2 Waste Disposal: What are the Impacts?

Contributions by Samuel Stucki, Jörg Wochele, Christian Ludwig, Helmut Brandl, and Zhao Youcai

Take 500 g of wet biomass, add 1200g of combustibles (paper, plastics, wood and textiles) and some 140g of minerals, season with some salts and top with 170 g of metals and you get the mix of roughly 2 kg of municipal solid waste (MSW) that an average American produces per day. The recipe is different in different parts of the world: If you reduce the amount of combustibles and metals in the American waste by roughly a factor of 10 you will end up with the composition of MSW in a Chinese City (section 3.5). MSW contents and quantities are a mirror of the material turnover of a society and reflect the consumption habits of the population. There is a clear correlation between the Gross Domestic Product of a country (GDP) and the amounts of waste it produces (section 2.1). Waste management in China and in particular in Shanghai faces the problems that are typical of an economy in transition (section 2.3). The development in Shanghai confronts the local authorities with the difficult task to adapt existing and to invest into new infrastructure in order to cope with the rapidly changing quantities and qualities of MSW.

Common to MSW of any origin is that it contains high proportions of organic compounds that are more or less easily bio-degradable. Normal practice throughout the world is to pile up the above cocktail of wastes in more or less organized landfills, or to just dump it wherever suitable. A MSW landfill is an uncontrolled bio-chemical reactor. The number of chemicals found to be released by landfills to the atmosphere and/or to the hydrosphere is huge (section 2.3). The ecological consequences of these emissions have local as well as global character. Emissions of polluted water from landfills to soil, surface and ground water are local, but can persist for centuries. With the potent greenhouse gas methane as the main component, gaseous emissions from landfills have a strong impact on a global scale. Although modern landfills attempt to collect, clean and use the methane resulting from anaerobic fermentation as a fuel, in most cases it is released to the atmosphere. Of the total global emissions of methane, estimated in 1999 at 535 million tons annually, 375 million tons are the immediate result of human activities, and 18% of those come from waste disposal. Methane emissions from landfills can be avoided if MSW is incinerated.

Figure 2.2 in section 2.1 shows that the composition of MSW deviates considerably from the composition of the geological formations it is discharged to ("average earth crust") for a number of elements. Next to carbon, chlorine and sulfur, associated mainly with food and vegetable wastes, the heavy metals, notably Zn, Cu, Cd, Pb, differ by one to two orders of magnitude from background. The release of these materials to the environment cannot be prevented unless they are efficiently separated and recycled (chapters 3 to 5). Even prohibition of certain toxic substances will have only an effect on the MSW composition on the mid- to long-term scale, because society has built up huge reservoirs of toxic objects, which will eventually become waste.

2.1. The Diversity of Municipal Solid Waste (MSW)

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Municipal solid waste (MSW) includes the solids discarded by the end consumers, i.e. private households, small business and public areas, and typically collected by public authorities for disposal. Normally, separately collected waste for recycling, such as paper, metals, aluminium, glass, etc. is included in the MSW quantities given. MSW refers specifically to that part of MSW which is sent to landfill, incineration, or other final treatment [6]. MSW is only a relatively small fraction of all the solid waste that is generated in an advanced economy. According to the OECD Environmental Outlook [5] the total solid waste generated in the OECD countries reached 4 billion tons in 1997, of which 14% were classified as MSW. Table 2.1 shows that the major sources of solid wastes are in primary production (agriculture, forestry and mining) and in manufacturing.

Most of the waste streams generated in primary production are dealt with locally on the site where they are generated (e.g. agricultural and forestry wastes are generally used as fertilizers or as fuels, most other production waste is being disposed of locally or recycled).

Table 2.1. Percent share of solid waste in OECD countries; Total amount: 4 billion tons in 1997

Manufacturing	25
Agriculture & Forestry	21
Mining & Quarrying	14
MSW	14
Construction & Demolition	14
Energy Production	4
Water Purification	2
Others	6

The present book is dealing with post-consumer waste, its prevention, treatment and disposal, i.e. specifically with the 14% of MSW and some of the wastes arising from building sites (Construction and Demolition).

2.1.1. Quantities of MSW Collected

MSW production in developed economies has grown continually, very much in line with economic growth. MSW production has increased by 40% between 1980 and 2000, matching very nearly the increase in Gross Domestic Product (GDP) (50 %) over the same time span [5], and illustrating the fact that so far the increase in prosperity has been linked with an increase in material throughput. Annual MSW production has reached an average of 500 kg/cap in OECD countries. Figure 2.1 shows that there are marked differences in the specific per capita waste production of different countries. The correlation with GDP in the same countries confirms the strong link between affluence and MSW quantities. A similar correlation is also seen in a comparison of waste quantities with GDP for different cities in China (section 3.5.1).

The collection and assessment of MSW data in developing countries is much more difficult, as rural areas of these regions are hardly connected to an organized waste management infrastructure. Even in the big cities of the developing world, especially in Asia, only a fraction of the population is connected to regular waste collection services. Much of the waste there is dealt with informally, i.e. it is dumped in an uncontrolled way, and/or recycled very efficiently by scavengers and waste pickers. Table 2.2, taken from data published in World Resources 1996-97, shows the MSW generation for a number of Indian cities, together with an estimated collection efficiency (% waste collected). A detailed analysis of the evolution of waste quantities and compositions in China, and in particular in the booming city of Shanghai is given in section 2.3 of this book.

As mentioned above, the amounts of waste are expected to rise further with increased economic development and very likely a near 1:1 correlation of the increases of MSW and GDP has to be expected. In fact, a growth of 50% MSW production is expected in the period 1995-2020 for OECD countries, and of 100% in non OECD countries. Some of this growth is expected to be offset by more efficient recycling [5].

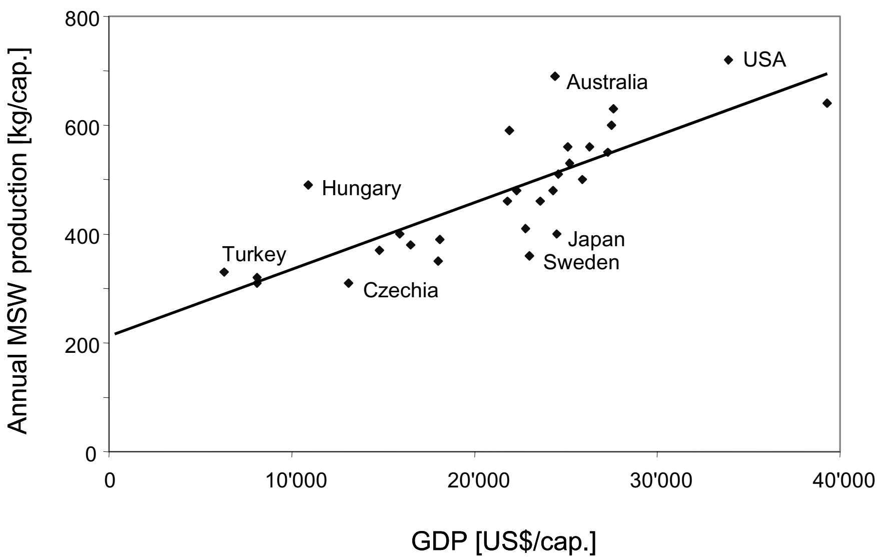

Fig. 2.1. Production of per capita MSW in OECD countries as a function of per capita Gross Domestic Product (GDP) [4].

Table 2.2. Waste collected in selected cities in India [35]

	MSW [kg/cap]	% collected
Bombay (India)	200	90
Delhi (India)	440	77
Bhiwandi (India)	100	40

2.1.2. The Composition of MSW

The composition of MSW is a mirror of consumption patterns, eating habits, social structure etc. of the societies producing the waste. Whereas in low income areas the main components of MSW are readily bio-degradable (food waste), this fraction is strongly reduced in highly developed cities. In section 2.3 it is shown that, in some of the less developed Chinese cities, food waste and coal ashes account for over 80% of MSW, whereas a significant increase of plastics and paper is seen in the cities of Shanghai and Beijing, which are rapidly developing towards western consumption standards. This trend is also reflected by the data given for the USA (Table 2.3). The shares of paper and especially of plastics increased over the period of 30 years between 1960 and 1990 in the USA, while, over the same period, the percentage od food waste declined drastically, although, in absolute numbers, it stayed constant.

Table 2.3. Development of average composition of MSW in USA (1960 to 1990). In million tons per year [19], and in %.

	Paper	Glass	Metals	Plastics, Rubber, Leather	Textiles	Wood	Food Waste	Garden waste	Total MSW
1960	29.90	6.70	10.50	2.40	1.70	3.00	12.20	20.00	86.4
1970	44.20	12.70	14.10	6.30	2.00	4.00	12.80	23.20	119.3
1980	54.70	15.00	14.50	11.20	2.60	6.70	13.20	27.50	145.4
1990	73.30	13.20	16.20	20.80	5.60	12.30	13.20	35.00	189.6
	in %								
1960	34.61	7.75	12.15	2.78	1.97	3.47	14.12	23.15	100
1970	37.05	10.65	11.82	5.28	1.68	3.35	10.73	19.45	100
1980	37.62	10.32	9.97	7.70	1.79	4.61	9.08	18.91	100
1990	38.66	6.96	8.54	10.97	2.95	6.49	6.96	18.46	100

Table 2.4. Composition of MSW to incineration (separately collected fractions not included) in Switzerland (1992/93) [11]

	[%]
Paper and cardboard	28
Vegetable matter	23
Plastics	14
Mineral materials	6
Wood, leather, bones,…	5
Composite materials	8
Composite packaging	3
Metals	3
Glass	3
Textiles	3
Fines (< 8 mm)	4

Table 2.4 shows the average MSW compositions as sent to incineration in Switzerland [11]. The numbers do not include separately collected recyclable fractions such as paper, glass, metals etc. for which a separate collection system has been introduced. The average composition of MSW delivered to incineration, of incinerator ashes, and of the earth crust are given in Table 2.5. The data for the elemental composition of waste is taken from a representative study carried out in Switzerland in1993 [8].

The composition is typical for waste from an affluent society. Figure 2.2 shows the relative concentrations of elements, normalized with the average earth crust composition as reference. It is striking to see that significant deviations of concentrations in MSW from average background concentrations exist for C, Cl and S and for heavy metals, notably Pb, Zn, Cu, Cd, Hg.

Table 2.5. Average composition of average dry MSW, incinerator bottom ash, air pollution control (APC) residues [8] and average earth crust (mg/kg) [28]

	MSW (CH) (Belevi)	Bottom ash (Belevi)	APC residues (Belevi)	Earth crust (Reimann)
C	334000	20000		200
H	40000			
O	257000			
N	3120			
S	1120	2000	30000	260
Cl	6870	4000	259000	130
P	890	3000	3600	1100
Si	48500	190000	53000	280000
Fe	30000	100000	8000	56000
Ca	14000	120000	184000	41000
Al	12400	50000	48000	82000
Na	5140	25000	40000	24000
Mg	3380	16000	8200	23000
K	2060	10000	29000	21000
Zn	1310	3000	22000	70
Pb	500	1600	6900	13
Cu	1200	2000	690	60
Cd	12	7	360	0.20
Hg	2	0.04	130	0.08
Cr	315	1100	760	100
Ni	107	190	50	80
Co	2	16	12	

A large fraction of MSW is actually organic material and water, which will eventually disappear, be it by fast mineralization in an incinerator, be it by s low mineralization in a landfill. The inorganic fractions, notably the ashes left over after incineration, contain concentrations of heavy metals which exceed background concentrations by two to three orders of magnitude. Clearly, the deposition of untreated MSW in a landfill causes the enrichment of potentially harmful substances in the landfill, which will eventually lead to emissions to water, soil or air. The emissions resulting from the deposition of untreated MSW are described in the following section.

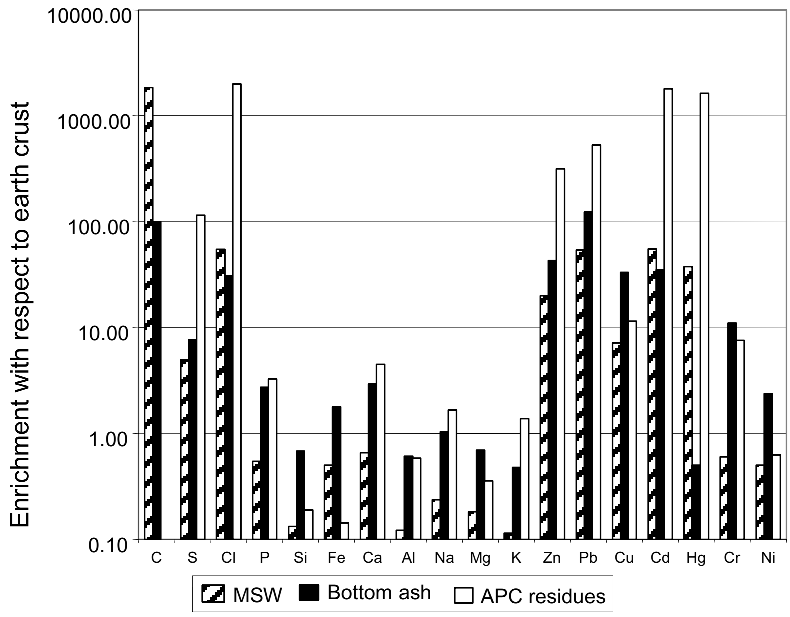

Fig. 2.2. Enrichment factors of chemical elements in MSW, incinerator bottom ash and air polution control (APC) residues, relative to the average composition of the earth crust, drawn from data in Table 2.5. Factor 1 = no enrichment.

2.2 Emissions from Municipal Solid Waste Landfills

Helmut Brandl

2.2.1 Emission to the Atmosphere

In principal, municipal solid waste (MSW) landfills represent anoxic environments with oxic conditions and aerobic processes occurring only at the surface [17]. MSW contains usually a large fraction of organic matter which can be metabolized by microorganisms. During biological mineralization of the organic material, electrons originating from the degradation have to be transferred to a terminal electron acceptor. Generally, electron acceptors have to be present in a sufficiently high concentration and thermodynamically favorable in such a way as to enable the microorganisms to gain energy for growth from the process. The development of a landfill system from an oxidizing to a reducing state is characterized by a typical series of electron acceptors which are microbially utilized in a typical sequential order (redox sequence): oxygen, nitrate, manganese, iron, sulfate, and carbon dioxide [31]. Under oxic conditions, oxygen offers the highest energy yield. After its depletion, anoxic conditions start to prevail and electron acceptors with lower energy yields are utilized. Additionally, a wide variety of fermentative microorganisms are active using organic metabolic intermediates as electron acceptors. Consequently, microbial communities and their activities are subject to gradual changes during the degradation of organic matter.

A general "classical" pattern of the degradation of organic matter is presented in Figure 2.3. Organic material is degraded in the presence of oxygen by fungi and heterotrophic bacteria resulting in the formation of simple organic compounds. As example, glucose is formed by the degradation of cellulose. These compounds can be further utilized under oxic conditions by heterotrophic microorganisms and carbon dioxide is formed as a final product, entering a "carbon dioxide pool" which is supplied by other metabolic reactions as well. In the absence of oxygen, denitrifying microorganisms can metabolize carbon compounds to form carbon dioxide and reduced nitrogen compounds such as dinitrogen or ammonium. In the absence of nitrate as electron acceptor, fermentation processes lead generally to the formation of acetate, other volatile fatty acids (such as e.g. propionate or butyrate), simple carbon compounds, carbon dioxide, and hydrogen. Although hydrogen is a main product in anaerobic mineralization, it can only be detected in significant amounts in anoxic systems in rare cases. It is only possible to detect high levels of hydrogen in a phase where fermentative microorganisms and processes are dominant. Hydrogen can be used as electron donor by a series of microorganisms (nitrate, iron, and sulfate reducers as well as autotrophic methanogens).

Generally, organic material is only completely degraded below a certain partial pressure of hydrogen due to thermodynamic reasons. High hydrogen concentrations can inhibit fermentation reactions (product inhibition). The inhibition can only be overcome by the activity of hydrogen scavenging microorganisms. In natural ecosystems (sediments, sewage, sludge, rumen) sulfate reducing and methanogenic bacteria keep the hydrogen partial pressure at values of <10 Pa ($<10^{-4}$ atm). Partial pressures of 10 to 100 Pa inhibit fermentative reactions which can be confirmed either by thermodynamic calculations or by experimental approaches [18]. However, in some cases the formation of microbial flocs or biofilms allows the functioning of fermentative reactions even at elevated hydrogen concentrations. Flocs and biofilms can reduce gas diffusion through these structures.

Volatile fatty acids (VFA) can be utilized by iron and manganese reducing bacteria. Again, carbon dioxide is formed as final product along with reduced iron and manganese. Sulfate is reduced by sulfate reducing bacteria (using e.g. acetate as carbon source) leading to the formation of hydrogen sulfide whereas in the absence of sulfate VFAs are converted to acetate by acetogenic bacteria. Acetate is used by acetoclastic methanogenic microorganisms to form methane (and also carbon dioxide) as product of the anaerobic mineralization. Additionally, methane can be formed by autotrophic methanogens using carbon dioxide from the "pool" and hydrogen from fermentation processes. As a consequence, methane and carbon dioxide are the main terminal products resulting from the degradation of organic matter in an ecosystem.

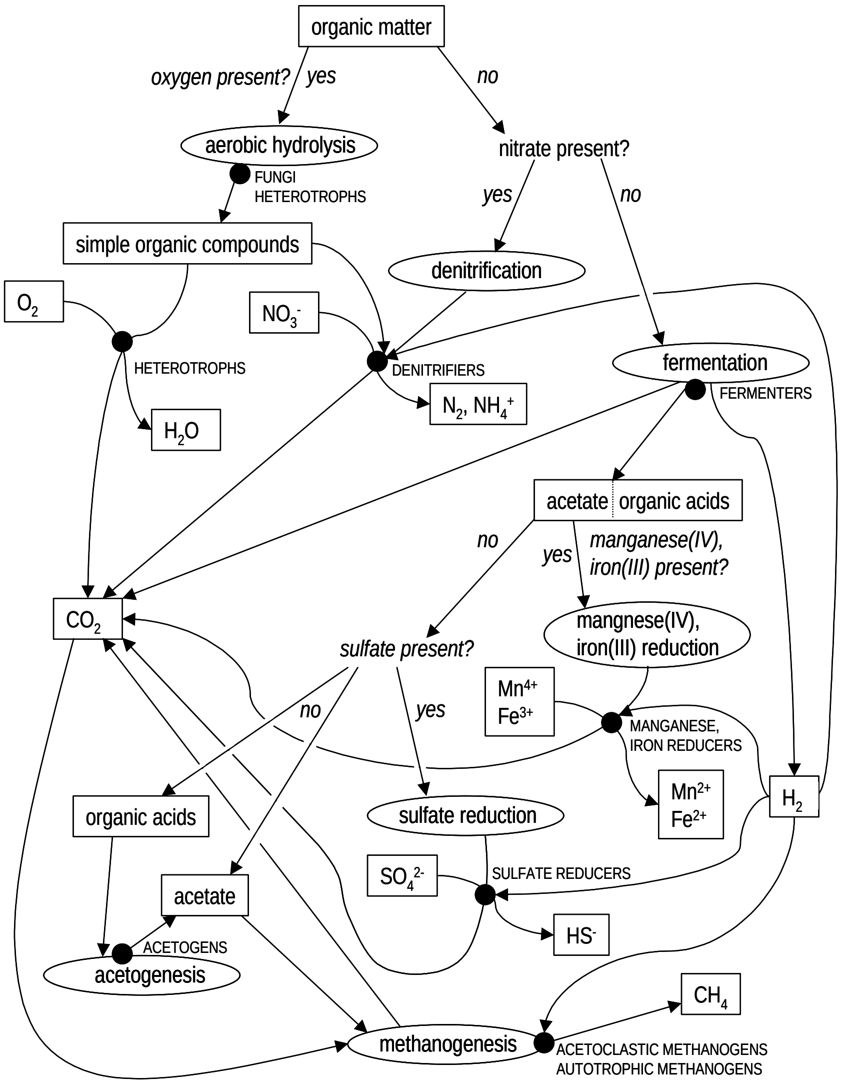

Fig.2.3. Schematic view of microbial degradation of organic matter in a landfill (adapted from Humphreys et al. [22]). Texts in boxes represent specific compounds (educts or products of the microbial metabolism). Texts in ellipses represent microbial processes. Solid symbols represent specific main functional groups of microorganisms.

Gas formation is one of the most important processes occurring in landfills. Methane and carbon dioxide are formed by microorganisms as terminal product during the anaerobic degradation of organic material. However, it is known that organic materials such as paper, cardboard, food and garden wastes are easily degraded under laboratory conditions or in compost heaps, whereas in landfills the process can take quite a long time [17].

Gaseous emissions from MSW landfills are characterized by their complex composition [16, 32]. Generally, landfill gas contains 40 to 60% methane (CH_4) and 60 to 40% carbon dioxide (CO_2) which is formed in the process of microorganisms degrading organic material [1]. Both of them are important greenhouse gases with methane possessing a global warming potential which is about 25 times greater than carbon dioxide. Regarding greenhouse gases, the concentration of nitrous oxide (N_2O) found in landfill gas is negligible. Methane is emitted at typical annual emission rates of about 10 m^3 of gas per ton of deposited waste, finally resulting in 150 to 300 m^3 of landfill gas [23, 32]. Factors affecting biological decomposition of MSW and landfill gas emission include presence and spatial distribution of microorganisms, moisture content of the waste, pH, temperature, redox potential, nutrient concentration as well as physical dimension of the landfill site, type and particle size of deposited waste, age of the waste, waste compaction, coverage, capping and so forth [2, 32].

Landfill gas formation is characterized by four to eight phases, depending on the point of view [12, 14]:

1. An oxidative phase dominated by oxic conditions where oxygen is gradually consumed by microbial activities and carbon dioxide is formed. Nitrogen concentration remain more or less constant.
2. Start of anoxic conditions after oxygen depletion where electron acceptors such as nitrate, iron, or sulfate are used instead of oxygen. Gaseous products are carbon dioxide and hydrogen. A series of short chain alkanoic acids are formed which are finally converted to acetate (acetogenesis). Nitrogen is displaced.
3. Start of methane formation where acetate, hydrogen and part of the carbon dioxide is consumed by methanogenic bacteria (“unstable” phase of methane formation). Methane concentrations gradually increase. The duration of the first three phases ranges from 180 to 500 days [14].
4. Methane and carbon dioxide are formed at a relatively constant rate (“stable” phase of methane formation) resulting in a constant gas composition over a certain period of time. This phase can last relatively long in comparison to the first three phases.
5. Gas formation rate starts to decrease. A significant portion of carbon dioxide is dissolved in the leachate. Ambient air start to intrude into the landfill body.
6. In this phase methane is aerobically oxidized resulting in the consumption of the intruding oxygen. Nitrogen concentrations increase along with carbon dioxide concentration originating from methane oxidation.
7. Methane oxidation terminates and oxygen concentration increase.
8. This phase is the final phase in which the waste has been fully degraded. Landfill gas more or less resembles interstitial air present in soil.

A schematic overview of the gas formation kinetics is shown in Figure 2.4. It has been stated that the duration and the relative amount of gases formed are affected by a variety of factors mentioned above [14]. Whereas for the first phases data are available, the later phases are only speculative [12].

Besides the major compounds methane and carbon dioxide, a wide variety of minor constituents such as hydrocarbons, halogenated hydrocarbons, alcohols, aldehydes, ketones, esters, ethers, and organosulfur compounds can be detected, sometimes in concentrations that are of toxicological significance [32]. However, a differentiation between their biotic and abiotic (purely chemical) formation is almost impossible. For instance, ethane can be abioticly produced by the reductive dehalogenation of chlorinated solvents, but a microbiological formation is also possible by the hydration of ethylene, which itself is microbially produced from sugars or ethanol. However, the source and microbial metabolism of non-methane hydrocarbons is not fully understood.

A series of compounds in landfill gas are already present in the MSW in their original form (e.g. propellant gases escaping from cans), whereas others are formed by chemical reactions. Alternatively, certain gaseous compounds from the sulfur and nitrogen cycle, such as e.g. H_2S, NH_3; $(CH_3)SH$; $(CH_3)_2S$, $(CH_3)NH_2$, or $(CH_3)_2NH$, can be microbially produced [7]. These substances are formed by the reduction and methylation of oxidized sulfur or nitrogen compounds. Elements other than sulfur and nitrogen can also be methylated by the metabolic activities of microorganisms [24]: A wide series of metals and metalloids are known to occur as methyl compounds which are characterized by a high volatility and mobility. In particular, volatile species of antimony, arsenic, bismuth, bromine, iodine, lead, mercury, silicon, tellurium, vanadium, and tin have been detected in gases released from domestic waste deposits in a concentration range of 0.1 ng to 10 µg per m^3 of gas [20]. It has been demonstrated that at least some of these compounds are formed under anoxic conditions by methanogenic, sulfate-reducing, and peptolytic bacteria [27]. This might also be the case in MSW landfills. Metals can, therefore, be emitted from MSW landfills not only as ionic, water soluble compounds, but also in gaseous forms [25].

Several models have been developed to simulate gas or leachate composition from MSW landfill emissions [36]. For example, net emissions of methane at individual landfill sites follow the following simplified equation [15]:

$$\text{net methane formation} = \Sigma(\text{methane emission} + \text{lateral migration} + \text{methane recovery} + \text{methane oxidation} + \text{methane storage})$$

A more complex model was used to simulate gas and leachate emissions [36]: waste composition, size and shape of site, water input, waste pretreatment, temperature, moisture level, pH, redox potential, bacterial population, and solute concentrations were used as input parameter for a mathematical model combining a number of subsystems. It has been stated that “all the mechanisms involved in regulating landfill degradation interact and cannot ultimately be considered in isolation from each other” [36].

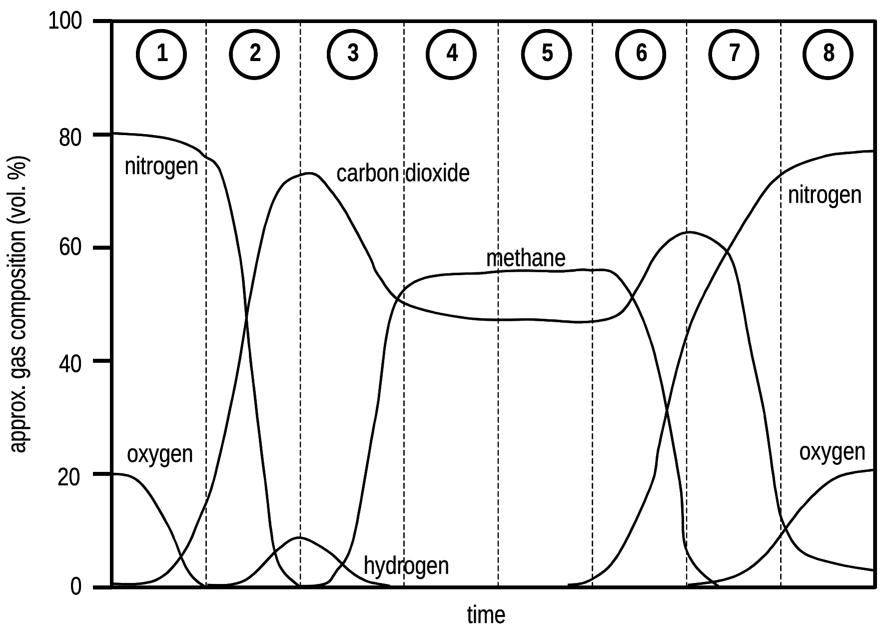

Fig.2.4. Kinetics of gas formation in a landfill (modified after Christensen & Kjeldsen [12]). Numbers represent distinct phases of gas formation (see text for explanation).

In a case study, a landfill composed of waste containing per m^3 150 kg of carbohydrates, 300 kg of protein, 20 kg of fat, and 450 kg of non-degradable materials was established [36]. Initial dissolved and gaseous concentrations of methane, hydrogen, oxygen, hydrogen sulfide, ammonia, acetate, carboxylic acids, alcohols, and glucose were taken as zero in an atmosphere of 20% carbon dioxide and 80% nitrogen assuming that all the oxygen has been rapidly consumed in the aerobic phase and that the anaerobic phase has just been started. The biomass of methanogenic bacteria was assumed as 10 mg per m^3. A 14-month simulation showed a pH decrease in the leachate from approximately 7.3 to 6.5 after 30 weeks, followed by a sharp increase to 7.5, and after 38 weeks a more or less constant value of 7.3. Correspondingly, the concentration of acetic acid increased to 1.3 g per liter after 30 weeks followed by a complete consumption within 8 weeks. The population of autotrophic methanogenic bacteria consuming hydrogen and carbon dioxide was large enough after 16 weeks to completely utilize hydrogen which had been formed in the early stages. At week 30, acetoclastic methanogens (consuming acetic acid) with a modeled biomass of 55 mg per liter started to use acetic acid until its full depletion after week 38. Consequently, biomass decreased to approximately 15 mg per liter after 14 months. Maximum methane concentrations of 70% could be found at around week 38 decreasing to approximately 50% after 40 weeks. Finally, it was concluded that methanogens eventually reach steady state conditions where substrates are utilized at the same rate as products are formed [36]. Waste is degraded to carbon dioxide and methane in a 1:1 molar ratio.

2.2.2 Emissions to the Pedosphere and Hydrosphere

In addition to uncontrolled gas release from landfills to the atmosphere, emissions can also occur affecting the pedo- and hydrosphere. Theses emissions are mostly in the form of liquid leachates which are generated by percolating rainwater and contain run-off of organic and inorganic compounds resulting in the contamination of soil, surface and groundwater. It has been estimated that groundwater pollution originating from landfills may be a risk even after several centuries [3, 21]: At medium rates of leachate formation (e.g. 200 mm per year), 300 years are needed until the final storage quality is reached and the leachate can be released into the hydrosphere without risk. With coverage systems allowing leachate formation at rates of only 100 mm per year or less, it is evident that, even for a very long time period, the quality criteria are not met.

Four groups of pollutants are characteristic for landfill leachates [13]:

1. dissolved organic matter, expressed as Chemical Oxygen Demand (COD) or Total Organic Carbon (TOC), also including methane and volatile fatty acids;
2. inorganic macro-compounds such as calcium, magnesium, sodium, potassium, ammonium, iron, manganese chloride, sulfate, and carbonate;
3. heavy metals such as cadmium, copper, chromium, lead, nickel, and zinc;
4. xenobiotic organic compounds such as aromatic hydrocarbons, phenols, and halogenated aliphatics.

Leachate composition varies depending on the waste type, rainfall conditions, landfill design and operation, and landfill age [32].

A major survey of landfill leachates showed that over longer periods of time ammonium in concentrations of up to 2.5 grams per liter has the highest hazardous potential to affect surface or ground water [13, 32]. By contrast, heavy metal concentrations (e.g. cadmium, chromium, copper, lead, mercury, nickel, or zinc) in leachates are generally low and present in amounts that are below those usually detected in household sewage. Recently, the biogeochemistry of landfill leachates affected aquifers has been critically reviewed. It was demonstrated that most contamination plumes are relatively narrow [13]. A spatial heterogeneity of leachate composition and concentrations can be observed with areas showing relatively low concentrations and some “hot spots” of high concentrations. Depending on climatic conditions (rainfall), temporal variations of compounds found in landfill leachates also occur [13]. In general, it has been found that natural physical, physico-chemical, chemical, and microbial attenuation processes such as dilution, sorption, ion exchange, precipitation, redox reactions, and degradation processes significantly contribute to natural remediation resulting in effects of the leachate in a distance from the landfill that does not exceed 1 kilometer [13].

With a leachate plume originating from a landfill, distinct redox zones are present with specific oxidation/reduction regimes. These zones are typical for certain microbiologcal metabolic activities and can also be found in other anoxic environments , e.g. in aquatic ecosystems such as freshwater lake sediments [10]. The typical redox sequence is present starting with a zone of methane formation closely located to the landfill body. Downgradient from this location, zones of sulfate reduction, iron reduction, manganese reduction, and nitrate reduction (denitrification) can be found, sometimes overlapping to a certain extent. Finally, at the edge region of the plume (furthest away from the landfill), oxic conditions prevail where aerobic processes occur. Additionally, fermentative reactions are possible where the electron acceptor is of organic nature. Fermentation can basically occur in the whole anoxic zone of the leachate plume. This sequence strongly depends on the presence (type, concentration) of terminal electron acceptors and the thermodynamic energy yield available for the microorganisms from each redox reaction [18].

The behavior of microbial communities involved in biogeochemical processes in each zone can be deducted from chemical thermodynamics [18]. Each redox zone in a leachate plume is the habitat of specific and typical microorganisms: It was shown that groundwater aquifers contaminated with landfill leachate are dominated by the presence of bacteria (eubacteria and archea) and that protozoae are absent [26]. Over a distance of approximately 300 meters, a total of 10^7 to 10^8 bacterial cells (determined by acridine orange direct counts) per gram of dry aquifer solids have been found [13]. Methane-forming bacteria were restricted to the most polluted part, closest to the landfill, showing the most reduced conditions. Around 10^5 cells per gram have been detected. On the other hand, highest cell numbers of nitrate-reducing bacteria (10^6 to 10^7 cells per gram) have been found at a distance of approximately 80 meters away from the landfill. On the basis of specific biomarkers (phospholipid fatty acids, PLFA), a decrease on viable microbial biomass as well as shifts of microbial community composition were detected along a horizontal gradient with increasing distance from the landfill body.

Biochemical and molecular techniques have been used to investigate the composition and the physiological capabilities of microbial communities present in aquifers contaminated by landfill leachates [29]. Anaerobic community-level physiological profiles (by BIOLOG multi-well plates) and DNA fragment analysis (by denaturing gradient gel electrophoresis) were applied to groundwater collected near a landfill site. With both techniques it was possible to differentiate microbial communities from the aquifer underneath the landfill as compared to sampling locations up- or downstream the aquifer. It was demonstrated that functional diversity of microbial populations regarding the range of metabolizable substrates was significantly enhanced in the plume of pollution resulting from the landfill [29]. Degradation of organic compounds occurred in the plume under iron-reducing conditions, whereas upstream of the landfill, nitrate reduction (denitrification) was the most important process [30]. Iron reduction was related to the presence of members of the family *Geobacteraceae* which strongly contributed to the microbial communities. Microorganisms of the class β-proteobacteria were dominating upstream of the landfill. Beneath the landfill, however, this group was not found and gram-positive microorganisms were mostly present. A profound effect of landfill effluents rich in organic matter on the chemistry and microbiology of aquatic environments underlying the landfill was clearly shown.

Table 2.6. Emissions of hazardous substances from landfills and their environmental impact (adapted from [15, 32]). COD = chemical oxygen demand.

path	group	compound	environmental impact	importance
gas	volatiles	methane	global climate change, explosive, asphyxia	high contribution of landfills to overall emissions
		carbon dioxide	global climate change	minor contribution of landfills to overall emissions
		hydrogen sulfide	odors, corrosion	minor impact due to fast oxidation in the presence of oxygen
		halogenated organics	human toxicity, cancerogeneity, ozone depletion	important for employees and local communities
		organics	human toxicity, cancerogeneity, nuisance	important for employees and local communities
		alkylated metals	human toxicity	importance unknown
leachate	salt	e.g. chloride	ecotoxicity	high contribution from landfill waste water treatment
	nitrogen	e.g. ammonia	eutrophication	important, due to local contamination of surface and groundwater
	metals	Cd, Ni	human toxicity, cancerogeneity	less important, small contribution to total emissions
		Cu, Hg, Pb, Zn	ecotoxicity	less important, small contribution to total emissions
	carbon	COD	eutrophication	less important, small contribution to total emissions from waste water treatment

In summary, Table 2.6 shows major emissions from landfills into the atmosphere, hydrosphere, and pedosphere and their corresponding impact on the environment [32]. At global levels, is has been estimated that methane emissions can contribute for approximately 18% to of total methane emissions. Regarding leachates, chloride is quantitatively the most significant compound. Approximately 2% of chloride discharged to the environment by waste water treatment systems originates from landfill leachates.

Case Study: Landfill 'Ritzer' Near the City of Aarau, Switzerland

An area near the city of Aarau (Switzerland) in the Jura Mountains was used until 1921 as a quarry for the production of raw materials (carbonate rocks) utilized in the cement industry. As early as 1959, landfilling of a variety of wastes already began to include the disposal of household, hospital, and industrial wastes, sludge from neutralization processes (iron chloride, calcium chloride), bitumen, soil excavated from gas works or from spills of chemicals or oil, and foundry sands. The volume of the landfill is approximately 360'000 m^3. After the opening of a waste incineration plant nearby in 1974, the landfill was closed and covered.

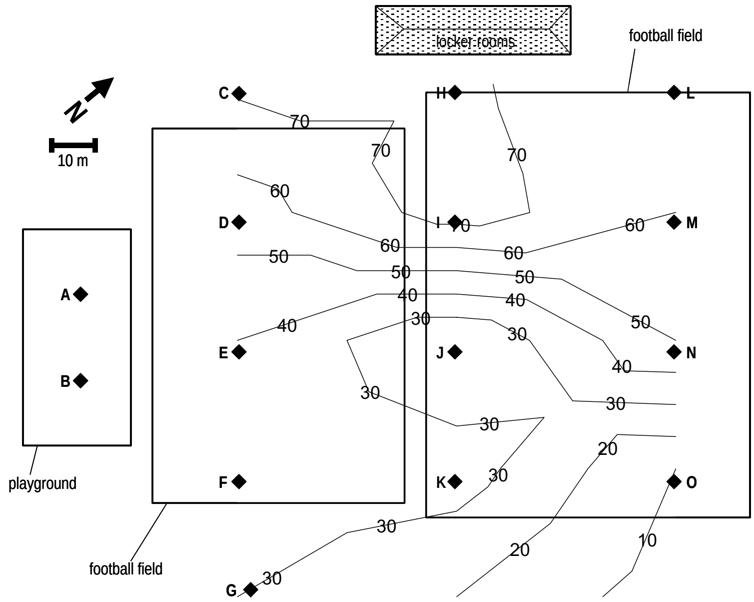

Fig. 2.5. Methane isopleths (% v/v) in a landfill which has been used as football field and playground since the closure of the landfill in 1974. Lines represent locations of equal methane concentrations. Diamonds (labelled with letters A to O) represent sampling wells.

Two years later, playground and football fields were established. After a certain time period, waste materials began to settle resulting in several depressions in the football fields which increased yearly by approximately ten centimeters. After rainfalls, water was retained in these depressions, preventing all sports activities. In addition, it was observed that from few spots in the field gas was emitting and that, therefore, the nearby locker rooms were endangered by possible explosion due to the gas penetrating the building.

During an investigation in 1998, physico-chemical characteristics, such as the composition of gas as well as leachate originating from the landfill, were determined (Eberhard & Partner AG, Aarau, Switzerland; personal communication). No microbiological studies were performed. Table 2.7 gives an overview of a series of landfill gas constituents. Methane concentrations of up to 75% (v/v) were determined to be highest in the area where the locker rooms are located (Fig. 2.5). Areas with the deepest depressions showed the highest methane concentrations. In addition to methane, butane was found as an important constituent of the trace compounds. Unfortunately, the high methane concentrations prevented a quantitative determination of ethane and propane. Benzene was detected as the main compound in the group of aromatic hydrocarbons (Tab. 2). Collected landfill gas showed elevated concentrations of halogenated hydrocarbons, mostly trichloroethene (TCE). Since a typical TCE profile could be determined, it was suggested that this compound was emitted from a point source, possibly from chemical solvents disposed in the landfill (Fig. 2.6).

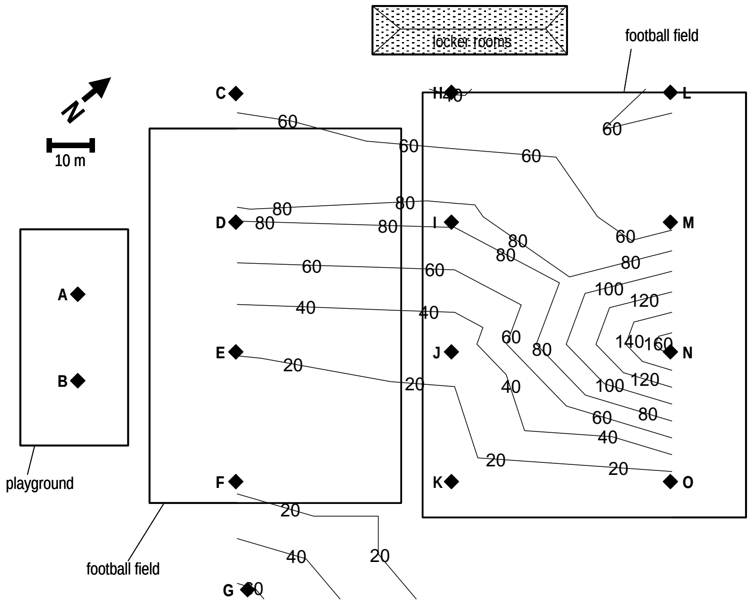

Fig. 2.6. Trichloroethene isopleths (μg/m^3) in a landfill which has been used as football field and playground since the closure of the landfill in 1974. Lines represent locations of equal trichloroethene concentrations. Diamonds (labelled with letters A to O) represent sampling wells.

Concentrations of heavy metals found in the landfill leachate were below the limit values of legal regulations. In one of the samples slightly elevated concentrations of sodium, potassium, nitrite, ammonium, sulfide, and boron were found. Additionally, traces of hydrocarbons were detected, suggesting the presence of residues originating from oil degradation. In contrast to the landfill gas, no trichloroethene was found. As a result from the investigation, it was calculated that between 22 and 54 million m^3 of methane could be formed in the landfill and that the gas formation would last up to twenty years until the waste in the landfill is consolidated (Eberhard & Partner AG, Aarau, Switzerland; personal communication). Important environmental effects resulting from gas emissions were identified, namely the damage of plant roots in the area due to oxygen depletion in the soil, the architectural instability of the locker room building due to the formations of depressions in the ground, and the danger of explosion due to gas/air mixtures containing high amounts of methane. As the primary measure to enhance the outgassing of the landfill, an active pumping and collection of the gas was suggested. The gas would serve as additional energy source for a nearby school.

Table 2.7. Composition of gas collected from a closed landfill site near the city of Aarau, Switzerland. For sampling locations A to O (figures 2.5 and 2.6). n.d. = not determined; n.a.=not applicable, because high methane concentrations matched ethane and propane; *=below detection limit. [1] Aliphatic hydrocarbon. [2] Aromatic hydrocarbon. [3] Halogenated hydrocarbon.

		Sampling location														
	unit	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O
Drilling depth	m	2.4	2.4	3.5	3.5	3.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
Sampling depth	m	0.7	1.5	3.3	3.3	3.3	2.0	2.3	2.3	2.3	2.3	2.3	2.3	2.3	2.3	2.3
Main compounds:																
Methane	%	0.005	0.04	75	n.d.	40	n.d.	30	70	70	20	35	60	60	50	10
Carbon dioxide	%	n.d.	n.d.	18	n.d.	20	5	16	20	20	15	16	30	25	23	18
Trace compounds:																
Ethane [1]	ppm	*	*	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Propane [1]	ppm	*	*	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Butane [1]	ppm	1.0	0.4	2.1	5.0	3.5	1.1	2.4	2.2	0.8	0.8	2.0	1.4	2.3	1.1	0.5
Pentane [1]	ppm	*	*	1.6	2.0	1.4	0.9	1.0	1.1	2.2	*	1.5	0.4	0.8	0.6	*
Hexane [1]	ppm	*	*	0.5	0.5	0.3	0.6	0.2	0.5	0.7	*	0.4	0.4	*	0.5	*
Heptane [1]	ppm	*	*	0.8	*	*	*	*	0.3	0.1	*	*	0.2	0.1	0.3	*
N-octane [1]	ppm	*	*	0.4	0.3	*	*	*	0.8	*	0.7	0.2	0.4	0.2	0.4	*
Iso-octane	ppm	*	*	0.6	*	*	*	0.4	*	*	*	*	*	*	*	*
Nonane [1]	ppm	*	*	0.3	0.5	0.2	*	*	0.4	*	*	*	0.3	*	0.5	*
Decane [1]	ppm	*	*	0.3	0.1	*	*	0.3	0.4	*	*	0.3	*	0.1	*	*
Benzene [2]	ppm	*	*	2.5	2.5	1.4	0.3	1.7	2.6	1.6	0.6	2.9	1.0	1.1	1.9	*
Toluene [2]	ppm	*	*	*	*	*	*	*	*	*	*	*	*	*	2.1	*
Ethylbenzene [2]	ppm	*	*	0.8	*	*	*	0.3	0.8	0.3	*	*	0.5	0.2	*	*
Ortho-xylene [2]	ppm	*	*	*	*	*	*	*	0.2	*	*	*	*	*	*	*
Propylbenzene [2]	ppm	*	*	*	*	*	*	*	0.4	*	*	*	*	*	*	*
1,2,4-trimethyl benzene [2]	ppm	*	*	*	*	*	*	*	0.4	*	*	*	*	*	*	*
Trichloroethene [3]	$\mu g/m^3$	*	*	55	82	21	*	67	38	85	25	*	63	47	171	20
Tetrachloroethene [3]	$\mu g/m^3$	*	*	*	*	*	*	*	*	*	*	*	*	*	208	35

2.2.3 Problems in Predicting the Long-Term Behavior of Landfills

Biotests for Toxicity Assessment

A series of ecotoxicological tests have been applied to determine the impact of MSW leachates on natural ecosystems. Under the term 'biotests' specific – mainly standardized - techniques are summarized which apply biological systems such as bacteria, protozoae, microalgae, small invertebrates, or fish to assess the impact of a sample or compound based on specific physiological reactions. Biotests find a wide application especially in aquatic ecotoxicology [9]. In general, acute and chronic toxicity of a sample or compound can be determined. Table 2.8 summarizes a selection of bacterial biotests which have been developed. A very popular and simple test is the determination of light emission inhibition by luminescent bacteria (*Vibrio fischeri*) after exposure to different amounts of an aqueous solution of the compound.

Biotests based on physiological reactions of higher organisms have been applied to investigate the toxicity of MSW leachates containing a variety of different organic and inorganic chemicals [34]. Invertebrates such as *Ceriodaphnia dubia* were used to test acute toxicity regarding the suppression of feeding activity by toxic compounds. It was suggested that toxicity of the leachates obtained form three sites was due mostly to organic compounds [34].

Zebrafish (*Brachydanio rerio*) were used as test organisms to study acute toxicity of MSW leachates in Brazil [33]. Leachates were differently treated (addition of EDTA or aluminium sulfate, aeration) to compare toxicities. It was found that the leachate was a highly toxic effluent potentially affecting aquatic species after discharge to aquatic ecosystems. The addition of aluminium sulfate significantly reduced toxicity.

Table 2.8. Selection of bacterial biotests for toxicity assessment

Type of assessment	Inhibition test	Standard norm	Organism	Physiological reaction	Duration (h)
acute toxicity	oxygen utilization	DIN 38412L27	*Pseudomonas putida*	substrate oxidation	0.5
acute toxicity	light emission	DIN 38412L34	luminescent bacteria (*Vibrio fischeri*)	energy metabolism	0.5
acute toxicity	respiration	OECD 209	sewage sludge culture	oxygen utilization	3
chronic	cell replication	DIN 38412L8	*Pseudomonas putida*	cell replication	16

2.2.4 Conclusions

Most of the problems related to landfill emissions are due to the amount of organic compounds in the waste which is microbially degraded, leading to soluble and volatile degradation products [32]. It has been suggested that proper landfill management (e.g. operational practices, controlling the waste type accepted for landfilling, appropriate leachate treatment prior to discharge) might reduce problems associated with landfills [32].

2.3 MSW Management and Technology in China

Zhao Youcai

China is one of the largest nations in the world, encompassing a vast area, with diversified nationalities and cultures, and a very large population. It is also the largest developing country and has relatively poor infrastructures and underdeveloped industry. From the viewpoint of MSW management, the country might be divided geographically into at least two sections, roughly the north and the south, with an approximate boundary along the Yangtze River. In the north of China, the weather is dry and cold for most seasons of the year, with a fragile ecological environment and a vast area of desert and high plateau. By contrast, it is humid and hot for nearly the whole year in the south of China, especially in the provinces along the East China Sea and South China Sea.

The south of China is densely populated and the available land seems to be very limited as nearly every inch of land has been used for agricultural, industrial, and living purposes. More land reserves might be available in the north, except for in the proximity of several big cities such as Tianjin, located in the Great Northern China Plain.

The agricultural and industrial sectors are also greatly varied from north to south. Rice is the main crop in the south, grain and corn in the north. Most heavy industries are located in the north, while the light industries are located in the south, though this situation is gradually changing.

There are differences in living and eating habits as well. While in the south, many kinds of soups are consumed because the weather is always hot, the food in the north is relatively dry. As a result, the MSW in the north and south differ in terms of humidity, composition, odor, etc. The humidity in MSW in the north is around 30-50%, compared with that in the south, which is around 40-60%. The humidity of MSW in the north disappears rapidly because of the dry weather, without strong odor or severe corruption. By contrast, the MSW in the south may degrade and corrupt very quickly, producing strong odor and leachate water. In the north, a great deal of coal is used for heat generation in the winter. Consequently, the proportion of coal ash in the MSW can be as high as 70%. Currently, the number of cities using natural gas or coal gas as fuel is increasing in the north (as well as in the southern cities), and the coal ash content is decreasing as a consequence.

The selection of treatment and disposal technologies of MSW in China should be flexible and adapted to local economical, social, geological and even cultural and historical conditions. Direct mechanical separation practices for the MSW in the south have proven difficult because of the high humidity. However, it is feasible in the north, where the MSW is relatively dry.

Around 120 million tons of municipal solid wastes were collected by the city authorities in China in 1999. Up to now, the MSW generated in rural areas has not been collected and is rather dumped into any available sites. About one quarter of the Chinese population are living in 700 cities and over 30,000 towns. The scale of these cities in population may vary from 0.1 to 9 million. It should be pointed out that, especially in small cities and towns, the service area for organized MSW collection by the relevant authorities usually covers only a small central part of the cities. The MSW generated in the suburbs and small towns is often not collected at all.

Table 2.9 shows the average composition of MSW in three typical cities in China. As mentioned above, the situation may vary greatly from one city to another. Generally speaking, the contents of plastics and papers are gradually increasing, while those of coal ash are decreasing. The construction and demolition wastes have increased in recent years as many families move into new housing. Food and ash wastes contents in Dalian are much higher than in the two other cities. Based on the *in situ* investigation, it was determined that the coal ash content in Dalian was still quite high in 1998.

The contents of recyclable wastes such as papers, plastics, metals, etc., are low. In fact, most of these wastes are collected and recovered by scavengers. The composition of MSW is determined *in situ* in landfills or dumping sites, not in the generation sites of the MSW. In large cities such as Shanghai and Beijing, the recyclable wastes are well recovered, including cans, cardboard, big pieces of woods, TV sets, nearly all kinds of plastics, and glass bottles. However, used batteries, lamps, thermometers, etc., have not been collected separately and are being mixed with MSW, ultimately entering the landfills or dumping sites.

In addition, the moisture in the MSW may vary from 30 to 60% in weight, depending on the seasons and locations. In the rainy seasons in the south of China, the moisture is so high (over 60%) that landfill operations become unacceptable .

Table 2.9. Average MSW composition in typical cities in China in 1998 determined in landfills [wt %]

Cities	Food wastes and ashes	Papers	Glass	Metals	Plastics	Textiles	Slag
Beijing	59.6	11.7	3.8	1.7	12.6	2.8	8.2
Shanghai	65.7	6.7	4.0	2.0	11.8	2.3	7.5
Dalian	82.1	3.4	2.6	0.5	5.7	1.6	4.1

The MSW yield per capita in China is shown in Table 2.10. Obviously, the values are relatively low for urban areas, as only 1.16 kg per day/person are generated. Lower yield in small cities (such as Maanshan in the Anhui Province in the south of China and Anshan in the Liaoning Province in the north of China) may be due to the relatively undeveloped economy and better recovery rates of recyclable wastes. The highest yield is found in Shenzhen, a newly developed city near Hong Kong. Table 2.11 presents the relationship between the GDP and the MSW generated in the large-scale cities, as reported in 1995. It can be found that the higher the GDP, the more the MSW, with nearly linear correlation between MSW quantity and the GDP, exclusive of the situation in Shenyang, a large city with a number of heavy industries (compare section 2.1). From this viewpoint, one may argue that the MSW yield per capita and the total quantity should increase as the economy in China develops. In this case, more and more treatment facilities for MSW will have to be planned and constructed.

Table 2.10. MSW yield [kg/day/person] in typical cities in China in 1996 based on investigations *in situ* in landfills or dumping sites

City	Beijing	Tianjin	Shanghai	Shenyang	Dalian
Yield	1.20	0.99	1.23	1.02	1.03
City	Hang-zhou	Shenzhen	Guangzhou	Maanshan	Anshan
Yield	0.92	2.62	1.20	0.66	0.76

There are three main alternative treatment methods for MSW in China; these are landfill, incineration, and composting. Landfill is the predominant method in China, while large-scale composting is limited and incineration is still being developed. The first large-scale incineration plant in Shengzheng was constructed in 1985. Meanwhile, an incinerator capacity of 2000 ton/d in Shanghai, 1000 ton/d in Ningbo, 600 ton/d in Zhuhai, 300 ton/d in Xiameng, and various capacities in Shengyang, Shengzheng, Tianjing, etc., will be constructed in the coming years. With increasing economic development, many of the existing landfills will soon be reaching their design capacity, and finding new landfill locations is becoming increasingly difficult. Although there are many treatment technologies being developed and applied in the world, the incineration technologies will be a method with priority for final MSW disposal in the coming years in China, especially in the more quickly developing big cities.

Presently, most MSW is dumped in the dumping sites around the cities, which results in serious environmental problems. Most so-called landfills have to be classified as dumping sites, but they can be restructured into sanitary landfills in the coming years. The statistic data shows that, over all, less than 10 % of MSW is treated in sanitary landfills, composting and incineration plants at present.

China is a developing country, not only economically but also in science and technology. However, for solid waste management in China, technology does not seem to be the key limiting factor in the obstruction of the development of solid waste management, although it should be improved in the future. In past years, a great deal of feasible and cost-effective technologies have been developed, and some of them have been applied in completed and ongoing projects.

Table 2.11. Quantitative relationship between GDP in a city and its MSW quantity in 1995 in China

City	City population [million]	Total population (incl. suburbs) [million]	GDP [billion Chinese Yuan]	MSW [million tons]
Maanshan	0.38	0.50	8.78	0.094
Anshan	1.44	3.31	39.5	0.401
Dalian	2.53	5.37	73.3	0.715
Shenyang	4.20	6.71	77.1	1.569
Hangzhou	1.96	6.03	90.6	0.660
Shenzhen	0.78	1.03	95.0	0.754
Tianjin	5.13	8.98	110.2	1.853
Guangzhou	4.03	6.56	144.5	1.764
Beijing	7.10	10.78	161.5	3.110
Shanghai	9.32	13.04	290.2	4.182

In China, almost all investments in solid waste management and operational costs are financed by local governments, which is usually not the case in industrialized countries. This situation has stifled the advancement of MSW treatment in contrast to the economic development. Moreover, many Chinese still think that solid waste management is the duty of the government. Action for *in situ* sorting and separation for MSW at home is also difficult to put into practice. Hence, the public environmental responsibility should be brought to task.

MSW Management in a Fast Developing Chinese City: Shanghai

MSW generated in Shanghai is increasing, with a total quantity of 11,620 tons per day in 2000. It is estimated that the MSW quantity in 2005 will reach 14,850 tons. In addition, the number of used TV sets, furniture, refrigerators, washing machines, bicycles, etc., as wastes have increased greatly in recent years. Table 2.12 shows the quantity of products sold in Shanghai in 1997. Theoretically, these products are expected to become bulky wastes in subsequent years. Nevertheless, most of this kind of waste has been reused or collected by the recycling plants as secondary materials when the users discard them. It is rare to see such wastes arrive at the landfills.

Used batteries and fluorescent lamps have not been collected and treated separately. At least 100 million pieces of small batteries are being used in Shanghai every year. None of them are treated properly. The main reason may be economical, in comparison to the primary raw materials. Investigations into recycling technologies for used batteries, relying on experience in Switzerland, USA, Germany, and France, have been conducted by the engineers, scientists and governmental officials in China, but implementing efficient recycling plants seems difficult. Currently, many private sectors have interests in the collection and treatment of used batteries, but progress is slow and there is the danger of potential secondary pollution.

Table 2.12. Quantity of products sold in Shanghai in 1997

Type	Used furniture	TV (set)	Refrigerator (set)	Washing machine (set)	Bicycle (set)
Quantity [ton]	440’000	390’000	240’000	235’000	475’000

There are two landfills in Shanghai. Liming Refuse Landfill, is a relatively small operation mainly used for the deposition of MSW collected in Pudong New Area. The large landfill, Shanghai Refuse Landfill Laogang, has been built and extended over the past 12 years, along the shore of East China Sea. Currently, there are around 6 km^2 of filling area available. An extension is planned which will increase the total area of the landfill to 12 km^2. Shanghai Refuse Landfill is not an ideal site. It was selected because there was no better site under consideration. Being on the shore of the sea, it is affected by the tides: It was found that the liners of the landfill are destroyed by the up and down motion of the tides. This problem has not been solved. In addition, the landfilling height is only 4 m, and the subsequent large area of placement leads to very high costs for liners. In addition to the landfills, there are still 12 large scale dumping sites in suburban Shanghai, of which 10 sites have been closed and 2 are still in use. Several million tons of refuse are stored in these sites.

Two incinerator plants are currently under construction. One is located in West Shanghai, another in East Shanghai. The key equipment was imported from Spain and France, with a loan from the foreign governments. It is claimed that the flue gases are treated at EU standard. 2000 tons of refuse can be incinerated, with a total investment of 0.75 billion Chinese Yuan for each plant.

There had once been a large scale composting plant in Shanghai. Unfortunately, it had to be closed because, on one hand, there was no market for the compost, and on the other, the composition of the waste became difficult to handle, with a large proportions of plastics, broken glass, textiles, etc. Currently, there is no composting plant in Shanghai.

Table 2.13 summarizes the current flows of waste materials in Shanghai. The MSW generated downtown, around 6,840 tons/day, is dumped in two controlled landfills and two dumping sites. The MSW in the suburb, around 4,010 tons/day, is simply dumped in the dumping sites without any pollution control measures. Planning for future MSW treatment facilities in Shanghai encompasses ‘Integrated Treatment Plants’, which consist of mechanical separation and sorting systems, composting systems, baling systems, perhaps also drying and compressing systems, and landfilling for the non-recoverable fraction. Shanghai is so large that reasonable planning for an economically feasible and technically viable MSW treatment system is quite difficult.

Table 2.13. Mass balances for Solid Wastes treatment in Shanghai in 1999

Items	Classified items	Quantity collected [ton]	Treatment methods
MSW	Total weight	6’840 in down-town	4500 tons in Laogang and Liming Landfills (with daily cover and drainage, and treatment of leachate, without liners)
			2’340 tons in two dumping sites
		4’010 in suburb	All in the various dumping sites without any pollution control facilities (leachate is directed to the sewage treatment plants in some sites)
	Bulky items (furniture, TV etc.)	230	Mostly recycled. Remnants are broken and deposited in landfills or dumping sites
	Plastics	990	Around 1/3 recycled, the remainder deposited in landfills or dumping sites
	Toxic waste, such as used batteries	3.4	Mixed with MSW and placed in landfills
Food origin wastes		1’100	Used as feed for 260,000 pigs until 1999. Prohibited since June 2000. Currently placed in landfills after dewatering, which makes the operation difficult, as the moisture is high.
Demolition and construction wastes		29’700	Mostly balanced in situ and partly recycled as feed for cement production, the remainder is deposited in the slag dumping sites
Human excrement		7’130	Mostly sewage, some recycled as organic fertilizer after digestion
Sludge		4’384	No way to go until 2002

Two viewpoints are always encountered when discussing the planning of MSW treatment establishments: centralized vs. decentralized facilities, which have their own individual advantages and disadvantages. According to the current political system in Shanghai, it is possible to establish a centralized facility. However, the problems may be the financial sources and sites selection. No Administration District or county wants to let the facility be constructed on its own land. In this regard, the Shanghai Government has to let every District construct its own ‘Integrated Treatment Plant’, in which mechanical separation, small scale landfill, composting, and perhaps incineration, should be located together at one site.

According to the experiences gained in the other large cities, such as in Guangzhou, a centralized treatment facility seems to be feasible if landfill space is available. In Shanghai, every inch of land has been used to a full extent and it has become increasingly difficult to find a sufficiently large site to host all the MSW.

Significant investments are required to construct the needed infrastructure mentioned above. In total, an estimated 4767.85 million Chinese Yuan (1 US dollar = 8.3 Chinese Yuan in 2001) in 6 years is required so that all the MSW in Shanghai can be treated to EU standards. However, it is impossible to get such a huge investment from the Shanghai Government. Currently, MSW in Shanghai (as in other Chinese cities) is collected, transported and treated by the Shanghai Environmental Sanitation Bureau, which acts as a company and an administration bureau. At the beginning of the year, the bureau receives all the funds from the Shanghai Government, based on the total expenses of the previous year. Generally, this fund can maintain only the lowest standard for MSW collection and treatment. If a new project is expected to be constructed, additional application must be presented and approved, which may take anywhere between some months and several years, depending on the scale of the investment.

Shanghai Government is the single investment source for MSW management. Nevertheless, the most important thing for local governments seems to be economic development, hence, the investment for environmental protection, including MSW and other waste treatment is usually put aside. Fortunately, many public and private companies are willing to invest in the treatment of MSW. Certainly, some profit should be guaranteed for these companies. One of most reliable financial sources is to collect payments from the MSW generators, including companies, households, and institutions from public and private sectors, etc. So far, in most cities in China such action has not been put into practice, as the local governments fear opposition from the households, especially ones with low income.

Many suggestions have been proposed for MSW management. For example, all the MSW facilities that are constructed by the governments can be rented to private companies, while relevant governmental organizations just act as regulator or supervisor. The governments should, of course, pay reasonable treatment fees to the companies. Landfills, incineration plants, waste water treatment plants, etc., can be sold and bought among interested customers. Currently, MSW collection and transportation operations still tend to be owned and operated by the local governments.

Private companies can construct their own treatment companies, and have the local governments pay the treatment fees. The prices can be negotiated. The dilemma facing the China cities is that everything is changing rapidly. Local governments are always reluctant to make any promises to private companies. It is very difficult to get a payment contract from a local government if one wants to treat MSW for the local community.

Hence, the Chinese government should speed up its reforms concerning MSW, including refuse fee collection, regulations for construction and operation treatment facilities.

Conclusions

The MSW management mechanism in China is basically centralized; local governments are responsible for MSW collection, transportation, treatment, facilities investment and construction, recruitment of all staff, and often affect ineffective productivity and heavy bureaucracy. The investment for MSW facilities construction is very constrained, as the governments are the single investors. Possibilities for the trading of MSW-related companies are under discussion. Private and public companies will be encouraged to invest and manage the MSW treatment facilities. Refuse tax may be levied in the future. Considering the advantages and disadvantages for the individual technologies, such as incineration, composting, landfill, the concept 'Integrated Treatment' may be adopted. It attempts to combine all the available technologies together at an optimum mode, in order to solve the difficulty of site selection and to facilitate the recycling of resources. The MSW management in Shanghai, in fact, involves a series of complex issues, e.g., adoption of centralized or decentralized manners, trading of facilities, maturing and developing of competitive and qualified companies and labors, etc. A significant amount of investment is required to create new facilities and upgrade the old ones. As most cities do not possess any modern treatment facilities for MSW, according to the experiences gained in recent years, landfilling seems to be the favored alternative for the rapid improvement of city sanitation, as the duration of construction of landfills is usually relatively short and the investment and operational costs are relatively low, provided that qualified liners be installed and leachate be properly treated. Nevertheless, the remediation of closed and functioning dumping sites should gain more attention from the public in general and from the local governments in particular, as the adverse long-term impacts the dumping sites have on human health and the environment have been clearly proven.

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