogas production tests <sup>‡</sup> (Nl kg <sup>-1</sup> dm)	
0§	32	6700	10500	-	34 269
5 weeks§	28	800	6100	-	90 162
9 weeks§	24	490	2700	-	141 153
20 weeks§	2.5	83	1500	-	16 19
0#	16.7	2711±637	8508±1617	-	- -
8 weeks <sup>#</sup>	-	529±36	2437±40	-	- -
20 weeks <sup>#</sup>	9.3	469±57	2642±231	-	- -
0 <sup>††</sup>	82.9	-	-	209.9	- -
4 weeks <sup>††</sup>	58.7/57.6	-	-	-	- -
8 weeks <sup>††</sup>	35.3	-	-	-	- -
16 weeks <sup>††</sup>	20.5	-	-	-	- -
21 weeks <sup>††</sup>	-	-	-	135.6	- -
25 weeks <sup>††</sup>	16	-	-	-	- -
3 months <sup>‡‡</sup>	13.1	422	1944	-	- -
6 months <sup>‡‡</sup>	6.9	6	561	-	- -
12 months <sup>‡‡</sup>	1.7	12	243	-	- -

<sup>†</sup> respiration activity: AT<sub>4</sub> test

<sup>‡</sup>Biogas production tests: GB<sub>21</sub>, GS<sub>90</sub>, GS<sub>240</sub>

<sup>§</sup>Data from: Zach et al., 2000 (MBTSF process)

<sup>#</sup>Data from: Trois et al., 2007 (MBTUF process)

<sup>††</sup>data from: Lornage et al., 2007 (MBTUF process)

<sup>‡‡</sup> data from: Münnich et al., 2006 (MBTUF process)

More available data (Table 19) allow an extensive comparison between untreated MSW or OFMSW and corresponding biological treated products, including the measurement of anaerobic biogasification potential (ABP).

Results indicated that short biological treatment if performed under optimal conditions, allow an effective reduction of potential impacts so that they can contribute to the realization of a sustainable landfill.

**Table 18. Dynamic respiration index (DRI), BOD<sub>5</sub>, and COD measured on water extract, determined for MSW and corresponding biological treated products: DiProVe data**

Waste typology	Processes number	DRI (mgO <sub>2</sub> kg dm <sup>-1</sup> h <sup>-1</sup> )	COD (mg O <sub>2</sub> l <sup>-1</sup> )	BOD <sub>5</sub> (mg O <sub>2</sub> l <sup>-1</sup> )
MSW <sup>†</sup>	6	1,649±265	2,622±1,186	1,881±962
Biostabilized MSW <sup>†</sup>		549±86	2,162±459	937±372
% reduction		66.70 <sup>‡</sup>	17.54 <sup>‡</sup>	50.19 <sup>‡</sup>

Waste typology	Processes number	DRI (mgO <sub>2</sub> kg dm <sup>-1</sup> h <sup>-1</sup> )	COD (mg O <sub>2</sub> l <sup>-1</sup> )	BOD <sub>5</sub> (mg O <sub>2</sub> l <sup>-1</sup> )
OFMSW§	4	1,675±1,106	4,777±274	3,258±1,090
Biostabilized OFMSW§		437±158	2,150±934	1,167±553
% reduction		73.91‡	54.99‡	64.18‡

†MBTFS processes

‡ Percent reduction

§MBTUF processes

**Table 19. Dynamic Respiration Index (DRI), BOD<sub>5</sub>, COD and anaerobic biogasification potential (ABP) measured on MSW and derived products (DiProVe data)**

Waste typology	DRI (mg O <sub>2</sub> kg dm <sup>-1</sup> h <sup>-1</sup> )	COD (mg O <sub>2</sub> l <sup>-1</sup> )	BOD <sub>5</sub> (mg O <sub>2</sub> l <sup>-1</sup> )	ABP (l kg dm <sup>-1</sup> )
MSW†	1,200±210 n=21	3,049±1,342 n=14	1,833±770 n=14	203±58 n=12
Biostabilized MSW†	377±215 n=17	1,950±710 n=12	828±365 n=12	90±15 n=10
% reduction	68.58‡	36.04‡	54.82‡	55.66‡
OFMSW§	1,675±1,110 n=38	6,636±2,026 n=22	3,756±1,656 n=22	308±71 n=12
Biostabilized OFMSW§	300±216 n=43	2,175±880 n=28	694±403 n=28	108±24 n=10
% reduction	82.08‡	67.22‡	81.52‡	64.93‡

†MBTFS processes

‡ Percent reduction

§MBTUF processes

### 3. CONCLUSION:

#### MECHANICAL BIOLOGICAL TREATMENT AND BIOLOGICAL IMPACTS REDUCTION IN LANDFILL

Mechanical biological treatments are effective in the reduction of impacts as the previous paragraph showed. Biological and chemical parameters developed, i.e. DRI, ABP, COD and BOD<sub>5</sub> are useful to measure the potential impact of waste, but they are not a direct measurement of the effective reduction of impact in landfills.

Unfortunately, at this time, not much data about the impact of biologically treated waste in a landfill are available. Many studies were performed on laboratory-scale in order to measure the biogas and leachate productions of pre-treated or untreated MSW, under different landfill simulating conditions.

Pre-treated wastes showed a strong biogas reduction (53% to 92%) with respect to untreated MSW (Table 20) (Bockereis and Steinberg, 2005; Raninger et al., 1999; Bidlingmaier et al., 1998; Stegmann et al., 1998). Generally the results obtained established, also, for pre-treated MSW a rapid methanogenic phase (Raninger et al., 1999). This because the aerobic pre-treatment degraded the readily degradable OM reducing the production of the volatile fatty acid (VFA) that at high concentrations inhibit the methane production (Vavilin et al., 2006) (Figure 9).

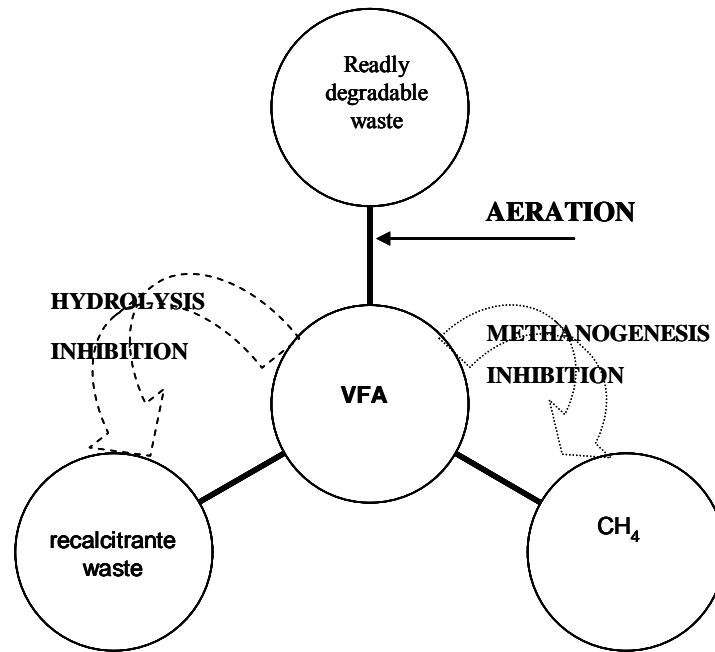


Figure 9. Scheme of the inhibition of the microbial processes caused by VFA concentration (Vavilin et al., 2006, modified).

**Table 20. Biogas reduction after one year of observation of biological treated waste, allocated in a simulated landfill**

	Time test (days)	Nm <sup>3</sup> /Mg	Biogas Reduction (%)§
MSW†		51	-
MSW pre-treated 1†		24	52.94
MSW pre-treated 2†	2500	13	80.39
MSW pre-treated 3†		8	84.31
MSW pre-treated 4†		5	90.19
MSW pre-treated 5†		4	92.15
MSW pre-treated 7 ‡	500	10	77

†Data from: Bockereis and Steinberg, 2005

‡Data from: Raninger et al., (1999)

§ biogas production percent reduction between MSW and MSW pretreated

**Table 21. BOD<sub>5</sub> and COD reduction after one year of biological treated waste allocated in a simulated landfill**

Leachate indicators	Time (days)	Starting point	After first year	reduction †
BOD <sub>5</sub> ‡ (mg/L)	365	748	522.3	30.17
COD (mg/L)	365	11,427	9,984	12.63
TOC (mg/L)	365	3,755	2,437	35.10
NH <sub>4</sub> (mgN/kg TS)	365	2,217	2,212	0.23

†100\*(value at the start-value after 365 days)/ value at the start.

‡Data from El-Fadel et al., 2002.

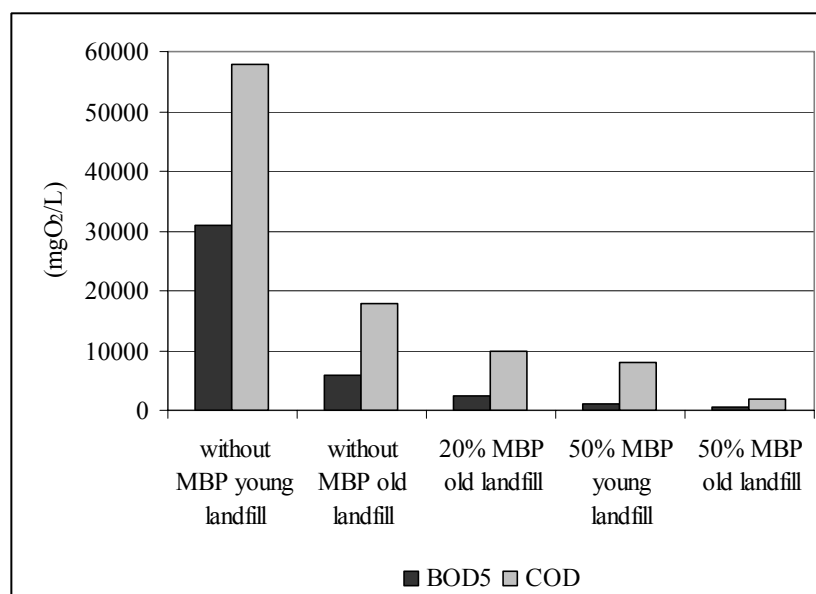


Figure 10. BOD<sub>5</sub> and COD values of MSW pre-treated disposed in landfill considering the landfill age (Binner, 2003; Lechner et al., 2001) (MBP = MBT).

The MBT treatment allowed the leachate pollution to be also reduced lowering waste impacts (Raninger et al., 1998). Simulated landfill experiment (volume of the simulated landfill of 75 m<sup>3</sup>) showed after one year, only a slight reduction of the leachate pollution (COD and BOD<sub>5</sub>) for biological treated waste in simulated landfill (Table 21).

When leachate from untreated waste were compared with leachate from MBT treated waste, a strong reduction of impact was observed. Pre-treated MBT waste impact was comparable with that of an old landfill (Figure 10).

All these simulated or full-scale experiences, suggested for MBT pre-treated waste, a low waste impact. The rate of reduction of impacts were very similar to those measured by the use of methods for the biological stability and leachate determination applied on untreated and treated wastes. In conclusion, mechanical biological treatments could be a way to reduce

waste impact in landfills. Analytical methods proposed to measure the degree of biological stability are effective on the determination of the potential impact of waste and so of the potential impact reduction after waste treatment. All this means a rapid degradation of residual organic matter in a landfill that allows the landfill to proceed in the shortest time to an “inert state,” of the landfill, i.e. a sustainable landfill.

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