VILNIUS GEDIMINAS TECHNICAL UNIVERSITY

FACULTY OF ENVIRONMENTAL ENGINEERING

DEPARTMENT OF ENVIRONMENTAL PROTECTION

Tadas Batavičius

THE RESEARCH AND EVALUATION OF THE DAILY COVERS INFLUENCE ON THE ODOURS PREVENTION IN LANDFILLS

ATLIEKŲ UŽDENGIMO SLUOKSNIŲ ĮTAKOS IŠSISKIRIANČIŲ KVAPŲ MAŽINIMUI TYRIMAI IR ĮVERTINIMAS

Master‘s degree Thesis

Environmental Management and Clean Production study programme, state code 621H17003

General Engineering study field

Vilnius, 2012

VILNIUS GEDIMINAS TECHNICAL UNIVERSITY
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Tadas Batavičius

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Vilnius, 2012

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Master Degree Studies **Environmental Management and Clean Production** study programme Final work

Title	**The Research and Evaluation of the Daily Covers Influence on the Odours Prevention in Landfills**
Author	**Tadas Batavičius**
Academic supervisor	**Prof Dr Saulius Vasarevičius**

Thesis language:
English

Annotation

Waste management is one of the most relevant problems of environmental protection. Landfills constitute an unavoidable component of the waste management system. The work discusses the problem of landfill odours.

Aims of this work are: to evaluate the influence of the thickness of construction and demolition debris cover on reduction of the odorous emissions from landfill, to evaluate the influence of the construction and demolition debris cover on reduction of the odorous emissions from landfill for short (1-2 days) and longer (more than 3 days) periods, to prepare the simulation of the odour dispersion around landfill site considering different odour control scenarios (rare covering of waste, application of building debris for daily waste covering and minimization of waste tipping area).

Concrete and brickwork building debris could be applied for more efficient odour reduction from municipal waste landfills. Upon covering waste with the 10 cm building debris layer, odour thresholds in 1-2 day period decrease 51.0-79.7 %. The building debris layer thinner than 10 cm is insufficient to cover waste for longer than 3 days period. Upon covering waste with the 5 cm layer, instability of odour thresholds was recorded.

Simulated odour control scenarios show that temporal waste covering and application of building debris effectively reduces landfill odour emissions. Odour concentrations at the same places decrease more than twice and do not exceed the sanitary protection zone of the landfill site.

Structure: introduction, review of literary sources, description of the methodology, results of the experimental tests, description of the dispersion model and input data, results of the odour dispersion modelling, conclusions and recommendations, references.

Thesis consist of: 78 p. without appendixes, 48 pictures, 17 tables, 86 bibliographical entries.

Appendixes included.

Keywords

waste management, landfill odours, waste covering, building debris, dynamic olfactometry, dispersion modelling

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Antrosios pakopos studijų **Aplinkos apsaugos vadybos ir švariosios gamybos** programos baigiamasis darbas

Pavadinimas **Atliekų perdengimo sluoksnių įtakos išsiskiriančių kvapų mažinimui tyrimai ir vertinimas**

Autorius **Tadas Batavičius**

Vadovas **prof. dr. Saulius Vasarevičius**

Kalba: anglų

Anotacija

Atliekų tvarkymas yra viena aktualiausių aplinkosaugos problemų. Neišvengiamas atliekų tvarkymo sistemos komponentas yra sąvartynai. Darbe aptariama sąvartyno kvapų problema.

Darbo tikslai: įvertinti statybinio laužo sluoksnio storio įtaką sąvartyno kvapų susidarymui, įvertinti statybinio laužo panaudojimo galimybes trumpalaikiam (1-2 dienos) ir ilgesniam (daugiau kaip 3 dienos) atliekų uždengimui, įvertinti sąvartyno kvapų sklaidą pagal skirtingus kvapų kontrolės scenarijus (retas atliekų uždengimas pagal poreikį, dažnas atliekų uždengimas naudojant statybinį laužą ir atliekų išpylimo plotų mažinimas).

Sutrupinto betono ir plytų mūro statybinis laužas gali būti panaudojamas efektyviam kvapų mažinimui sąvartynuose. Uždengus atliekas 10 cm storio statybinio laužo sluoksniu po 1-2 dienų kvapų koncentracija sumažėja 51.0-79.7 %. Statybinio laužo sluoksnis, plonesnis nei 10 cm, nėra efektyvus ilgesniam nei 3 dienų atliekų uždengimui. Uždengus atliekas 5 cm storio sluoksniu pastebimas kvapų koncentracijos nestabilumas.

Sumodeliuoti scenarijai rodo, kad laikinas atliekų uždengimas naudojant statybinį laužą efektyviai mažina sąvartyno kvapų susidarymą. Kvapų koncentracijos tose pačiose vietose sumažėja daugiau kaip du kartus ir neviršija sanitarinės apsaugos zonos lyginant su retu atliekų uždengimu pagal poreikį.

Darbą sudaro 6 dalys: įvadas, literatūros apžvalga, metodikos aprašymas ir rezultatų analizė, sklaidos modelio aprašymas ir rezultatų analizė, išvados ir rekomendacijos, literatūros sąrašas.

Darbo apimtis – 78 p. teksto be priedų, 48 iliustr., 17 lent., 86 bibliografiniai šaltiniai.

Atskirai pridedami darbo priedai.

Prasminiai žodžiai

atliekų tvarkymas, sąvartyno kvapai, atliekų uždengimas, statybinis laužas, dinaminė olfaktometrija, sklaidos modeliavimas

(the document of Declaration of Authorship in the Final Degree Paper)

VILNIUS GEDIMINAS TECHNICAL UNIVERSITY

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DECLARATION OF AUTHORSHIP IN THE FINAL DEGREE PAPER

May 31, 2012

I declare that my Final Degree Paper entitled „The Research and Evaluation of the Daily Covers Influence on the Odours Prevention in Landfills" is entirely my own work. The title was confirmed on November 3, 2010 by Faculty Dean's order No. 401ap. I have clearly signalled the presence of quoted or paraphrased material and referenced all sources.

I have acknowledget appropriately any assistance I have received by the following professionals/advisers: Dr Eglė Zuokaitė, Dr Mantas Pranskevičius.

The academic supervisor of my Final Degree Paper is Prof Dr Saulius Vasarevičius.

No contribution of any other person was obtained, nor did I buy my Final Degree Paper.

	Tadas Batavičius
(Signature)	(Given name, family name)

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LIST OF ABBREVIATIONS

µGC – microgas chromatograph

ADC – alternative daily cover

ADMS – atmospheric dispersion modelling system

AHP – analytic hierarchy process

ANN – artificial neural network

C&D – construction and demolition debris

CFCs – chlorofluorocarbons

CH_4 – methane

D_{max} – maximum particle size

DT – detection threshold

ET – evapotranspiration

GC-MS – gas chromatography-mass spectrometry

H_2S – hydrogen sulphide

LFG – landfill gas

MSW – municipal solid waste

NMOC – non-methane organic compound

OEF – odour emission factor

OER – odour emission rate, OU/s

OIA – odour impact assessment

OPF – odour perception frequency

OU – odour unit

OU_E – European odour unit

PDF – probability density function

q_{max} – potential methanotrophic activity

RT – recognition threshold

SOER – specific odour emission rate, $OU/m^2/s$

TOC – total organic carbon

TSIFT – transportable selected ion flow tube

VFAs – volatile fatty acids

VOC – volatile organic compound

INTRODUCTION

Problem

Human economic activity is accompanied by the generation of waste. It is considered that industrial output every 25 years increases by 10 times (Jaskelevičius 2009). Industry uses a lot of mineral raw materials and the quantity is constantly increasing. In 1910, average consumption of raw materials was 5 tons per capita. Today this level has reached 45 to 50 tons per capita. Only about 2 % of used materials are final product, while 98 % are wastes that pollute the environment. Therefore, waste disposal is one of the most important environmental problems known from ancient times.

Waste management requires a lot of money, but looking further into the future it is cheaper to manage waste today than liquidate the consequences of pollution after few decades. Solving waste disposal problems three main directions can be chosen: waste recycling, waste incineration and storage of waste in specially engineered landfills. Disposal of wastes to land is an inevitable component of every solid waste management system. Even if facilities are provided for processing the waste to recover materials or energy, there will always be a need for land disposal of a residual proportion of the waste originally produced.

Anaerobic degradation of the biodegradable fraction of the landfilled municipal solid waste causes several environmental problems such as the production of methane, volatile organic compounds, odours and leachate, the presence of vectors (insects, rodents and birds), public health hazard, explosions and plant toxicity (Scaglia and Adani 2008). All these negative impacts and the long time required to stabilize the materials are the major issues that make landfills unsustainable. Odorants in landfill sites are mainly generated by landfill gas (trace compounds), fresh wastes and leachate. Some typical odorous compounds in landfill gas have very low detection thresholds and can be detected by human nose at very low concentrations.

Actuality of the work

The best way to avoid problems of waste disposal to landfills is to provide a number of preventive actions in the planning of the landfills.

Landfill odours and landfill gas composition are widely investigated in France (Bogner, Lambolez, Taramini, Senante), Greece (Loizidou, Kapetanious), Turkey (Dincer), Australia (Bowly). Landfill covers are investigated in many aspects: composition, hydro-physical properties, water balance, methane oxidation, adsorption characteristics of H_2S, short and long term performance, and cost (Kavazanjian, Dobrowolski, Plaza, USA; He, China; Cossu, Italy; Solan, Ireland; Scheutz, Denmark; Tham, Sweden). Odour perception complexity, number of compounds, low sensitivity and subjectivity are the main limitations of odour evaluation techniques (Bonoli, Capelli, Sironi, Italy; Nicolas, Belgium; Karnik, UK).

Aim of the work

The aim of this research work is to evaluate the influence of the thickness of construction and demolition debris cover on minimization of odorous emissions in landfill for short (1-2 days) and longer (more than 3 days) periods.

Objectives

The main objectives of this research work are:

1. To evaluate the influence of the thickness of construction and demolition debris cover on reduction of the odorous emissions from landfill.
2. To evaluate the influence of the construction and demolition debris cover on reduction of the odorous emissions from landfill for short (1-2 days) and longer (more than 3 days) periods.
3. To prepare the simulation of the odour dispersion around landfill site according to data obtained during the investigation.

Novelty of the work

Intermediate and final landfill covers are investigated in many aspects: composition, methane oxidation, adsorption characteristics of H_2S, etc. In this work daily covers are evaluated with respect to variety of odorous compounds, characteristics of used materials.

Practical meaning of the work

In this work alternatives of conventional daily cover used in landfills are recommended. Investigation results could be put in practice to improve landfill operation and mitigate public concerns related to the odorous emissions.

Acknowledgements

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1. LANDFILL ODOURS AND POSSIBILITIES OF ODOUR CONTROL

By the turn of the 20th century, a variety of waste disposal practices were adopted by municipalities, ranging from land disposal, water disposal (including ocean dumping), incineration, reduction, or some combination of methods (Pichtel 2005). Landfills are and will be a significant part of municipal solid waste (MSW) management for the coming years (Benavides and Craik 2003).

Anaerobic degradation of the biodegradable fraction of the landfilled MSW causes several environmental problems such as the production of methane, VOCs, odours and leachate, the presence of vectors (insects, rodents, and birds), public health hazard, explosions and plants toxicity (Scaglia and Adani 2008). All these negative impacts and the long time required to stabilize the materials (after care period) are the major issues that make landfills unsustainable.

Potential hazards for MSW landfill workers include (Gillett 1993): primary human pathogens, secondary pathogens and their toxins, volatile and semi-volatile organic chemicals, persistent, lipophilic organic chemicals, metals, other inorganic materials (e.g., asbestos), organometallics, allergens, corrosive, caustic, explosive, and sharp materials.

People's concerns about various impacts related to municipal solid waste management facilities were analyzed by performing a questionnaire survey and statistical analysis (Matsuto *et al.* 2003). Pollution and health effects, reliability of facility owners, and damage to nature are highly worrying impacts of MSW management facilities. Among the nuisance factors, flies, rodents, and crows were the most annoying. Two alternative disposal methods, incinerator and landfill, were evaluated by using value systems identified by AHP (Matsuto *et al.* 2003). Except for the case of giving a higher weight to incinerators for dioxin, landfills were not preferred. This is because residents had a negative image of landfills in most impact categories.

1.1. CHARACTERISTIC OF LANDFILL ODOURS

Municipal wastes are highly heterogeneous and include durable goods, nondurable goods, packaging and containers, food wastes, yard wastes, and miscellaneous inorganic wastes. For ease of visualization, MSW is often divided into two categories: garbage and rubbish. Garbage is composed of plant and animal waste generated as a result of preparing and consuming food. This material is putrescible, meaning that it can decompose quickly enough through microbial reactions to produce bad odours and harmful gases. Rubbish is the component of MSW excluding food waste, and is nonputrescible. Some, but not all, of rubbish is combustible (Pichtel 2005).

"Landfill" means a waste disposal site for the deposit of the waste onto or into land (i.e. underground), including: internal waste disposal sites (i.e. landfill where a producer of waste is carrying out its own waste disposal at the place of production), and a permanent site (i.e. more than one year) which is used for temporary storage of waste, but excluding: facilities where waste is unloaded in order to permit its preparation for further transport for recovery, treatment or disposal elsewhere, and storage of waste prior to recovery or treatment for a period less than three years as a general rule, or storage of waste prior to disposal for a period less than one year (1999/31/EC; Lietuvos Respublikos… 2000).

Like any other type of waste treatment, landfill should be adequately monitored and managed to prevent or reduce potential adverse effects on the environment and risks to human health (1999/31/EC). Measures should be taken to minimize nuisances and hazards arising from the landfill through: emissions of odours and dust, wind-blown materials, noise and traffic, birds, vermin and insects, formation of aerosols, fires.

The basic unit of a landfill is a "cell", which includes daily deposits of compacted waste and daily layers of cover material (Hilger *et al.* 2009). A cell is typically 3 m high, although heights of 10 m have been employed. Cells typically have a rectangular surface area and steeply sloping sides. Waste is deposited into a cell each day and compacted to 710-950 kg/m^3. At the end of each work day, the waste is covered by soil, which serves to exclude disease vectors, rodents, and rainwater, minimize odour and windborne litter (Rushbrook and Pugh 1999). A given cell is filled to a designated height, after which a new cell is begun. After adjacent cells in a sector are filled to the same height, they are collectively referred to as a lift (Fig. 1.1). A lift is often covered with an additional 15 cm layer of soil or combination of soil and compost that provides a more permanent barrier to odour and storm water. New cells are then established over the intermediate cover until the landfill section has reached a pre-determined height.

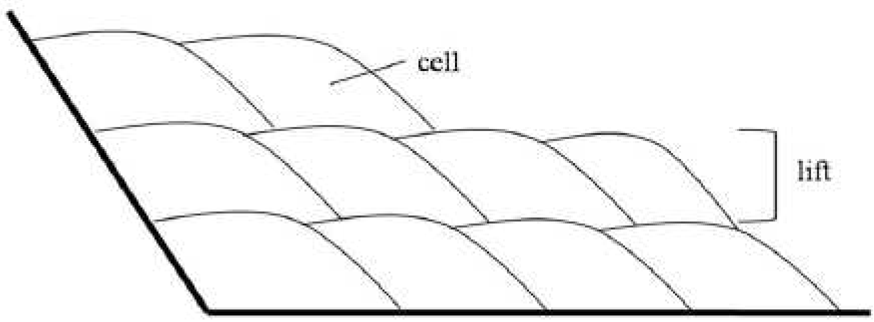

Fig. 1.1. Scheme of typical landfill cell formation (Hilger *et al.* 2009)

After a series of lifts has been completed, but before final capping occurs, it is common practice to place a layer of intermediate cover on top of the cells (Hilger *et al.* 2009). This typically 30 cm layer of soil is seeded for erosion control. A final cover is a highly engineered system that overtops the intermediate cover of a completed landfill sector to minimize infiltration of rain water and dispersion of waste. This cover also aids in the long-term maintenance of the landfill. The final cover typically consists of a gas control layer that routes gas to flares or a collection system, a filter and drainage layer, and a layer of seeded topsoil for erosion control.

Methane, a basic component of the landfill biogas contributes significantly (6-20 %) to its total anthropogenic emission to the atmosphere. Reduction of methane emission from the landfills can be achieved by its practical utilization or by its microbial oxidation in the landfill cover soil layer (Stepniewski and Zygmunt 2003; Abichou *et al.* 2009).

The landfill soil cover is capable to oxidize 40 to 100 % of methane diffusing through it under the considered conditions (Stepniewski and Zygmunt 2003). The degree increases with the potential methanotrophic activity and decreases with the gas diffusion coefficient. Methane and carbon dioxide concentrations in the landfill cover soil are not highly dependent on D/D_0. The amount of methane emitted from the landfill soil cover to the atmosphere increases with D/D_0 but decreases with the increase of q_{max}. Gas diffusion coefficient is a very important soil physical parameter decisive for the methanotrophic capacity of the soil layer covering landfill.

The changes in the hydro-physical properties of a MSW landfill owing to an intermediate soil layer were studied (Olayiwola 2009). Key parameters, including dry density, drainable porosity, and saturated hydraulic conductivity of waste samples with and without an intermediate soil layer were measured in conventional test cells under increasing overburden stresses. The waste-only fill was more permeable than waste incorporating a soil cover; however, this reduced with increased vertical stresses applied to the fills. The measured and calculated values of the saturated hydraulic conductivity of the composite fills differed up to a factor of 100 at low vertical stresses. The moisture routing, undertaken with a modified Hydrologic Evaluation of Landfill Performance (HELP) model, suggests that the use of daily cover soil may reduce leachate drainage, thus increasing the degree of moisture saturation in waste fills.

All aspects of landfill – delivery of waste, the type of waste and its placement, the installation of pollution control infrastructure for LFG and leachate management, operation of pollution control plant, especially LFG flares and gas engines – are potential sources of odour (McKendry *et al.* 2002).

The gaseous compounds emitted from landfills have various impacts on their surroundings and act on different scales, as illustrated by Fig. 1.2 (Fischer *et al.* 1999). In addition to having impacts over a large spatial scale, gaseous emissions also act on different time scales. Odours and dust, for example, are mainly transient phenomena, while some of the anthropogenic trace compounds in LFG may persist and accumulate in organisms or natural ecosystems over very long periods of time.

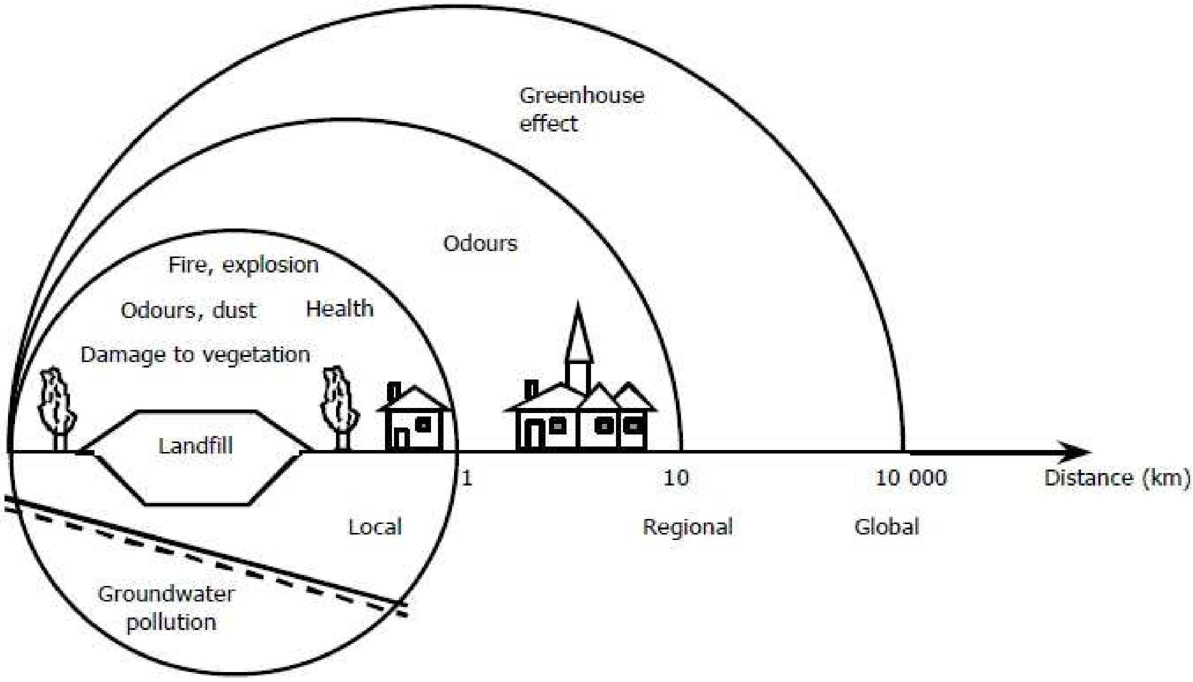

Fig. 1.2. The different scales of the impacts of gas from landfills (Fischer *et al.* 1999)

Organosulphur compounds, i.e. mercaptans and carbon or methyl sulphides, are important contributors to the foul smell of MSW landfill gas (Fischer *et al.* 1999). They most probably arise from the degradation of proteins, which typically form some 6 % of food wastes. Hydrogen sulphide is highly toxic and affects the nervous system. It also has a repugnant odour and is highly flammable. Its odour threshold is comprised between 5 and 40 ppm. Above 50 ppm it paralyses the olfactory system. Concentrations above 400 ppm affect the nervous system and above 700 ppm there is risk of death by respiratory failure.

Odorous compounds in landfill sites are mainly generated by landfill gas (trace compounds), fresh wastes and leachate (Senante *et al.* 2003). Some typical odorous compounds in landfill gas have very low detection thresholds and can be detected by human nose at very low concentrations.

Landfill gas is a complex mixture of many compounds: principal compounds which are odourless (CH_4, CO_2, N_2) and many trace compounds among which some of them are responsible for odours (Senante *et al.* 2003). All these compounds can be classified in the following main families: sulphurous, nitrogenous, aldehydes, acids, ketones, alcohols, aromatics, esters and chlorinated compounds.

The following areas have been identified as odour sources in landfill sites (Senante *et al.* 2003; Sironi *et al.* 2003):

- Tipping area: odours due to waste discharged from trucks and the first operations with engines to lay out fresh wastes.
- Active cells: this source includes the fresh municipal solid waste odours and landfill gas odours due to the beginning of aerobic processes.
- Non active cells: this source is due basically to landfill gas and reveals the limits of cell covers and the collection network efficiency.
- Landfill gas: this term gathers odours from the flare area, odours from leaks or failures of the gas collection network and the cells which are not connected to the collection network.
- Leachate pond: it deals with odours spreading from leachate in general and storage pond in particular, except on site leachate treatment plant emissions.

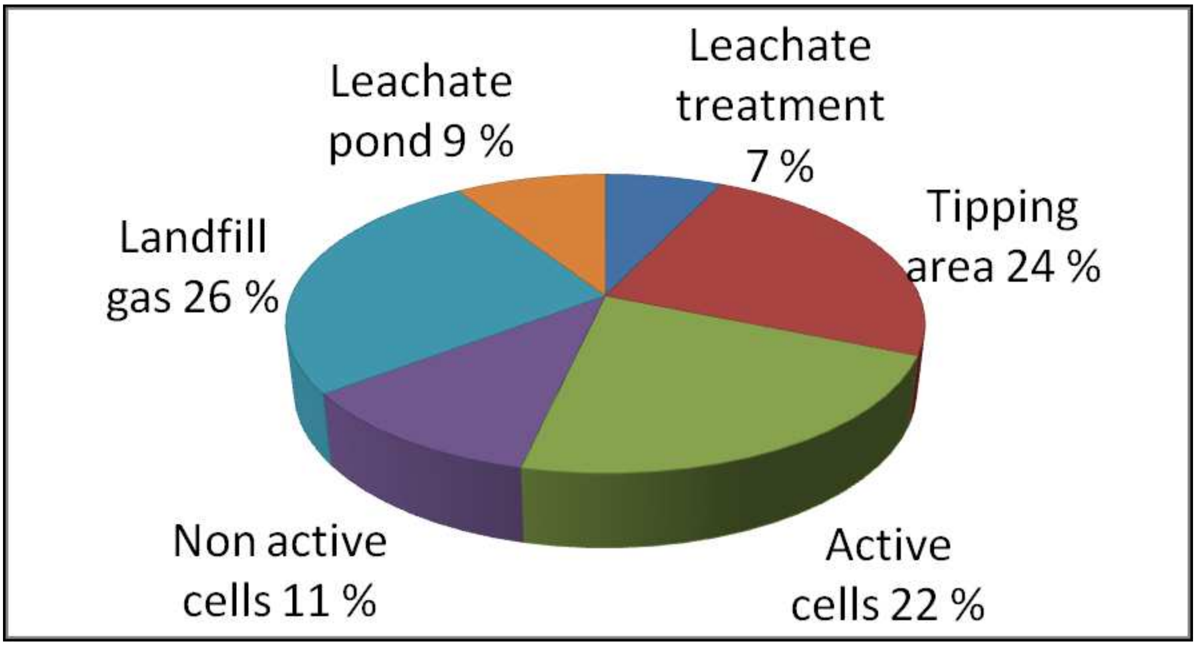

Fig. 1.3. Odour sources in landfill sites (Senante *et al.* 2003)

Fig. 1.3 shows that:

- Odours associated with landfill gas (landfill gas, active cells and non active cells) are the main sources of odours from a sanitary landfill (59 %),
- Odours associated with fresh MSW (unloading area and active cells) represent the second source of odours (46 %),
- Odours associated with leachate (leachate pond and treatment) are the third odour source in landfill sites (16 %).

Waste deposited in a landfill will undergo anaerobic decomposition resulting in generation of landfill gas consisting mainly of methane (55-60 %) and carbon dioxide (40-45 %). Landfill gas contains more than 100 different trace gases including halogenated and aromatic hydrocarbons, sulphur and nitrogen containing compounds. Typical trace gas concentrations are in the range of 10-250 $\mu g/m^3$. The trace components originate from hazardous materials deposited in the landfill or from biological or chemical degradation of materials disposed in the landfill (Scheutz *et al.* 2003).

The potential for natural attenuation of non-methane organic compounds (NMOCs) in landfill covers was investigated in soil microcosms incubated with methane and air, simulating the gas composition in landfill soil covers (Scheutz *et al.* 2003). In total, 18 NMOCs were investigated, including chlorinated methane, ethane, ethene, fluorinated hydrocarbons, and aromatic hydrocarbons. Mass balance calculations using the maximal oxidation rates obtained demonstrated that landfill soil covers have a significant potential for not only methane oxidation but also degradation of selected volatile organics. At old landfills with lower gas production, or at engineered landfills with gas collection systems, methane oxidation and degradation of NMOCs in cover soils may play a very important role in reducing the emission of both methane and trace components into the atmosphere.

Emissions of methane and more than 30 non-methane organic compounds were quantified at two French landfills: Lapouyade (near Bordeaux) and Grand'Landes (near Nantes) (Bogner *et al.* 2003). At Lapouyade, three areas were investigated: the final cover area, the temporary cover area, and a forest control area. At Grand'Landes, emissions from three areas were measured: the final cover area with a geomembrane overlying an innovative gas collection layer, the final cover area, and a field control area. Based on the emission measurements and complementary laboratory experiments, a general coherence was observed between emissions and biodegradability of various NMOCs. The emissions consisted mainly of compounds which are not degradable or slowly degraded under aerobic conditions (CFCs and higher chlorinated compounds), while low to negative emissions were observed for compounds more readily degradable under aerobic conditions (especially the aromatics and lower chlorinated compounds).

Table 1.1. List of parameters selected for completion of exhaustive analyses (Lambolez *et al.* 2003)

Pollutants	Toxicity criteria
Monocyclic aromatic hydrocarbons	Neurotoxicity
Ketones, aldehydes, alcohols	Irritants, neurotoxicity
Cyclic hydrocarbons, phenols, halogen compounds	Neurotoxicity
Polycyclic aromatic hydrocarbons	Diesel emission tracers
Organic sulphur compounds (mercaptans)	Irritants
Mineral and organic acids (HCl, HF)	Irritants
Phtalates	Potential carcinogenicity
H_2S, NH_3	Acute toxicity
CO, CO_2, and O_3	Irritants, acute toxicity
NO_2, NO_x, and SO_x	Chronic respiratory toxicity
Gaseous and particulate metals: As, Pb, Mn, Cd, Ni, Cr	Systemic toxicity
Total (Ø 0,5-20 μm) and alveolar (Ø 0,5-5 μm) dust	Inhalable dust
Asbestos	Carcinogenicity

Potential risk to health from household or industrial waste landfills is an issue of continuing public concern (Lambolez *et al.* 2003). The study implemented on two sites by the French Health and Waste Network includes exhaustive chemical and microbiological analyses of sources, air analyses of selected pollutants and micro-organisms at several places of work, and survey of health parameters on workers. Table 1.1 shows a list of pollutants adopted due to their concentration levels and their toxicity criteria.

The measured concentrations of chemical pollutants in the environment at the two landfill sites monitored are low and below the threshold values (Lambolez *et al.* 2003). Two parameters of concern for workers have been identified because of their capacity to cause respiratory problems and immuno-allergic reactions: the dust and the microorganisms' concentrations. Their spreading outside the site boundaries has not been showed.

Three municipal solid waste landfills in Australia were investigated to assess landfill odour emissions (Bowly 2003). The tools used in this investigation are tools currently used in odour impact assessment, namely dynamic olfactometry, gas chromatography – mass spectrometry (GC-MS) and artificial neural networking (ANN).

The top 10 landfill odorants, based on individual odour threshold estimates, are shown in Table 1.2. This table also separates the tipface and the LFG samples to show the difference in odorous chemicals from each source.

Table 1.2. Contribution of top 10 chemical odorants (Bowly 2003)

Landfill	Percentage odour contribution	Tipface	Percentage odour contribution	Landfill gas	Percentage odour contribution
Ethylbenzene	24.3	2,3-butanedione	31.6	Ethylbenzene	40.8
2,3-butanedione	21.6	Methyl-mercaptan	24.7	Methyl-mercaptan	20.5
Methyl-mercaptan	20.0	Hydrogen sulphide	22.5	Hydrogen sulphide	18.9
Hydrogen sulphide	18.5	Ethylbenzene	11.3	Dimethyl-sulphide	7.5
Dimethyl-sulphide	5.3	Dimethyl-sulphide	3.4	Sulphur dioxide	4.7
Sulphur dioxide	2.0	2-methylpropanal	1.2	Carbon disulphide	2.8
Dimethyl-disulphide	1.9	i-propyl-mercaptan	1.2	Benzene	1.8
Carbon disulphide	1.8	Dimethyl-disulphide	0.7	2,3-butanedione	0.6
Benzene	0.8	Ethylmercaptan	0.7	m,p-xylenes	0.6
i-propyl-mercaptan	0.6	Ethanol	0.5	2-methylpentane	0.3

Odour emission rates were found to be 0.335 OU/m^3/m^2/s for landfill tipfaces and less than 0.002 OU/m^3/m^2/s for the covered landfill surfaces and it was found that landfill gas odour and tipface odour had "distinct" odour intensity at an odour concentration of 1.4 OU and 1.0 OU, respectively. Methyl-mercaptan, ethylbenzene, hydrogen sulphide and dimethyl-sulphide were found to be the primary odorants (Bowly 2003).

Surface emission investigations have been undertaken on two MSW landfills in France (Taramini *et al.* 2003). This study visualizes methane flux cartography of two different sites for different types of cover (Fig. 1.4). High flux locations were identified: sides, slopes, local heterogeneities (wells and edges) and discontinuities in the capping system.

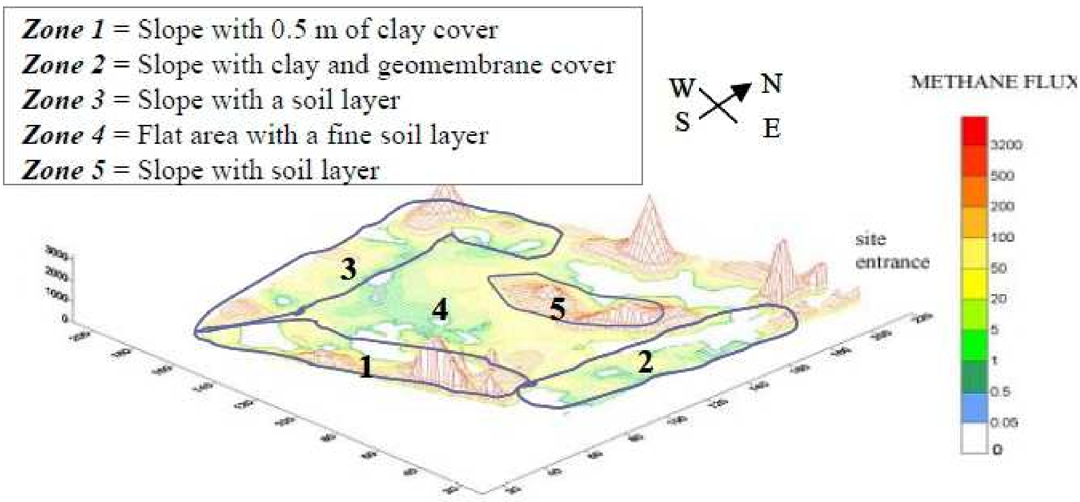

Fig. 1.4. Linear interpolation results and peak locations of methane emissions (Taramini *et al.* 2003)

A linear interpolation of measurements for the whole site shows that emissions are localised on the slopes, at the end of the LFG collection system, on the edge due to the "edge effect", and on the limits separating two different covers (Taramini *et al.* 2003).

Table 1.3. Emissions of the different capping systems (Taramini *et al.* 2003)

Areas	CH_4 emission in $m^3/h/ha$	Collection rates
Slope with clay	61 (no liner influence)	0.78
Slope with clay and liner	28	0.90
Slope with a fine soil layer	85	0.69
Plateau with a fine soil layer	75	0.73

Collection rate for each cover type (Table 1.3) is difficult to determine because of the different waste ages, LFG collection system efficiencies and covers thickness (Taramini *et al.* 2003). Nevertheless the assessment of methane emissions through cover with different methods was possible.

H_2S production from landfill with and without added sulphur from plasterboard, the influence of water content in the waste and the effectiveness of different reactive layers to remove H_2S have been studied (Haarstad *et al.* 2003). The lab experiments were conducted in 30 l plastic containers with controlled water level. The organic waste produced H_2S in the order of 40 ppm over a period of 80 days. When plasterboard was added, the H_2S production with a high water level increased to 800 ppm and with low water content to about 100 ppm. The experiments show that H_2S production is significantly increased if the waste is rich in sulphate under water saturation.

Odorous compounds from a landfill have been characterised by gas chromatography – mass spectrometry, identifying about 100 volatile organic compounds (Davoli *et al.* 2003). Since the perception levels of the human nose are often very low, at ppt level, analytical techniques are not sensitive enough to ensure direct detection of all malodour compounds at such levels (Young and Parker 1983). As a consequence, a preconcentration step is required. Among the methods available for trace enrichment from air samples, solid-phase microextraction (SPME) is a fast, inexpensive, and solvent free sample preparation technique (Martos and Pawliszyn 1997).

Air samples coming from the most significant emission sources of the landfill have been analysed by GC-MS after SPME preconcentration, and the chromatographic data have been submitted to chemometric analysis in order to identify specific markers of the emission sources. Fig. 1.5 shows the total ion chromatogram for 13 standards chosen as representative of VOC emissions from MSW landfill.

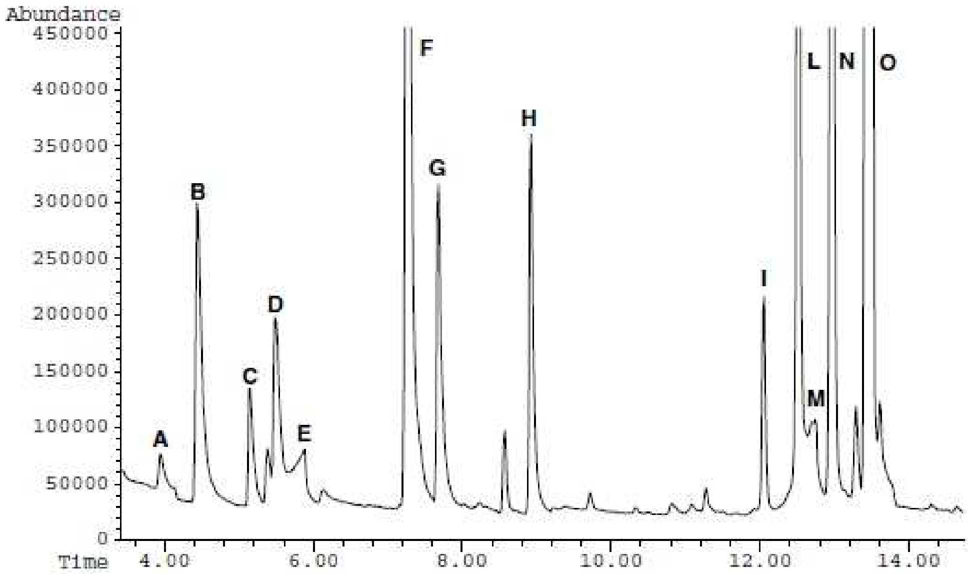

Fig. 1.5. Chromatogram obtained for 13 VOC standards at the concentration of 50 ppt: A - dimethyl disulfide; B - toluene; C - 2-hexanone; D - ethyl butanoate; E - butanoic acid; F - internal standard; G - hexanal; H - trichloroethylene; I - α-pinene; L - ethyl hexanoate; M - hexanoic acid; N - limonene; O - α-terpinene (Davoli *et al.* 2003)

The automation of sampling and analysing procedures was realised with a microgas chromatograph (µGC Agilent P200). The peak area trend vs. time obtained for all the identified compounds is shown in Fig. 1.6. For the most abundant ones, limonene, toluene, p-cymene, α-pinene, and xylenes, a quantitative determination was carried out to estimate their concentrations (Davoli *et al.* 2003).

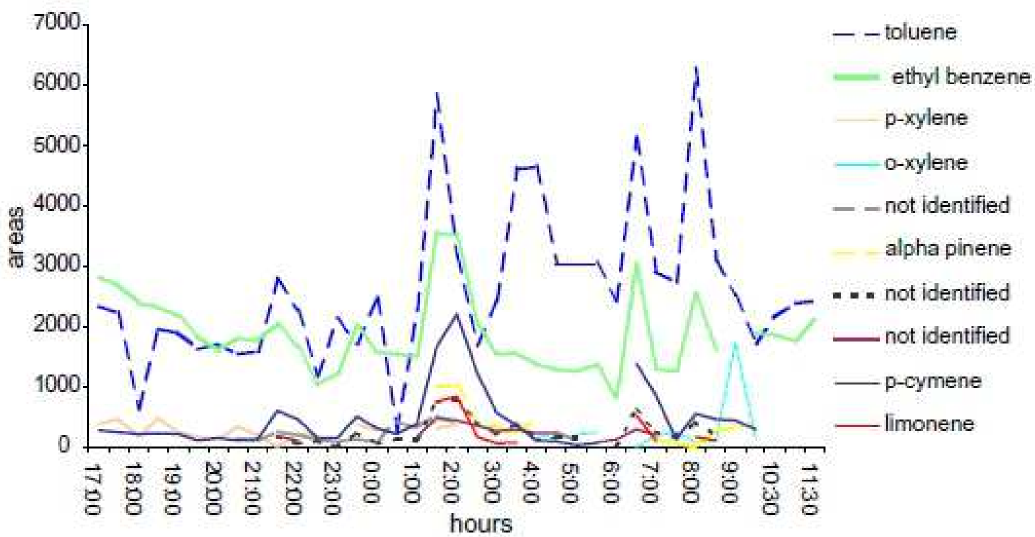

Fig. 1.6. Peak area trend vs. time for the compounds detected by the on-site system (Davoli *et al.* 2003)

However, instrumental characterisation of environmental odour annoyance is still an open field not only because of the complex relationship between odorants and individual odour perception, but also because of the instrumental limitations, such as inability to detect reactive inorganic gases, sensitivity problems, total number of compounds and degradation in the wind plume (Davoli *et al.* 2003).

The composition of odorous gases emitted from a municipal landfill in Turkey was investigated using gas chromatography – mass spectrometry (Dincer *et al.* 2006). Several volatile organic compounds were identified and quantified at five sampling sites in May and September 2005 (Fig. 1.7).

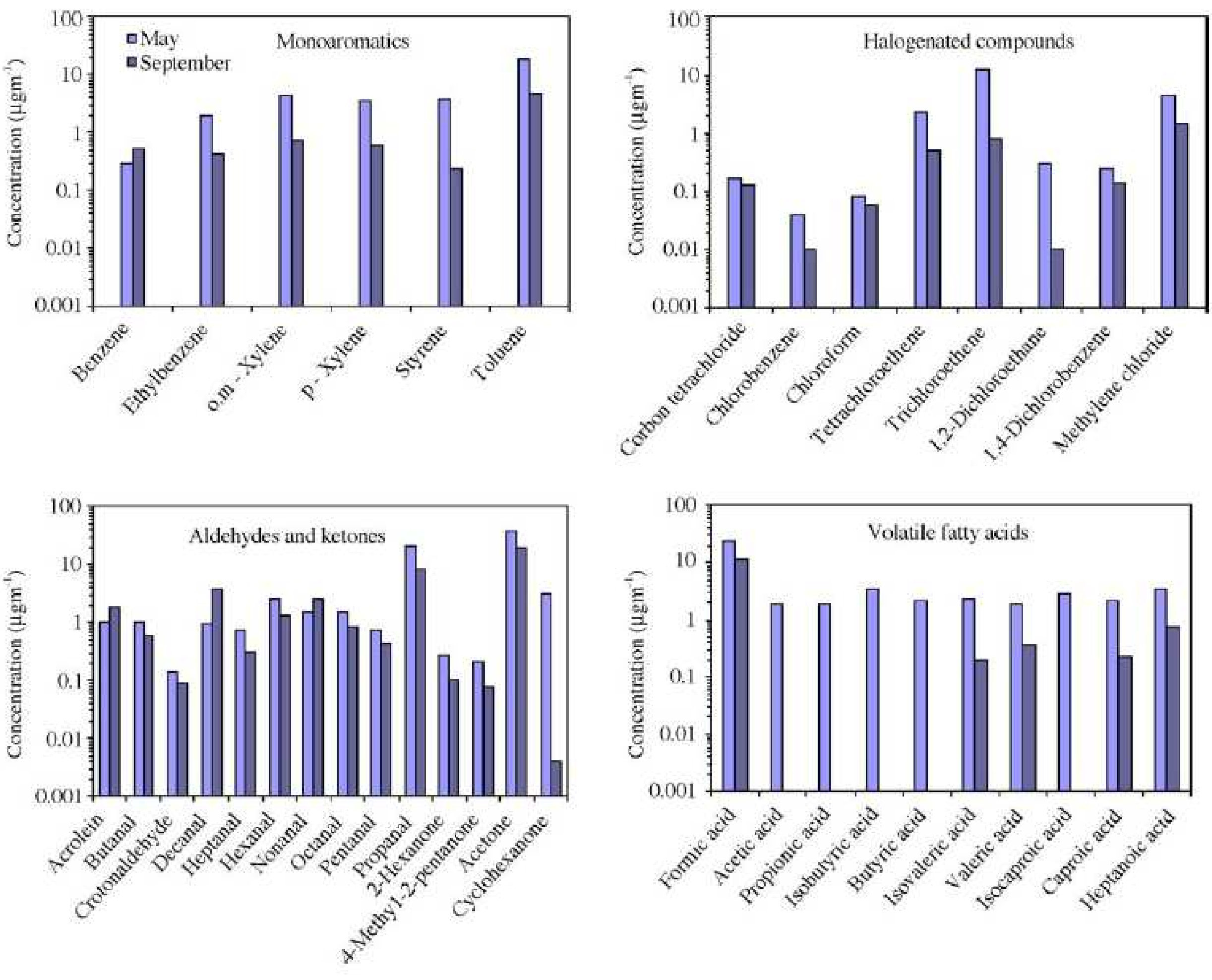

Fig. 1.7. Seasonal variations of concentrations (Dincer *et al.* 2006)

Detected VOCs were monoaromatics (0.09-47.42 µg/m^3), halogenated compounds (0.001-62.91 µg/m^3), aldehydes (0.01-38.55 µg/m^3), esters (0.01-7.54 µg/m^3), ketones (0.03-67.60 µg/m^3), sulfur/nitrogen containing compounds (0.03-5.05 µg/m^3), and volatile fatty acids

(0.05-43.71 μg/m^3). High levels of aldehydes (propanal up to 38.55 μg/m^3) and VFAs (formic acid up to 43.71 μg/m^3) were measured in May. However, VOC concentrations were relatively low in September (Dincer *et al.* 2006).

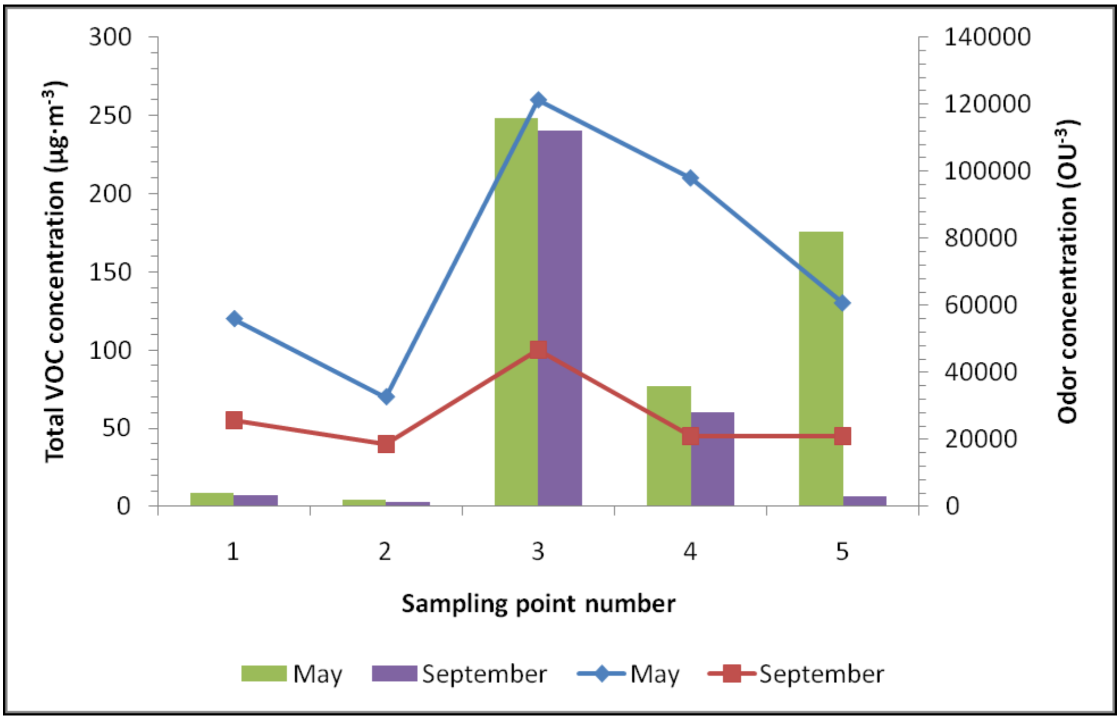

Fig. 1.8. Variations of odour and total VOC concentrations (Dincer *et al.* 2006)

A statistically significant linear relationship was found between odour concentrations determined by olfactometry and total VOC concentrations (Fig. 1.8). The relationships of odour concentrations with the different groups of chemicals were also examined and it was found that the concentrations of aldehydes, ketones, and esters are the best estimators, explaining 96 % of the variability in odour concentrations (Dincer *et al.* 2006).

Gaseous emissions from a landfill in the Athens area were studied (Loizidou and Kapetanious 1991; Parker *et al.* 2002). Mercaptans (organosulphur compounds) were noted over the odour limit at the site boundary. Odour thresholds (Table 1.4) for possible trace components in landfill gas were reported by Ruth (1986).

The Jerubaičiai landfill in Lithuania was selected for the investigation of H_2S emissions (Vasarevičius 2011). It was determined that the amounts of H_2S varied from 0.9 ppm (in February) to 8.6 ppm (in August) in different places of the landfill. The largest amounts of H_2S were identified in the areas of freshly-tipped waste. H_2S formation is influenced by aerobic and anaerobic conditions (Zdeb *et al.* 2008). H_2S generation in the top layer of waste is mainly influenced by air temperature.

Table 1.4. Human detection limits of odorous gases associated with landfills (Ruth 1986)

Compounds	Formula	Odour (Description)	Detection Limits ($\mu g/m^3$)	Detection Limits (ppb_v)	Boiling Point (°C)	Molecular Weight (g)
Sulphur Compounds						
Hydrogen sulphide	H2S	Rotten eggs	0.7	0.5	-60.7	34.1
Carbon disulfide	CS2	Disagreeable, sweet	24.0	7.7	46.3	76.1
Dimethyl sulphide	CH3-S-CH3	Rotten cabbage	2.5	1.0	37.3	62.1
Dimethyl disulfide	(CH3)2S2	Rotten cabbage	0.1	0.026	109.7	94.2
Dimethyl trisulfide	(CH3)2S3	Rotten cabbage	6.2	1.2	165.0	126.2
Methyl mercaptan	(CH3)SH	Rotten cabbage	0.04	0.02	6.2	48.1
Ethyl mercaptan	CH3CH2-SH	Rotten cabbage	0.032	0.01	35.0	62.1
Allyl mercaptan	CH2=CH-CH2-SH	Garlic coffee	0.2	0.1	NA	74.15
Propyl mercaptan	CH3-CH2-CH2-SH	Unpleasant	0.2	0.1	NA	76.16
Amyl mercaptan	CH3-(CH2)3-CH2-SH	Putrid	0.1	0.02	NA	104.22
Benzyl mercaptan	C6H5CH2-SH	Unpleasant	1.6	0.3	NA	124.21
Thiophenol	C6H5SH	Putrid garlic	1.2	0.3	NA	110.18
Sulphur dioxide	SO2	Irritating	1175.0	449.3	NA	64.07
Carbon oxysulphide	COS	Pungent	NA	NA	-50.2	60.1
Nitrogen Compounds						
Ammonia	NH3	Pungent, sharp	26.6	38.3	-33.4	17.0
Aminomethane	(CH3)NH2	Fishy, pungent	25.2	19.5	-6.3	31.6
Dimethylamine	(CH3)2NH	Fishy, amine	84.6	46.0	7.4	45.1
Trimethylamine	(CH3)3N	Fishy, pungent	0.1	0.046	2.9	59.1
Skatole	C6H4C(CH3)CHNH	Feces, chocolate	0.00004	0.00001	265.0	131.1
Volatile Fatty Acids						
Formic	HCOOH	Biting	45.0	24.0	100.5	46.0
Acetic	CH3COOH	Vinegar	2500.0	1019.1	118.0	60.1
Propionic	CH3CH2COOH	Rancid, pungent	84.0	27.8	141.0	74.1
Butyric	CH3(CH2)2COOH	Rancid	1.0	0.3	164.0	88.1
Valeric	CH3(CH2)3COOH	Unpleasant	2.6	0.6	187.0	102.1
Ketones						
Acetone	CH3COCH3	Sweet, minty	1100.0	463.9	56.2	58.1
Butanone	CH3COCH2CH3	Sweet, minty	737.0	250.4	79.6	72.1
2-pentanone	CH3COCH2CH2CH3	Sweet	28000.0	7967.5	102.0	86.1
Acetaldehyde	CH3CHO	Green sweet	0.2	0.1	20.8	44.1
Methanol	CH3OH	Alcohol	13000.0	9953.1	NA	32.0
Ethanol	CH3CH2OH	Alcohol	342.0	342.0	NA	60.0
Phenol	C6H5OH	Medicinal	178.0	46.0	181.8	94.1

The presence of odour in the landfill depends on MSW composition, MSW putrescence and biogas production (Scaglia and Adani 2008). The anaerobic biodegradation of MSW determines the formation of odorous molecules (mercaptans, thiophenols, thioalcohols, thioacids, and aliphatic amines) that during the biogas production are emitted. The odour concentration depends on the landfill's age and microbial populations (Young and Parker 1984).

1.2. LANDFILL COVERS IN TERMS OF ODOUR REDUCTION

The use of natural materials in landfill covers results in high costs and strain on the environment through the exploitation (Travar *et al.* 2005).

An option may be to use secondary construction materials such as ashes, bed sand and sludge (Travar *et al.* 2005). Results of the first year monitoring of the full-scale landfill cover built from secondary construction materials illustrate how the construction materials impact on the quality of infiltrating water as it passes through the layers of the cap. Both leachate and drainage water need treatment before discharge into the local recipient. Nitrogen is identified as one of the major pollutants in both leachate and drainage water. The drainage water also contained elevated concentrations of Cu, Ni and Zn. Cd, Cu, As and Ni are the most important pollutants in the leachate that has passed through fly ash, while Cd and Cu are the main pollutants in the leachate that has passed the mixture of bottom ash and clay.

A complex series of biological and chemical reactions begin with the burial of MSW in a landfill (Benavides and Craik 2003). The effects of four different soils on the anaerobic degradation of municipal solid waste and the development of methanogenesis were tested. Four soils (a clay-loam, an organic rich peat, a well limed sandy soil and a chalky soil) were used. The incorporation of soil into MSW had a significant effect on buffer capacity. Over the acidic range buffer capacities of the mixtures were much greater than values measured for MSW and soils separately. Results show that, in terms of pH, buffer capacity, VFA turnover, the inclusion of peat and chalky soil has a significant effect to methanogenesis from MSW.

Lab-scale tests were set up at the IMAGE Department of the University of Padua to investigate the effectiveness of different kinds of landfill cover soil in the removal of methane and sulphurated compounds from a biogas stream (Cossu *et al.* 2003). The following materials considered as possible landfill cover soil were employed: mechanical-biological pretreated municipal solid waste (MBP); mechanical-biological pretreated biowaste (PB); fine (PBS_f) and coarse (PBS_c) mechanical-biological pretreated biowaste and sewage sludge mixtures; natural soil usually used as landfill cover (NS).

The maximum methane oxidation rate was observed for MBP (Table 1.5). Similar results were obtained for PB and PBS_c. Natural soil showed lower oxidation rate. Hydrogen sulphide was completely adsorbed by activated carbon membranes. Activated carbon membranes do not influence the methane oxidation process. Carbonyl sulphide and ethyl mercaptan, present in the biogas in concentrations up to 0.7 ppm resulted lower than instrumental limits (0.01 ppm). The same happened for the isopropyl mercaptan, present in the biogas in concentrations up to 4.7 ppm and instrumental limit (0.1 ppm). Probably they are absorbed by filling materials (Cossu *et al.* 2003).

Table 1.5. Minimum, maximum and mean methane oxidation rates, $gCH_4/m^2/h$ (Cossu *et al.* 2003)

Material	Minimum oxidation rate	Maximum oxidation rate	Mean oxidation rate
MBP	6.9	17.4	10.3
PB	5.7	13.7	9.5
PBS_f	0.9	8.1	3.6
PBS_c	2.7	14.3	8.8
NS	4.1	8.9	6.7

The adsorption characteristics of H_2S on waste biocover soil, an alternative landfill cover, were investigated (He *et al.* 2010). The results showed that the adsorption capacity of H_2S increased with the reduction of particle size, the increase of pH value and water content of waste biocover soil. The optimal composition of waste biocover soil, in regard to operation cost and H_2S removal performance, was original pH value, water content of 40 % and particle size of ≤4 mm.

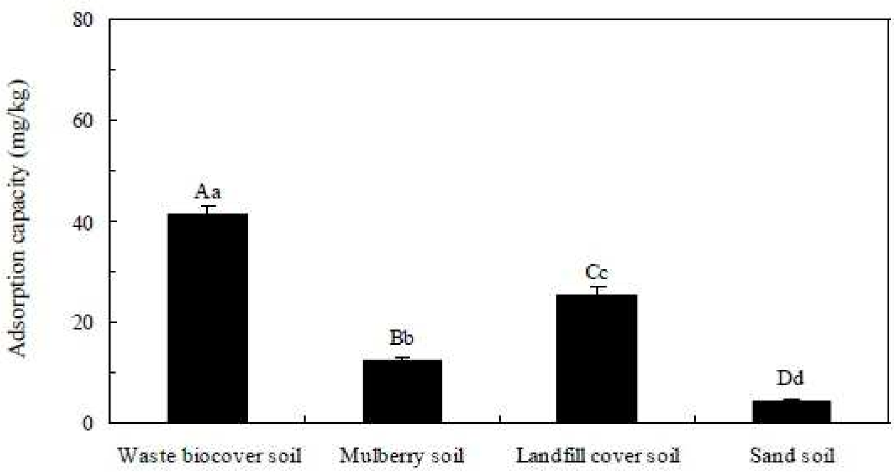

Fig. 1.9. The adsorption capacity of H_2S on different soils (He *et al.* 2010)

The adsorption capacity of H_2S on waste biocover soil with optimal composition reached the maximum value of 59.6±1.3 mg/kg at oxygen concentration of 10 % (He *et al.* 2010). Among the four experimental soils, the highest adsorption capacity of H_2S was observed on waste biocover soil, followed by landfill cover soil, mulberry soil, and sand soil, which was only 9.8 % of that of waste biocover soil (Fig. 1.9).

The suitability of construction and demolition fines, commercial and industrial fines and wood chip as potential landfill cover materials in terms of odour control was assessed (Solan *et al.* 2010). Background odour analysis was conducted to determine if any residual odour was emitted from the cover types. The odour reduction performance of each of the materials was also examined on an area of an active landfill site. The background odour emissions from construction and demolition fines, commercial and industrial fines and wood chip were considered negligible, even when moisture content was varied. Results indicate that the 200 mm deep layer of construction and demolition fines and wood chip is adequate for odour abatement.

The attenuation potential of methane and selected volatile organic compounds of four types of compost materials were compared in dynamic laboratory column experiments over a period of 255 days (Scheutz *et al.* 2005). The column containing compost made of garden waste mixed with wood chips showed the highest steady state methane oxidation rate of 161 $g/m^2/d$. All the tested VOCs were degraded. Overall the highest removal of VOCs was observed in the column containing the compost/wood chips mixture. This study demonstrates that biocovers consisting of compost materials have a potential of attenuation of trace gas emissions from landfill sites.

The ability of municipal waste compost as a daily cover material to reduce the odorous emissions associated with landfill surfaces was investigated (Hurst *et al.* 2005). Gas samples were taken from the inlet, outlet and at varying column depths and examined using a combination of sensory analysis (olfactometry) and a novel analytical method (Transportable Selected Ion Flow Tube – TSIFT). Results of the trials using landfill gas showed a 69 % odour reduction through the column for compost with a bulk density of 590 kg/m^3, and a 97 % reduction using compost with a bulk density of 740 kg/m^3. TSIFT analysis showed an overall decrease in the concentration of terpenes and sulphurous compounds in the outlet gas from the column for both bulk densities.

The feasibility of immobilizing methane oxidizing bacteria into a tarp-like matrix for alternative daily cover at open landfill cells to prevent methane emissions was assessed (Hilger *et al.* 2009). Prototype biotarps made with geotextiles plus adsorbed methane oxidizing bacteria were tested for their responses to temperature, intermittent starvation, and washing (to simulate rainfall). While laboratory landfill simulations showed that four-layer composite biotarps made with two different types of geotextile could remove up to 50 % of influent methane introduced at a flux rate of 22 $g/m^2/d$, field experiments did not yield high activity levels.

An innovative synthetic paste of waste tire chips and paper sludge was developed for landfill daily cover applications. The engineering properties and behaviours of the proposed paste were studied through a series of laboratory tests. When compared to traditional soil covers, the paste was 2-3 times lighter in weight, at least two orders of magnitude more impermeable and comparable in shear resistance. An optimal proportion of tire chips in the paste is about 55 %. The environmental benefits of the paste were demonstrated using column tests. An equilibrium deterministic transport model was used to fit the transport parameters from the breakthrough curves of Pb. Analysis of the effluent concentrations displayed retardation effect in all cases, with fitted retardation factors from 19.9 to 59.0 (Wai 2008).

Odour control performance can be quantified via the ASTM E679 procedure (Kittle 1993). In conjunction with the EPA Flux Chamber Technology, the ASTM technique can yield information on both odour control as well as total non-methane hydrocarbon control. These two testing procedures have been used to quantify odour and non-methane hydrocarbon control performance for Rusmar long duration foam, conventional soil cover, and three commonly used tarpaulins/geotextiles (Table 1.6).

Table 1.6. Odour and hydrocarbon emission control for various ADC materials (Kittle 1993)

Cover Material	**Measurement Timing**			
	Immediately		**Next Day**	
	Odour, %	**NMHC, %**	**Odour, %**	**NMHC, %**
Rusmar Foam (15 cm)	98	100	99	100
Soil (23 cm)	99	93	99	93
Griffolyn	99	100	99	98
Air Space Saver	100	100	99	98
Fabrisoil	82	0	82	0

All alternate daily cover materials claim to control odours; however, some vendors do not have evidence to support the claim (Kittle 1993).

The ability of M1 steel, a by-product from shredded tires, iron rich soil, wood mulch, and compost to attenuate odour of landfill gas was investigated (Anunsen 2007). The most efficient filter design observed for reducing H_2S was the open filter. Results indicate that an opened 5-gallon bucket filled with rusted M1 steel could attenuate 43 grams of sulfur. Red soil, wood mulch, and compost are materials that are appropriate for interim covers for landfill slopes. Interim covers are eventually covered with more soil and a geomembrane liner for final closure. It was observed that red soil has great potential to reduce H_2S in landfill covers.

Alternative final covers evaluated for the Lebec landfill included evapotranspirative, geomembrane, geosynthetic clay liner, and asphalt cement concrete configurations (Kavazanjian and Dobrowolski 2003). These configurations were evaluated with respect to short and long term performance, compatibility with the post-closure use, regulatory and community acceptance, and cost. The Lebec landfill is located about 20 km north of Los Angeles.

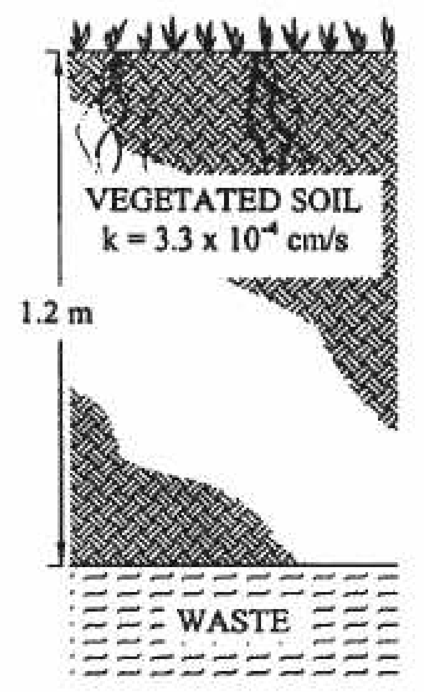

Fig. 1.10. Evapotranspirative final cover (Kavazanjian and Dobrowolski 2003)

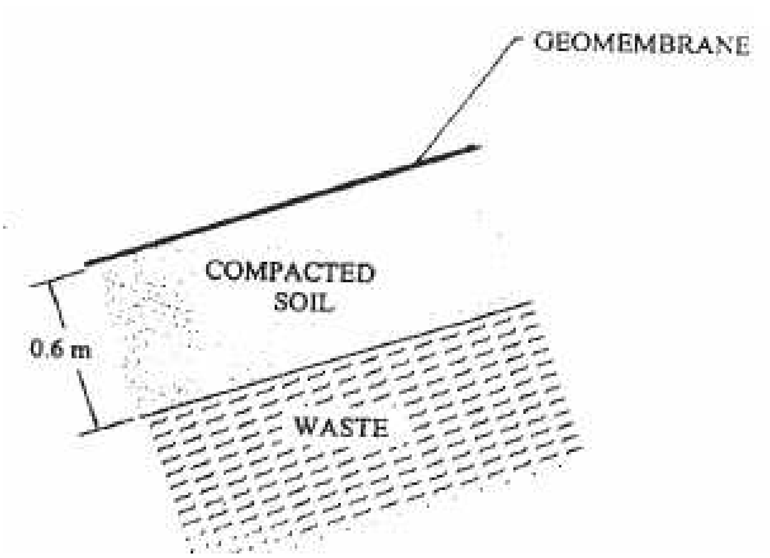

Fig. 1.11. Exposed geomembrane cover (Kavazanjian and Dobrowolski 2003)

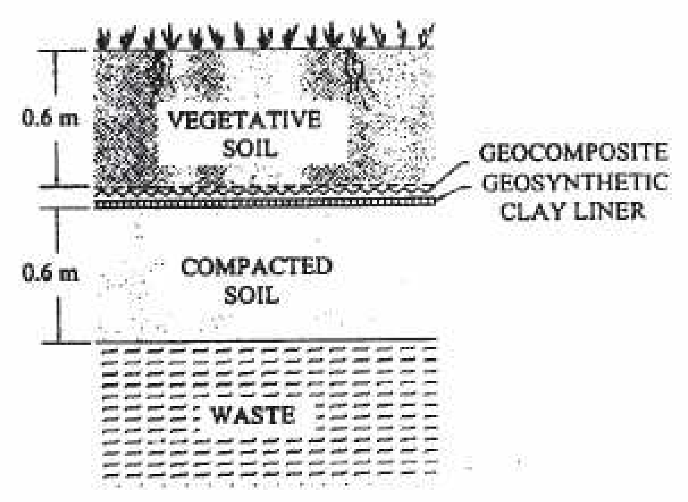

Fig. 1.12. Geosynthetic clay liner cover (Kavazanjian and Dobrowolski 2003)

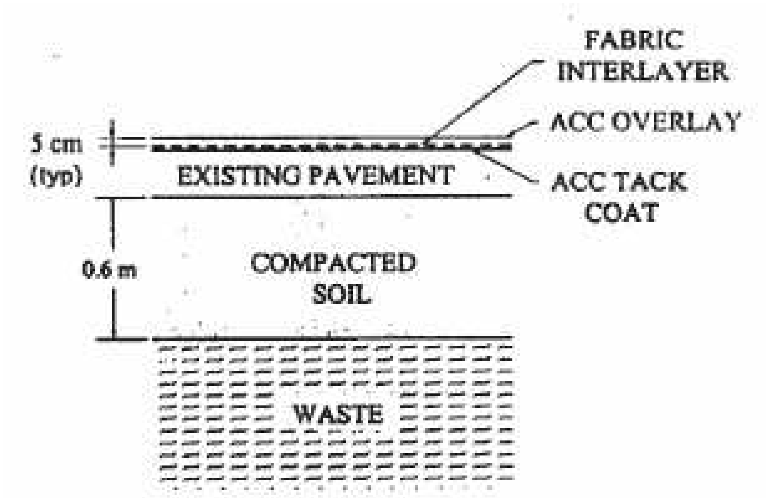

Fig. 1.13. Asphalt cement concrete cover (Kavazanjian and Dobrowolski 2003)

Though an evapotranspirative final cover (Fig. 1.10) constructed using on-site borrow soil was the most cost effective option, it was unacceptable based upon regulatory approval considerations. Community acceptance concerns due to aesthetic considerations ruled out an exposed geomembrane final cover (Fig. 1.11). Therefore, the geosynthetic clay liner configuration (Fig. 1.12) was employed over most of the landfill. However, an asphalt cement concrete cover (Fig. 1.13) was employed in areas designated for transfer station operations in the post-closure period (Kavazanjian and Dobrowolski 2003).

ET covers are also known as store and release covers, vegetative covers, sponge and pump covers, alternative final covers, alternative final earthen covers, and other names (Evapotranspiration… 2010). They include various combinations of earthen materials and plants, and generally can be categorized into the following cover types:

- Monolithic (Fig. 1.14): any precipitation water is stored in a layer of soil and later removed through evapotranspiration.
- Capillary break (Fig. 1.15): this cover uses a two layer system to increase the water storage capacity of the cover. A capillary break is formed by two layers – a layer of fine soil over a layer of coarser material (e.g. sand or gravel). Capillary force causes the layer of fine soil overlying the coarser material to hold more water than if there were no change in particle size between the layers.
- Dry barrier: the dry barrier cover uses wind-driven airflow through the layer of coarse material to remove water from a storage layer.

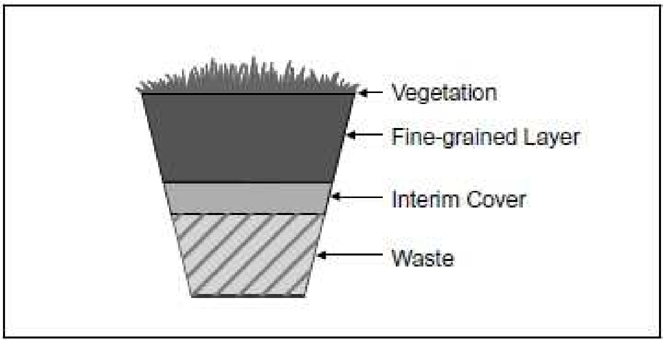

Fig. 1.14. Design of a monolithic barrier ET final cover (Evapotranspiration… 2003)

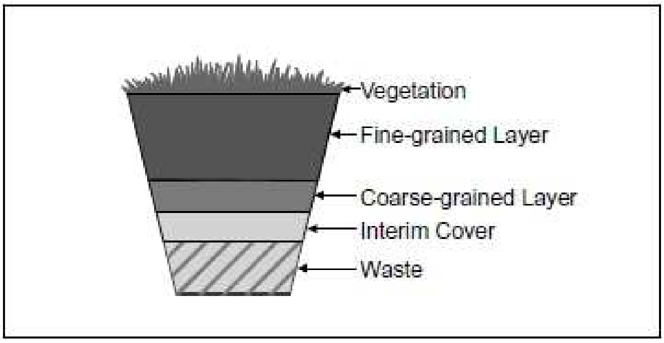

Fig. 1.15. Design of a capillary barrier ET final cover (Evapotranspiration… 2003)

Capillary barriers for landfills and waste dumps are an alternative to conventional surface sealing systems (Wawra and Holfelder 2003). A layer of coarse material beneath the top cover layer is needed for gas distribution. The sealing effect is based on the principal of unsaturated flow and permeability for gas. The influence of hydraulic stress variations on the gas permeability and gas distribution capacity of capillary barriers has been investigated in laboratory tests and supplementing numerical simulations (Fig. 1.16). The model TOUGH2 has been used as numerical simulator for flow of multicomponent and multiphase fluids in porous media, to investigate the flow processes in the capillary barrier and the top cover layer.

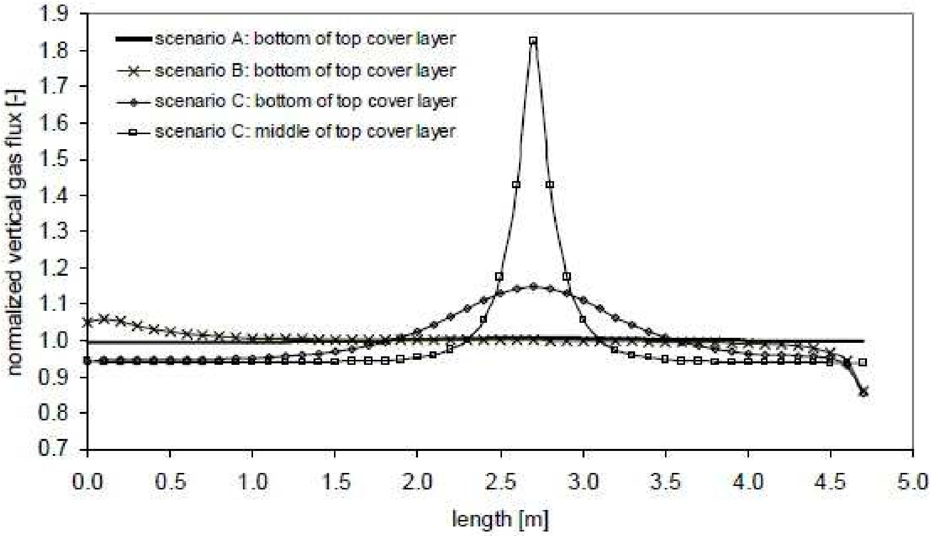

Fig. 1.16. Simulated distribution of vertical gas flow through the cover layer (Wawra and Holfelder 2003)

Independent of the hydraulic stress (scenario A – low hydraulic stress in the capillary layer, and B – high hydraulic stress in the capillary layer) of the capillary layer there is a well distributed vertical flux at the bottom of the top cover layer. Gas flow within the landfill covers are strongly depending on their soil properties. As shown in scenario C (preferential pathway in the top cover) a macro pore in the top cover layer causes a concentration of gas flow, which may lead to a hot spot and an inefficient methane oxidation (Wawra and Holfelder 2003).

Capillary barriers are a suitable sealing system in combination with a top cover layer for methane oxidation. No extra gas distribution facilities are required.

Evapotranspirative final covers offer environmentally superior, cost effective alternatives to prescriptive barrier layer final covers for municipal solid waste landfills in arid and semi-arid climates (Hadj-Hamou and Kavazanjian 2003). Water balance analyses demonstrate that a properly configured ET cover can provide superior resistance to percolation of surface water into the waste compared to prescriptive barrier layer covers. An ET cover is also more resistant to cracking due to differential settlement and desiccation and less likely to induce gas migration problems than a prescriptive cover. Evaluation of the measured moisture contents show that, generally, there is less than 5 percent change in the relative volumetric moisture content near the bottom of the evapotranspirative soil cover compared to nearly 90 percent near the surface, suggesting that most of the water infiltrating into the cover is removed by evaporation and transpiration and does not percolate through the cover into the waste.

The successful landscape integration of a landfill generally begins with the agronomic restoration of the local topsoil used on site (Morcet *et al.* 2003). The objective is to provide to the plants all the nutritive and structural elements necessary to their good development. The Environment, Energy and Waste Research Center has run an experimental program aimed at using green and MSW compost as a topsoil on two French landfills in order to test growing media. The two case studies have been monitored for three years and a comparison between all the created growing media was assessed. The addition of 20 % to 40 % volume of green compost to the local sandy soil increases its stability, its organic content (from 1 % to 3-4 % dry matter) and its fertiliser content. The incorporation of compost does not disturb the composition of the gas phase of the growing media and does not have any influence on the conditions of development of the plants' root system.

Different ashes were tested in the laboratory alone and in various combinations with clay minerals (Tham *et al.* 2003). Based on this, a selection of lining mixtures have been proposed and tested on a 4 ha large testing area at the Tveta landfill in Sweden. The landfill cover construction also included various waste materials in other functions. The Tveta landfill is situated approximately 50 kilometres south-west of Stockholm. The composition of the layers in the cover system is shown in Fig. 1.17. The thickness of the layers and the materials used are given in Table 1.7.

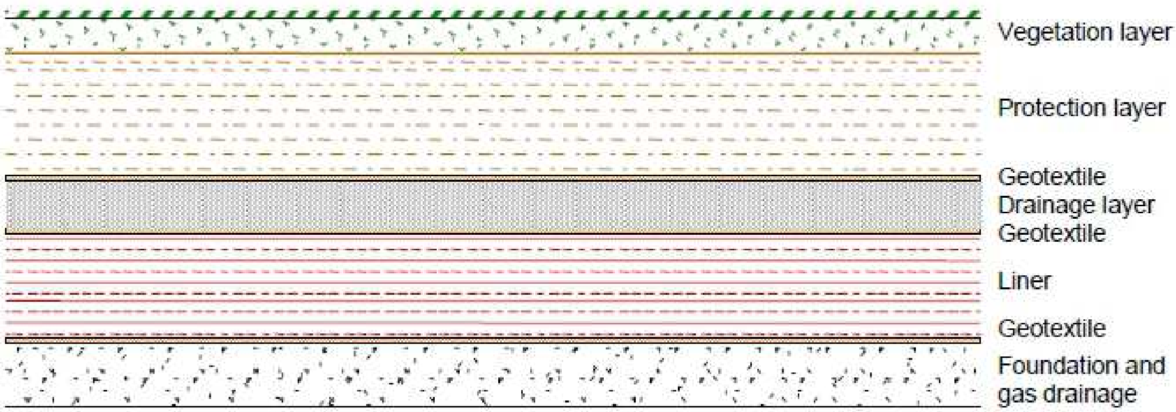

Fig. 1.17. Layer composition for the ash based cover system (Tham *et al.* 2003)

Ashes and a variety of other waste materials can potentially be used in landfill cover systems (Tham *et al.* 2003). Both bottom ash and fly ash can function as liner materials under certain conditions. The chemical interaction of materials and the resilience to differential settlement are among the most critical factors with regard to the long term integrity of the liners and the overall function of the capping system.

Table 1.7. Thickness of the layers in the cover system and materials used (Tham *et al.* 2003)

Layer	Thickness	Materials used
Vegetation layer	0.3 m	Compost, treated soil
Protection layer	≥ 1.5 m	Sewage sludge, foundry sand, crushed and sieved slag, wood chips
Geotextile	0.2 mm	Geotextile
Drainage layer	0.3-0.5 m	Bottom ash (non-capillary), broken glass, gravel
Liner	≥ 1.0 m	Fly ash, bottom ash mixed with clay
Foundation layer	> 0.3 m	Bottom ash (without fine fraction)

Landfills in Texas have the potential to consume millions of pounds of tire shreds by spreading a layer of tire shreds over the facility's waste (Using … 1999). Tire shreds can also be used as part of the leachate collection system, alternative of protective cover over landfill liner, in the primary drainage layer for a liner system, in the drainage layer within final cover system, and media within landfill gas vents.

Research was performed to evaluate the performance of various cover materials as control measures for H_2S emissions from C&D debris landfills (Plaza *et al.* 2007). Five different cover materials were placed on top of the waste inside laboratory-scale simulated landfill columns: (1) sandy soil, (2) sandy soil amended with lime, (3) clayey soil, (4) fine concrete (particle size less than 2.5 cm), and (5) coarse concrete (particle size greater than 2.5 cm). H_2S concentrations measured from the middle of the waste layer ranged from 50000 to 150000 ppm. The sandy soil amended with lime and the fine concrete were the most effective for the control of H_2S emissions. Both materials exhibited reduction efficiencies greater than 99 %. The coarse concrete was found to be the least efficient material as a result of its large particle size.

Sand, gravel, wood-pulp with polymers, construction and demolition debris are the most widely used daily cover materials in Lithuanian landfills (Dumpių… 2007).

Emissions of the hydrogen sulphide whose odour threshold equals 0.012-0.03 mg/m^3 are reduced with the help of crushed wood bark (Zigmontienė and Zuokaitė 2010). H_2S emissions from compost mixed with bark are lower than those from compost covered with bark.

Where biodegradable waste is layered using expanded clay, methane emissions are lower compared to pure waste (Baltrėnas *et al.* 2005). The amount of oxygen decreases slowly and the concentrations of generating CH_4 fall due to air spaces.

1.3. ODOUR EVALUATION TECHNIQUES

The human perception of odours consists of more than just "smell", it represents a complex series of psychological and physiological responses to the quality of the odorant detected (Kehoe *et al.* 1996).

Controlling odours from landfill sites has become an important regulatory issue, requiring accurate and reproducible sampling and measurement (Nicolas *et al.* 2006). But the monitoring of the odour annoyance generated by a landfill area is difficult. Many authors mention that the main landfill odour problem is caused by the handling of fresh waste (Karnik and Parry 2001; Stretch *et al.* 2001). As this is an intermittent activity, the sampling of the gas that is emitted at the landfill working face is particularly problematic.

Two kinds of approach are available for odour detection: the analytical methods (chemical analysis, electronic nose) and the sensorial method of olfactometry which measures the human odour perception (Odorizzi *et al.* 2003). The application of olfactometry to assess the odour impact of waste treatment plants showed that it was possible to estimate the total emission rate in terms of odour concentration (Table 1.8) expressed as odour unit per hour (OU/h) and to use this value as input for the dispersion models applied to estimate the environmental impact on the considered area.

Table 1.8. Results of the olfactometric analysis and emission rate calculation (Odorizzi *et al.* 2003)

Emission source	Velocity of odour emission (m/s)	Odour concentration (OU/Nm^3)	Emission rate (OU·10^6/h)
Biogas collector 1 Biogas collector 2	0.13 0.13	>735.738 >717.871	>43 >3.8
MSW storage platform	0.13	1050 994	2.4
Packed waste platform	0.13	60 45 76 63	0.03

The dispersion model of odour emissions in orographically and meteorologically very complex Trento-North area can be useful to the realization of similar models for other provincial or extra-provincial areas (Odorizzi *et al.* 2003).

The emissions of bulk and trace gases from landfills created either directly from waste decomposition or from the combustion of landfill gas during flaring and/or gas utilisation, have the potential to impact the global atmosphere, the local environment and expose humans to potential health risks (Gregory *et al.* 2003). GasSim has been developed for the Environment

Agency of England and Wales as a risk assessment tool to aid in the evaluation of these impacts, and help landfill operators comply with the new regulations and guidance.

GasSim considers the uncertainty in input parameters using a Monte Carlo Simulation. Parameter uncertainty allows specifying a range of values for each input parameter rather than a single number, using probability density functions (PDFs). GasSim is divided into 5 main modules (Fig. 1.18):

- gas generation (the source term);
- gas emissions (fugitive surface and lateral emissions and combustion plant emissions);
- environmental transport through atmospheric dispersion;
- environmental transport through terrestrial lateral migration;
- human exposure and other environmental impacts.

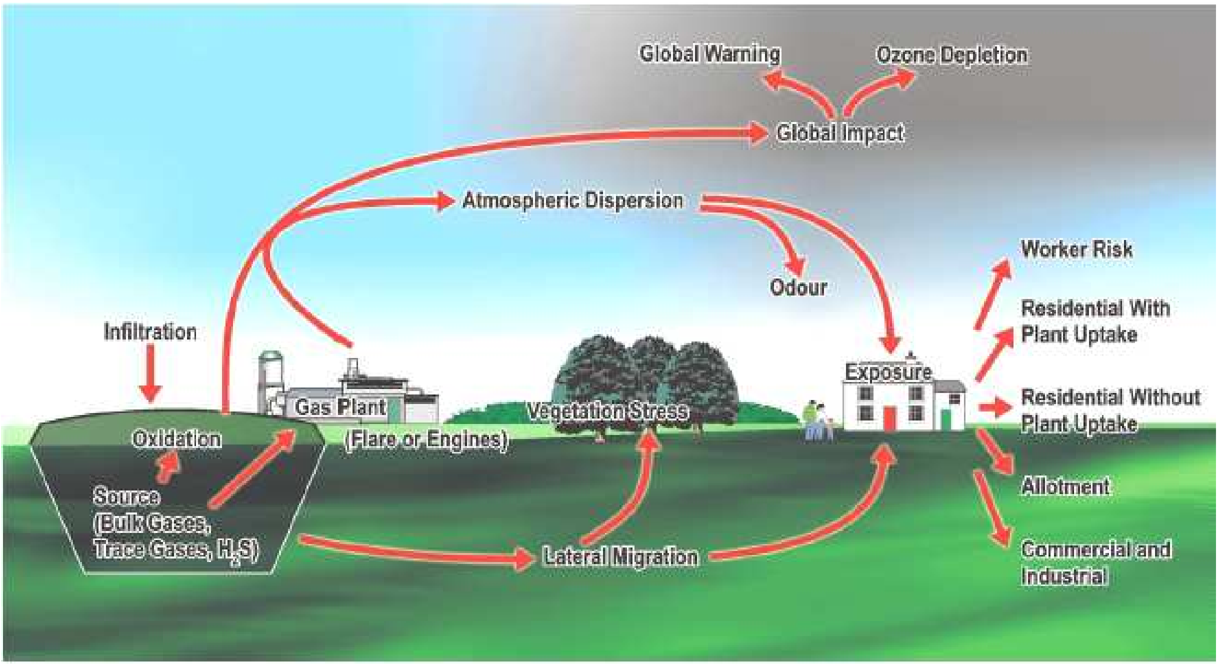

Fig. 1.18. The GasSim conceptual model (Gregory *et al.* 2003)

The impact of odour can be assessed in three ways (Gregory *et al.* 2003). The first determines the point at which the concentration of an odorous trace gas component falls below its odour threshold value. The second method simulates the emissions of European odour units (OU_E) from the site. Thirdly, GasSim allows the simulation of measured OU_E from various parts of the site, including uncapped, capped and discrete features.

The verification trials have demonstrated that the GasSim model produces results that agree with other models, namely LandGem and the equations used in HELGA framework (Gregory *et al.* 2003).

The understanding of landfill odour emissions, which have difficult sources to measure compared to other industrial odours, requires assessment using appropriate techniques and procedures (Bowly 2003). The techniques and the reasons for the choice are shown in Table 1.9.

Table 1.9. Assessment tools and reasons for use (Bowly 2003)

Tool	Reason
Olfactometry	To determine emission rates and obtain odour intensity data
Chemical analysis	To identify the chemicals responsible for the odours by comparing the chemicals to their individual odour threshold
Artificial neural networking	To provide a predictive tool for assessing odour concentrations based on chemical concentrations

One issue of odour impact assessment (OIA) is the relationship between chemicals and odours (Bowly 2003). A receptor, for example a resident near an industrial odour source, may complain of a low concentration odour. Current validation of the low concentration odour received by the complainant is usually by subjective assessment of a single authorised government representative. This validation technique may be considered inconsistent or imprecise, depending on the representative(s) involved. Artificial neural networks (ANN) were used to predict odour concentrations based on 79 VOC concentrations in Australia. Odour concentrations were predicted within 50 % of the input odour concentration.

The electronic nose, inspired by physiology, reproduces the human olfactory process of sniffing, receiving and recognizing of odorant molecules, exploiting the reactivity of chemical sensors and the computational abilities of modern personal computer (Bonoli *et al.* 2003; Nicolas *et al.* 2007). The use of an electronic nose in environmental odour recognition is in the phase of development of suitable instrumentation and data processing software, in order to discriminate between various odours directly in the field and to monitor them continuously. Artificial neural networks have been used to analyze complex data and combined with gas sensor array.

The tests conducted in the field with electronic nose lead to very promising results. The instrument allows the identification of the biogas odour and it is able to monitor biogas continuously (Bonoli *et al.* 2003).

Olfactometry employs a panel of human assessors to characterise the odour in terms of their perceived effect and is the usual method for measuring odours (Karnik *et al.* 2003). Although this methodology can now be based on a European odour standard, EN 17325, it is strongly influenced by subjectivity and disadvantages of time and cost, but can be quicker and cheaper than GC-MS. Spectrometry is often used to characterise odour samples chemically. In many cases, a particular odorant may be dominant and can give an indication of the overall odour concentration.

The development of sensor array technology (so called “electronic nose”) for odour classification may offer an objective and on-line instrument for assessing environmental odours (Karnik *et al.* 2003; Capelli *et al.* 2008). Typically an electronic nose consists of three elements: a sensor array which is exposed to the volatiles, conversion of the sensor signals to a readable format, and software analysis of the data to produce characteristic outputs related to the odours encountered. The output from the sensor array may be interpreted via a variety of methods such as pattern recognition algorithms, principal component analysis, discriminant function analysis, cluster analysis and artificial neural networks to discriminate between samples.

Software techniques and material science are important aspects of the development of the instrumentation and the technology is still under development for applications in environmental odour measurement. A big problem is that the sensor technology available is not yet as sensitive to unpleasant odours as the human nose, and so it is difficult to correlate sensor responses with olfactometric odour measurements (Karnik *et al.* 2003).

To be usable for the real-time odour monitoring, the electronic nose has to deal with the lack of long term stability of chemical sensors (Romain and Nicolas 2010). The instrument has to automatically compensate the time drift and the influence of ambient air parameters such as temperature or humidity. The chemical sensors alter over time and therefore they produce different responses for the same odour. Two identical sensors with the same history have different time stability. The sensor signals can drift during the learning phase too.

Another frequent problem associated with long term stability is the sensor failure or irreversible sensor poisoning (Romain and Nicolas 2010). Sensor replacement is generally required to address such issue, but the previous calibrated model is no longer applicable for the same odorous emissions. The replacement of an old or broken sensor by a new one corresponds to having a new electronic nose that requires new models of classification and quantification (e.g. signal pre-processing, univariate sensor correction and multivariate array correction).

Operating municipal solid waste landfills sited in Italy were monitored (Sironi *et al.* 2003; Sironi *et al.* 2005). The odour emission rate of every relevant odour source was monitored in order to establish the overall odour emission rate of the landfill. Then an atmospheric dispersion model was applied to quantitatively assess the odour impact of the landfill on the population living in the surroundings.

Fig. 1.19 gives the sum of the OERs of all odour sources of the landfill, as a function of time, according to the variability of the SOER with the season and to the amount of waste disposed on each parcel. It depicts the relative contribution of each source to the total odour emission rate.

Surface emissions from closed parcels covered by the final cap (i.e. parcels 1A, 2A, 2B, 3A) are much lower than emissions from operating parcels (i.e. parcels 2C, 1C, 1B, S2A), especially in summer. Surface emissions from operating parcels are much stronger in summer than in the other seasons. This is due to the higher SOER of partially and totally covered waste in summer (Sironi *et al.* 2003).

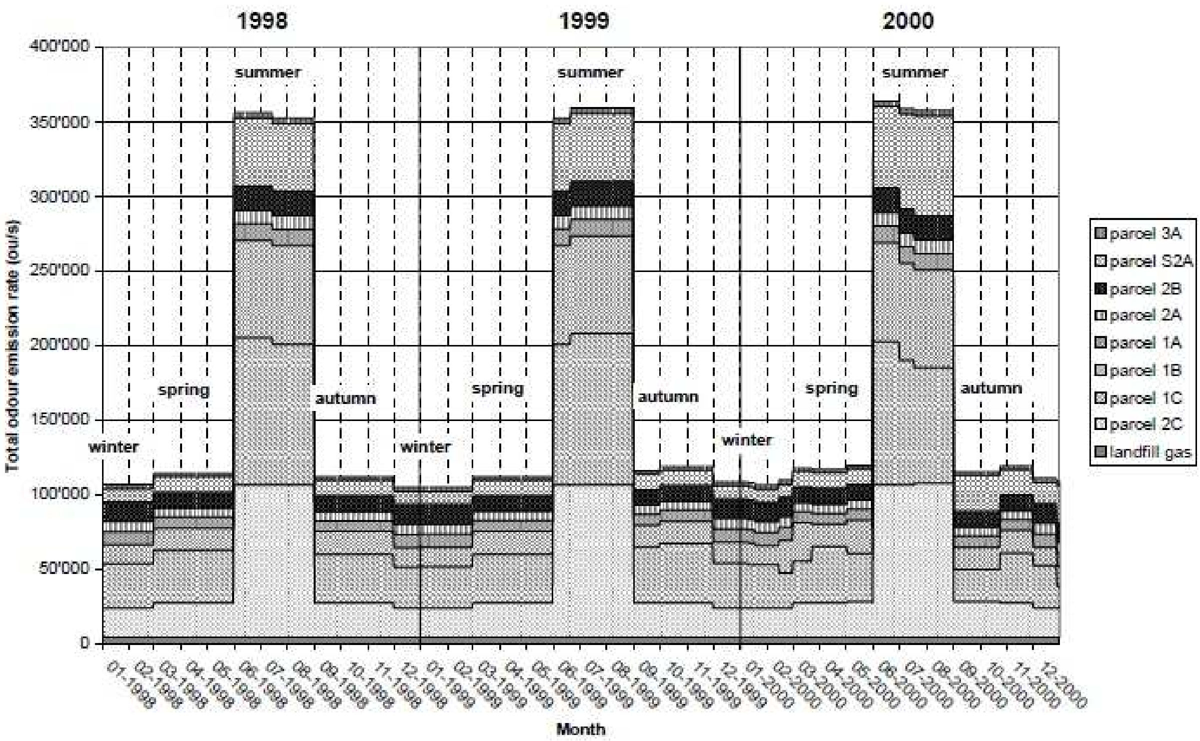

Fig. 1.19. Overall landfill odour emission rate (Sironi *et al.* 2003)

In this study the CALPUFF dispersion model was employed. Fig. 1.20 shows the isopleths of the odour perception frequency (OPF), with an exceedance level of 1 OU/m^3 (Sironi *et al.* 2003).

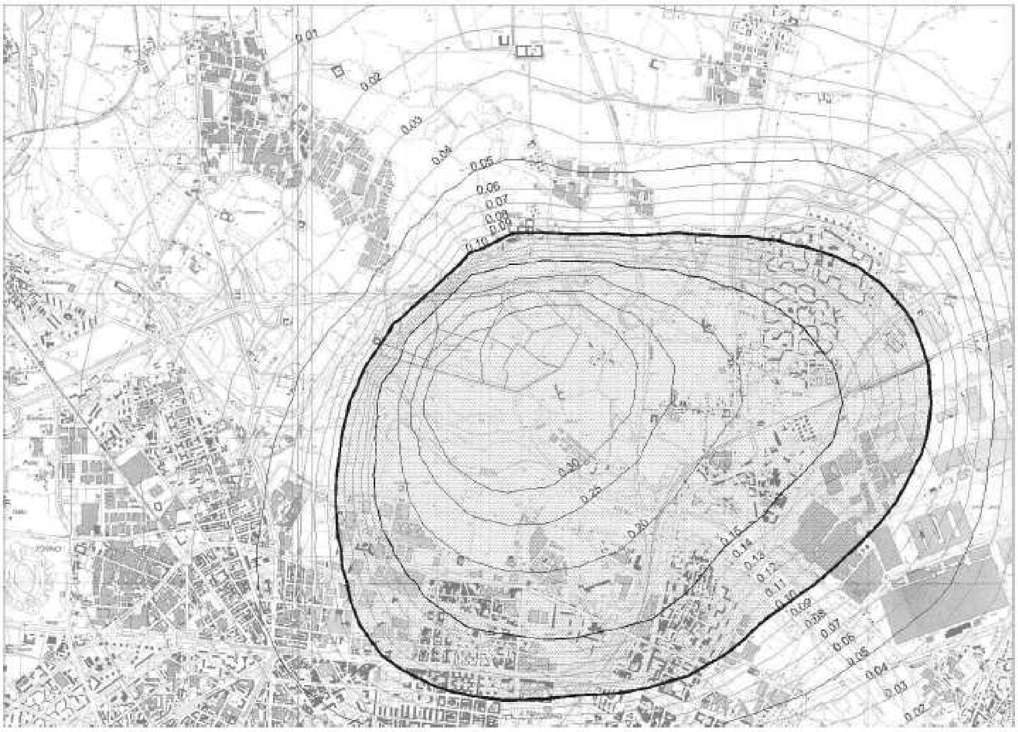

Fig. 1.20. Isopleths of the odour perception frequency (Sironi *et al.* 2003)

The thickest contour indicates the region exceeding the OPF of 0.10 (i.e. the 90^{th} percentile), prescribed by the German directive as the maximum value for residential settlements. CALPUFF computes ground level concentrations in each hour of the simulation period, so we can easily find the hours of the day or the seasons when a control on the OERs give the best reduction of the OPFs and improve the air quality (Sironi *et al.* 2003).

The total OERs of landfills A and B, their OEFs related to the landfill surface, and the percentage contributions of each odour source to both landfills' total OER are illustrated in Fig. 1.21 (Sironi *et al.* 2005).

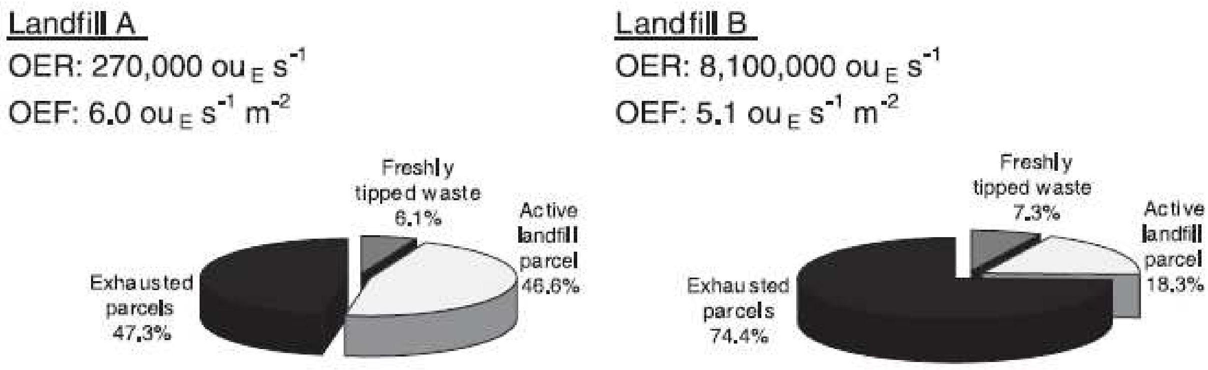

Fig. 1.21. Total OER, percentage contribution of single odour source and OEF (Sironi *et al.* 2005)

Results show that freshly tipped waste does not represent a particularly relevant odour source in a landfill, as its percentage contribution to the landfill total OER is less than 10 % (Sironi *et al.* 2005).

The primary technical issue associated with the evapotranspirative final cover (Fig. 1.10) is equivalence to the conventional final cover (Fig. 1.22) with respect to infiltration (i.e. groundwater protection). A water balance evaluations were conducted using an unsaturated flow model UNSAT-H to evaluate the infiltration performance of the evapotranspirative final cover compared to that of the conventional final cover. Fig. 1.23 presents the cumulative amount of water that percolates through these final cover alternatives for the 10-year modelling period (Kavazanjian and Dobrowolski 2003; Hadj-Hamou and Kavazanjian 2003).

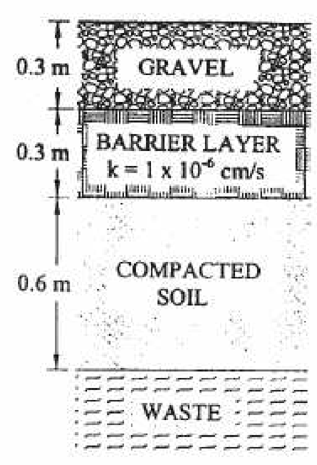

Fig. 1.22. Conventional final cover (Kavazanjian and Dobrowolski 2003)

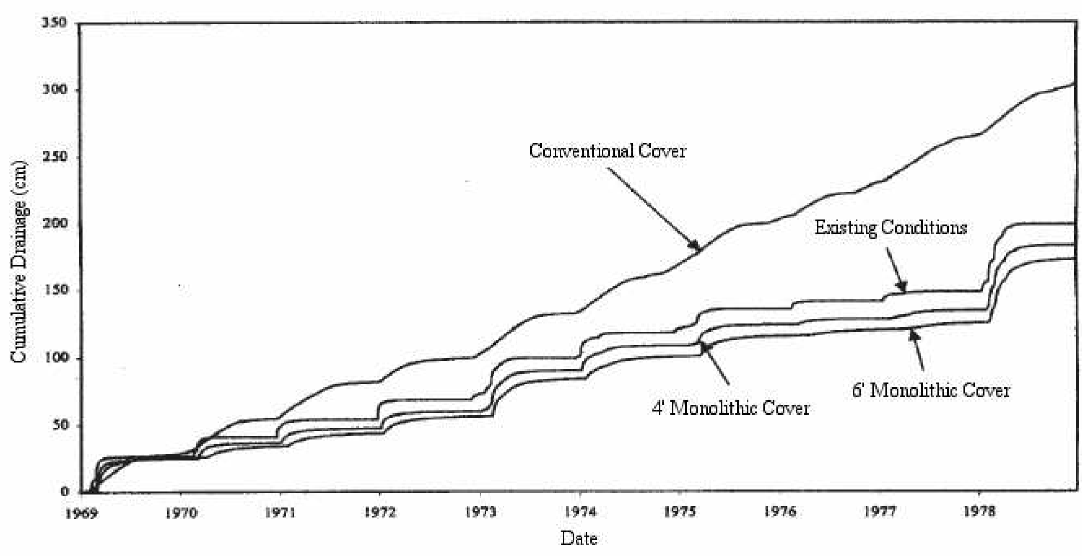

Fig. 1.23. Infiltration performance of evapotranspirative and conventional covers (Kavazanjian and Dobrowolski 2003)

The modelling indicates that both 1.2 m and 1.8 m evapotranspirative final covers are more effective than the conventional cover and that the performance of the 1.8 m evapotranspirative cover is only slightly superior to that of the 1.2 m evapotranspirative cover (Kavazanjian and Dobrowolski 2003; Hadj-Hamou and Kavazanjian 2003).

The results of the chemical analyses, which are useful to determine the chemical composition of odours, show no correlation with the odour concentration values measured by dynamic olfactometry (Capelli *et al.* 2008). Olfactometric analyses enable to measure odour concentration and quantify the sensory impact of odours. Finally, the continuous ambient air monitoring by electronic nose allows quantifying the time percentage in which odours are detected at the landfill boundaries and at a receptor.

A micrometeorological model has been developed based on the estimation of footprints of scalar odour concentration measurements in the atmospheric surface layer (Sarkar and Hobbs 2003). The model is based on an analytical solution of the Eulerian advection-diffusion equation for vertical diffusion. Lindvall hoods are commonly used for measuring odour fluxes from ground based fugitive sources. Lindvall hoods are portable flux chambers with an induced-draft fan-hose connection. Typical results show an average odour flux of ±25.91 $OU/m^2/s$ from freshly tipped wastes for an upwind fetch of 45.0 m and with the sensor at a height of 1.5 m from the ground. The overall accuracy and precision of Lindvall hood measurements depend on the biases and variability associated with the emission source, the sampling method and the analytical methods for analysing the odour samples.

Dispersion modelling was used to quantify the potential odour strength causing an impact on the community around a MSW landfill site in North London (Sarkar *et al.* 2003). The case studies were completed with COMPLEX-I developed by the US-EPA. The year 1998 was chosen as a source of baseline data. In 2004, concentrations as high as 25.0 OU_E/m^3 were observed with 3 min averaging time in the south westerly areas. All other surrounding farms and small villages were exposed to the concentration of 3.0 OU_E/m^3 on certain occasions. In 2008, the maximum odour concentration around the landfill site for 1 h averaging time was approximately 3 OU_E/m^3 about 1.0 km north and 0.5 km west of the landfill. For 3 min averaging time, the stretch of 5 OU_E/m^3 band was up to 2.5 km towards the north of the landfill.

Two approaches of the Gaussian atmospheric dispersion model were employed to predict odour impact from a projected landfill area and a waste treatment facility (Ubeda *et al.* 2010). The first approach was a simplified bi-Gaussian atmospheric dispersion model developed by the authors. Calculated odour concentrations were represented using GIS tools. Regarding the second approach, a commercial bi-Gaussian atmospheric dispersion one was used.

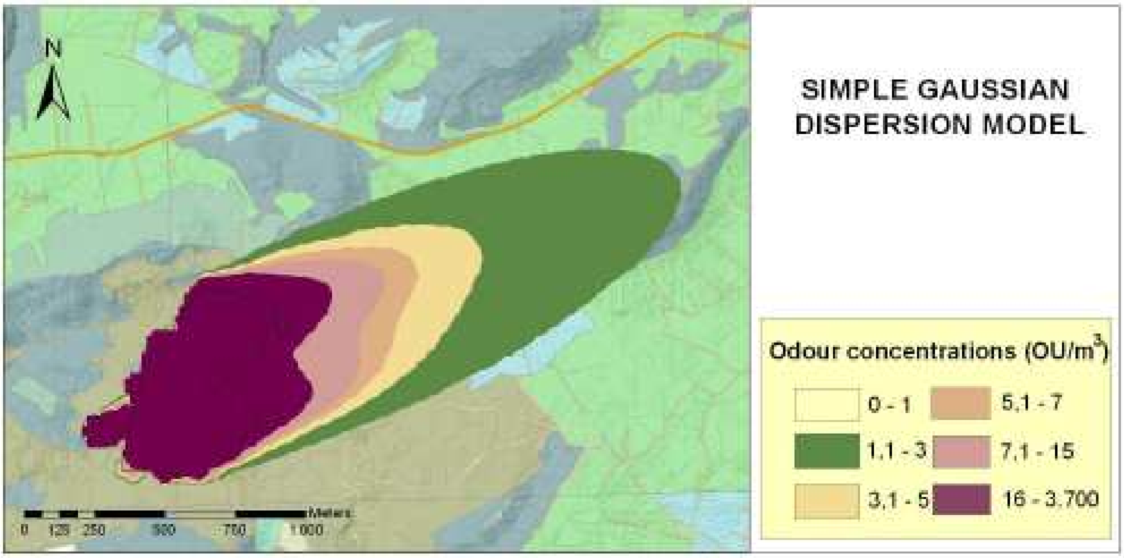

Fig. 1.24. Predicted odour concentrations with the simple model (Ubeda *et al.* 2010)

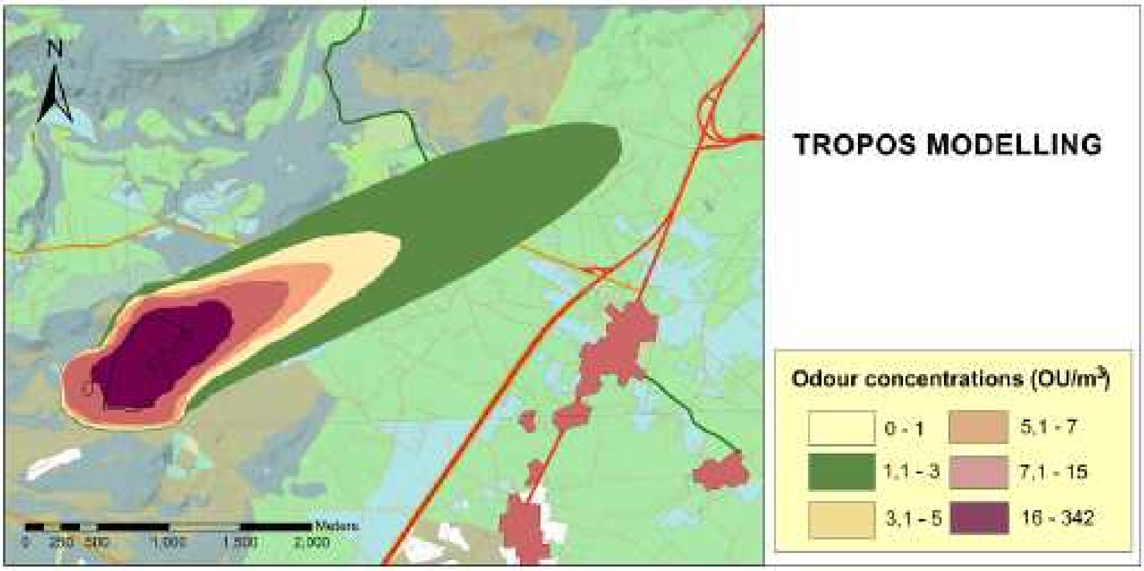

Fig. 1.25. Predicted odour concentrations with the Tropos model (Ubeda *et al.* 2010)

Odour concentrations were modelled for the prevailing winds. The maximum distance obtained by the simple dispersion model was 1.5 km (Fig. 1.24), modelled by the commercial one was 3.3 km (Fig. 1.25). Meteorological conditions in Mediterranean areas typically present a high proportion of calm winds, and in these situations Gaussian models may present high errors. Field measurements are required when landfill installation becomes operational, in order to determine the real reach of odour (Ubeda *et al.* 2010).

Odours can be quantified by five parameters that profile the human response (Odour… 2007). These parameters include: odour thresholds, odour intensity, odour persistency, hedonic tone, and odour characterization. The most common measure of odours is the odour threshold value (OTV), also referred to as the odour concentration or odour strength. Odour strength is quantified by determining the amount of dilution needed to bring the odorous air sample to its threshold. The higher the threshold value, the more dilution is needed to bring the odour to threshold, thus the stronger the odour.

The dilution ratio is an estimate of the number of dilutions needed to make the actual odour emission just detectable (Odour… 2007). This is known as the detection threshold (DT). The recognition threshold (RT) is the dilution ratio at which the assessor first detects the odour's character ("smells like…"). The recognition threshold value is always lower than the detection threshold value. It takes more dilution to bring an odour to its detection threshold (no odour present) compared to its recognition threshold (odour is not recognizable).

The odour threshold is reported as a dimensionless dilution ratio; however, often the pseudo-dimensions of "odour units" (OU) are used. Units of "odour units per cubic meter" (OU/m^3) are also commonly applied in order to calculate odour emission rates (Odour… 2007).

The pseudo-dimensions of "odour units per cubic meter" are commonly used for odour dispersion modelling, taking the place of "grams per cubic meter." The odour concentration can be multiplied by the air flow rate, cubic meters per second, resulting in a pseudo-dimension of "odour units per second," analogous to grams per second. Because "odour concentrations" from different source types cannot be "added" nor can they be "averaged," odour modelling must be conducted with caution (McGinley *et al.* 2000).

1.4. CONCLUSIONS

After a review of literary sources the following conclusions can be drawn:

1. The gaseous compounds emitted from landfills have various impacts on their surroundings. Typical odorants (e.g., mercaptans, hydrogen sulphide, etc) have relatively low detection thresholds. Tipping area, active and non active cells are identified as the main odour sources in landfill sites. Higher concentrations of odorous compounds occur during warm season.
2. Reduction of the operating area and temporary waste covering are the main techniques to avoid or reduce emissions of odorous compounds at landfill sites. Intermediate and final landfill covers are investigated in many aspects: composition, hydro-physical properties, water balance, methane oxidation, adsorption characteristics of H_2S, short and long term performance, and cost. Investigations were conducted on both laboratory-scale and field conditions.
3. Construction and demolition materials, ashes, compost, wood chips can potentially be used as alternatives for conventional daily cover systems. In terms of odour reduction these alternatives still need further investigation with respect to variety of odorous compounds, characteristics of used materials, economic aspects.
4. Performance of various cover materials for odour reduction at construction and demolition debris landfills is widely evaluated. There is lack of information about suitability of the C&D debris for odour reduction at MSW landfills.
5. The most common odour impact evaluation methods are: olfactometry, chemical analysis and electronic nose. Odour perception complexity, number of compounds, low sensitivity and subjectivity are the main limitations of odour evaluation techniques. Atmospheric dispersion models (e.g., GasSim, TROPOS, COMPLEX-1, CALPUFF, ADMS, etc) are applied to assess the odour impact of the landfill on the population living in the surroundings.

2. EXPERIMENTAL TESTS ON THE INFLUENCE OF WASTE COVERING LAYER ON ODOUR REDUCTION

In Lithuania, from 2010 the amount of biodegradable municipal waste going to landfills must be reduced to 75 %, from 2013 – 50 % and from 2020 – 35 % of the total amount produced in 2000 (Misevičius and Baltrėnas 2011).

Reduction of waste tipping areas and provisional waste covering are the most efficient measures for the reduction of odours from landfills (Senante *et al.* 2003).

It is required that thickness of the daily cover and frequency of the waste covering must be set according to season, weather conditions and waste disposal rate to ensure protection against the emission of odours and dust (Lietuvos Respublikos… 2000). The use of efficient materials for periodic waste covering would not only reduce the effect of adverse factors on the environment but would also help to more efficiently deal with the problem of quick completion of the landfill site.

The aim of this research is to evaluate the possibilities of using building debris for the reduction of odours in municipal landfills.

The section includes description of the methodology, results of the experimental tests and its statistical analysis.

2.1. METHODOLOGY OF THE EXPERIMENTAL TESTS ON THE WASTE COVERING LAYER PERFORMANCE

During the experiment mixed kitchen waste was placed into a 1.0 m^3 capacity box and covered with a layer of building debris (Fig. 2.1). The box was lined with low density polyethylene film, 0.15 mm thick. In order to prevent waste and building debris from being mixed up, a 350 mm layer of compacted waste was covered with 17 g/m^2 agri film.

Granulometric composition, bulk density and humidity content of the building debris (crushed concrete and brickwork) were evaluated. C&D debris was collected from a local building lot.

Granulometric composition of the building debris was determined according to partial residues on standard sieves.

Minimum mass of the sample is chosen according to maximum particle size D_{max}. When D_{max} of the debris is 4(5) mm (4 means rectangular mesh and 5 means circular mesh), then the mass of the sample required is 1 kg. When D_{max} of the debris is 8(10) mm, then the mass of the sample required is 5 kg. When D_{max} of the debris is 16(20) mm, then the mass of the sample required is 10 kg. When D_{max} of the debris is 31.5(40) mm, then the mass of the sample required is 20 kg. When D_{max} of the debris is 63(70) mm, then the mass of the sample required is 40 kg.

After removing of large particles and desiccation at the temperature of 110±5 °C, 2000 g samples of the C&D debris were sieved through 0.4 mm, 1 mm, 4 mm, 10 mm meshes and named as particle size <0.4 mm, 0.4-1 mm, 1-4 mm, 4-10 mm, >10 mm in the study.

The partial residue on each sieve a_i (%) is calculated as:

$$a_i = \frac{m_i}{m} \cdot 100 \tag{2.1}$$

in which m_i is the mass of residue on the sieve, m is the mass of the sample. Total residue on each sieve A_i (%) is calculated as sum of partial residues on the selected sieve and all sieves above. Respectively, wastage through each sieve B_i (%) is calculated as:

$$B_i = 100 - A_i \tag{2.2}$$

in which A_i is the total residue on the sieve.

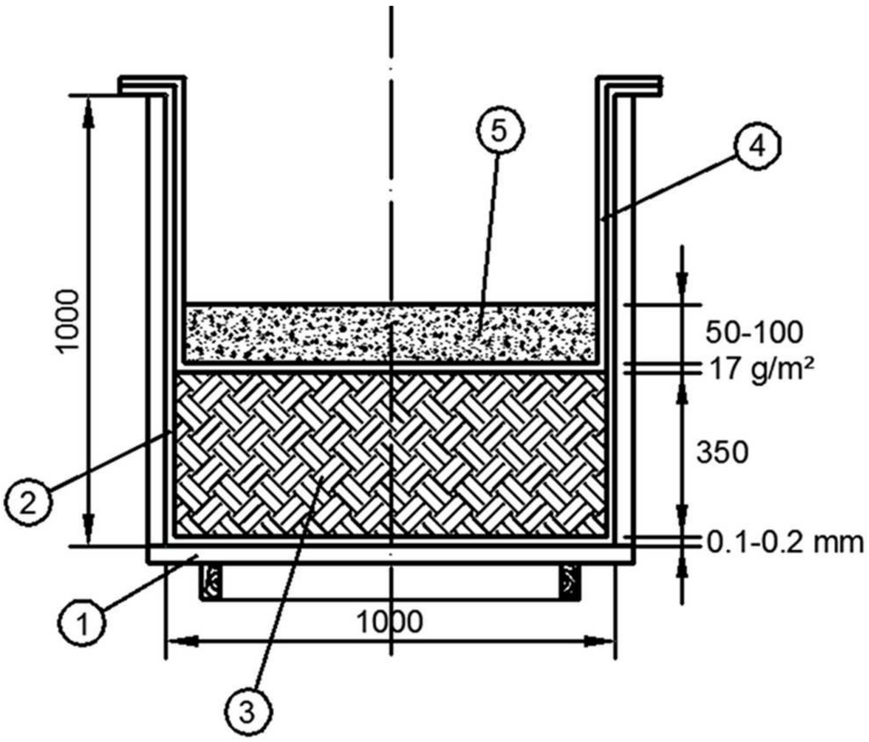

Fig. 2.1. Experimental display: 1 – wooden box, 2 – 0.15 mm polyethylene film of low density, 3 – biodegradable waste, 4 – 17 g/m^2 agri film, 5 – building debris

Bulk density of the building debris was determined by the mass of dry material placed into known volume container.

Minimum volume of the container is chosen according to maximum particle size D_{max}. When D_{max} of the debris is 4(5) mm (4 means rectangular mesh and 5 means circular mesh), then the volume of the container required is 1 dm^3. When D_{max} of the debris is 31.5(40) mm, then the volume of the container required is 10 dm^3. When D_{max} of the debris is 63(70) mm, then the volume of the container required is 20 dm^3.

Dry and clean container is weighed and placed on the horizontal surface. The container is filled with the C&D debris from maximum 5 cm height. Full container is weighed with accuracy of 0.2 %. 3 samples of the C&D debris are evaluated.

Humidity content of the building debris was determined by the mass reduction after desiccation process.

The sample mass depends on maximum particle size D_{max}. When D_{max} of the debris is 63(70) mm (63 means rectangular mesh and 70 means circular mesh), then the mass of the sample required is 5 kg. When D_{max} of the debris is 31.5(40) mm, then the mass of the sample required is 2.5 kg. When D_{max} of the debris is 16(20) mm, then the mass of the sample required is 1.5 kg. When D_{max} of the debris is 8(10) mm, then the mass of the sample required is 1 kg.

Dry and clean container is filled with the sample of the C&D debris and weighed. The sample is desiccated at the temperature of 110±5 °C until constant mass. The container with dry sample is weighed. 2 samples of the C&D debris are evaluated.

Mixed kitchen waste was selected as the source of odours. It accounts for 14 % of the total municipal waste amount (Municipal Solid… 2011). The processes of biodegradable waste degradation predetermine the formation of odours (Pichtel 2005). Density, humidity content and total organic carbon (TOC) content of waste were evaluated.

The total organic carbon content was established with the Shimadzu instrument TOC-VCSN (Fig. 2.2). During the ignition process the apparatus measures CO_2 gas. The total organic carbon content is calculated as:

$$w_{C,t} = 1000 \cdot \frac{m_2}{m_1} \cdot 0.2727 \qquad (2.3)$$

in which $w_{C,t}$ is the total organic carbon content (mg/kg), m_1 is the mass of tested sample (mg), m_2 is the mass of released CO_2 (mg), 0.2727 is the conversion factor for CO_2 to C.

Fig. 2.2. Total organic carbon analyzer TOC-VCSN

With the aim of avoiding the influence of precipitation and the formation of leachate, testing was performed in indoors. The testing comprised two stages. In the first stage odour samples were taken every second day from uncovered waste, a 5 cm thick layer of building debris and a 10 cm thick layer of building debris. In the second stage samples were taken every day from uncovered waste (1 day), a 5 cm thick layer of building debris (3 days) and a 10 cm thick layer of building debris (3 days).

Odour samples were taken in the second half of the day (after 12:00). Upon taking the first sample, air temperature and relative humidity in the room were measured with the multi-functional meter for measuring air parameters METREL MI6401. The instrument's ambient temperature measurement range is -20÷60 °C, resolution 0.1 °C, accuracy ±0.5 °C (±0.2 °C at 25 °C). Relative humidity measurement range 0÷100 %, resolution 0.1 %, accuracy ±3 % (0÷10 %), ±2 % (10÷90 %), ±3 % (90÷100 %). The second sample was taken after one hour. Samples were taken at a height of several centimetres from the waste or building debris surface with the vacuum chamber AC'SCENT and collected to Tedlar bags of 10 litre capacity and analysed in a laboratory on the same day (McGinley *et al.* 2000).

Vacuum chamber has the integral pump powered by 4 D-size batteries (Fig. 2.3). Gas samples are kept in 10 litre capacity Tedlar bags. The sample bag is first filled with the odorous air for "conditioning" the bag. The bag is filled to 1/3 full and held for at least one minute. The bag is then emptied using the pump.

Samples are collected using the vacuum case. When the sample bag is 2/3 full, the vacuum is released from the case and the sample flow stops. The sample line is disconnected and the bag is removed from the vacuum case after the bag valve is closed. It is required the odour testing to be conducted within a nominal 24 hour time period after sample taking (McGinley *et al.* 2000).

Fig. 2.3. Vacuum chamber with integral pump

Odour concentration measurements were carried out with the AC'SCENT International Olfactometer. The odour thresholds are determined by trained human assessors observing presentations of the odorous air samples dynamically diluted with the olfactometer (Odor… 2007).

The AC'SCENT International Olfactometer (Fig. 2.4) is a dynamic dilution venturi-nozzle style olfactometer used for determination of detection and recognition thresholds of odorous air samples, including odorant mixtures or pure compounds.

Fig. 2.4. AC'SCENT International Olfactometer with the test administrator and the assessor (St. Croix Sensory 2005)

The olfactometer mixes odorous air samples with the odour-free air in specific ratios (Table 2.1) for the presentation to a panel of human assessors.

Table 2.1. Olfactometer air flow rates and dilution ratios (St. Croix Sensory 2005)

Dilution Level	Odorous Air Flow	Dilution Ratio
1	0.31 cm^3/min	64000
2	0.63 cm^3/min	32000
3	1.25 cm^3/min	16000
4	2.50 cm^3/min	8000
5	5.00 cm^3/min	4000
6	10.0 cm^3/min	2000
7	20.0 cm^3/min	1000
8	40.0 cm^3/min	500
9	80.0 cm^3/min	250
10	160 cm^3/min	125
11	320 cm^3/min	63
12	630 cm^3/min	32
13	1250 cm^3/min	16
14	2500 cm^3/min	8

The AC'SCENT International Olfactometer meets all requirements of internationally accepted standards of olfactometry:

- EN 13725:2003 Air Quality – Determination of Odour Concentration by Dynamic Olfactometry (European Community),
- ASTM E679-04 Determination of Odor and Taste Thresholds by a Forced-Choice Ascending Concentration Series Method of Limits (North America),
- AS/NZS 4323.3:2001 Stationary Source Emissions – Determination of Odour Concentration by Dynamic Olfactometry (Australia & New Zealand).

These olfactometry standards allow the use of two analysis methods (Forced-Choice or Yes/No), which are both available for use on the AC'SCENT International Olfactometer. For both test methods, after responding to the first dilution level, the assessor is then presented with the next dilution level. In the "Forced-Choice" method the assessor is again presented with three sample choices, one of which is the odour sample. In the "Yes/No" method the assessor again tests six randomly presented samples. However, in both cases this next dilution level presents the diluted odour at a higher concentration (e.g. two times higher concentration, one-half the dilution ratio). The assessor continues to additional higher concentration levels (lower dilutions ratios) of sample presentation following these methods. This statistical approach of increasing concentration levels of sample presentation is called "ascending concentration series."

The olfactometer is self contained in a podium style enclosure. The lower compartment of the enclosure contains the zero air system (Fig. 2.5), which supplies the odour-free dilution air. The upper compartment of the enclosure contains the electrical components and the valve control system (Fig. 2.6).

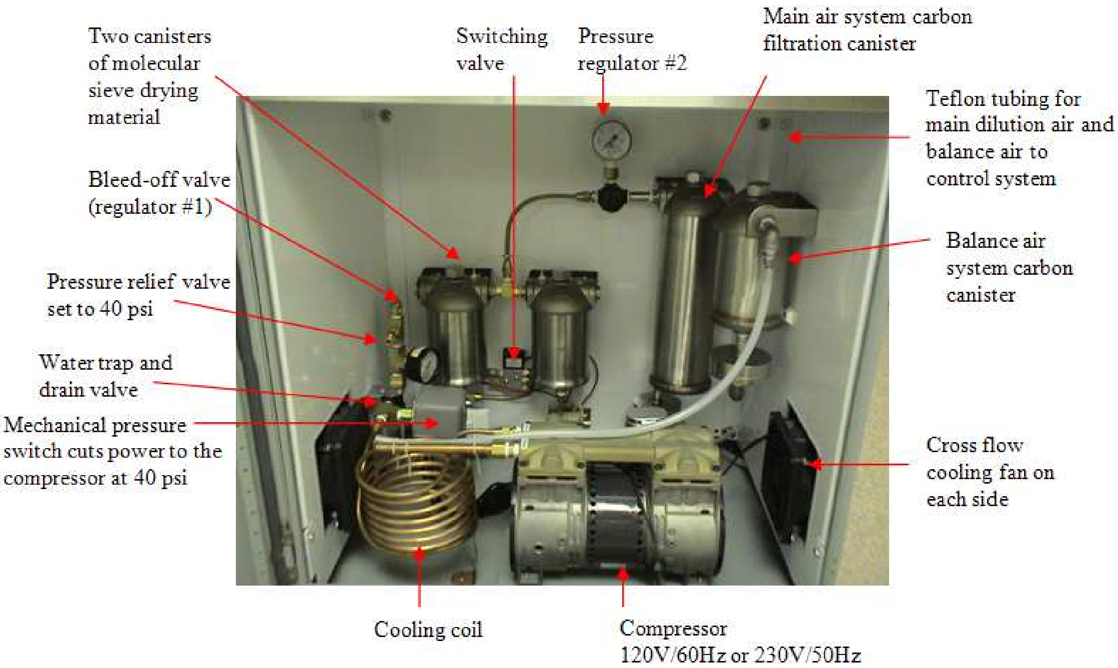

Fig. 2.5. Olfactometer zero air system (St. Croix Sensory 2005)

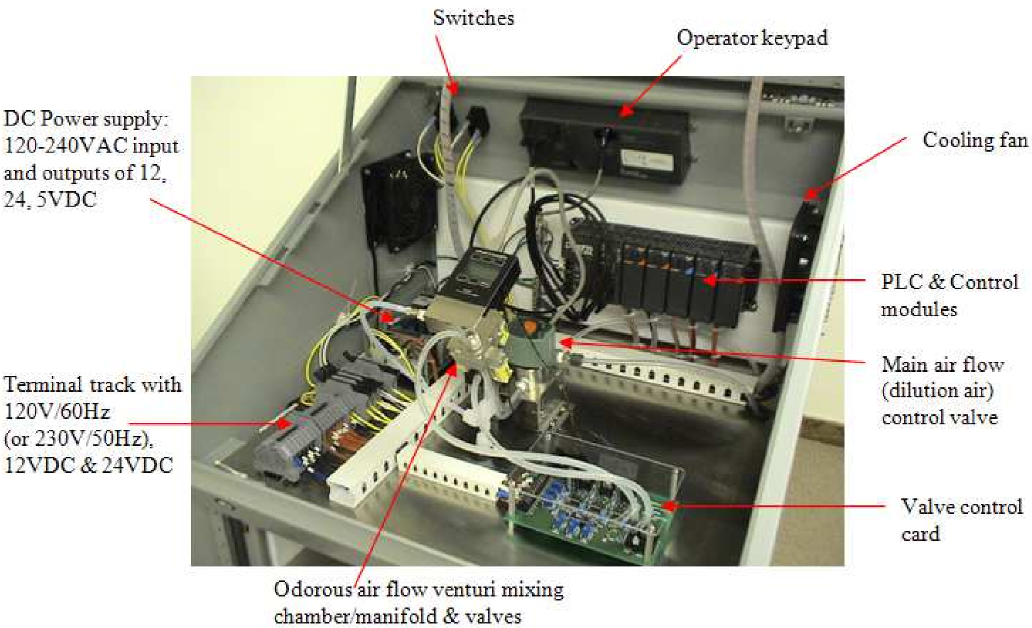

Fig. 2.6. Olfactometer electrical and valve control system (St. Croix Sensory 2005)

The olfactometer contains a venturi style air manifold system (Fig. 2.7). A main air system provides the odour-free air to the venturi manifold. The venturi acts as a flow controller for the main air flow, as well as providing suction pressure for mixing volumes of balance air and/or odorous air without the aid of pumps. The balance air system is controlled to provide a continuous, proportional flow of odour-free air to balance the system pressures in the manifold when switching between blank and odour presentations.

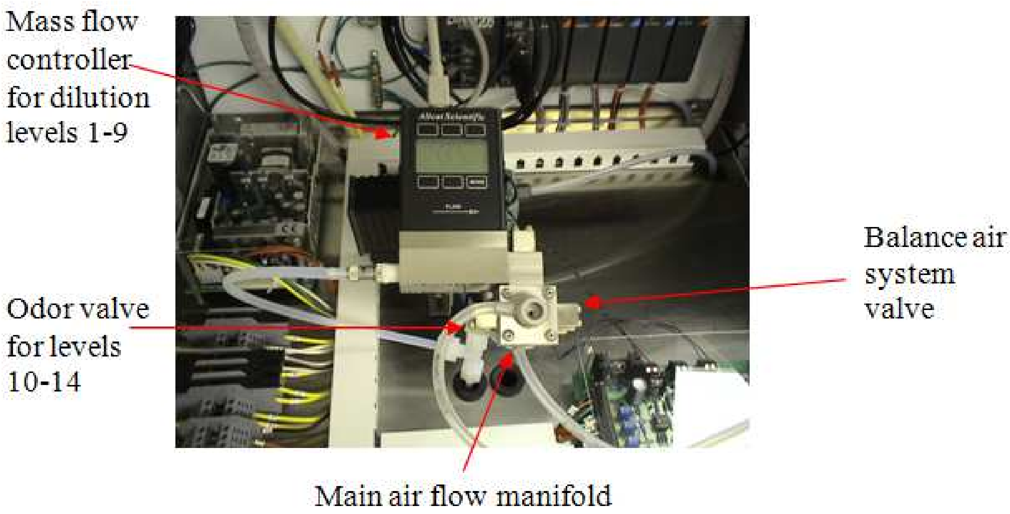

Fig. 2.7. Olfactometer main air flow venturi manifold (St. Croix Sensory 2005)

Results are computed for each assessor based on the dilution levels where correct “detection” or “recognition” responses are recorded (Odor… 2007). The responses of all assessors are averaged to determine the sample’s detection and recognition thresholds.

The analysis was performed by the method of forced choice. A researcher evaluates three deliveries of each dilution. One of them contains the diluted sample at issue, others – clean air. The researcher selects “guess” (one of the deliveries may contain an odour samples), “detection” (odour is sensed in one of the deliveries) and “recognition” (the nature of odour is recognised in one of the deliveries) answers. The levels of dilution differ from each other by a twofold lower dilution ratio and a twofold higher odour concentration. Each sample is evaluated by 4 researchers.

2.2. RESULTS OF THE DAILY COVER PERFORMANCE EVALUATION

A granulometric composition of the building debris (crushed concrete and brickwork) used for waste covering is presented in Table 2.2 and Fig. 2.8. In accordance with partial residues on sieves the main fractions of building debris are 1÷4 mm (22.5 %) and 4÷10 mm (30.7 %). Debris' bulk density – 1.32 g/cm^3 (Table 2.3), humidity content – 1.7 % (Table 2.4). Error of the weighing is 0.01 g.

Table 2.2. Granulometric composition of the building debris

Sieve size, mm	Sieve weight, g	Sieve and residue weight, g	Residue weight m_i, g	Partial residue a_i, %	Total residue A_i, %	Wastage B_i, %
0.0	335.19	647.24	312.05	15.6	99.7	0.3
0.4	298.57	555.14	256.57	12.8	84.1	15.9
1.0	303.64	754.36	450.72	22.5	71.3	28.7
4.0	373.03	987.85	614.82	30.7	48.7	51.3
10.0	141.69	501.63	359.94	18.0	18.0	82.0
Sample size m, g			2000.00			
$\sum m_i$			1994.10			
$100 \cdot (m-\sum m_i)/m$			0.29			

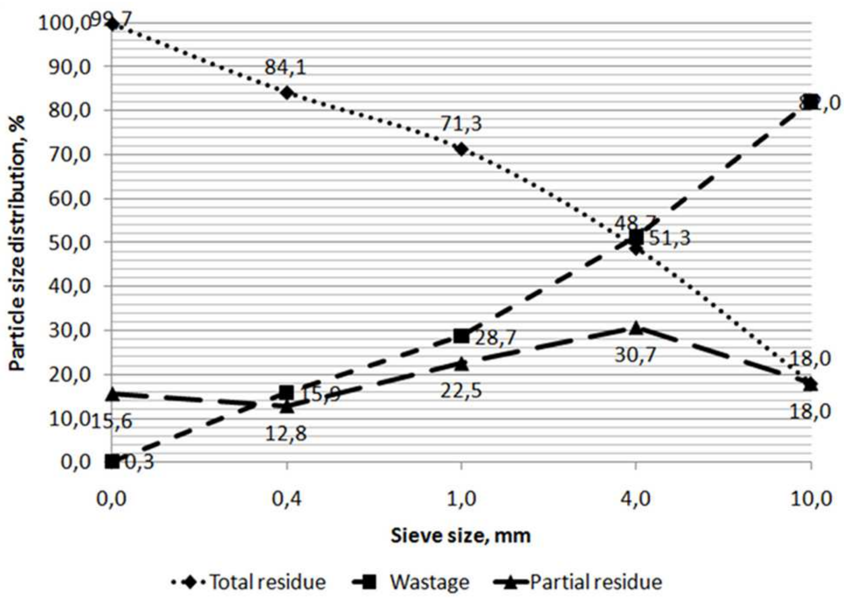

Fig. 2.8. Building debris granulometric composition curve

Table 2.3. Bulk density of the C&D debris

Parameters	Equations/Dimensions	Samples		
		1	2	3
Empty container weight	m, g	285.52	285.52	285.52
Full container weight	m_1, g	1586.72	1610.78	1615.20
Container volume	V, cm^3	1000	1000	1000
Bulk density	$\rho=(m_1-m)/V$, g/cm^3	1.30	1.33	1.33
Average	ρ, g/cm^3	1.32		

Table 2.4. Moisture content of the C&D debris

Parameters	Equations/Dimensions	Samples	
		1	2
Container and wet sample weight	m_1, g	970.27	964.34
Container and dry sample weight	m_2, g	953.84	948.12
Moisture content	$w=100\cdot(m_1-m_2)/m_1$, %	1.7	1.7
Average	w, %	1.7	

The waste used for the tests had 0.48 g/cm^3 density, 74.8 % humidity content and 34.6 % total organic carbon (TOC) content. Results are presented in Table 2.5, Table 2.6 and Table 2.7 respectively.

Table 2.5. Density of the food waste

Parameters	Equations/Dimensions	Samples		
		1	2	3
Empty container weight	m, g	12.43	12.43	12.43
Full container weight	m_1, g	37.68	35.09	36.14
Container volume	V, cm^3	50	50	50
Density	$\rho=(m_1-m)/V$, g/cm^3	0.51	0.45	0.47
Average	ρ, g/cm^3	0.48		

Table 2.6. Moisture content of the food waste

Parameters	Equations/Dimensions	Samples	
		1	2
Container and wet sample weight	m_1, g	120.16	114.37
Container and dry sample weight	m_2, g	31.72	27.45
Water content	$w=100\cdot(m_1-m_2)/m_1$, %	73.6	76.0
Average	w, %	74.8	

Table 2.7. TOC of the food waste

Sample number	Sample weight, mg	Carbon content, mg	Carbon content, % TS	Average, % TS
1	99.62	34.87	35.0	34.6
2	70.14	22.87	32.6	
3	70.03	25.28	36.1	

Assessment in the first stage covered the influence of thickness of a building debris layer on the formation of odours (Fig. 2.9 and 2.10). In the event of uncovered waste, the highest odour detection thresholds reached 419 and 347 odour units.

Upon covering waste with a building debris layer of the thickness of 5 cm, the odour detection threshold fell by 75.7 % to 102 OU (the first samples) and by 51.0 % to 170 OU (samples after one hour). Upon covering waste with a building debris layer of the thickness of 10 cm, the odour detection threshold decreased by 79.7 % to 85 OU compared to uncovered waste and by 16.7 % compared to 5 cm thick layer (the first samples). The odour detection threshold of the second samples remained unchanged (Fig. 2.9).

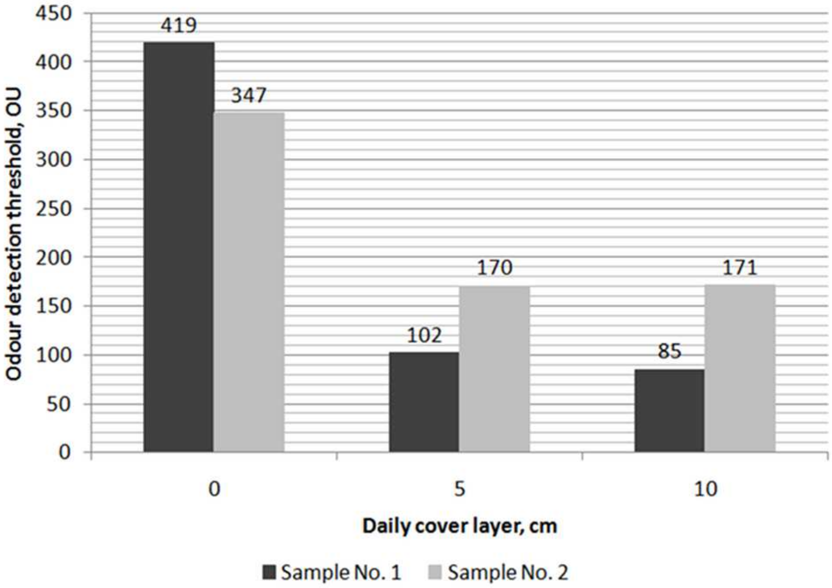

Fig. 2.9. Dependence of odour detection threshold on thickness of a building debris layer

The dependence of odour recognition thresholds on the thickness of a building debris layer is similar to that of the odour detection threshold but the values themselves are lower. In the case of uncovered waste the highest odour recognition thresholds reached 205 and 244 odour units (Fig. 2.10).

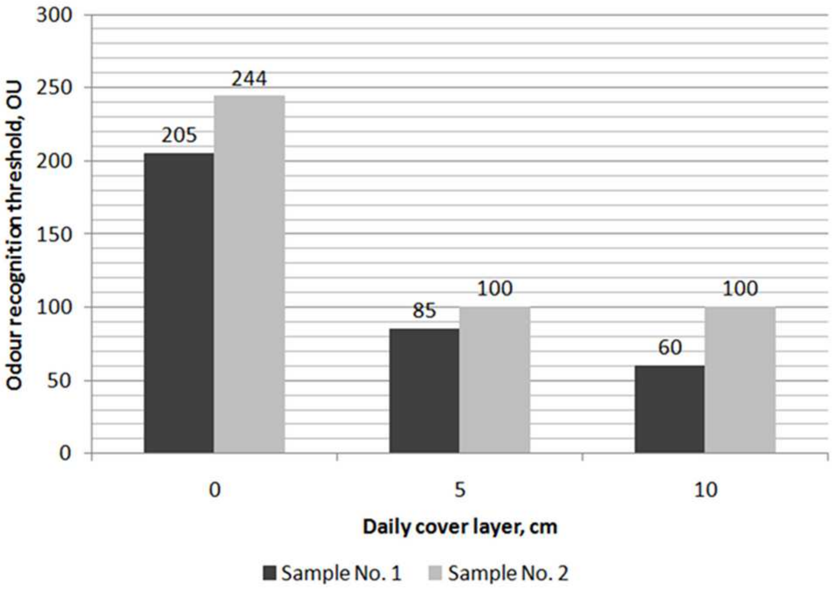

Fig. 2.10. Dependence of odour recognition threshold on thickness of a building debris layer

Upon covering waste with a building debris layer of the thickness of 5 cm, the odour recognition threshold decreased by 58.5 % to 85 OU (the first samples) and by 59.0 % to

100 OU (samples after one hour). Upon covering waste with a building debris layer of the thickness of 10 cm, the odour recognition threshold decreased by 70.7 % to 60 OU compared to uncovered waste and by 29.4 % compared to 5 cm thick layer (the first samples).

The specific odour emissions from the first samples determined during tests reached 1.26 OU/m^2/s in the case of uncovered waste, 0.31 OU/m^2/s in the case of the layer 5 cm thick and 0.26 OU/m^2/s in the case of the layer 10 cm thick. The specific odour emissions of the second samples reached 1.04 OU/m^2/s in the case of uncovered waste, and 0.51 OU/m^2/s in the cases of the layer 5 and 10 cm thick.

In accordance with Sironi *et al.* 2003, the specific odour emissions of uncovered waste vary from 1.7 OU/m^2/s (in summer) to 2.0 OU/m^2/s (in autumn). The specific odour emissions of covered waste vary from 0.2 OU/m^2/s (in winter) to 1.4 OU/m^2/s (in summer). In accordance with Sironi *et al.* 2005, the average specific odour emissions in Italian landfills reach 5.5±3.4 OU/m^2/s. As the tests performed by Odotech Inc. in Canadian landfills in 2001 show, odour concentrations in old sections amount to 2.6 OU/m^2/s, in waste tipping sites – 5.4 OU/m^2/s, and in truck parking areas – 3.5 OU/m^2/s (Nicolas *et al.* 2006).

In accordance with Ubeda *et al.* (2010), the average concentration of municipal waste landfill odours reaches 1.375 OU/m^2/s. The average odour concentrations presented by Bowly (2003) and Romain *et al.* (2007) vary from 0.3 to 0.5 OU/m^2/s. Odour emissions from landfills normally depend on (Sironi *et al.* 2003):

- waste characteristics;
- frequency of waste covering;
- area of waste tipping sites;
- the quantity of waste per time unit;
- meteorological conditions (wind speed, solar radiation, air temperature, relative humidity).

Assessment in the second stage covered the influence of thickness of a building debris layer on the formation of odours in the course of time (Fig. 2.11 and 2.12). The ambient air temperature and relative humidity were also determined (Fig. 2.13).

In the first samples the highest values of the odour detection threshold were recorded in the case of uncovered waste (85 OU). Upon covering waste with a building debris layer of the thickness of 5 cm, the odour detection threshold fell by 29.4 % to 60 OU and remained stable for all three days. Upon covering waste with a building debris layer of the thickness of 10 cm, the odour detection threshold decreased from 60 OU to 37 OU (38.3 %) on the second day. On the third day the odour detection threshold reached 44 OU. The total decrease in the odour detection threshold within the testing period (7 days) is 48.2 % (Fig. 2.11).

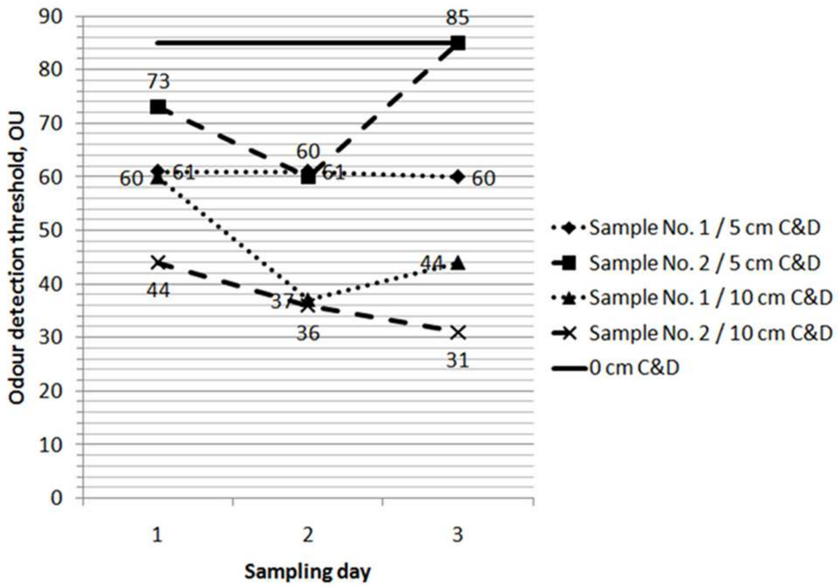

Fig. 2.11. Dependence of the odour detection threshold on the thickness of a C&D debris layer vs time

The highest values of the odour detection threshold in the second samples stood at 73 OU (on the first day in the case of the layer 5 cm thick) and 85 OU (on the third day in the case of the layer 5 cm thick). The total decrease in the odour detection threshold accounts for 63.5 %. The dependence of odour recognition thresholds on the building debris layer thickness vs time is similar (2.12).

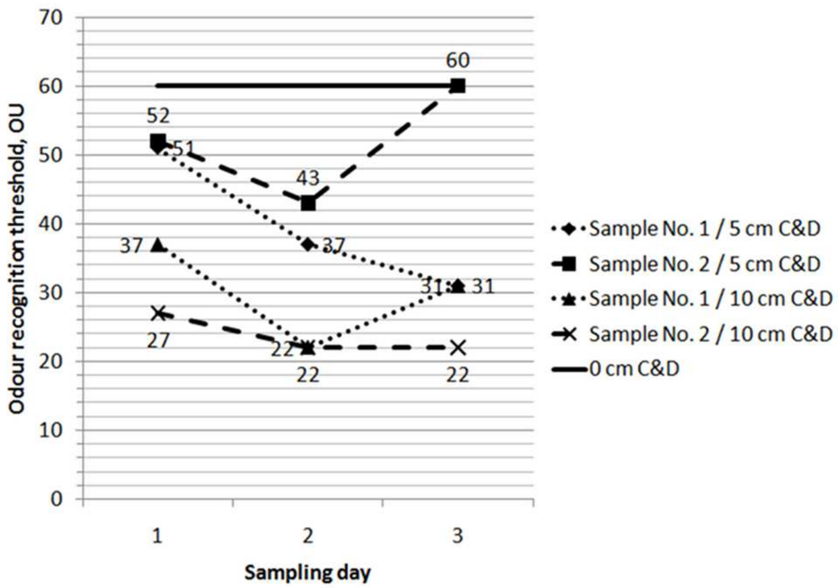

Fig. 2.12. Dependence of the odour recognition threshold on the thickness of a C&D debris layer vs time

When municipal waste is periodically covered with compost the concentrations of landfill odours decrease from 69 to 97 % (Hurst *et al.* 2005). Odour concentrations in waste tipping places reaching 4000-30000 OU/m^2/h decrease to 120-900 OU/m^2/h. In order to reduce gaseous emissions, biodegradable waste is composted with wood sawdust, green (garden) waste, or peat (Zigmontienė and Zuokaitė 2011). However, as these materials are non-homogenous their use is limited.

The odour concentration of the waste covered with a layer consisting of 50 mm of building debris and 150 mm of sawdust decreases by more than 50 % (Solan *et al.* 2010). Finely fractioned building debris is distinguished by low concentrations of background odours. This is insignificantly influenced by a waste collection site and methods of treatments.

Where waste is covered with the building debris layer of the thickness of 5 cm instability of the odour threshold can result from insufficient thickness of the building debris layer. Already on the third day of the testing odour thresholds reached the level of uncovered waste. In the meantime upon covering waste with a 10 cm thick layer the thresholds gradually decreased or insignificantly fluctuated. In this case fluctuations could have been determined by changes in the relative humidity of the ambient air (Fig. 2.13 and 2.14). The correlation coefficient of the odour detection thresholds in the first samples was equal to 0.9988, that in the second ones – 0.8003. The correlation coefficient of the odour recognition thresholds in the first samples was equal to 0.9262, that in the second ones – 0.9684.

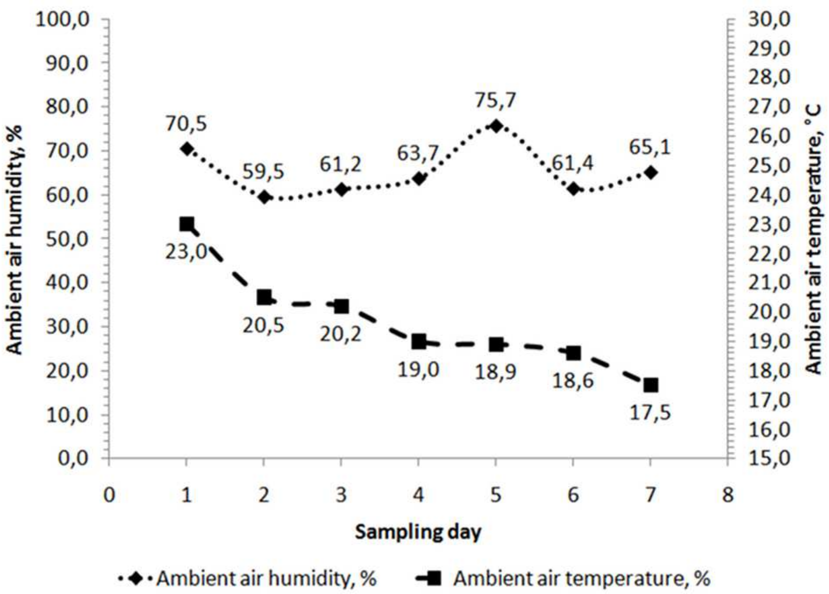

Fig. 2.13. Alternation of ambient air temperature and relative humidity during the experiment

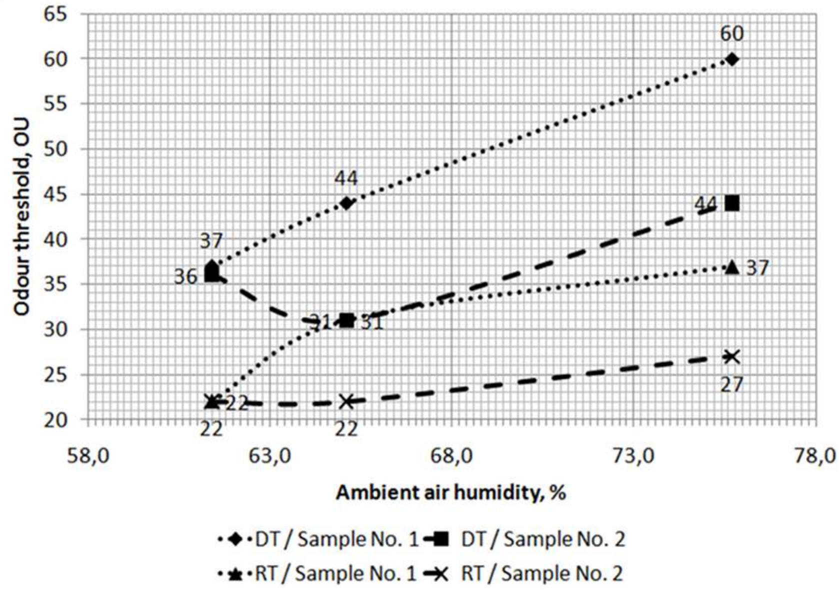

Fig. 2.14. Dependence of odour thresholds on relative humidity of the ambient air: DT – odour detection threshold, RT – odour recognition threshold

Certain fluctuations in the concentrations of background odours resulting from the humidity of materials were determined by Solan *et al.* 2010. Tests covered wood sawdust, building-demolition and industrial dust. When the content of humidity in materials significantly increases, the specific odour emissions also increase. The increased emission of hydrogen sulphide at a higher humidity of the ambient air was identified by He *et al.* 2010. At higher temperature water vapour pressure grows, which deteriorates the adsorption properties of the filler resulting in more intensive emissions of H_2S.

Throughout the testing period (7 days) ambient air temperature was gradually falling. The highest temperature (23.0 °C) was recorded on the experiment's first day when waste was uncovered. The lowest temperature (17.5 °C) was identified on the last day of the test in the case of the building debris layer of 10 cm thickness (Fig. 2.13).

On the basis of tests on hydrogen sulphide degradation activity (Zdeb *et al.* 2008), the average value of H_2S degradation activity at 28 °C temperature was by only 1.85 times above the average value at 6 °C temperature.

During the testing period a difference in temperatures was a mere 5.5 °C and therefore temperature changes did not have a significant influence on testing results.

Results are computed for each assessor based on the dilution levels where correct "detection" or "recognition" responses are recorded (Fig. 2.15). The responses of all assessors are averaged to determine the sample's detection and recognition thresholds.

Dilution Level	1	2	3	4	5	6	7	8	9	10	11	12	13	14
Sample Volume	0,37	0,62	1,22	2,50	5,02	10,1	20,2	40,1	80,3	170	340	620	1260	2550
Total Volume	20.000	20.000	20.000	20.000	20.000	20.000	20.000	20.000	20.000	20.000	20.000	20.000	20.000	20.000
Dilution Ratio	54.054	32.258	16.393	8.000	3.984	1.980	990	499	249	118	58,8	32,3	15,9	7,8
Geometric Mean	76.444	41.757	22.996	11.452	5.646	2.809	1.400	703	352	171	83	43,6	22,6	11,2
Log (Geo. Mean)	4,88	4,62	4,36	4,06	3,75	3,45	3,15	2,85	2,55	2,23	1,92	1,64	1,35	1,05

Calibration Date : 2011.02.21

THRESHOLDS
G = Guess
D = Detection
R = Recognition

Assessor/Round																Log G	Log D	Log R
001	1							1	2	6	8					2,85	2,55	2,23
004	1							1	1	6	8					2,55	2,55	2,23
007	1							2	6	8						3,15	2,85	2,55
002	1							1	1	6	8					2,55	2,55	2,23

Sample Comments : ____________________

Specific Chemical Concentration Data

Chemical : Landfill odors

Concentration (ppm) : ____________

Response Key:
1 = Incorrect Guess
2 = Correct Guess
5 = Incorrect Detection
6 = Correct Detection
7 = Incorrect Recognition
8 = Correct Recognition

Final Results

	G	D	R
Avg. Log Value	2,77	2,62	2,31
Std. Dev.	0,29	0,15	0,16
Threshold	591	419	205

Fig. 2.15. Statistical analysis of the odour thresholds

The assessor did not indicate "detection" of the odour at dilution level 8 which is a dilution ratio of 500, but did indicate detection at the next higher odour concentration (lower dilution ratio of 250 and two times more odour). The assessor's individual estimated detection threshold is the geometric mean between 500 and 250, or 352. The result of this statistical method is called the "best-estimate" threshold.

$$(\log 500 + \log 250)/2 = (2.7 + 2.4)/2 = 2.55;$$

$$10^{2.55} = 352.$$

The geometric mean is used when calculating the "best estimate" threshold due to the lack of "equal variance" along the dilution ratio scale.

The individually estimated thresholds of four assessors are averaged to determine the detection threshold for which 50 % of individuals will observe the presence of an odour. In the Fig. 2.15 this average of 4 assessors' detection threshold is 2.62 or 420 odour units (antilog of 2.62 = 420 OU). The "detection threshold" value that is obtained from odour testing is actually derived from dilution ratios, and is therefore dimensionless.

2.3. CONCLUSIONS

1. Concrete and brickwork building debris could be applied for more efficient reduction of odour emissions from municipal solid waste landfills.
2. Upon covering waste with the 5 cm building debris layer, odour thresholds in 1-2 day period decreased 51.0-75.7 % compared to uncovered waste. Upon covering waste with the 10 cm building debris layer, odour thresholds decreased 51.0-79.7 %.
3. The specific odour emission rates determined during tests (1.04-1.26 $OU/m^2/s$ in the case of uncovered waste and 0.26-0.51 $OU/m^2/s$ upon covering waste with building debris) comply with the standard concentrations of odours from municipal waste landfills.
4. Changes in odour intensity result from waste characteristics, composition of covering material and environmental parameters.
5. A building debris layer thinner than 10 cm is insufficient to cover waste for a longer period (more than 3 days).
6. Upon covering waste with the 5 cm layer, instability of odour thresholds was recorded. Already on the third day of the test they reached the level of uncovered waste. Upon covering with the 10 cm layer, thresholds were gradually decreasing or varied insignificantly. In this case variations could have been determined by changes in the relative humidity of the ambient air.

3. MODELLING OF THE ODOUR DISPERSION AROUND MUNICIPAL SOLID WASTE LANDFILL SITE

Landfill odour impact is the negative effect, given in numerical form, of odorants emitted into the atmosphere by waste delivering or treating operations (Sironi *et al.* 2003). This effect is usually expressed by an odour concentration level and a percentile of occurrence of odour hours per year. For example, the 98th percentile of 5 OU_E/m^3 means that odour concentration in the air exceeds the concentration level of 5 OU_E/m^3 for the 2 % of the hours in a year.

As olfactometric odour concentration is given by a dilution factor, it can be viewed as a scalar property of turbulent atmosphere, exactly like any other gaseous pollutant (Sironi *et al.* 2003). However odour dispersion models have to meet unordinary requirements: odour impact is reflected by a peak concentration rather than hourly mean concentration and odour exposure is reflected by a percentile of concentration rather than a long-term-averaged concentration.

There are two types of dispersion models that meet these requirements and can currently be used to predict a map of the odour concentration frequency caused by odour emissions (H4 Odour… 2011):

- Steady state Gaussian models (e.g. Aermod, ADMS). These models represent a good mathematical approximation of odour plume behaviour when the odour source is located in relatively simple terrain; where the winds are relatively evenly distributed; and where the frequency of low wind speeds (< 1.5 m/s) is below 2 % for each compass direction.
- Non-steady state Lagrangian models (e.g. Calpuff, Austal). These models are capable of simulating a wider range of dispersal conditions than steady state models. They are therefore useful for odour assessments at sites which are characterized by complex air flow/dispersion conditions.

It should be noted that the dilution of the actual odour emission is the physical process that occurs in the atmosphere downwind of the odour source (McGinley *et al.* 2000). The "receptor" (people in the community) sniffs the diluted odour. If the receptor detects the odour, then the odour in the atmosphere is above the receptor's detection threshold level.

The aim of the study is to evaluate the potential odour impact around MSW landfill considering different odour control scenarios: rare covering of waste (minimum landfill operation requirements), daily covering of waste (application of construction and demolition debris for waste covering) and minimization of waste tipping area.

The section includes description of the dispersion model and input data, results of the odour dispersion modelling and its analysis.

3.1. DESCRIPTION OF THE DISPERSION MODEL ADMS AND INPUT DATA

Dispersion modelling was used to quantify the potential odour impact around MSW landfill site to the south east of Klaipėda in Lithuania. The case studies were completed with Atmospheric Dispersion Modelling System (ADMS), the software developed by Cambridge Environmental Research Consultants (CERC).

ADMS is a dispersion model using two parameters, the boundary layer height h and the Monin-Obukhov length L_{MO}, to describe the atmospheric boundary layer and using a skewed Gaussian concentration distribution to calculate dispersion under convective conditions (CERC 2010). The model is applicable up to 60 km downwind of the source and provides useful information for distances up to 100 km.

The Monin-Obukhov length is defined as:

$$L_{MO} = \frac{-u_*^3}{\left(\frac{\kappa g F_{\theta 0}}{\rho c_p T_0}\right)} \qquad (3.1)$$

in which u_* is the friction velocity at the Earth's surface, κ is the von Karman constant (0.4), g is the acceleration due to gravity, $F_{\theta 0}$ is the surface sensible heat flux, ρ and c_p are, respectively, the density and specific heat capacity of air and T_0 is the near-surface temperature.

In unstable or convective conditions, the Monin-Obukhov length is negative. The magnitude of the length is then a measure of the height above which convective turbulence, i.e. turbulent motions caused by thermal convection, is more important than mechanical turbulence, i.e. turbulence generated by friction at the Earth's surface.

In stable conditions, the Monin-Obukhov length is positive. It is then a measure of the height above which vertical turbulent motion is significantly inhibited by the stable stratification.

In the different regions of the boundary layer different mechanisms are important in generating turbulence. These are (CERC 2010):

1. surface heating or convectively generated turbulence (the convective eddies increase in energy as they rise through the boundary layer),
2. turbulence mechanically generated by shearing at the surface,
3. local shear, for instance at the top of the boundary layer, that can be a weak source of turbulence.

This approach to boundary layer stability, whereby the boundary layer structure is defined in terms of two variables (z/L_{MO} and z/h) supersedes the Pasquill-Gifford formulation, and differs crucially from the Pasquill formulation in allowing the variation of boundary layer properties with height to be included.

Two types of odour unit commonly used are “OU” and “OU_E” (CERC 2010):

- The odour unit (OU) strength of a release is the number of times the mixture must be diluted, at standard temperature and pressure, to reach the detection limit of 1 OU. The 1 OU contour shows the area where the model predicts the odour threshold is exceeded.
- An alternative unit for odour studies is the European odour unit (OU_E). One OU_E is the mass of pollutant that, when evaporated into 1 m^3 of odourless gas at standard conditions, has the same odour nuisance as 1 OU of a reference odorant.

The *Odours* option in ADMS enables the user to input emissions and calculate output in either of these units. European odour units (OU_E) are more commonly used and olfactometry measurements give odour concentrations in OU_E/m^3 (Carruthers and Kāla 2012).

For non-odour calculations, ADMS calculates mass concentrations in g/m^3 from mass emission rates in g/s, g/m/s, $g/m^2/s$ or $g/m^3/s$, for point, line, area and volume sources respectively (Gray and McHugh 2009). For odour calculations, if units of OU are used for odour calculations, the user specifies emissions in OU and ADMS produces output in OU. It is therefore necessary to convert the OU release strength to an “emission rate” in order to obtain output in OU. When using units of g, the concentration at the release point in g/m^3 is equal to the mass emission rate in g/s divided by the volume flow rate V. Hence the “emission rate” Q for the odour calculation can be calculated as follows:

$$Q = Q_{OU} \cdot V \cdot \frac{T_{STP}}{T_R} \tag{3.2}$$

where Q_{OU} is the strength of the release (OU), V is the volume flow rate at actual temperature and pressure (m^3/s), T_R is the release temperature (K) and T_{STP} is the standard temperature (288.15 K). Here the temperature ratio is included because the OU release strength is defined at standard temperature and pressure. Modelling in OU cannot be done for sources that have no plume rise (since V=0).

Since OU_E are mass measures, they can be treated identically to grams (Gray and McHugh 2009). The user specifies emissions in OU_E/s, $OU_E/m/s$, $OU_E/m^2/s$ and $OU_E/m^3/s$ for point, line, area and volume odour sources respectively; and results are obtained in OU_E/m^3.

Dumpiai MSW landfill was opened in 2009 (Fig. 3.1). Total area of operated landfill sections is 65047 m^2. Total capacity of the landfill for 20 years is 1.65 million m^3 of waste. About 120000 tons of municipal solid waste are treated annually.

Fig. 3.1. Location of the Dumpiai MSW landfill

Three different landfill operation and odour control scenarios were evaluated:

- The 1st scenario means that wastes are tipped in the area of operated landfill sections (Fig. 3.2). Temporary covering of waste is rare (minimum landfill operation requirements). Average odour emission rate is 1.15 $OU_E/m^2/s$.
- The 2nd scenario means that wastes tipped in the area of operated sections are temporary covered at the end of each working day. Construction and demolition (C&D) debris is applied as a covering material. Average odour emission rate is 0.41 $OU_E/m^2/s$.
- The 3rd scenario combines two odour reduction techniques (Fig. 3.3). Only one cell of the 2nd landfill section is used for waste tipping. Average odour emission rate is 1.15 $OU_E/m^2/s$. The remaining area is covered with different layers of C&D debris. Average odour emission rates are 0.38 $OU_E/m^2/s$ for the 1st landfill section and 0.41 $OU_E/m^2/s$ for the other cell of the 2nd section.

Fig. 3.2. Location of odour sources for the 1st and 2nd scenarios

Fig. 3.3. Location of odour sources for the 3rd scenario

Odour emission rates were obtained by dynamic olfactometry measurements during previous experimental investigation (Fig. 3.4). According to experimental data and landfill operation conditions some assumptions were made: height of all odour sources is 12 m, volume flux – 0.003 m^3/s, temperature – 23 °C; odour emission rates are constant with time, background concentrations are not considered; surface roughness of agricultural areas is 0.3 m, and meteorological data are hourly sequential (Fig. 3.5).

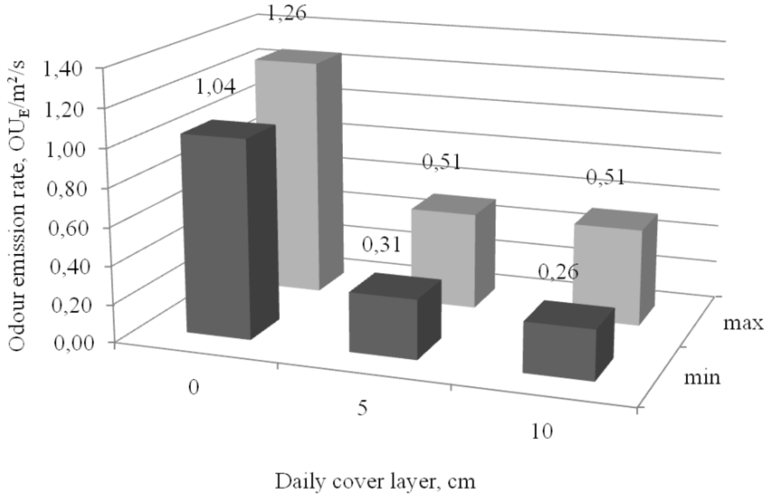

Fig. 3.4. Experimental odour emission rates

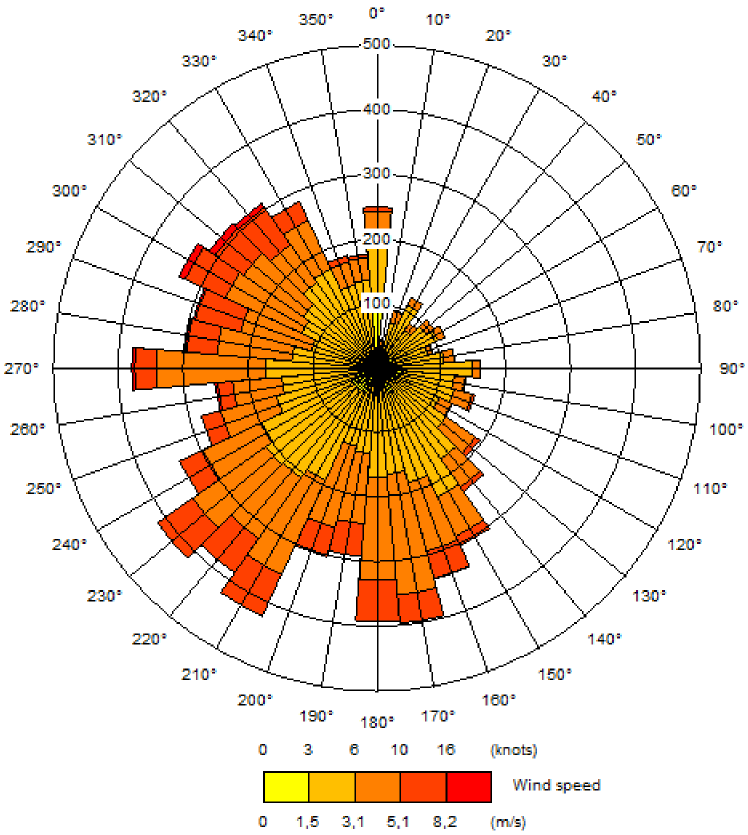

Fig. 3.5. Wind rose of hourly sequential meteorological data

The resulting "odour concentration" value of "1", calculated by a dispersion model, represents the odour detection threshold that was determined using the "best estimate criteria". A value of less than 1 represents "no odour" or "sub-threshold". A value of greater than 1 represents "odour" at a "supra-threshold" level (McGinley *et al.* 2000).

3.2. RESULTS OF THE ODOUR DISPERSION MODELLING AND ITS ANALYSIS

Knowing that 1 OU_E/m^3 corresponds to the odour detection threshold, the 98th percentile of such concentration shows the limit of the area beyond which the odour is perceived less frequently than 2 % of the time (Nicolas *et al.* 2006).

Lithuanian legislation (HN 121:2010) defines the maximum allowable odour concentration of 8 OU_E/m^3 at living areas. Sanitary protection zone of 500 m around MSW landfill site is required.

The human perception of odour consists of more than just “smell”. It represents a complex series of psychological and physiological responses to the quality of the odorant detected. Therefore, it is important to evaluate not only odour concentration and perception frequency but odour character also.

British Horizontal Guidance for Odour Management (April 2011) classifies relative “offensiveness” of odours arising from different processes as High, Medium and Low (Table 3.1). Odour exposure criteria are expressed in terms of 98th percentile of hourly mean (equivalent to 175 exceedences per year).

Table 3.1. Categories of odour offensiveness (Carruthers and Kāla 2012)

Offensiveness category	Indicative criterion (below which there is no reasonable cause for annoyance)
High	1.5 OU_E/m^3
Medium	3.0 OU_E/m^3
Low	6.0 OU_E/m^3

MSW landfills are categorized as high offensiveness odour sources and indicative criterion is the 98th percentile of 1.5 OU_E/m^3.

Results of all evaluated scenarios showed lower odour concentrations than the maximum allowable odour concentration defined in Lithuanian legislation. Considering the 1st scenario with the highest odour emission rates contours of 1 OU_E/m^3 and 1.5 OU_E/m^3 exceeded the sanitary protection zone (Fig. 3.6). The maximum distances of 1.5 OU_E/m^3 contour were about 900 meters to the north east and more than 1200 meters to the south of the landfill site. The high offensiveness odour could be perceived more frequently than 2 % of the time (more than 175 exceedences per year) by people living about 550 meters to the south of the landfill site. Only 0.5 OU_E/m^3 concentrations reached the surrounding villages Ketvergiai, Kaspariškiai and Gručeikiai.

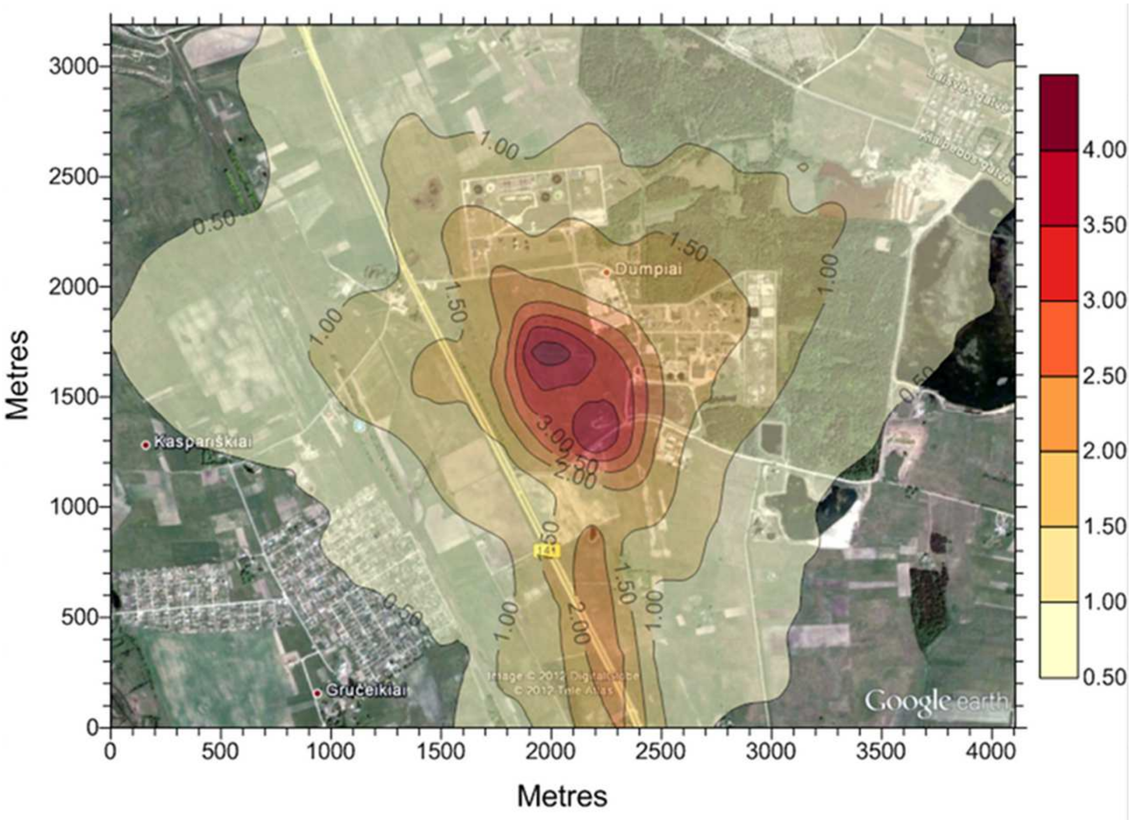

Fig. 3.6. 98th percentile of hourly averaged odour concentration for the 1st scenario

Results of the 2nd scenario showed that temporal covering of waste tipping areas with the C&D debris reduced odour concentrations effectively (Fig. 3.7). Odour concentrations at the same places decreased more than twice compared to the 1st scenario. Contour of 1 OU_E/m^3 did not exceed the sanitary protection zone. The maximum distances of 1 OU_E/m^3 contour reached only about 250 meters to the north and to the south of the landfill site. The odour concentration of 1.5 OU_E/m^3 could be exceeded only in the area of the 1st landfill section.

The 3rd scenario combined minimization of waste tipping areas and temporal waste covering (Fig. 3.8). One cell of the 2nd landfill section was used for waste tipping with rare covering; therefore odour concentrations around landfill site were higher compared to the 2nd scenario. Only the contour of 1 OU_E/m^3 exceeded the sanitary protection zone. The maximum distances of 1 OU_E/m^3 contour were about 600 meters to the north west and about 1200 meters to the south of the landfill site. The maximum distances of 1.5 OU_E/m^3 contour were only about 250 meters to the north west and to the north east of the landfill site.

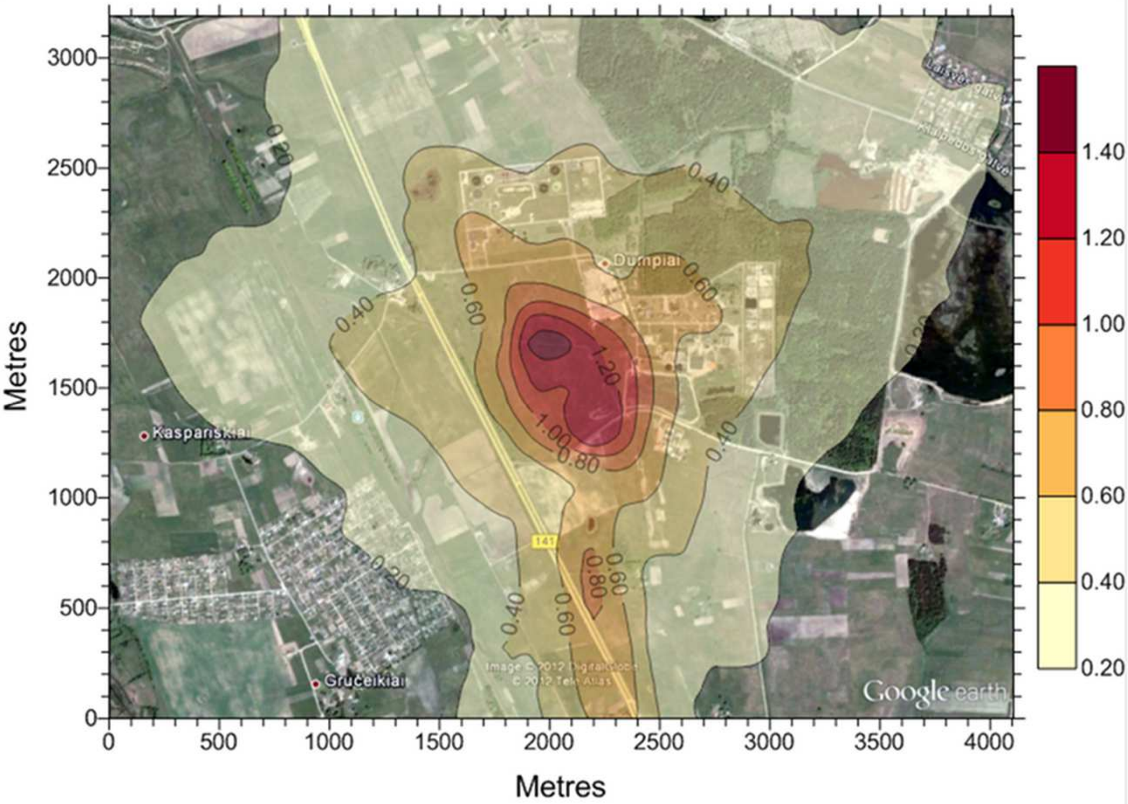

Fig. 3.7. 98th percentile of hourly averaged odour concentration for the 2nd scenario

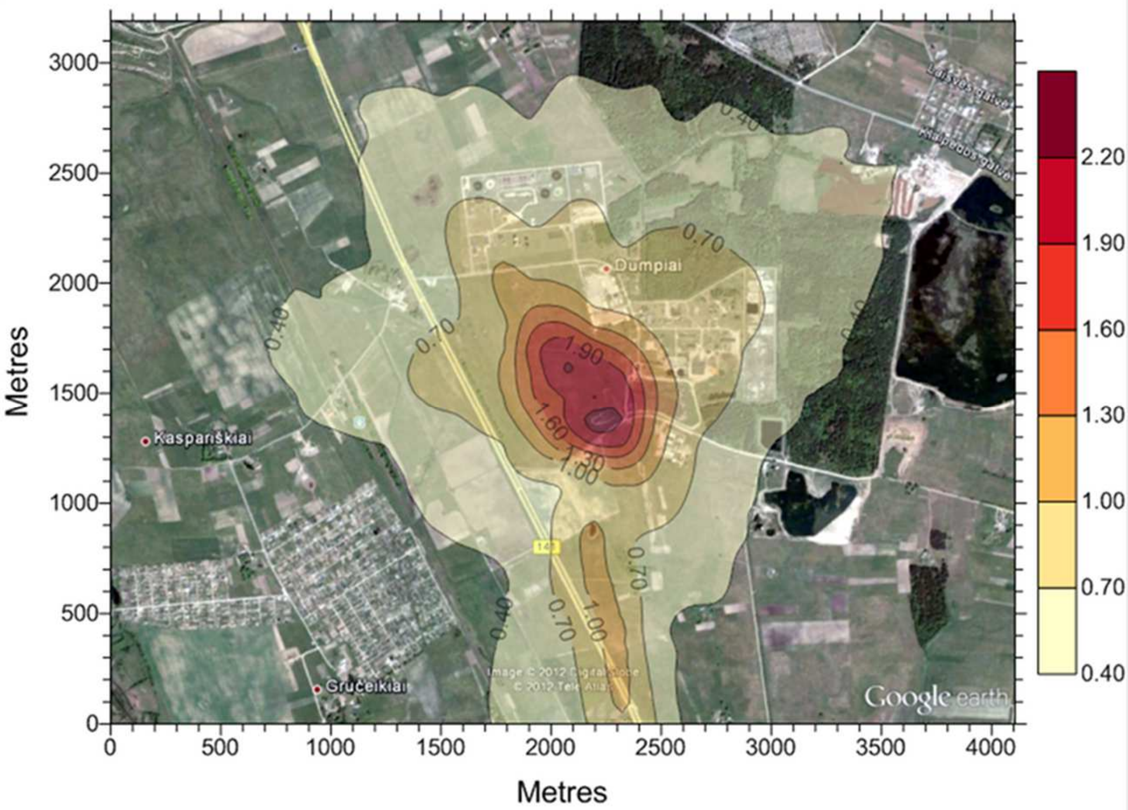

Fig. 3.8. 98th percentile of hourly averaged odour concentration for the 3rd scenario

Distances reached by odour to the south of landfill site seem to be overestimated, because meteorological data contain relatively high proportion of calm (<1.5 m/s) northern winds. Gaussian dispersion models may present high errors in these situations (Ubeda *et al.* 2010).

The reach of odour was predicted in 1.5 km from the odour source with the simple Gaussian dispersion model, whereas with the Tropos model the odour reached 3.3 km (Ubeda *et al.* 2010). Wind data employed in the modelling was close to 1 m/s, and both models were inversely dependent on this parameter. Tropos model only considered meteorological data higher than 1 m/s, so wind speed values lower than that limit were assumed to be 1 m/s. This assumption could be a strong source of errors.

The Gaussian model seems to be unsuitable for odour modelling, because the human response to the odour perception is very fast, in the order of 1 s, and odour modelling provide average hourly concentrations (Ubeda *et al.* 2010).

Sarkar *et al.* (2003) evaluated odour concentrations around the landfill site for short averaging times. The same locations were affected more with 3 min averaged concentrations (concentrations as high as 25 OU_E/m^3) as compared to hourly and 10 min averaged (concentrations as high as 20 OU_E/m^3) ones. The percentage frequency of occurrence of such critical events having ranges of odour concentrations as high as 10-25 OU_E/m^3 are very low (Sarkar *et al.* 2003).

3.3. CONCLUSIONS

1. Dispersion modelling was found useful for odour impact assessment within communities near landfill sites.
2. Considering the 1st scenario with the highest odour emission rates contour of 1.5 OU_E/m^3 exceeded the sanitary protection zone. The maximum distances were about 900 meters to the north east and more than 1200 meters to the south of the landfill.
3. Considering the 2nd scenario temporal waste covering and application of concrete and brickwork building debris effectively reduces landfill odours. Odour concentrations at the same places decreased more than twice compared to the 1st scenario.
4. The 3rd scenario combined minimization of operation area and temporal waste covering. The maximum distances of 1.5 OU_E/m^3 contour were only about 250 meters to the north west and to the north east of the landfill.
5. Choice of the dispersion model influences accuracy of the odour impact assessment. Gaussian dispersion models are not suitable for evaluation of calm winds and complex terrain.

GENERAL CONCLUSIONS

1. Concrete and brickwork building debris could be applied for more efficient odour reduction from municipal waste landfills. Upon covering waste with the 10 cm building debris layer, odour thresholds in 1-2 day period decrease 51.0-79.7 %.
2. Variation of the specific odour emission rates determined during tests result from waste characteristics, composition of covering material and environmental parameters.
3. A building debris layer thinner than 10 cm is insufficient to cover waste for a longer period (more than 3 days). Upon covering waste with the 5 cm layer, instability of odour thresholds was recorded. Upon covering with the 10 cm layer, odour thresholds varied insignificantly because of changes in the relative humidity of the ambient air.
4. Dispersion modelling was found useful for odour impact assessment within communities near landfill sites. Modelled scenarios show that temporal waste covering and application of building debris effectively reduces landfill odour emissions. Odour concentrations at the same places decrease more than twice and do not exceed the sanitary protection zone of the landfill site.
5. Choice of the dispersion model influences accuracy of the odour impact assessment. Gaussian dispersion models are not suitable for evaluation of calm winds and complex terrain.

RECOMMENDATIONS

1. 5 cm layer of the concrete and brickwork building debris could be applied to reduce landfill odour emissions only for 1-2 day period. Upon covering waste with the 5 cm layer, odour thresholds decrease 51.0-75.7 %.
2. In order to avoid instability of landfill odour emissions for a longer period (more than 3 days) minimal 10 cm layer of the building debris should be applied. Upon covering waste with the 10 cm layer, only ambient air parameters influence odour thresholds.
3. In terms of landfill capacity limits temporal waste covering with building debris and minimization of operating area should be combined. Considering this scenario odour concentration of 1.5 OU_E/m^3 should not exceed the sanitary protection zone of 500 m.
4. For the evaluation of calm winds and complex terrain non-steady state Lagrangian models (e.g., CALPUFF) should be used instead of Gaussian dispersion models (e.g., ADMS).

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Batavičius, T.; Vasarevičius, S. 2012. Experimental tests on the influence of waste covering layer on odour reduction (given for publication to Polish Journal of Environmental Studies).

Batavičius, T.; Vasarevičius, S. 2012. Construction and demolition debris as a waste cover for odour reduction (given for publication to 15th Conference of Junior Researchers "Science – Future of Lithuania", section of Environmental protection engineering, 12 April 2012, Vilnius).

LIST OF PRESENTATIONS

Batavičius, T.; Vasarevičius, S. 2011. Landfill odours and possibilities of its evaluation and reduction in Lithuania. Poster presentation in 14th Conference of Junior Researchers “Science – Future of Lithuania”, section of Environmental protection engineering, 14 April 2011, Vilnius.

Batavičius, T.; Vasarevičius, S. 2012. Construction and demolition debris as a waste cover for odour reduction. Poster presentation in 15th Conference of Junior Researchers “Science – Future of Lithuania”, section of Environmental protection engineering, 12 April 2012, Vilnius.

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