Universität für Bodenkultur
Department für Wasser, Atmosphäre und Umwelt
Institut für Abfallwirtschaft

Quantification of landfill gas emissions in biocovers – an experimental simulation in lysimeters

Diplomarbeit
zur Erlangung des akademischen Grades
Diplomingenieurin

eingereicht von
DI (FH) Marlies Hrad
Studienkennz.: 416/Matr. Nr.: 0740966

Wien, Oktober 2010

ACKNOWLEDGEMENTS

An dieser Stelle möchte ich mich bei Prof. Peter Lechner und dem ABF-Team für die Möglichkeit bedanken, diese Arbeit am Institut für Abfallwirtschaft zu schreiben.

Ganz besonders möchte ich mich bei DI Dr Marion Huber-Humer bedanken, die mir mit Ratschlägen und konstruktiver Kritik stets hilfreich zur Seite stand.

I have furthermore to thank Graeme Buchan for reviewing this thesis, especially for correcting both my English style and grammar and for offering suggestions for improvement.

Mein Dank gilt auch Ing. Thomas Ebner und DI Oliver Gamperling, die mich immer mit Rat und Tat unterstützten und für meine Anliegen und Fragen stets ein offenes Ohr hatten.

Weiters möchte ich mich bei DI Dr Bernhard Wimmer für die Bereitstellung sämtlicher Daten und Geräte bedanken, die zur Ausarbeitung der Diplomarbeit notwendig waren.

Danke auch an Lukas Egle und Andreas Schuh für die Unterstützung bei den Haubenmessungen.

Ganz besonderer Dank gilt meinen Eltern, ohne die dieses Studium nie möglich gewesen wäre. Durch ihre moralische und finanzielle Unterstützung ermöglichten sich mir neben der Fachhochschule auch einen universitären Abschluss, sowie einen unvergesslichen Auslandsaufenthalt in Neuseeland. Abschließend möchte ich auch meinem Freund Florian für seine Motivation und Unterstützung während der Diplomarbeit danken.

Marlies Hrad

Himberg, Oktober 2010

KURZFASSUNG

Autor: Marlies Hrad

Thema der Diplomarbeit: Quantifizierung von Deponiegas in Methanoxidationssystemen („biocovers“) – eine experimentelle Simulation in Lysimetern

Im Rahmen des Forschungsprojektes „Nutzraum“ („I**N**novative In-Sit**U** Me**T**hoden **Z**ur Sanie**R**ung von **A**ltablagerungen **U**nd konta**M**inierten Standorten“) wurden Groß-Lysimeterversuche am Standort Seibersdorf (AIT Austrian Institute of Technology) konzipiert und errichtet. Ziel dieses Projektes ist es, einen optimalen technischen Aufbau einer Deponieabdeckschicht, die gleichzeitig als Methanoxidations- und Wasserhaushaltsschicht dient, für das Verfahren der In-Situ Aerobisierung zu finden. Insgesamt wurden drei Methanoxidationssysteme („biocovers“) mit unterschiedlichen Substraten zur Forcierung der methanoxidierenden Prozesse sowie eine Kontrollschicht parallel in Lysimetern eingebaut.

Inhaltlich beschäftigt sich die vorliegende Diplomarbeit mit der relativen Emissionsminderung der passiven Deponieabdeckschichten vor der In-Situ Aerobisierung. Die Oxidationsleistung der Abdeckschichten wurde anhand von Gaskonzentrations- und Temperaturprofilen sowie mithilfe von Haubenmessungen und oberflächennahen FID-Vermessungen bestimmt. Darüber hinaus wurde an ausgewählten Messtagen die Oxidationsleitung basierend auf der Methode der Isotopenfraktionierung quantifiziert. Die zum Einsatz gekommene Messhaube wurde eigens für dieses Projekt konstruiert und beruht auf dem Prinzip der geschlossenen, dynamischen Kammermethode. Die Schwierigkeit bestand darin, die bestehende Belüftungslanze (D = 6 cm) in das Design der Messeinrichtung zu integrieren um die gesamte Lysimeteroberfläche (4 m²) abzudecken und dabei die Messungen nicht zu beeinträchtigen. Die Eignung des Messsystems wurde für bekannte Methan- und Kohlendioxidströme zuvor im Labor validiert.

Aus den Ergebnissen geht hervor, dass nur bei mineralischen Abdeckschichten (Lysimeter C und D) Methanemissionen nachgewiesen werden konnten, wohingegen bei Abdeckschichten aus reifem Klärschlammkompost bzw. einer Sand-Kompost-Mischung (Lysimeter A und B) keine Methanaustritte detektiert wurden. Die Abdeckschichten wurden direkt mit Deponiegas aus den darunterliegenden Abfallkörpern versorgt, wodurch eine Kontrolle der Methanfracht nicht möglich war. Des weiteren ist zu beachten, dass die Methanbildung in den Abfallschichten sehr unterschiedlich verlief und es zu keiner einheitlichen Gasbildung in den Lysimetern A, B, C und D kam. Die Isotopenfraktionierung sowie die Gaskonzentrations- und Temperaturprofile von Lysimeter A und B weisen darauf hin, dass ein Großteil der Methanfracht bereits in der Gasverteilungsschicht und im Abfall selbst reduziert wurde. Die Abdeckschicht aus Klärschlammkompost in Lysimeter A zeigte aber grundsätzlich vielversprechende Ergebnisse im Bezug auf die Umgebungsbedingungen für methanotrophe Bakterien. Im Vergleich zu den anderen Abdeckschichten, konnte im Klärschlammkompost während des gesamten Untersuchungszeitraumes ein optimales Feuchte- und Temperaturmilieu für die Methanoxidation gehalten werden.

ABSTRACT

Author: Marlies Hrad

Topic of the thesis: Quantification of landfill gas emissions in biocovers – an experimental simulation in lysimeters

Within the framework of the joint project on remediation of old sites and dumps ("Nutzraum"), a lysimeter experiment was set up at the Austrian Institute of Technology in Seibersdorf, Austria, in February 2008. The project aims to derive design criteria for optimised composition of landfill covers regarding water balance and methane emission mitigation, in combination with in-situ aeration. Therefore, three experimental CH_4 oxidation biocovers using different substrates to enhance methane uptake as well as one control cover have been installed in parallel in lysimeters.

The diploma thesis presents data concerning the relative emission reduction effect of the passive biocovers prior to in-situ aeration. The methane oxidation efficiency of the biocovers was determined by gas composition and temperature profiles measured within the lysimeters, and by surface flux measurements as well as surface methane concentration screenings. In addition, methane oxidation was quantified at selected dates using stable isotope methods. The surface flux measurements were conducted using a closed dynamic chamber, which was constructed specially for this project. The challenge was to include the aeration pipe (D = 6 cm) of the lysimeter into the design of the accumulation chamber in order to cover the whole lysimeter surface (4 m^2) without disturbing the measurement procedure. Laboratory tests were conducted to verify the performance and accuracy of the measurement system with different known methane and carbon dioxide fluxes.

Among the four biocovers studied, CH_4 emission fluxes could only be detected from the biocovers with mineral soil covers (lysimeters C and D), whereas no CH_4 emissions were measured on the biocovers with mature sewage sludge compost and sand/compost mixture, respectively (lysimeters A and B). The biocovers were fed with biogas directly from the waste mass; therefore, it was not possible to control the upward biogas flux. It should be noted that methane production developed differently in the solid waste layers and methane load of lysimeters A and B did not indicate the same range as in lysimeters C and D. Stable isotope analyses as well as gas composition and temperature profiles of lysimeters A and B revealed that methane was reduced in the gas distribution system and in the waste itself. However, the mature sewage sludge compost (SSC) placed in lysimeter A showed, in principle, very promising results regarding the optimal ambient conditions for methanotrophic bacteria. In contrast to the other biocovers, the SSC-cover was capable of retaining the moisture content and the temperature profiles at an optimum level during the investigation period.

TABLE OF CONTENTS

1 INTRODUCTION

1.1 Motivation

Methane (CH_4) is an important greenhouse gas (GHG), being 25-fold more radiatively active than carbon dioxide (CO_2) over a 100-year period (IPCC, 2007a). Landfills are regarded as a major source of anthropogenic methane emissions, globally releasing between 30 and 70 Tg CH_4 per year (Reay et al., 2007) or 10 – 20 % of the total global annual anthropogenic CH_4 emissions (IPCC, 2007b). In developed countries landfill CH_4 emissions have been stabilised during the last 20 years as a result of increased landfill CH_4 recovery, decreased landfilling and waste generation, increased composting of source-separated bio-waste as well as the tendency towards either thermal or mechanical-biological pretreatment of municipal solid waste (MSW) (Bogner et al., 2007). For example, due to the implementation of the EU Landfill Directive (CEC 1999) landfilling of organic waste in Europe has progressively been reduced. On the other hand, methane generation from landfill sites is likely to rise in future as a result of rapid increase in population and urbanisation in developing countries (Bogner et al., 2007) (see Figure 1.1).

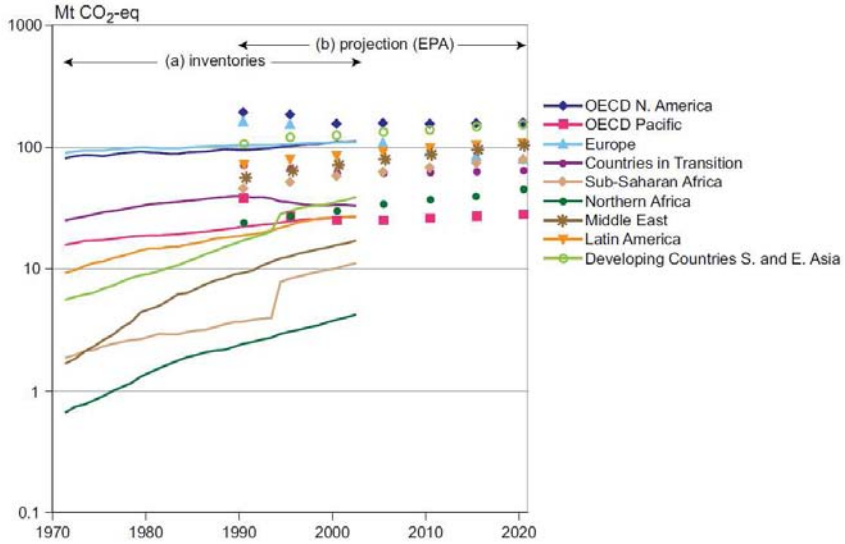


Figure 1.1: Comparison of regional emissions estimates for five-year intervals from 1990-2020 to annual historical estimates from 1971 – 2002 (from Bogner et al., 2007)

Depending on the landfilling technology applied, the waste and site characteristics as well as the environmental conditions, landfill gas will continue to be generated for periods between 15 and 30 years after final depositions of the waste (Williams, 2005). However, low-level gas production may continue up to 100 years after waste emplacement (Huber-Humer et al., 2009; Williams, 2005). Figure 1.2 shows the time-dependent methane production and recovery over a landfill lifetime. Engineered landfills with landfill gas (LFG) utilisation and control systems lead to considerably reduced CH_4 emissions and recovery of energy. According to Haubrichs and Widmann (2006) and Mosher et al. (1999) gas recovery systems can reduce about 80 – 90 % of CH_4 emission to the atmosphere depending on the boundary conditions of each landfill and its gas collection and control system. In contrast, Humer and Lechner (1999a) reported that in practice only 40 to 60 % of the landfill gas can be

controlled over the entire landfill lifetime by a gas recovery system as emission may also escape preferentially from and around wells and along the routes of installed landfill equipment.

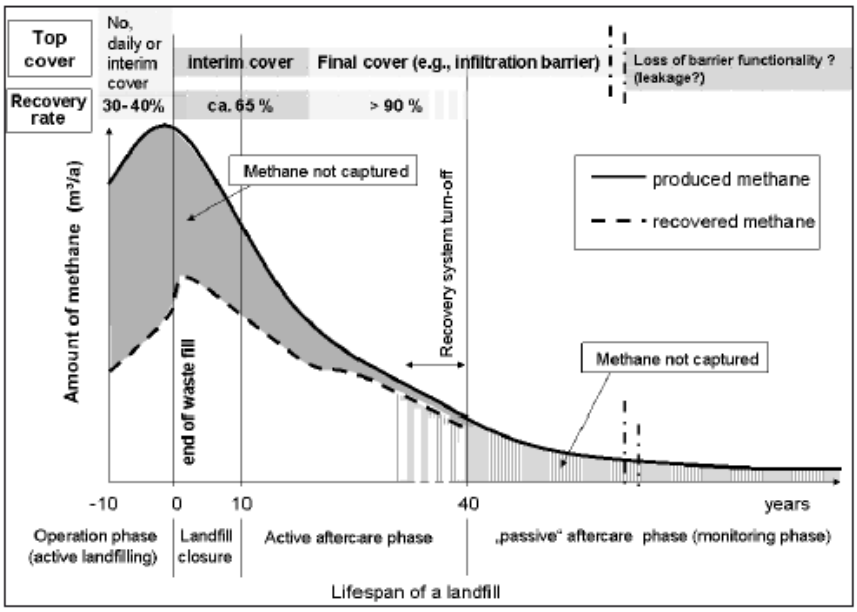


Figure 1.2: Time-dependent methane production and recovery over a landfill lifetime. Methane emissions (methane not captured) are shown as a function of cover type and do not include methane oxidation removal (from Huber-Humer et al., 2008b)

In the long term recovery facilities are not economically and technically suitable as only high methane concentrations (> 45 %) can be used for heat and power production (Haubrichs and Widmann, 2006). Once CH_4 concentrations reach 20 – 25 % and total gas production rates are 10 – 15 m³/h, the gas is flared with risk of producing toxic combustion products, or is just escaping to the atmosphere (Scheutz et al., 2009; Huber-Humer et al., 2008b). In addition, installation of a gas to energy system at old and small landfills is technically difficult and in most cases not feasible as LFG emissions are too low for energetic use (Damman et al., 1999).

Microbial methane oxidation, as a natural biogeochemical process, seems to be a preferable option to reduce, in particular, low-level CH_4 emissions with a simple, passive landfill cover system. A considerable amount of methane can be biologically oxidised in the landfill cover to carbon dioxide, which has a lower global warming potential than methane. In recent years, specialised landfill “biocover” designs engineered to optimise CH_4 oxidation have demonstrated huge potential to mitigate residual landfill CH_4 emissions (Stern et al., 2007; Barlaz et al., 2004; Huber-Humer, 2004). Engineered biocovers usually consist of a coarse gas distribution layer to homogenise and distribute landfill gas to the overlying substrate layer. Such landfill cover systems can be potentially applied in the absence of a gas collecting system or as a complementary strategy for the control of CH_4 emissions escaping gas collection, as well as to mitigate residual emission from mechanically and biologically pretreated (MBT) waste or old and small landfills with low CH_4 production potential. Moreover, biocovers can also be combined with forced in-situ aeration as a postclosure remediation method for closed, old landfills. Importantly, biocovers offers a low-cost alternative for methane reduction especially required in developing countries.

The Austrian joint project on remediation of old sites and dumps ("Nutzraum"[1]) aims to derive design criteria for optimised composition of landfill covers in combination with in-situ aeration regarding water balance and emission mitigation. Within this framework, three experimental CH_4 oxidation biocovers using different substrates to support growth of methanotrophic bacteria as well as one control cover have been installed in parallel in lysimeters. The lysimeters, each measuring 2 m x 2 m x 3 m, consisted of 1 m thick substrate layer underlain by a 0.2 m gas distribution layer and 1.5 m layer of fresh municipal solid waste. In contrast, the "control" lysimeter included only a 0.5 m substrate layer (see section 3.3.1).

This diploma thesis covers only a part of the above mentioned project and presents the evaluation of the passive biocover performances under natural conditions by monitoring of the relative emission reduction effect. Therefore, the remaining methane and carbon dioxide emissions are determined and compared to the control cover.

1.2 Tasks and Objectives

The present diploma thesis aims to assess the efficiency of different passive biocovers in reducing methane emission before the underlying waste is actively aerated for rapid waste stabilisation. Therefore, the residual landfill gas emission will be quantified. Further subgoals are:

- the development of a measuring system to quantify CH_4 and CO_2 surface emission from the different landfill covers in the lysimeters
- monitoring the development of the LFG generation and remaining CH_4 and CO_2 surface emission as well as CH_4 oxidation in the cover

[1] see more information at www.wau.boku.ac.at

2 BACKGROUND INFORMATION

2.1 Landfill gas generation

Landfill gases (LFG) are produced during the microbial anaerobic decomposition of waste containing degradable organic components. The biological decomposition of one tonne of deposited municipal solid waste produces roughly 160 to 250 m³ of landfill gas (Abichou et al., 2004; Humer and Lechner, 1999). These gases are comprised typically of approximately 55 vol. % CH_4, 44 vol. % CO_2 and numerous trace gases including hydrogen sulphide (H_2S) and volatile organic compounds (VOC) resulting from microbiological as well as physico-chemical processes. Hence, one tonne of deposited municipal solid waste results in approximately 88 to 138 m³ of CH_4. However, the generation of landfill gas depends heavily on the landfilled waste volume, the solid waste properties, waste age, landfill operation and water content of the landfill waste (Steinlechner et al., 1994; Scheutz et al., 2009).

2.1.1 Waste degradation processes

Waste degradation in landfills is a multi-stage process and can be divided into five main stages (Williams, 2005). Every stage is accomplished by a certain group of bacteria and has different requirements regarding pH and temperature. Figure 2.1 shows the different degradation steps and corresponding by-products during the decomposition of biodegradable waste. Until the waste has reached the final stage, the different stages may be proceeding simultaneously because of the heterogeneous nature of waste (Williams, 2005).

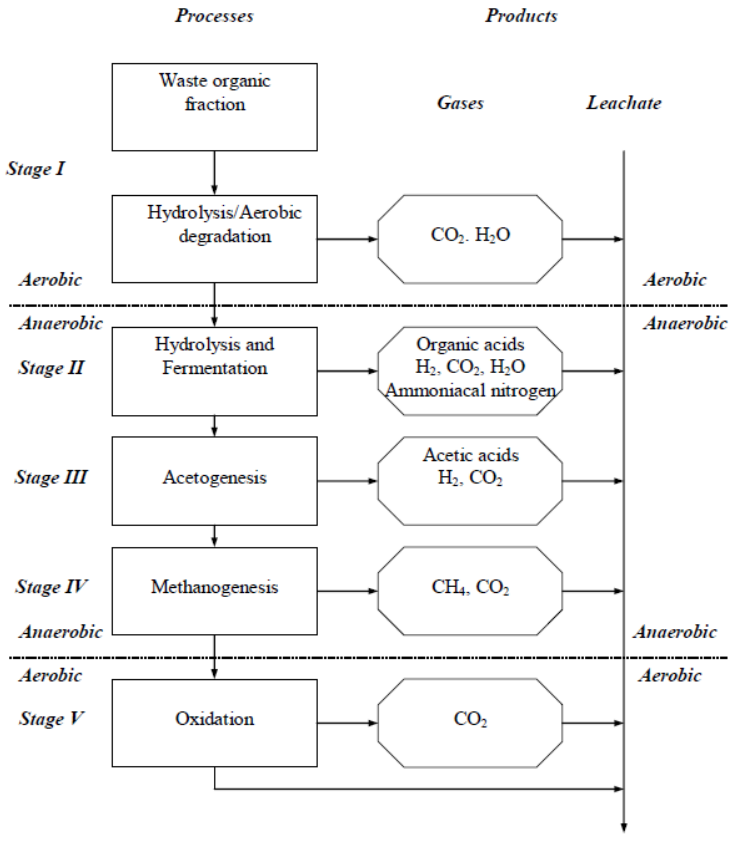


Figure 2.1: Major stages of waste degradation in landfill (Williams, 2005)

Aerobic degradation in the landfill occurs during the emplacement of waste and for a period thereafter. In this stage, aerobic bacteria, which only live in the presence of oxygen, are responsible for decomposition of the waste organic fraction. Thereby readily degradable organic matter is decomposed to CO_2, water and heat (Steinlechner et al., 1994). CO_2 may be released as a gas or is transformed to carbonic acid (H_2CO_3) when it dissolves in water. However, the aerobic stage lasts only for a short time (days or weeks) (Williams, 2005), as the ability of oxygen to diffuse into the waste mass is lost within the waste body. The predominant part of the landfilled waste will soon become anaerobic.

During the second stage, carbohydrates (cellulose, starch and sugar), lipids and proteins are hydrolysed to glucose, amino acid, fatty acids etc. by means of extra-cellular enzymes, which are produced by facultative and obligatory anaerobic bacteria (fermenters). The hydrolysis process is a very important process in the landfill environment, as the solid organic compounds must be solubilised before the microorganisms can convert them. The resulting dissolved organic fragments are then fermented to CO_2, hydrogen (H_2), ammonia and organic acids like butyric acid, propionic acid, formic acid, valeric acid, etcetera. As ammonia largely derives from the deaminisation of proteins, the leachate contains high concentrations of ammoniacal nitrogen.

In the acidifying stage, organic acids formed in the second stage are transformed into acetic acid (CH_3COOH), CO_2 and H_2 by acetogen microorganisms. Some bacteria are also able to directly decompose carbohydrates to acetic acid in the presence of CO_2 and H_2. During the anaerobic stages, hydrogen sulphide may also be produced by the reduction of sulphate compounds by sulphate-reducing microorganisms. Due to the presence of organic acids, a pH level of 4 or even less is generated (Williams, 2005).

During the fourth stage, which is also the longest time stage, CO_2, H_2 and acetic acid are consumed by methanogenic microorganisms in the following way:

$4H_2 + CO_2 \rightarrow CH_4+2H_2O$ (reductive methane formation) and (2.1)

$CH_3COO^- + H^+ \rightarrow CH_4 + CO_2$ (splitting up of acetic acid/decarboxylation) (2.2)

About 70 % of the methane produced originates from splitting up of acetic acid although the responsible bacteria are the most sensitive ones in the whole process (Le Mer and Roger, 2001; Jeris and McCarty, 1965). During the methanogenic stage LFG can be generated over a temperature range of 30 – 65 ℃ as two classes of microorganisms (mesophilic and thermophilic) are active (Williams, 2005). Ideal for methanogenic microorganisms is a pH range from 6.8 to 7.5. However, microbial activity seems to continue down to pH 5 and up to pH 9 (Williams, 2005).

In the final stage, new aerobic microorganisms slowly replace the anaerobic forms as the acids are used up in the production of the LFG (Williams, 2005). Methanotrophic microorganisms utilise CH_4 as their only source of carbon and energy. While some of the CH_4 is consumed for energy yield gained from its oxidation to CO_2, another fraction is incorporated into biomass. The exothermic process is catalysed by enzymes (see section 2.3) and can be simplified to the following stoichiometric equation:

$CH_4 + (2\text{-}x)\ O_2 \rightarrow (1\text{-}x)\ CO_2 + x\ CH_2O + 2\ H_2O$ + heat (210,8 kCal/mol) (2.3)

where x is the fraction of carbon that is assimilated into biomass (CH_2O). Methane oxidation causes a net decrease in the number of gas molecules as two molecules of

oxygen are consumed per oxidised methane molecule. The resulting volume reduction enhances the ability of oxygen to diffuse deeper into the waste mass (Huber-Humer et al., 2008a). Under aerobic conditions also hydrogen sulphate may be formed in waste with high concentration of sulphate.

2.1.2 Landfill gas composition over time

The gas composition changes with each of the five phases of degradation between different landfills (see Figure 2.2). The first aerobic phase continues until available oxygen is depleted. Depending on how much oxygen is present during the emplacement of waste in the landfill, the first stage lasts only for a short period of time. In the hydrolysis and fermentation stage carbon dioxide is the main constituent of landfill gas, which may rise to levels up to 80 %. The presence of CO_2 and H_2 reduces the content of nitrogen (N_2). H_2 and CO_2 levels start to decrease throughout the third stage, as methanogenic bacteria start to grow producing CH_4. The anaerobic methanogenic phase is characterised by a fast increase in the production of CH_4 followed by an extended period where gas production is relatively stable. This stage is the main LFG production phase, with typical LFG composition of approximately 55 % CH_4 and 44 % CO_2 and low concentrations of H_2. The maximum of LFG production is reached when the majority of degradable refuse will decompose (Thomas and Ferguson, 1999). The methanogenic stage of a landfill site is typically established within a half to one year and can last a few years or decades, depending upon environmental conditions as well as site and waste characteristics and landfilling technology used (Steinlechner et al., 1994, Williams, 2005; Scheutz et al., 2009). The last stage marks the end of the degradation reactions and a return to aerobic conditions. N_2 is now present in significant concentrations in the gas.

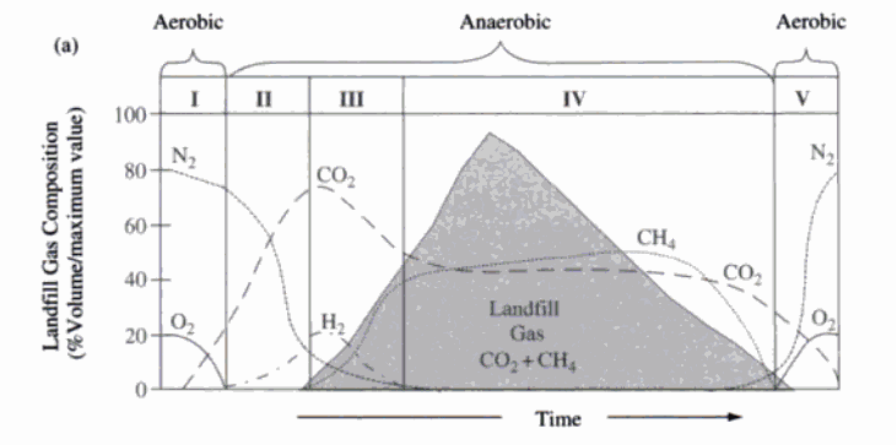


Figure 2.2: Landfill gas composition over time (Williams, 2005)

2.2 Landfill gas emission

Emissions from landfills vary roughly by seven orders of magnitude, from less than 0.0004 to more than 4000 g CH_4/m^2 per day (Bogner et al., 1997a). This natural variability range of net emissions results from production (methanogenesis), consumption (methanotrophic oxidation) (see section 2.3), temporary storage (e.g. in pores, dissolved in water) and gaseous transport processes in a landfill (Bogner et al., 1997a). Several factors affect gaseous transport processes and control LFG emissions (Kjeldsen, 1996), as follows:

1. Meteorological conditions (e.g. barometric pressure, precipitation, temperature, wind);
2. Soil conditions (e.g. cracks and fissures, permeability, diffusivity, porosity, water content, organic matter content); and
3. Waste and landfill condition (e.g. gas production rate)

The major transport mechanisms for gas emissions from landfills include diffusion and advection. While diffusion is driven by variations in gas concentrations in the soil, advection is caused by pressure gradients induced by changes in atmospheric pressure (Gebert and Gröngröft, 2006; Christophersen and Kjeldsen, 2001; Christophersen et al., 2001; Latham and Young, 1993), landfill gas production (Kjeldsen, 1996) and wind turbulence (Poulsen, 2005). A decline in atmospheric air pressure will draw out gas from the landfill body, while an increase in barometric pressure will lead to reduced gas release. Figure 2.3 depicts the release of CH_4 through the top cover of a landfill following alteration in barometric pressure (Pirkle et al., 1993, cited in Kjeldsen, 1996). Kjeldsen and Fischer (1995) investigated the changes in gas composition in a well of a Danish landfill under a decrease in barometric pressure from 1022 mbar to 1010 mbar over a 33 h period. The authors observed that CH_4 concentration increased at 80 cm depth from below 1 % to near 40 % while CO_2 concentration showed only an increase of approximately 20 %.

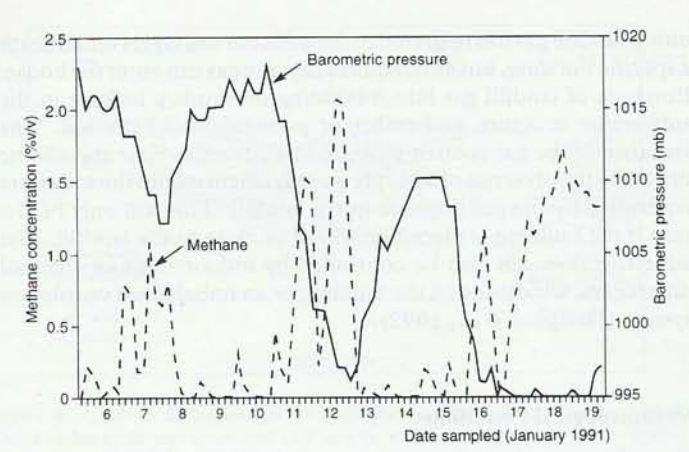


Figure 2.3: Methane concentration measured under a groundsheet placed on a landfill top cover and barometric pressure vs time (Pirkle et al., 1993, cited in Kjeldsen, 1996)

Czepiel et al. (2003) found a robust negative relationship between landfill CH_4 emissions and barometric pressure, as can be seen in Figure 2.4.

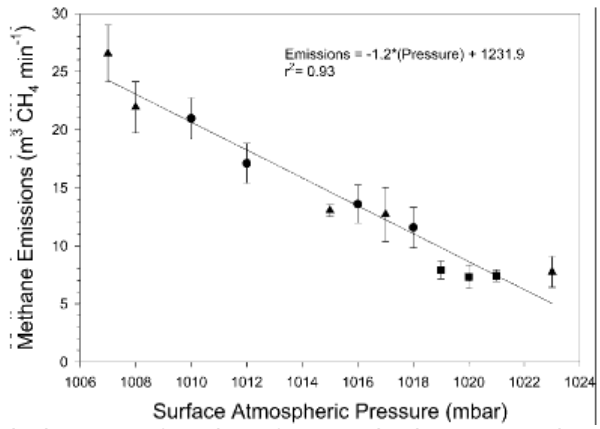


Figure 2.4: CH_4 emissions as a function of atmospheric pressure (Czepiel et al., 2003)

In some cases surface LFG emission even becomes negative during rising barometric pressure, which causes an influx of atmospheric air through the biofilter (Gebert and Gröngröft, 2006) and the top layer of landfill (Nastev et al., 2001), respectively.

Investigations by Poulsen (2005) showed that wind induced emissions can substantially contribute to total gas emissions, especially in moist soils and at wind speeds exeeding 5 m/s (see Figure 2.5).

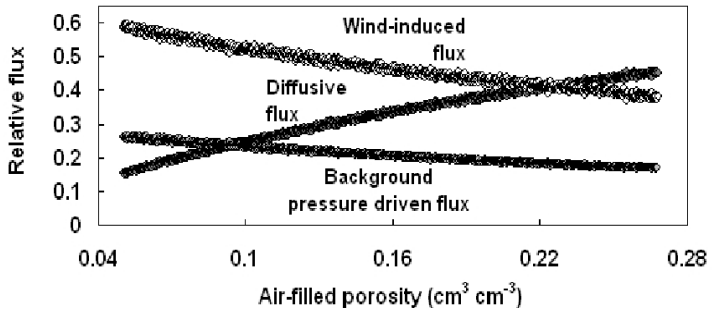


Figure 2.5: Relative fluxes of CO_2 caused by background pressure gradients, molecular diffusion and wind induced gas transport at a Danish landfill (Poulsen, 2005)

These results confirmed that the amount of gas emission from landfills is also dependent on both moisture content and physical components of the soil cover such as grain size, soil texture and porosity which affect the mobility of gas within the soil cover (Tecle et al., 2009; Poulsen, 2005; Börjesson and Svensson, 1997). At low air-filled porosities advective gas transport is the most important transport mechanism while diffusive flux dominates under higher air-filled porosity at drier conditions (Poulsen, 2005). The amount of precipitation, therefore, influences gaseous transport processes both in the long and the short term since high moisture content reduces the permeability and diffusivity of soil. In the short term, large amounts of precipitation can lead to water saturation of the top soil resulting in increased lateral gas migration. Heavy rainfalls often follow a substantial decrease in barometric pressure. This unfavourable combination has been reported as the main factor at a landill gas explosion in a house close to a Danish landfill (Kjeldsen and Fischer, 1995).

Air temperature has direct effects on soil temperature, which has a close relationship to biological CH_4 oxidation in landfill covers (see section 2.4.5). Soil temperature also controls soil physical characteristics such as swelling and contracting (Tecle et al., 2009). These are important factors on regulating moisture content and LFG emissions throughout the soil cover.

Cracks and fissures, resulting from waste settlement or desiccation of the landfill cover during dry periods, can serve as pathways for methane. Previous studies have found that main emissions occurred at a few areas or hot spots associated with heterogeneities in the cover soil (Nozhevnikova et al., 1993, cited in Börjesson and Svensson, 1997). Bergamaschi et al. (1998) observed that 70 % of CH_4 emissions through landfill cover soils in Germany and The Netherlands were attributed to cracks.

In general, the gaseous transport process is governed by several processes and factors in an often complex network of connections. The transport mechanism is controlled by both diffusive and advective forces. It is difficult to evaluate which of the two mechanisms dominates in a specific case (Kjeldsen, 1996). In some cases pressures in landfills could be substantially higher than barometric pressure,

especially under saturated cover soils or under low permeability covers (e.g. clay or geomembrane composite) (Scheutz et al., 2009). Pressures as high as 2 kPa with respect to atmosphere have been recorded (Hartless, 1995). There have been also instances in a coarse sandy top cover where only very small pressure differences are needed for advective flux to dominate the diffusional flux (Kjeldsen, 1996; Christophersen and Kjeldsen, 2001). Several authors claim that the assumption of a preponderance of an advective flux is supported by observations of a robust inverse relationship between CH_4 emissions and barometric pressure at several landfills. (Stern et al., 2007, De Visscher et al., 2004; Czepiel et al., 2003). In particular, this is true for landfill sites without a gas collection system.

Kjeldsen (1996) concluded that the most important factors controlling CH_4 emission from landfills, besides the gas production intensity, are barometric pressure changes and microbial methane oxidation.

2.3 Microbial methane oxidation – general principles

Microbial CH_4 consumption by methanotrophic bacteria is an important biogeochemical sink for the removal of methane from the biosphere thereby reducing the amount of methane released to the atmosphere (Topp and Hanson, 1991). Methanotrophic bacteria, therefore, play an important role in the global methane-cycle (Hanson and Hanson, 1996) as they control the balance between CH_4 sources and sinks in both natural and anthropogenic settings such as landfills (Hanson and Hanson, 1996). Microbial CH_4 oxidation generally occurs at an interface of aerobic and anaerobic zones, where combined conditions of temperature, moisture content, CH_4 and O_2 concentrations are suitable (Zeiss, 2006).

The methane oxidising microorganisms, or methanotrophs, are a subset of a highly diverse group of bacteria known as methylotrophs which are able to metabolise a variety of different one-carbon compounds including methane, methanol, methylamines and halomethanes (Hanson and Hanson, 1996). Most of these bacteria are obligate methanotrophs and strict aerobes (Hanson and Hanson, 1996). However, a genus of facultatively methanotrophic bacteria (Methylocella) has been recently identified which also can use two-carbon compounds (Theisen and Murrell, 2005). Beside methanotrophs there are also some ammonia oxidising nitrifying bacteria (nitrifiers) and yeasts that are able to oxidise CH_4 (Bedard and Knowles, 1989; Wolf and Hanson, 1979).

In general, there are two known pathways used by methanotrophs for the conversion of methane to energy and biomass: the ribulose monophosphate path (RuMP) and the serine path (Hanson and Hanson, 1996). The following Figure 2.6 depicts the complete pathway for the oxidation of CH_4 to CO_2 and the assimilation of formaldehyde (CH_2O) of both pathways. The first step of methane oxidation by methanotrophic bacteria is the conversion of dissolved methane to methanol (CH_3OH) which is catalysed by a key enzyme known as methane monooxygenase (MMO). The MMO exists in two forms, a soluble, cytoplasmic form (sMMO) and a membrane-bound or particulate form (pMMO). While all known methanotrophs (except Methylocella sp.) are capable of forming pMMO, only certain methanotrophs have the ability to produce sMMO linked to low copper concentration (Hanson and Hanson, 1996). Formaldehyde produced from the oxidation of methane and methanol is then used either for biosynthesis or for catabolism (Lengeler et al., 1999). During biosynthesis, formaldehyde enters the serine or the RuMP pathway. In the

serine pathway 2 mol of formaldehyde and 1 mol of CO_2 are utilised for the synthesis of multicarbon compounds whereas in the RuMP cycle 3 mol of formaldehyde are assimilated. In the catabolic reactions, formaldehyde is further oxidised via formate into CO_2, which constitutes the last step of the CH_4 metabolism.

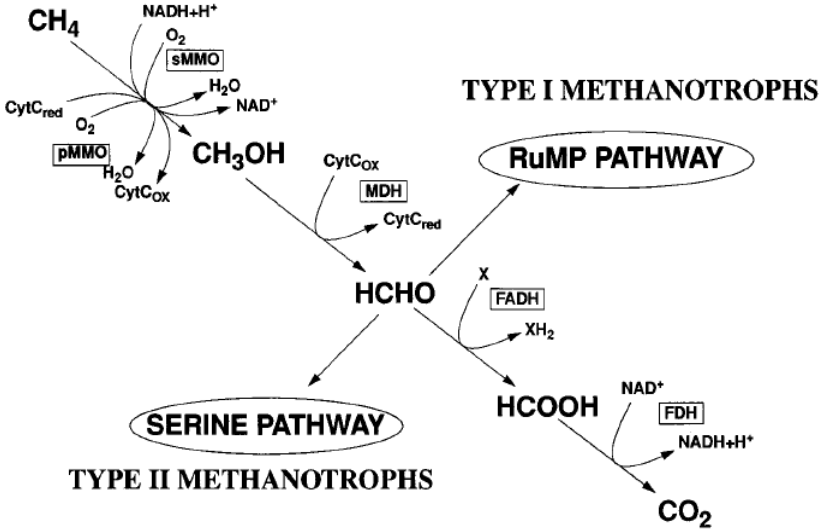


Figure 2.6:. Pathways for the oxidation of methane and assimilation of formaldehyde. (Abbreviations: CytC, cytochrome *c*; FADH, formaldehyde dehydrogenase; FDH, formate dehydrogenase), (Hanson and Hanson, 1996)

Taxonomically, methanotrophs can be classified into two main distinct groups: type I and type II, belonging to the alpha and gamma subdivision of the Proteobacteria (Hanson and Hanson, 1996). Methanotrophs of the Proteobacteria differ in their carbon assimilation pathway, morphology, biochemistry, affinity for CH_4, membrane type and composition (Hanson and Hanson, 1996; Börjesson et al., 2004; Vishwakarma et al., 2009). Type I methanotrophs are represented by ten genera including Methylomonas, Methylobacter, Methylomicrobium, Methylosarcina, Methylosphaera, Methylosoma, Methylohalobium, Methylococcus, Methylocaldum and Methylothermus (Vishwakarma et al., 2009). Type II includes the genera Methylosinus, Methylocystis, Methylocella and Methylocapsa. While type I methanotrophs assimilate carbon via the RuMP cycle, type II methanotrophs use the serine pathway.

According to methane oxidation kinetics and affinities, two methanotrophic populations have been observed (Dunfield et al., 1999). The first group, commonly referred to as the upland soil cluster (USC) alpha, are considered as 'high-affinity' bacteria because of their ability to oxidise CH_4 at concentrations close to that of the atmosphere (< 12 ppm) (Le Mer and Roger, 2001; Huber-Humer et al., 2008b). The second group, known as 'low-affinity' methanotrophs, is only active at CH_4 concentrations higher than 40 ppm (Le Mer and Roger, 2001).

Hanson and Hanson (1996) suggest a classificiation of the methanotrophs according to the CH_4 and O_2 concentrations. Type I methanotrophs, which are mainly found in the upper parts of the soil, prefer to grow at low CH_4 levels and high O_2 concentrations (Hanson and Hanson, 1996). By contrast, the growth of some type II methanotrophs is favoured in deeper zones when CH_4 concentrations are high together with low levels of nitrogen and oxygen (Hanson and Hanson, 1996). These methanotrophs are more typical of those encountered in landfill cover soils (Kightley

et al., 1995). However, type II methanotrophs have also been detected in an organic soil oxidising methane at atmospheric level (Dunfield et al., 1999)

Due to the multiplicity of methane sources, such as wetlands, microbial CH_4 uptake is ubiquitous (Hanson and Hanson, 1996). Biological CH_4 oxidation has been reported to occur in variety of ecosystems, as in temperate (Bradford et al., 2001), tropical (Butterbach-Bahl et al., 2004), desert (Angel and Conrad, 2009; Striegl et al., 1992) and arctic soils (Liebner et al., 2009; Whalen and Reeburgh, 1996) as well as in aquatic environments (Le Mer and Roger, 2001). However, large methanotrophic populations and the highest rates of CH_4 oxidation have been observed in landfill cover soils (Scheutz et al., 2003; De Visscher et al., 1999; Kightley et al., 1995; Whalen et al., 1990). One of the first journal publications to report methane oxidation activity in landfill cover soils was made by Whalen et al. (1990) using laboratory batch tests. The authors found the highest rates of CH_4 consumption observed in any soils before that time (45 g/m²d). Recent studies have observed the capacities for CH_4 oxidation of landfill cover soil in column experiments, with rates as high as >200 g CH_4/m²d (Scheutz et al., 2003; De Visscher et al., 1999). Even higher steady state CH_4 removal rates (> 400 g CH_4/m²d) have been found in simulated landfill covers rich in organic matter such as mature compost materials (Haubrichs and Widmann, 2006; Humer and Lechner, 2001b). Numerous studies, therefore, have supported the concept of microbial CH_4 removal by oxidation as an effective tool for greenhouse gas reduction in landfill emission (Stern et al., 2007; Zeiss, 2006; Hilger and Humer, 2003; Humer and Lechner, 1999a). Reported values for whole landfill CH_4 oxidation rates had a wide range which can be attributed to physical heterogeneities in the cover, different CH_4 levels, and seasonal climate change (Hilger and Humer, 2003; Börjesson et al., 2001; Chanton and Liptay, 2000). Rates of CH_4 removal in landfill covers can range from negligible to 100 % (Börjesson et al., 2001, 2007; Stern et al., 2007; Abichou et al., 2006; Huber-Humer, 2004). Under certain conditions, landfill covers even take up rather than emit methane, resulting in the landfill cover functioning as a sink for atmospheric methane (Barlaz et al., 2004; Bogner et al., 1995 and 1997a, b; Börjesson and Svensson, 1997; Boeckx et al., 1996). In general, steady state CH_4 consumption rates for landfill cover soils are between 100 to 150 g CH_4/m²d, indicating 30 to 60 % CH_4 removal, whereas maximum rates of 200 to 250 g CH_4/m²d can contribute to 80 – 100 % CH_4 oxidation (Scheutz et al., 2009).

Methanotrophs can also co-metabolise a variety of non-methane compounds including some halogenated hydrocarbons and aromatics (Bogner et al., 2010; Scheutz and Kjeldsen, 2004) due to the broad substrate specificity of the MMO enzyme (Bogner et al., 2010).

There have also been instances of methanotrophic activity contributing to production of nitrous oxide (N_2O) (Mandernack et al., 2000), itself a greenhouse gas with a global warming potential of 289 over a 100-year period (IPCC, 2007). Additions of nitrogenous fertiliser or nitrogen-rich environments (i.e. covers made of organic soil substrates or compost) and alternating aerobic and anaerobic zones have resulted in increased N_2O emission rates (Lee et al., 2009; Zhang et al., 2009; Huber-Humer, 2008b; Mandernack et al., 2000; Boeckx and Van Cleemput, 1996). In contrast, no or little N_2O emissions were measured in organic landfill cover materials consisting of a mixture of biowaste compost and gravel (Watzinger et al., 2005) and mechanically–biologically treated municipal solid waste (Einola et al., 2008). However, it is still unclear what the relative contributions of methanotrophs are to N_2O formation in landfill covers or how to minimise them. Besides nitrification by methanotrophs, also

nitrification and denitrification by ammonia-oxidising bacteria or archea play a part in the N_2O production processes (Lee et al., 2009).

2.4 Controlling factors on methane oxidation

Several laboratory and field studies have shown a wide range of CH_4 removal rates (Stern et al., 2007; Zeiss, 2006; Huber-Humer, 2004; Chanton and Liptay, 2000; Whalen et al., 1990). This wide variation can be attributed to numerous factors including, among others, soil temperature and moisture, methane and oxygen supply, nutrient level, organic matter content and soil physical properties such as permeability and particle size which control microbial CH_4 uptake (Boeckx and Van Cleemput, 1996; Boeckx et al., 1996; Kightley et al., 1995; Castro et al., 1994; Whalen et al., 1990). Although methanotrophic microorganisms are fairly adaptable and resilient to changing conditions, landfill covers have to ensure optimum ambient conditions in order to achieve sufficient CH_4 oxidation rates. Thus, management of these physical and environmental factors in engineered biocover systems can contribute to enhanced methanotrophic performance, resulting in reduced GHG emissions. In this chapter the controlling factors on CH_4 oxidation are presented in more detail.

2.4.1 Moisture content

The moisture content in substrates appears to be a crucial factor controlling CH_4 oxidation because it affects both the movement of gases through the cover soil and microbial activity. Soil moisture influences both gas permeability and diffusivity of soil as the available pore space for gaseous transport and diffusion are affected at different moisture contents (Tecle et al., 2009). Higher moisture levels may decrease the CH_4 oxidation capacity of the landfill cover due to limiting of gaseous transport processes of O_2 and CH_4. As a consequence, gas-phase diffusion is shifted to aqueous-phase diffusion, which is 10^4 fold less rapid, resulting in limited substrate delivery to the microbes (Boeckx et al., 1996). Nesbit and Breitenbeck (1992) showed that the average rate of CH_4 oxidation was reduced by 56 % at water saturation. Sitaula et al. (1995) reported a significant decrease in CH_4 uptake rate by 35 % to 50 % when increasing the soil moisture content from 32 % to 42 % in silty sand forest soil. In general, O_2 can penetrate much deeper into unsaturated substrate layers compared to saturated ones. Hence, unsaturated zones serve as a barrier for residual CH_4 emissions. On the other hand, low moisture levels can also decrease CH_4 uptake due to microbial desiccation resulting in lower methanotrophic activities. CH_4 consumption is substantially reduced when soil moisture content decreases below 5 % (Scheutz and Kjeldsen, 2004; Stein and Hettiaratchi, 2001; Whalen et al., 1990). CH_4 oxidation will be at is maximum when there is both maximum gas phase molecular diffusion and sufficient soil moisture content (Scheutz et al., 2009) (see Figure 2.7).

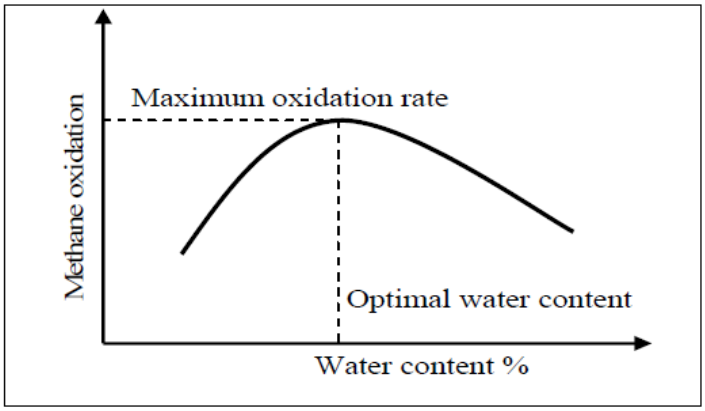


Figure 2.7: Response of CH_4 oxidation to soil water content (Yuan, 2006)

The optimum moisture content varies for different soil types and is further dependent on temperature (see section 2.4.5) and other environmental factors. Experimental studies showed that optimal moisture content in mineral landfill covers ranged between 10 % and 20 % w/w (Boeckx and Van Cleemput, 1996; Boeckx et al., 1996; Whalen et al., 1990) whereas higher optimum water contents have been observed in organic landfill cover materials (Albanna et al., 2007; Mor et al., 2006; Humer-Huber, 2004; Scheutz and Kjeldsen, 2004; Humer and Lechner, 1999b). A high CH_4 oxidation was reported at a moisture content of 45 % w/w in composted municipal solid waste (24 % organic matter) used as landfill cover (Humer and Lechner, 1999b). Mor et al. (2006) found optimum moisture content of garden waste compost (31 % and 52 % organic matter) ranging from 45 to 110 % on a dry weight basis. However, under wet conditions, the high water level may block gas transport causing a reduction of CH_4 consumption or even contributing to CH_4 production in extreme cases (Scheutz et al., 2009). Hence, a balanced grain size distribution is a prerequisite for sufficient gas permeability at high moisture contents (Scheutz et al., 2009). Humer-Huber (2004) measured oxidation rates up to 100 % even at water saturation (57 % w/w) in mature sewage sludge compost mixed with roughly shredded wood chips (1:1 w/w). The author concluded that high performance could be maintained at high moisture contents because of the adequate porosity and high water-holding capacity (WHC) of the waste material. As a consequence, compost covers control water infiltration into the landfill which enhances waste degradation and establishes optimal conditions for the methanotrophic community (Scheutz et al., 2009; Stern et al., 2007; Huber-Humer, 2004).

In numerous studies, soil moisture has been suggested as a major controlling factor of CH_4 oxidation (Jugnia et al., 2008; Albanna et al., 2007; Gebert et al., 2003; Boeckx et al., 1996; Castro et al., 1994). Boeckx et al. (1996) stated that the moisture content largely controls methanotrophic activity. A multiple linear regression analysis indicated that moisture content has more influence than temperature explaining much of the variation of CH_4 emission (Boeckx et al., 1996). The authors concluded that the optimal moisture content is situated at about 50 % of the WHC supporting both acitivity of methanotrophic bacteria and gas permeability. In forest soils, the soil moisture content was a key variable influencing CH_4 oxidation rates, where 78 % of the variability in CH_4 oxidation rates has been correlated to soil moisture variability (Castro et al., 1994).

Investigations by Spokas and Bogner (2010) indicated that soil moisture potential, which expresses soil moisture in terms of the physical force with which water is held

in soil, provides a more robust parameter than gravimetric or volumetric moisture for examining the dependency of landfill CH_4 oxidation rates on soil moisture. The authors found that optimal soil moisture potential in landfill cover soils was close to field capacity (50 kPa) whereas the minimum threshold soil moisture potential was at approximately 1500 kPa (soil wilting point). Using soil moisture potential, differences in soil texture and structure could be normalised and, thus, a direct comparability to other soils would be achieved.

2.4.2 Organic content and nutrient supply

The organic matter in substrates strongly influences the CH_4 oxidation capacity. On the one hand, organic matter improves the substrate properties such as soil structure and aggregation, WHC and aeration and it further serves as a carrier for microorganisms (Sparks, 1995). On the other hand organic matter provides nutrient supply which is a prerequisite for cellular metabolism of methanotrophic bacteria and other bacteria in a substrate.

In general, CH_4 oxidation rate increases with increasing content of organic matter in a substrate. Composted materials with high organic content (of up to 35 % w/w) show 10 to 100 fold higher CH_4 oxidation potentials compared to soils with organic contents of 1 to 10 % (He et al., 2008; Zeiss, 2006). Huber-Humer (2004) reported 100 % steady-state CH_4 oxidation in fully matured sewage sludge compost used as landfill cover exceeding performance of conventional soil covers (40 – 45 % uptake). Compost maturity (7-day oxygen demand < 8 mg O_2/g DM) proved to be important to ensure minimum competition for O_2 from other microorganisms as it provides biochemically stable organic matter (Scheutz et al., 2009; Kettunen et al., 2006; Humer and Lechner, 1999). Compared to conventional soils, composts have considerably higher content of long-term available nutrients (Nitrogen: 0.85 – 1.25 % DM; Phosphorus: 0.43 – 3.06 % DM) (Huber-Humer, 2004). Therefore, several studies have suggested that high organic materials such as composted sewage sludge, biowaste, yard waste, municipal solid waste or mechanically-biologically pretreated municipal solid waste and biowaste may be used as a support medium for CH_4 oxidation in landfill covers to mitigate greenhouse gas emissions from landfills (Bohn and Jager, 2009; Einola et al., 2008; Jugnia et al., 2008; Stern et al., 2007; Mor et al., 2006; Humer-Huber, 2004; Cossu et al., 2003).

The addition of nutrients such as sewage sludge, phosphate, lime and commercial fertiliser has also been found to enhance CH_4 oxidation rates in landfill cover soils (Albanna et al., 2007; Hilger et al., 2000; Börjesson et al., 1998; Boeckx and Van Cleemput, 1996; Kightley et al., 1995). Kightley et al. (1995) documented that amendment of the coarse sand with sewage sludge (2.5 g/kg of soil) raised CH_4 oxidation rate by 26% whereas mineral fertiliser showed no increase or even a decrease in the CH_4 uptake rate. According to investigations by Albanna et al. (2007), commercial fertiliser (0.3 g N_2:0.13 g P:0.249 g K for each kg soil) could be a source of nutrients for methanotrophs when the moisture in the landfill cover is adequate. Adding fertiliser to a landfill cover soil (1.5 g/kg of soil) that contained 30 % moisture content increased the CH_4 oxidation rate from 38 % to 81 % while adding nutrient to the soil with low level of moisture content (15 %) negatively affected the methanotrophic performance. Based on a statistical design model higher maximum CH_4 oxidation rates could be achieved when nutrients are added to the cover soil with 45 % moisture content, as can be seen in Figure 2.8. These results again confirmed the importance of moisture content for CH_4 oxidation in landfill cover soil.

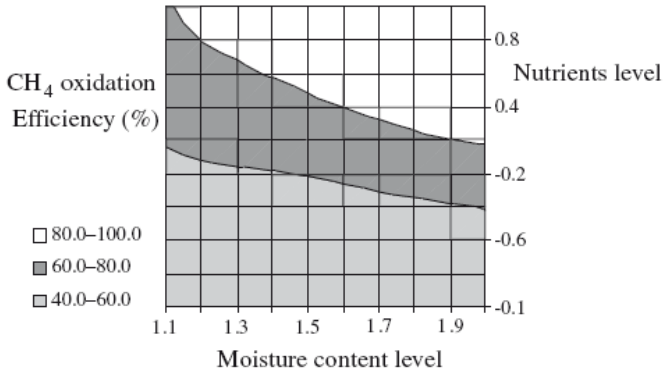


Figure 2.8: Plot of the modelled maximum CH_4 oxidation efficiency (%) for moisture content levels between 30 % and 45 % and nutrient additions (-1 code represents no nutrients added, +1 code, added nutrients of 1.5 g fertiliser/kg of soil dry weight) (Albanna et al., 2007)

Also the C:N ratio of the nutrient amendment or the used substrate seems to considerably affect CH_4 oxidation in landfill cover soil. Addition of crop residues (potato and sugar beet leaves) with low C:N ratios (13.5 and 11.3) produced high and inhibitory concentrations of ammonium (NH_4) and nitrite (NO_2^-) due to nitrogen mineralisation resulting in a strong inhibition of the CH_4 oxidation (Boeckx and Van Cleemput, 1996) (see also section 2.4.7). In contrast, crop residues (wheat and maize straw) with a high C:N ratio (97.2 and 24.8) stimulated N-immobilisation and, thus, minor amounts of ammonium and nitrite were produced, which caused no inhibition.

2.4.3 pH-value

Methanotrophs are mainly neutrophilic and their optimum growth and activity rate lies between pH 5.5 and 8.5 (Figueroa, 1993). Some methanotrophs, however, tolerate high pH values up to 9 and oxidation acitivity seems to continue down to pH 2 (Islam et al., 2007; Huber-Humer, 2004). Two genera of moderately acidophilic methanotrophs have been characterised and the optimal value for their development is between pH 5.0 – 5.5 (Islam et al., 2007). Hilger et al. (2000) observed enhanced CH_4 oxidation when adding lime to landfill covers which raised pH from 6.3 to 7.4. According to Scheutz and Kjeldsen (2004), it is questionable whether significant pH gradients will develop in “natural” landfill covers as they often have high buffer capacities. Hence, the CH_4 oxidation capacity will be less affected by pH.

2.4.4 Particle size and porosity

Substrate physical properties such as soil texture and porosity can influence the supply of essential elements (O_2, water, nutrients) to methanotrophic microorganisms and thereby control microbial CH_4 oxidation. In general, a topsoil of coarse texture shows a higher CH_4 removal rate than a fine one due to its better aeration property (Mor et al., 2006; Watzinger et al., 2005, Boeckx et al., 1997). For instance, higher oxidation rates have been found in coarse sand (61 %) than in fine sand or clay (40 – 41 %) (Kightley et al., 1995). In addition, coarse substrates have shown a better insulation from atmospheric temperature fluctuations than fine substrate (Humer-Huber and Lechner, 2009; Humer-Huber, 2004). Huber-Humer (2004) observed that during winter months the temperature produced by microbial activity was retained much better in coarse sewage sludge compost mixed with big wood chips than the fine-sieved MSW-compost. Moreover, materials with a high air-filled pore space such as composts (on the order of 50 % compared to 20 – 30 % for soils) will allow more O_2 to penetrate into the cover material creating a more extended aerobic zone (Mor

et al., 2006). Humer and Lechner (1999a) and Börjesson et al. (2004) observed that soils with high porosity and large particle size distribution can retain CH_4 and O_2 for a longer period of time, leading to higher oxidation rates. However, Pawlowska et al. (2003) measured slightly higher CH_4 uptake rates in coarse sand (0.5 – 1 mm) compared to coarse gravel (2 – 4 mm). The lower specific surface area of the gravel fraction might have led to lower potential contact between CH_4 molecules and microogranisms. Gebert and Gröngröft (2009) suggested a gas-filled porosity of at least 17.5 vol. % to ensure a rate of diffusive oxygen ingress that enables complete oxidation of a CH_4 flux of 0.5 l CH_4/m²h.

Since substrate physical properties constitute the main design variables of engineered covers, they should be optimised to allow almost the full cover depth to support CH_4 oxidisers.

2.4.5 Soil temperature

The activity of methanotrophic microorganisms is dependent on the soil temperature. Typically, CH_4 uptake rates increase with rising temperatures while at low temperatures inhibition effects have been observed (De Visscher et al., 2001; Visvanathan et al., 1999; Börjesson and Svensson, 1997; Whalen et al., 1990). As of most methanotrophs are mesophilic, the optimum temperature for CH_4 oxidation is usually in the range of 20 – 38 °C (see Figure 2.9) (Gebert et al., 2003; Visvanathan et al., 1999; Börjesson and Svensson, 1997; Boeckx et al., 1996; Figueroa, 1993; Whalen et al., 1990).

Huber-Humer (2004) reported that a complete CH_4 oxidation was achieved between 8 °C and 30 °C in mature sewage sludge. At a lower temperature of 4 °C CH_4 oxidation decreased to about 70 – 80 %. Experiments by Zeiss (2006) showed even a halt in biological CH_4 oxidation in yard waste compost during the winter period. Similar observations have been made by Börjesson et al. (2001) in a Swedish landfill cover soil. Boeckx and Van Cleemput (2000) suggested that CH_4 oxidation does not occur at temperatures below 0 °C. The susceptibility of methanotrophs to extreme temperatures may be associated with a small cover depth (Börjesson et al., 2001; Chanton and Liptay, 2000). In contrast, CH_4 oxidation continued in biofilters during winter (Streese and Stegmann, 2003). Methanotrophic bacteria in the cover soil of a boreal landfill were also able to oxidise CH_4 at increasing rates even at temperature below 10 °C and close to freezing point (Einola et al., 2007). The increase in CH_4 uptake rates was due to growth or activation of psychrophilic (cold tolerant) methane oxidisers. According to Hanson and Hanson (1996) and Börjesson et al. (2004) psychrophilic methanotrophs are mainly of type I which prefer low CH_4 and high O_2 concentrations. Results from Börjesson et al. (2004) and Gebert et al. (2003) showed that enriching methanotrophic mixed cultures at low temperatures (3 – 10 °C) lead to a shift towards more methanotrophs of type I indicating that temperature response in support media such as soil or compost might be time-dependent.

Spokas and Bogner (2010) demonstrated that CH_4 uptake was amplified by increasing temperatures up to 30 °C and then rates declined and dropped to zero by 55 °C, as can be seen in Figure 2.9. However, there have been also some instances, where thermophilic species have been detected with optimal growth temperature at 55 – 59 °C (Islam et al., 2007).

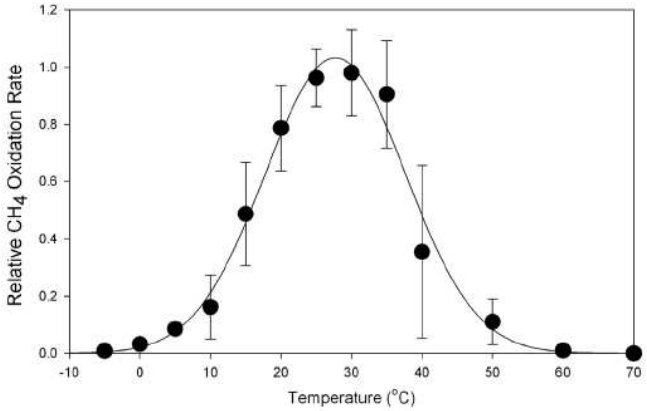


Figure 2.9: Effects of temperature on relative rates of CH_4 oxidation with associated standard deviations and fit to a 3-parameter Gaussian curve (n = 3456) (Spokas and Bogner, 2010)

In order to quantify the temperature influence on microbial CH_4 oxidation, the so called Q_{10} value might be used which gives the number of times the oxidation rate increases when the temperature is increased 10 °C at temperatures below the optimum temperature. The Q_{10} value between the temperatures of 10 °C and 30 °C was found to be in the range of 1.7 – 4.1 (Börjesson et al., 2004; De Visscher et al., 2001; Boeckx et al., 1996; Whalen et al., 1990). Also Q_{10} values as high as 4.10 – 7.26 have been observed in landfill cover soils (Christophersen et al., 2000). In general, a high Q_{10} value indicates that temperature controls the CH_4 oxidation more than other factors. Börjesson and Svensson (1997) stated that soil temperature is the key factor of CH_4 oxidation, explaining 85 % of the variation in measured CH_4 oxidation. According to De Visscher et al. (2001), the influence of temperature on CH_4 oxidation is more pronounced at high CH_4 concentrations than at low concentrations. In contrast, experiments by Boeckx et al. (1996) indicated that temperatures had only slight effects on CH_4 oxidation. As can be seen in Table 2.1, the optimum incubation temperature (30 – 20 °C) declined with increasing moisture contents. The authors concluded that equilibrium exists between CH_4 diffusion rate at certain moisture content and the uptake rate by the methanothrophic bacteria. Hence, at high moisture contents substrate supply of CH_4 and O_2 becomes the limiting factor and not the bacterial activity due to temperature.

Table 2.1: Correlation between moisture content and optimum temperature (Boeckx et al., 1996)

Moisture content (%)	Optimum temperature (°C)
5	30.0
10	27.1
15	24.9
20	23.5
25	21.2
30	20.1

Not surprisingly, air temperature has direct effects on soil temperature, especially in the upper horizons, leading to seasonal temperature dependence of methanotrophic activity in the landfill cover (Mor et al., 2006; Hilger and Humer, 2003). However, methanotrophic bacteria themselves can influence ambient temperature as CH_4 oxidation is an exothermic process releasing 210.8 kCal per mol of converted CH_4. Berger et al. (2005) reported that methanotrophic activity in the landfill cover (mixture of compost and sand, 0.3 m, over a layer of loamy sand, 0.9 m) was able to produce temperatures which were 5 – 8 °C higher than the ambient temperature. Field

experiments by Humer and Lechner (2001) showed that temperatures > 15 °C were maintained in a biocover made of compost materials at a depth of 0.5 m downwards even during winter months. However, heat generation also occurs inside the waste due to anaerobic exothermic reactions leading a temperature range from 30 – 60 °C (Williams, 2005).

2.4.6 Air pressure

As already mentioned, a rapid drop in atmospheric pressure could pull CH_4 through a CH_4 oxidation zone faster than methanotrophs can consume it resulting in occasional peaks of CH_4 emissions. On the other hand, a rapid atmospheric pressure rise enables O_2 to penetrate into deeper regions of relatively inactive sites where it could oxidise CH_4 (Latham and Young, 1993).

In addition, aerobic methane oxidation itself shows a “gas-pumping-effect”. The resulting volume reduction during aerobic CH_4 oxidation (according to equation 2.1) creates a low pressure enhancing the ability of air to diffuse deeper into the oxidation layer (Huber-Humer 2004; De Visscher, 2001; Kjeldsen, 1996). While the oxygen from the air-flow is immediately consumed by the methanotrophs, nitrogen accumulates. Moreover, the upward gas flow slows down as it reaches the top (DeVisscher, 2001). This “gas-pumping-effect” may increase oxygen transport into the substrate, thereby enhancing methane oxidation. Experiments by Huber-Humer (2004) confirmed this phenomenon using Ar as a tracer in an actively methane oxidising column and a sterile column without methanotrophic activity. The tracer-component clearly accumulated in the “active column” compared to the “sterile” one, indicating that the “pumping effect” triggered a higher flow of air into the cover.

2.4.7 Inorganic Nitrogen

As already stated nutrient supply, especially nitrogen, is a prerequisite to build up methanotrophic biomass. Methanotrophs take up nitrogen in the form of ammonium or nitrate. In addition, some strains (type II) even can fix atmospheric nitrogen. According to Anthony (1982; cited in Scheutz et al., 2009) 0.25 mole of nitrogen is required for every mole of assimilated carbon. The effect of inorganic nitrogen (ammonium, nitrate) on CH_4 oxidation is quite complex and can be both stimulatory and inhibitive depending on several factors including the species and concentration of nitrogen, soil pH and type of methanotrophs present (Scheutz et al., 2009; Bodelier and Laanbroek, 2004). Several studies have shown that elevated NH_4^+ concentrations in soil may inhibit CH_4 oxidation rates due to possible substrate competition between CH_4 and NH_4^+ at the level of MMO enzymes (Reay and Nedwell, 2004; Boeckx and Van Cleemput, 1996; King and Schnell, 1994). For instance, Boeckx and Van Cleemput (1996) observed inhibition of the CH_4 oxidation rate in landfill cover soil upon NH_4^+ addition (25 mg N/kg). Scheutz and Kjeldsen (2004) found unaltered CH_4 oxidation rates with the addition of NH_4^+ up to 14 mg N/kg (added as NH_4Cl) to landfill cover soil, whereas soil showed decreasing CH_4 oxidation rate upon increasing NH_4^+ application rates (see Figure 2.10).

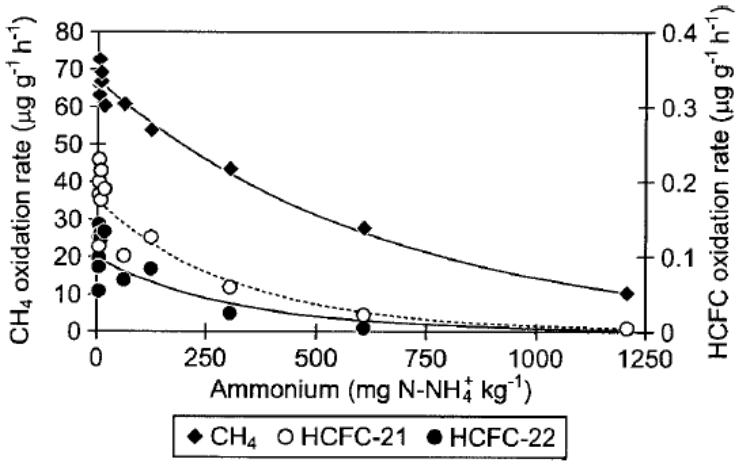


Figure 2.10: Influence of ammonium on the oxidation rate of methane and hydrochlorofluorocarbons (HCFCs) (Scheutz and Kjeldsen, 2004)

In contrast, stimulation of CH_4 oxidation following NH_4^+-based fertilisation has been reported (Lee et al., 2009; Hilger et al., 2000). As can be seen in Figure 2.11, the CH_4 uptake rate in landfill cover soils amended with 100 mg-N NH_4/kg soil (added as NH_4Cl) was increased by about 60 % (Lee et al., 2009). However, the ammonium application also led to N_2O production, which increased over 16-fold compared to the baseline conditions.

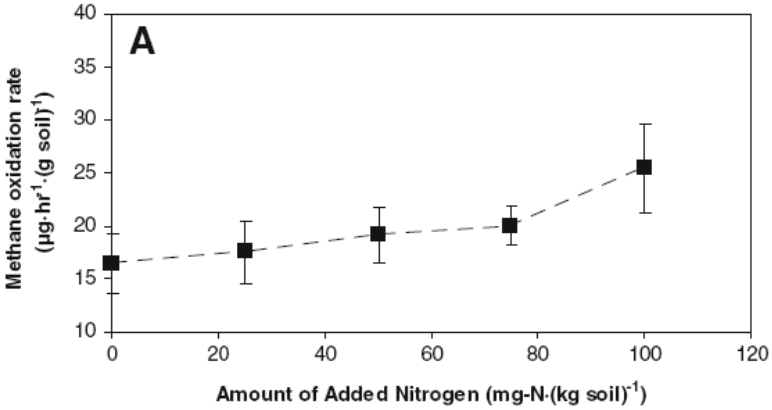


Figure 2.11: Effect of ammonium on the rate of methane consumption (Lee et al., 2009)

De Visscher et al. (1999) concluded that NH_4^+ inhibition becomes more pronounced at low atmospheric CH_4 concentrations in soil. De Visscher and Van Cleemput (2003) reported that at steady state NH_4^+ inhibited CH_4 oxidation at low CH_4 concentration while at high concentrations CH_4 oxidation was stimulated. The authors further observed that after an initial peak of methanotrophic activity, growth rates, probably of type I bacteria, declined considerably as a response to N shortage and later on methanotrophic activity, probably of type II methanotrophs, increased in spite of the low inorganic N content, due to their independence from inorganic N.

Boeckx et al. (1996) and Boeckx and Van Cleemput (1996) suggested that the inhibition by NH_4^+ seems to be considerably related to its nitrification rate or N-turnover rate rather than to its actual concentration. In addition, the authors found that ammonium has a greater effect on CH_4 oxidation than the soil moisture content.

Experiments by Wilshusen et al. (2004) indicated that high NH_4^+ concentrations within fresh compost materials were negatively correlated with CH_4 oxidation

suggesting that highest methane oxidation efficiency could be achieved with a low NH_4^+ content in compost. Similar observations have been made with fresh sewage sludge showing inefficient CH_4 oxidation rates, most probably because nitrification suppressed CH_4 oxidation (Huber-Humer, 2004).

Nitrate showed inhibiton of CH_4 oxidation only at very high concentrations through osmotic effects which usually do not occur in landfill environments (Scheutz et al., 2009; Bodelier and Laanbroek, 2004).

2.4.8 Vegetation

Previous studies have indicated that vegetation may have an important effect on the growth and activity of methanotrophs as well as on the soil physical properties (Xiaoli et al., 2010, Bohn and Jager, 2009, Wang et al., 2008; Hilger and Humer, 2003). The influence on soil physical properties is still not well known. Generally, vegetation stabilises and protects the top cover from erosion by spreading roots and controls moisture infiltration due to its evapotranspiration capacity (Bohn and Jager, 2009). Since roots provide a channel for oxygen penetration and build up different pore sizes, gas diffusion of O_2 and CH_4 to the methanotrophs is improved and, therefore, CH_4 oxidation is likely to be enhanced. In addition, the root system of vegetation releases exudates into the surrounding soil and induces a more suitable environment for methanotrophs. Stralis-Pavese et al. (2004) observed that all tested plant covers on sewage sludge compost considerably enhanced CH_4 oxidation potential compared to bare substrate covers. These results were related to a higher diversity of methanotrophs in the vegetated landfill covers than in the substrate alone. Similar observations have been made by Hilger et al. (2000) and Bohn and Jager (2009). There is evidence that the efficiency of CH_4 oxidation also depends on plant species and physiological status of the vegetation (Xiaoli et al., 2010; Reay et al., 2005; Stralis-Pavese et al., 2004). On the other hand vegetation is closely related to the depth and moisture retentiveness of the cover material (Xiaoli et al., 2010). Experiments by Huber-Humer and Lechner (2003) and Huber-Humer (2004) indicated that the evapotranspirative capacity of vegetation combined with the high water retention capacity of compost material used in a biocover (1 m high) will be sufficient to prevent high levels of infiltration and leachate production. The tests revealed a water infiltration rate of < 10 % annual precipitation (700 mm) within the first years of operation.

Various studies have indicated that the plant growth in landfill covers is severely inhibited by the presence of high landfill gas concentrations in the root zone (Xiaoli et al., 2010; Chan et al., 1997). High CO_2 concentrations are directly toxic to the roots even at sufficient O_2 availability (Nagendran et al., 2006). In addition, the root penetration depth might be shortened due to a possible suppression of O_2 as a result of high O_2 demand by methanotrophs (Gerzabek and Reichenauer, 2006). Consequently, the plant growth will be inhibited resulting in reduced transpiration. However, some tree, shrub and grass species seem to tolerate high landfill gas concentrations (Wang et al., 2008; Chan et al., 1997). Wang et al. (2008) studied the effect of a LFG tolerant plant (Chenopodium album L.) on CH_4 oxidation in landfill cover soil. Methanotrophs population and CH_4 oxidation activity in soils exposed to landfill gas were significantly higher than in soils without landfill gas.

Some plant species (e.g. Typhaceae) are even able to obtain aerenchyms (air-filled spaces in stem and roots), which facilitates the movement of O_2 to the roots (Bosse and Frenzel, 1997). In addition to their own consumption, roots release O_2 to the

surrounding anoxic soil. In this way, plants may provide O_2 for methanotrophic bacteria, thus reducing CH_4 emissions (Zak, 2008; Mainiero and Kazda, 2004; Bosse and Frenzel, 1997). However, the complex interactions between these plants and methane-oxidising bacteria should be subjected to further investigation due their revelance for CH_4 oxidation.

In contrast to the above mentioned advantages, plant-mediated transport mechanisms may also affect localised fluxes (Chanton, 2005). There is also the potential for increased gas emissions through the creation of preferential channels by plant roots (Scheutz et al., 2009). In addition, vegetation could also compete with CH_4 oxidisers for nutrients and water resulting in decreased CH_4 oxidation (Hilger and Humer, 2003; De Visscher et al., 1999).

2.4.9 Methane and oxygen supply

Methanotrophic activity depends on the presence of sufficient concentrations of both CH_4 and O_2 at the same time. Hence, their habitat is confined to a narrow layer limited by the downward diffusion of atmospheric O_2 and the upward diffusion of CH_4 from the underlying waste (Scheutz et al., 2009). In addition advective forces (see section 2.2) and the “gas pumping effect” (see section 2.4.6) may be important as well.

Previous studies have indicated that CH_4 removal rates are sensitive to CH_4 concentrations. Higher CH_4 supply increased CH_4 oxidation in investigations of Visvanathan et al. (1999). However, the effect was not proportional to the supply rates and at certain rates a limiting CH_4 oxidation capacity of the soil was observed. Jones and Nedwell (1993) found that counts of methanotrophs increased with rising CH_4 concentrations suggesting that the methanotrophic community had adapted to the presence of elevated CH_4 concentration. In contrast, high CH_4 flow rate from underlying waste can hinder the diffusion of O_2 in the landfill cover resulting in reduced CH_4 oxidation (Abichou et al., 2004). In general, microbial CH_4 oxidation occurs in a zone, where optimal conditions for methanotrophs growth, O_2:CH_4 ratio, retention time and suitable environmental conditions exist. Kightley et al. (1995) and Jones and Nedwell (1993) reported that CH_4 oxidation potentials were greatest where the vertical profiles of O_2 and CH_4 overlapped.

Huber-Humer (2004) investigated the CH_4 oxidation capacity in mature sewage sludge compost exposed to a range of CH_4 supplies. The results showed an upward displacement of the CH_4 oxidation zone in the soil column upon increasing CH_4 loads. At low CH_4 supply (25 l/m²d) the main CH_4 oxidation zone was situated at a depth between 40 and 60 cm whereas at high CH_4 loads (355 l/m²d) all of the O_2 was already depleted between 0 – 15 cm.

Several researches reported different optimum CH_4 oxidation zones at different depths depending on the moisture content, temperature, O_2 availability and CH_4 supply. For instance, Kightley et al. (1995) found that the maximum zone of CH_4 oxidation is situated between 20 and 30 cm in coarse sand, while Visvanathan et al. (1999) located it between 15 and 40 cm in different landfill cover soils. Generally, the methanotrophic active zone is situated in the upper part of the soil profile (30 – 40 cm), with maximum CH_4 oxidation between 15 – 20 cm below the surface (Scheutz et al., 2004, 2009; He et al., 2008). Figure 2.12 depicts a gas concentration profile measured at a Danish landfill compared with maximum CH_4 oxidation rates obtained in batch incubation experiments, indicating that the optimal CH_4 oxidation zone is located in the upper part of the soil profile.

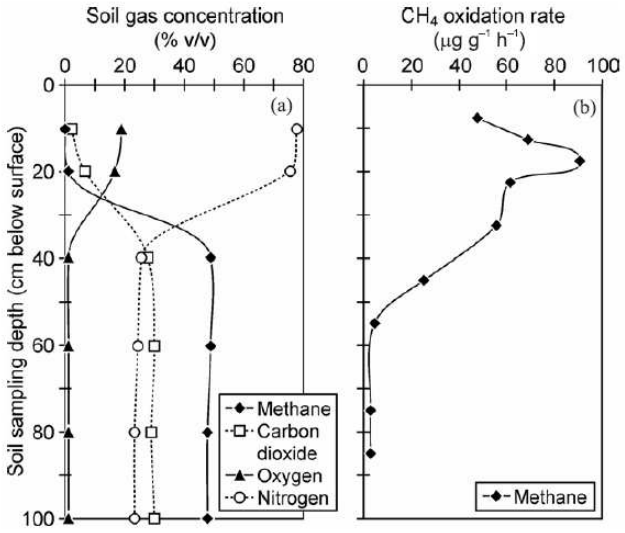


Figure 2.12: Soil gas concentration profiles measured at a Danish landfill and maximum methane oxidation rates obtained in batch incubation experiments vs sampling depth (Scheutz et al., 2009)

Scheutz et al. (2009) concluded that methanotrophic activity is limited by low O_2 concentrations at depth below 60 cm (Scheutz et al., 2009). However, Humer and Lechner (2001c) found in their field experiment that the optimum zone extended from 40 and 90 cm in sewage sludge and MSW compost.

Czepiel et al. (1996, cited in De Visscher et al., 1999) found that O_2 concentration of about 3 % v/v is a threshold for CH_4 oxidation to occur, which means changing O_2 concentration above 3 % v/v will have very little influence on CH_4 oxidation but oxidation will decrease dramatically when the concentration is lower 3 % v/v. The insensitivity to O_2 concentration above 3% v/v can be the reason for the sharp slope in CH_4 concentration profiles with depth.

Based on the assumption that no carbon is converted into biomass, the required O_2:CH_4 ratio for CH_4 oxidation is 2:1 (see stoichiometric equation 2.3), which is equivalent to 2 l O_2: 1 l CH_4 and 4 g O_2:1 g CH_4, respecitvely (Huber-Humer, 2004). This would suggest that O_2 availability in landfill cover soils is most likely the limiting factor. However, as biomass is accumulated, generally less O_2 is required for CH_4 oxidation (Huber-Humer, 2004).

2.5 Construction and design of a biocover

A biologically active cover (biocover) or methane oxidation layer is a landfill cover system that has been designed to mitigate CH_4 emissions by optimising conditions for microbial CH_4 oxidation. In general, a biocover is a layered system with different layers serving different purposes. Typically, the cover consists of a basal gas distribution layer with high gas permeability to homogenise and distribute landfill gas to the overlying oxidation and vegetative layer, where living conditions for methanotrophic populations are optimised. An appropriate vegetative layer may also serve as an evapotranspiration cap to reduce leachate. Figure 2.13 shows a simple conceptual scheme of a methane oxidation layer.

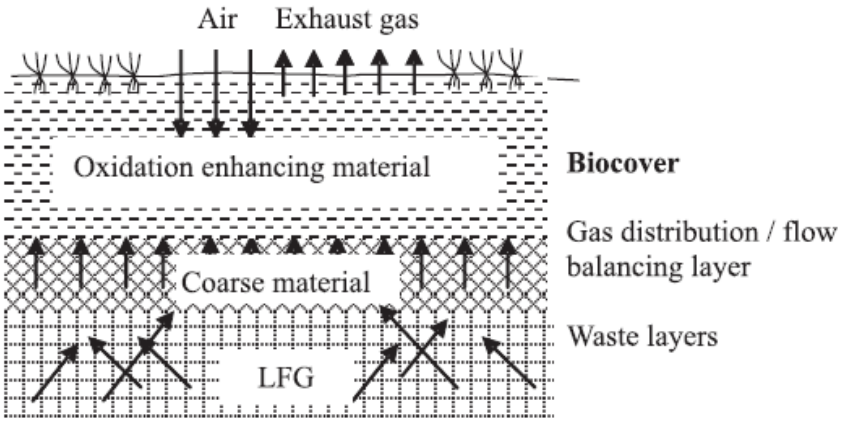


Figure 2.13: Schematic illustration of a methane oxidation layer (Huber-Humer et al., 2008b)

Generally, the biocover effectiveness relies on the use of a suitable carrier and a proper technical design and dimensioning of the landfill cover in order to oxidise variable rates of CH_4 (Bogner et al., 2010; Huber-Humer et al., 2008a; Humer and Lechner, 2001). As CH_4 fluxes typically exhibit high spatial inhomogeneity across a landfill site, the installation of a coarse gas distribution layer is crucial to achieve a more uniform supply and a slowing down of gas fluxes. Importantly, biocovers must maintain high CH_4 oxidation rates even during different seasonal climatic situations leading to changes in soil moisture, temperature and other variables (Bogner et al., 2010; Humer and Lechner, 2001a). According to recommendations from laboratory and field studies an efficient biocover will provide a long-term nutrient supply (N, P), a high temperature-insulating capacity, good porosity and gas permeability as well as a high water holding capacity (Scheutz et al., 2009; Kettunen et al., 2006; Humer and Lechner, 2001a). The design of the biocover system should accommodate a methane load of at least 4 l CH_4/m²d to ensure efficient mitigation (Huber-Humer et al., 2008a). Moreover, a biocover should be structurally stable and settle very little to maintain its porous cover (Scheutz et al., 2009; Humer and Lechner, 1999a). As already mentioned mature or specific compost materials have proven to be a suitable substrate carrier enhancing microbial CH_4 uptake (see section 2.4.2).

Engineered approaches

Currently designed biocover systems engineered to optimise CH_4 oxidation have demonstrated huge potential to mitigate residual landfill CH_4 emissions (Stern et al., 2007; Barlaz et al., 2004; Huber-Humer, 2004). In general, biocover design and dimensioning can vary in order to meet local site-specific conditions and depend on the respective climate conditions, expected gas fluxes, the purpose of the cover (final or temporary), characteristics of substrate carrier (settling behaviour, oxygen-penetration depth) and the intended after-use of the site (Humer-Huber et al., 2008b). Biocovers can be potentially applied in the absence of a gas collecting system or as a complementary method to an active system to capture escaping emissions. In addition, such systems can be used to mitigate residual emission from MBT waste or closed old landfills as emissions are too low for energetic use. Natural attenuation could be especially favourable at smaller landfill sites where the installation of a gas collection system is not technically and economically feasible. Moreover, biocovers can also be combined with forced in-situ aeration as a postclosure remediation method for old landfills. Figure 2.14 shows various specialised landfill biocovers using compost materials as a substrate carrier during different climatic conditions.

Yard waste compost (1 m)

Tire chips (gas distribution) (0.15 m)

Clay (0.15 m)

Barlaz et al. (2004)

Mature compost (1.2 m) (sewage sludge/wood chip mixture)

Non-calcareous gravel (gas distribution) (0.5 m)

Huber-Humer et al. (2008a); Huber-Humer (2004); Humer and Lechner (1999b, 2001a)

Sand and compost mixture (0.8 m)

6.4 mm gravel (transitional layer) (0.1 m)

12.7 mm gravel (gas distribution) (0.2 m)

Ait-Benichou et al. (2009); Cabral et al. (2007, 2009); Jugnia et al. (2008)

Yard or garden waste compost (0.5 m)

Crushed recycled glass (0.10 – 0.15 m)

Sandy clay and sandy loam (interim cover) (0.65 – 0.75 m)

Bogner et al. (2005, 2010); Stern et al. (2007); Abichou et al. (2006)

Figure 2.14: Engineered biocover designs

A field trial was conducted on the Outer Loop Landfill in Louisville, Kentucky, to assess a biocover system in parallel with 1 m clay soil cover for mitigating CH_4 emissions (Barlaz et al., 2004). The biocover system consisted of 0.15 m thick base layer of clay over the waste, followed by a 0.15 m gas distribution layer of tyre chips and 1 m ground yard waste compost (35 – 50 % brush) previously windrow-composted for three months. A variety of broad leaf vegetation became spontaneously established on both covers. CH_4 measurements using static chambers (see section 2.6.1) were performed while the gas collection system was operating and after the system was turned off. The biocover consumed atmospheric CH_4 or had almost zero CH_4 emissions (-1.73 – 1.33 g/m²d) during the 14 months measuring campaign even when the gas collection system was turned off. CH_4 emissions from the soil cover were more variable with relatively high fluxes (> 15 g/m²d) due to deactivation of the gas collection system and were mainly associated with desiccation cracks. Both covers showed atmospheric CH_4 uptake half of the time when the the gas collection system was operational. With the gas collection system off, the biocover continued to consume atmospheric CH_4 about 60 % of the time compared to only 12 % on the conventional soil cover. When positive emissions were measured, stable isotope measurements (see section 3.3.6) showed that an average of 55 % of CH_4 was oxidised in the biocover but only 21 % in the soil cover. In addition, measurements of trace organic compounds indicated that the biocover reduced these emissions to a greater extent than the soil cover. Barlaz

et al. (2004) concluded that compost-made biocovers serve both to reduce emissions through biotic mitigation and as deterrents to soil cracking.

Humer and Lechner (1999b, 2001a) developed some of the first prototype biocovers using various composted materials. The field trial assessment of the CH_4 removal effectiveness of compost as a landfill cover for middle-European climatic conditions was carried out on two different Austrian MSW landfills between spring 1999 and winter 2002 (Humer and Lechner, 2001a; Humer-Huber, 2004). Based on these experiences a simple two-part cover system proved to be most effective. The biocover system consisted of a 0.3 – 0.5 m coarse gravel layer (minimum particle size 16/32 mm) overlain by a layer of up to 1.2 m of mature, well-structured compost. Results showed almost 100 % CH_4 capture when 1.2 m compost was used in conjunction with a gravel under bed, and 68 – 74 % CH_4 capture when only compost (about 30 – 40 cm) was used (Humer and Lechner, 2001a). In general, the coarse mixture of sewage sludge compost with wood chips had a much better temperature insulating capacity during winter than a fine-sieved MSW-compost (Huber-Humer, 2004). The authors concluded that high CH_4 oxidation rate was mostly associated with the installation of a coarse gas distribution layer for uniform CH_4 fluxes and the good temperature insulation effect due to the sufficient dimensioning of the cover and the adequate porosity of the used material. Presently, at least five closed MSW-landfills or section of landfills in Austria are fitted with this biocover design, which has been officially approved as an acceptable interim MSW landfill cover for a period of about 20 years. These biocovers are serving either as the sole means to mitigate CH_4 emissions or complement an operating gas extraction system (Huber-Humer et al., 2008b).

Two field-scale biocovers (PMOB-1 and PMOB-3B) composed of a mixture of sand and compost have been installed at the St-Nicéphore landfill in Quebec in order to assess the efficiency of the covers in reducing CH_4 emissions (Ait-Benichou et al., 2009; Cabral et al., 2007, 2009; Jugnia et al., 2008). The biocover PMOB-1 included a 0.8 m thick substrate layer consisting of a sand/compost mixture (1:5 v/v) underlain by a 0.1 m thick transitional layer of 6.4 mm net gravel and 0.2 m thick gas distribution layer of 12.7 mm net gravel above a 3.5 year old buried waste mass. The compost was composted from a mix of municipal sewage sludge and sludge from the pulp of paper and agri-food industries. The biocover PMOB-3B consisted only of a 0.3 m thick substrate layer above 0.1 m of 6.4 mm net gravel as a transitional layer and 0.8 m of 12.7 mm gravel layer. For the substrate layer a coarser material was used that resulted from mixing one volume of the same material used as substrate in PMOB-1 with one volume of 6.4 mm gravel. In contrast to PMOB-1, PMOB-3B was fed with biogas coming from a well. In order to isolate the biocover from the existing silty cover and the waste mass, PMOB-3B was lined with a 1 mm thick HDPE geomembrane. During the first four months of monitoring (Jul. – Oct. 2006), CH_4 emissions from PMOB-1 remained low and varied between 2.5 to 30 g CH_4/m^2d (Jugnia et al., 2008) following the closed dynamic chamber method (see section 2.6.2). A high CH_4 emission flux of 210 g CH_4/m^2d was recorded once when the barometric pressure was the lowest during this study. The higher water content (51 – 64 % v/v) through the depth profile acted as a physical barrier to O_2 penetration and therefore the uppermost 0 – 10 cm layer appeared to be the most important layer for CH_4 oxidation. Stable isotope analyses were performed for both biocovers on selected dates from July to September 2007 (Cabral et al., 2009). The results showed that the substrates used in the two biocovers were able to promote CH_4 oxidation which ranged from 2.9 % to 89.7 % (at a depth of 0.1 m) in PMOB-1 and

was equal to 88.7 % for a representative profile in a relatively dry period in PMOB-3B. Quite high CH_4 emission fluxes (21.4 – 458.2 l/m²h) were measured on five occasions while the surface flux from PMOB-3B was relatively low (4.3 l/m²h). Only on two selected dates PMOB-1 had almost zero fluxes (not detectable – 1 l/m²h). In some cases, poor aeration of the substrate in PMOB-1 was observed leading to quite low efficiencies which may be linked to the degree of water saturation, the magnitude of the biogas flux and the substrate temperature.

Experiments were performed at the Leon County Landfill in Florida to study biocover effectiveness in a subtropical environment and the use of multiple recycled materials for biocover construction (Bogner et al., 2005, 2010; Stern et al., 2007; Abichou et al., 2006). Two test areas were established. On the first site (S1) a 0.5 m substrate layer of a 3 year old composted garden waste was placed over 0.1 – 0.15 m of crushed recycled glass distribution layer overlying a 0.65 – 0.75 m existing clay cover (sandy clay and sandy loam). Test area S4 included a shallow (0.3 m) or deep (0.6 m) substrate layer consisting of freshly ground garden waste (to compost in-situ). Both substrate layers were laid on top of 0.15 m of crushed fluorescent glass tubes as a gas distribution layer and a very thin interim cover consisting of 0.15 m of compacted sandy clay. On both test areas (S1 and S4) the interim covers acted as a control. Based on stable isotope analysis the biocover at site S1 consumed twice as much CH_4 (64 %) as the control cells (30 %) (see Figure 2.15).

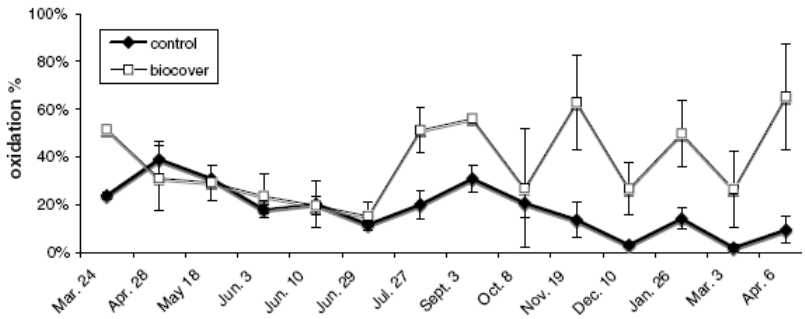


Figure 2.15: Percent oxidation of CH_4 in the control and the biocovers cells, calculated from isotope data only (Stern et al., 2007)

Following static chamber measurements, mean CH_4 emission rates from the control cells (10.6 g/m²d) were significantly greater than flux from the biocover (2 g/m²d) (see Figure 2.16) leading the authors to conclude that the thickness and moisture-holding capacity of the biocover increased the retention times for transported CH_4 and provided better protection against desiccation (Stern et al., 2007). As a consequence a higher fraction of CH_4 could be oxidised.

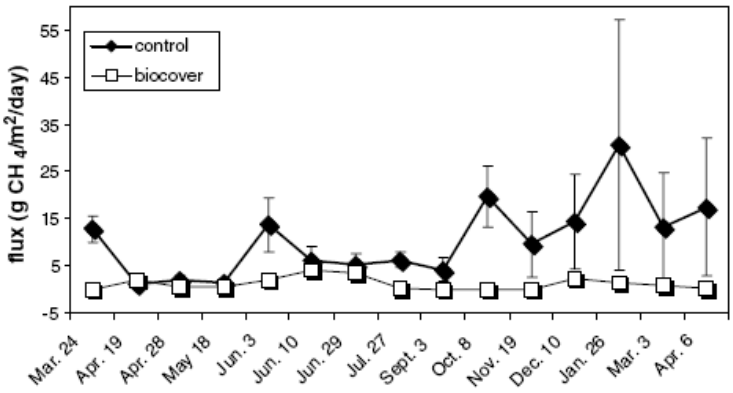


Figure 2.16: Mean CH_4 emission rates (g CH_4/m²d) from control and biocover cells (Stern et al., 2007)

At the second test area (S4) it was determined that CH_4 uptake of the deep biocover varied between 20 % and 70 %, with lower but overlapping values for the shallow biocover and the control cells (0 – 40 %) (Bogner et al., 2010). Gross CH_4 fluxes from the shallow biocover and the controll cells (up to 300 – 700 g/m²d) were 10-fold higher than the 30 – 70 g CH_4/m²d calculated for the deep biocover. These results indicate that low-dimensioned and poorly designed covers do not achieve as high CH_4 oxidation capacities as properly designed biocovers, as also demonstrated by Humer and Lechner (2001a).

2.6 Determination of landfill gas emissions

To deal with the difficulties of surface gas flux measurements, a number of methods have been designed, each with associated strengths and weaknesses. Several methods are used to quantify the emission rate for small areas, while others are used for large surface areas (e.g. for the entire landfill). For small area measurements, techniques are used, such as the chamber method and the concentration-gradient method based on Fick's law, whereas for large area measurements, micrometeorological methods, the trace method and infrared spectroscopy are applied (Abushammala et al., 2009; Fowler, 1999; Tregoures et al., 1999). Up to now, no single technique is recognised as 'the standard' methodology. In the available literature the different methods used for measuring landfill gas emission rate from landfill sites have been reviewed (Tregoures et al., 1999), however each technique has unique advantages and disadvantages. The choice will depend on economic constraints, measurement objectives and the sampling site.

Following this, the research in this study concentrates on the chamber methods which are the most common technique for the measurement of gas fluxes from different ecosystems. Chamber techniques have been widely used in wetlands (Van Huissteden et al., 2008), agricultural and semi-natural soils (Kusa et al., 2008), landfills (Chen et al., 2008; Stern et al., 2007; Barlaz et al., 2004), forest ecosystems (Von Arnold et al., 2005), but also active volcanic areas (Carmada et al., 2009, Cardellini et al., 2003, Chiodini et al., 1998) measuring various gaseous emissions, including CO_2, N_2O, NO_x, CH_4, selenium (Se) and volatile organic compounds (VOCs) such as volatile pesticides (Reichmann and Rolston, 2002; Laville et al., 1997; Hutchinson and Mosier, 1981).

Chamber techniques are based on the principle to restrict the volume of air with which gas exchange occurs so as to magnify chanbes in gas concentration in the

headspace. The aim is to have a linear increase indicating a constant flux from the surface. In general, chamber techniques can either measure (i) the rate of gas accumulation (closed system) or (ii) the instantaneous gas flux (open system). Two types of closed accumulation chambers have been developed, which are characterised by the absence (static) or presence (dynamic) of airflow inside the chamber. In Figure 2.17, the different types of chamber methods are depicted.

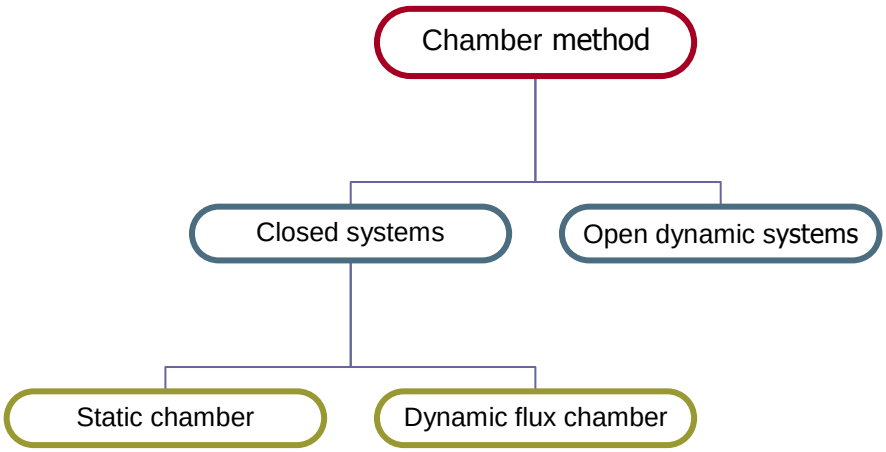


Figure 2.17: Different systems of chamber methods

In previous research studies chamber systems have been either:

- Process orientated (investigation of production, emission or mechanisms of exchange),
- Designed for long-term flux measurement (traceability of the seasonal or annual time course in fluxes of trace gases which show large temporal cycles); and
- Complementary flux measurements (Fowler, 1999).

The advantages, limitations, assumptions and computation of exchange rate differ greatly among the different designs.

Current types of chamber used usually vary in basal sampling area, from 0.1 to 100 m². The upper limit represents the megachamber approach such as that used by Smith et al. (1994) and Galle et al. (1994). Currently, chambers < 1 m² are the most common tools since larger ones are difficult to transport. However, considerations of the basal area depend on the practicality of installing chambers as determined by the terrain of the sampling site (Livingston and Hutchinson, 1995). Closed systems have been used more often than open dynamic chambers as they are mechanically simpler and small gas fluxes are easier to detect. In addition to the choice of general enclosure type, decisions are required concerning the deployment, chamber geometry, fabrication materials, temperature control and measurement equipment to monitor concentration changes within the enclosure (Livingston and Hutchinson, 1995).

Results obtained by chamber methods, both closed and open, can be statistically evaluated to determine whole landfill fluxes by establishing statistically-based sampling schemes. However, a large number of samples (≥ 100) are needed for the descriptive statistics of spatial data based on the arithmetic mean as only point emissions can be measured (Bogner et al., 1997a). Given the high spatial and temporal variability in LFG fluxes, it might be helpful to use several enclosures

simultaneously or a higher number of replicates in order to obtain a representative estimate. General errors in gas flux measurements are related to the chamber effects on modifications of the microclimate (e.g. temperature, humidity), physical disruption of the surface, pressure-induced gas flows in open chamber designs, inhibition of fluxes through concentration and pressure build-up in closed chambers and perturbations of the natural conditions (Denmead, 2008; Asman et al., 1999).

2.6.1 Closed static chamber

Kunz and Lu (1980, cited in Bogner et al., 1997a) reported the first use of a closed static chamber measuring CH_4 fluxes from a New York site. Nowadays, this enclosure method has been recognised as the simplest technique to measure gas emission fluxes at selected points on the surface of a landfill (Bogner et al., 1997a; Livingston and Hutchinson, 1995). Generally, an open-bottom chamber (cylindrical or rectangular) made, for instance, of plastic or metal is placed over a small area of soil surface so that the gas concentrations increase continually in the head space. A sealing has to be done, otherwise air can infiltrate into the chamber compromising the sample. The gas emission flux (F) at the surface is calculated from the relationship:

$$F = \frac{V}{A} \cdot \frac{dC}{dt} \tag{2.4}$$

where *V* is the volume of the head space, *A* the surface covered by the chamber, *C* the change in gas concentration and *t* is the sampling time. In addition to the gas concentration, temperature (T) in the chamber and barometric pressure (P) have to be measured to convert the change in volumetric concentration to a mass flux (g/m^2d). When temperature and pressure correction are applied, the following equation results:

$$F_{cor} = F \cdot \frac{273.15}{273.15 + T} \cdot \frac{P}{1013} \tag{2.5}$$

Typically, air samples are taken periodically from the head space with a gas syringe and the gas concentrations are measured later in the laboratory, for example, using a gas chromatograph (Chanton and Liptay, 2000; Bogner et al., 1997b). The enclosure period should be no longer than 40 min and at least three time points are required for flux calculation (e.g. 0, 20, 40 min) (Livingston and Hutchinson, 1995). Fans are often used to obtain better mixing of the air inside the chamber ensuring representative sampling. Two closed static chambers are shown in Figure 2.18.

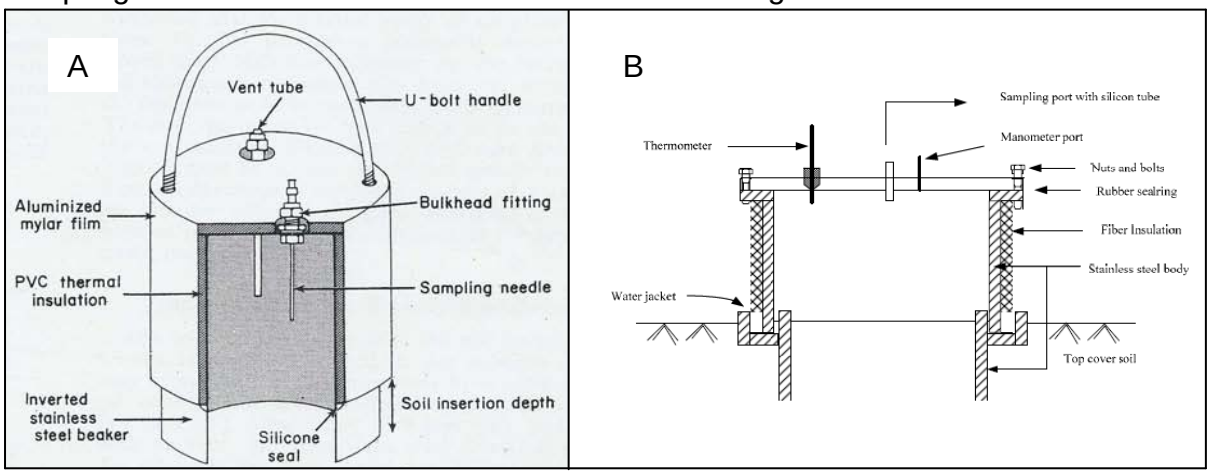


Figure 2.18: Closed static chamber A: (Hutchinson and Mosier, 1981), B: Visvanathan et al. (2004)

Strengths and weaknesses

The static chamber technique is low in cost, simple in concept and operation and requires no power in the field. It is especially useful for addressing research objectives needing spatial and temporal variability of fluxes at a small scale (Abichou et al., 2006; Bogner et al., 1997a). Further advantages include portability and adaptability to a wide variety of field conditions. Since gas collected inside the static chamber is not being continually diluted with external air (as is the case with open dynamic chambers), this method can be highly sensitive detecting small fluxes or even net uptake of atmospheric CH_4 (Denmead, 2008; Perera et al., 2002; Whalen and Reeburgh, 1990).

However, the static chamber may not give reliable results as the chamber itself influences actual emission rates from the soil surface. Once the chamber is placed on top of a landfill cover during a measurement, the gas concentration gradient is altered reducing diffusional flux and thereby actual emission rate is suspected to be underestimated (see Figure 2.20). Matthias et al. (1978, cited in Perera et al., 2002) reported that the actual flux could be as much as 55 % higher than the calculated flux from the chamber measurement. If the flux into the chamber is controlled by pressure difference, a build-up of pressure above barometric pressure will occur in the chamber decreasing the gas flux with time (Rolston, 1986, cited in Bogner et al., 1997a). Due to these reasons, the chamber can only be used for relative comparison of fluxes from place to place. In addition, natural wind conditions, an increase of temperature inside the chamber and gas sampling can induce pressure differences between the chamber and outside air creating artefact gas fluxes. Denmead (1979), for example, demonstrated that a pressure deficit of only 100 Pa between the atmosphere inside and outside the chamber could alter the gas emission rate by a factor of 10. Moreover, closed chambers may cause a by-pass and enhanced emission fluxes outside the measuring chamber when placing it on top of a coarse-textured cover material with high gas permeability (Huber-Humer et al., 2009). Therefore, installation of a vent tube in a chamber wall to permit pressure equilibration is a wise precaution. Hutchinsons and Mosier (1981) provided guidelines for appropriate vent dimensions for effectively transmitting ambient pressure fluctuations to the enclosed space (for wind speeds up to 4 m/s) while minimising loss of the accumulating gas by diffusion to the outside (< 1 % diffusion loss). Optimum vent tube diameter and length for selected wind speeds and enclosure volume are summarised in Figure 2.19.

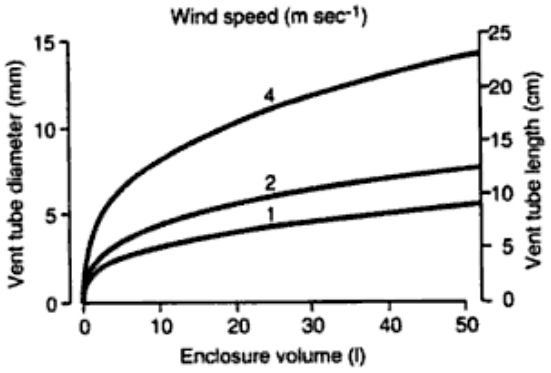


Figure 2.19: Optimum vent tube diameter and length for selected wind speeds and enclosure volume as described by Hutchinson and Mosier (1981)

The errors in flux measurement by closed chambers largely depend on the configuration of the chamber and the gas flux rate. Based on experiments by Perera et al. (2002), the smaller the chamber and higher the flux rate, the higher the percentage error in measurement. Most of the artefacts introduced by the chamber itself can be overcome with sophistication of the equipment or quantified by experiment using a test bench (Fowler, 1999).

2.6.2 Closed dynamic chamber

Compared to static chamber, closed dynamic chambers are more sophisticated. Their operating principle involves air circulation in a closed loop between the head space and a gas analyser. The rate of gas enrichment of the recirculated mixture can be monitored continuously and, thus, any inhibition of the flux through a build-up in head space concentration can be detected. The increase of gas concentration over time will be calculated with the equation used for static chamber systems.

Strengths and weaknesses

As air is pumped between chamber and gas analyser the gas inside the chamber is mixed minimising negative effects of a possible temperature increase inside the chamber. However, this configuration can also cause pressure effects. Apart from that, the closed dynamic chamber has the same advantages and disadvantages as static chambers except that power is needed for air pumping.

Figure 2.20 shows the simultaneous time course of the concentrations of CO_2, CH_4 and N_2O in the head space of a closed dynamic chamber installed in a fertilised field (Breuer et al., 2000, cited in Denmead, 2008). The plot indicates that towards the end of the 18-min measurement period the gas fluxes (CO_2, N_2O) are inhibited, even over this short enclosure period, due to high gas concentration in the head space. This is a particular problem for closed chamber methods. Hence, the time series of the concentration change should be investigated in order to establish an appropriate sampling period (Denmead, 2008).

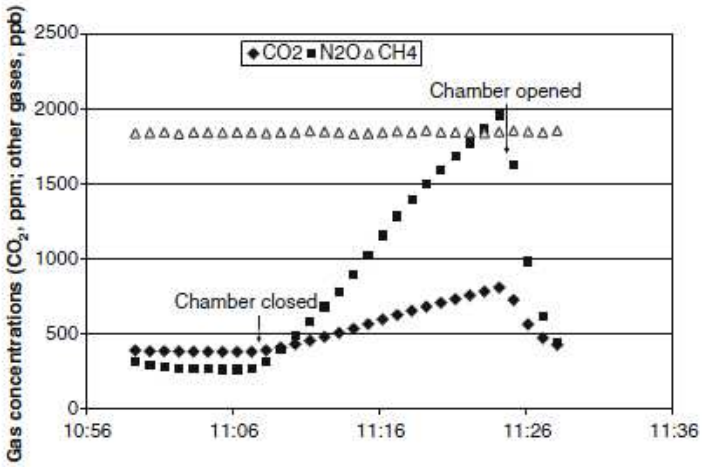


Figure 2.20: Time course of gas concentrations in the head space of a closed dynamic chamber (Breuer et al., 2000, cited in Denmead, 2008)

2.6.3 Open dynamic systems

An alternative flux measurement method is the open dynamic system in which a constant flow of outside air is maintained through the head space of the chamber and the difference in concentration between inlet and outlet is measured. Usually, air is

blown or pumped into the enclosure at a fixed rate and gas concentration is measured using appropriate detection instruments. The gas emission flux is calculated as follows:

$$F = \frac{Q}{A} \cdot \left(C_{out} - C_{in}\right) \tag{2.6}$$

where F and A are as defined previously, Q is the volume flow rate, C_{out} is the gas concentration in the air leaving the chamber and C_{in} the gas concentration in the air entering the chamber. Various possible configurations and operating features of open dynamic chambers have been reported in the literature (Müller et al., 2009; Röder et al., 2004; Gao et al., 1997; Pokryszka et al., 1995), as can be seen in Figure 2.21.

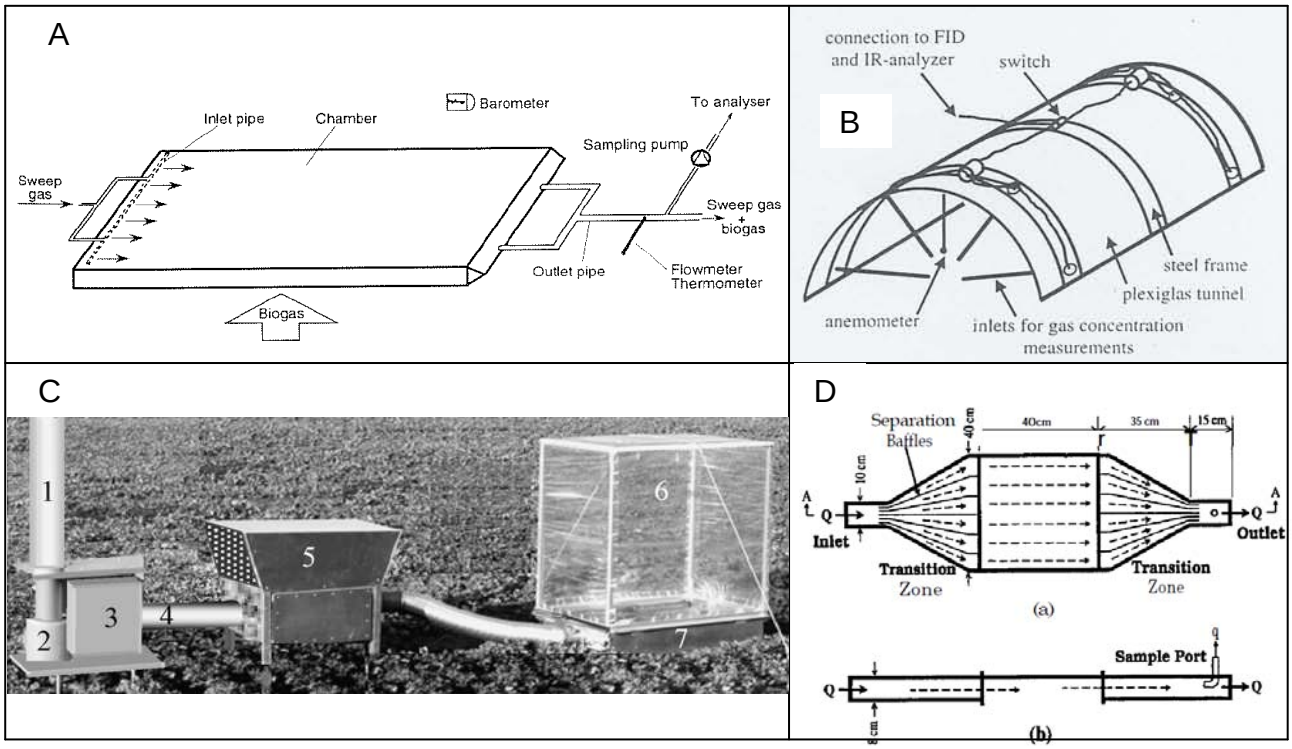


Figure 2.21: A: Dynamic flux chamber (Pokryszka et al., 1995); Open wind tunnel (Röder et al.; 2004), C: Canopy chamber system (Müller et al., 2009); Top-view and cross-section of the flow-through chamber system (Gao et al., 1997)

In the dynamic flux chamber introduced by Pokryszka et al. (1995) a sweep gas flow (inert gas) is injected via a perforated pipe at the entry of the measuring chamber. At the outlet the sweep gas/biogas mixture is collected passively and the flow rate is measured by a hot-wire anemometer. Gas samples from the outlet pipe are pumped to the gas analyser where concentrations of biogas constituents are measured.

An open wind tunnel was developed by Röder et al. (2004), in which the natural wind velocity is used instead of a pump or blower to carry the air and the emitted gases through the tunnel. In the rear and the front section of the tunnel gas concentrations are detected while at the outlet wind speed is measured.

The canopy chamber is an automated system which consists of a tube for supplying ambient air to the system, a radial blower, a central measuring and controlling unit and the canopy enclosure (Müller et al., 2009). Air and soil temperature as well as soil moisture can be measured within the chamber.

The flow-through chamber system consists of an inlet transition zone, a square main body, and an outlet transition zone (Gao et al., 1997). In order to conduct and spread the flowing air uniformly across the soil surface, six equally-spaced channels are installed in both inlet and outlet transition zones. Gas samples can be taken at the inlet and outlet of the chamber.

Strengths and weaknesses

In recent studies, open dynamic chambers have been preferred as these minimise the difference between ambient conditions and those within the enclosure since the ambient air is continuously flowing through the chamber (Fowler, 1999; Gao et al., 1997). The increase in gas concentration above background can be controlled by the volume flow rate and, thus, the risk of inhibiting the surface flux can be reduced. In addition, open systems can detect a wider range of gas concentration (Pokryszka et al., 1995).

However, because of the small magnitude of the concentration increase (emitted gas is diluted with air), open dynamic chambers may not be able to detect small fluxes. Besides, the air flowing in the chamber may change the pressure gradient between soil-gas phase and the chamber interior (Gao et al., 1997) depending on the degree to which the chamber is open to the atmosphere (Fang and Moncrieff, 1998). A change in the pressure gradient may create an additional advective mass flow of the target gas or an influx of gases from outside the enclosure, in particular, when the air in the enclosure flows relatively fast compared to the wind outside the chamber. This pressure deficit may cause an overestimation of the emission rate (Fang and Moncrieff, 1998; Gao et al., 1997). Another potential problem with open dynamic chambers is associated with the air-flow pattern within the enclosure (Gao et al., 1997). If the inlet and outlet of the chamber are not designed properly, the air flowing through the enclosure may not sweep over the entire covered surface. As a result, local stagnant air zones are created inside the chamber which can cause a spatially variable emission flux. This is also true for large closed dynamic chambers.

In practice, both closed and open chamber methods are subject to a wide range of errors, including modification of environmental conditions within the chamber relative to those in the field (Fowler, 1999). Anomalous pressure effects seem to pose major challenges for accurate chamber-based measurements given their difficulty to measure or control them (Perera et al., 2002).

2.7 Existing standards and legal ordinances

Currently, there are no specific technical standards or legal specifications on the setup of biocovers in Austria or in the European Union. In spite of the numbers of research studies regarding the applicability of biocovers, standardised implementation and construction criteria within the European Union are still missing. However, a national “Technical Guideline for Biocovers” has been drafted under the auspices of the “Austrian Association for Management of Contaminated Sites” (see Huber-Humer et al., 2008a).

In Austria, implementation of biocovers may be affected by some specific rules and standards including the “Austrian law on remediation of inherited waste (“Altlastensanierungsgesetz”), the “Austrian Ordinance on Composting” and the “Austrian Landfill Ordinance”.

According to paragraph § 2 in the “Austrian law on remediation of inherited waste” (ALSAG, BGBl. Nr. 299/1989, amended by BGBl. I Nr. 40/2008), waste materials (including waste composts) used for constructing landfill covers, including methane oxidation layers and reclamation layers, are subject to “contaminated site liability” (Altlastenbeitragspflicht). The “contaminated site liability” is a tax for each ton of waste material put on a landfill site, which has to be paid by the landfill operator. However, reclamation layers and temporary covers (which also serve the purpose of oxidising methane) will be excluded from the “contaminated site liability”, if they are implemented in full compliance with criteria defined in Appendix 3, sections 4.5 and 6.1 of the “New Landfill Ordinance 2008” (BGBl. II 39/2008).

In certain circumstances, the use of compost for constructing a biocover is subject to the “Austrian Ordinance on Composting” (BGBl. II 292/2001), in particular paragraph § 6 (3) which limits the quantities of compost applied for the construction of a reclamation layer. This is the case, if high-quality compost is used meeting the quality criteria of the Compost Ordinance. As this compost is no longer classified as waste, the “contaminated site liability” according to ALSAG can be disregarded.

The “Austrian Landfill Ordinance“ (BGBl. II 39/2008) provides requirements for intermediate covers as well as for temporary permeable covers (acting as a biocover for a max. period of 20 years), which can be used to minimise methane emissions from landfills (or compartments) with high amounts of biodegradable waste. According to paragraph § 29 (4), the use of compost for constructing an intermediate cover is prohibited whereas it is allowed for temporary covers, except compost made of municipal solid waste. The use of compost for constructing a final reclamation layer is only allowed in accordance with regulations in the “Austrian Ordinance on Composting”. Appendix 3, section 6.1 provides specifications regarding biocover effectiveness and a monitoring program for temporary covers. Gaseous emissions from a whole landfill site must not exceed an average of 5 kg CH_4/m²year and single values occurring at “hot spots” may exceed this up to a maximum of 10 kg CH_4/m²year. According to the monitoring program, the temporary cover surface has to be surveyed every quarter for possible methane emissions using a grid-based FID-mapping (Flame-Ionisation-Detector). Emission rates (CO_2 and CH_4) have to be quantified at least twice a year taking into account the grid-based FID-survey. In addition, landfill operators are committed to submit a project to the local authorities concerning the covering of the landfill site after the temporary period of 20 years. (Huber-Humer et al., 2008a)

3 MATERIALS AND METHODS

3.1 Development of a measuring system

The primary considerations in the choice of method for measuring gas fluxes include design of the lysimeters under study (see section 3.3.1), gas species in question (CO_2 and CH_4), availability of analytical instruments and cost. After extensive literature research, it was decided to develop a measuring system based on the closed dynamic chamber technique. Specific efforts included design and development of the measuring system and testing of this system under different known CH_4 and CO_2 fluxes to demonstrate the feasibility of the measurement technique.

Prototype

The technique used in this work was inspired by the accumulation chamber method developed by INERSIS (Institut National de l´Environnement et des Risques) for estimating CH_4 emissions from MSW landfills (Savanne et al., 1995; Tregoures et al., 1999). In the first instance a prototype was developed to test the operating principle only for CH_4 fluxes, as illustrated in Figure 3.1.

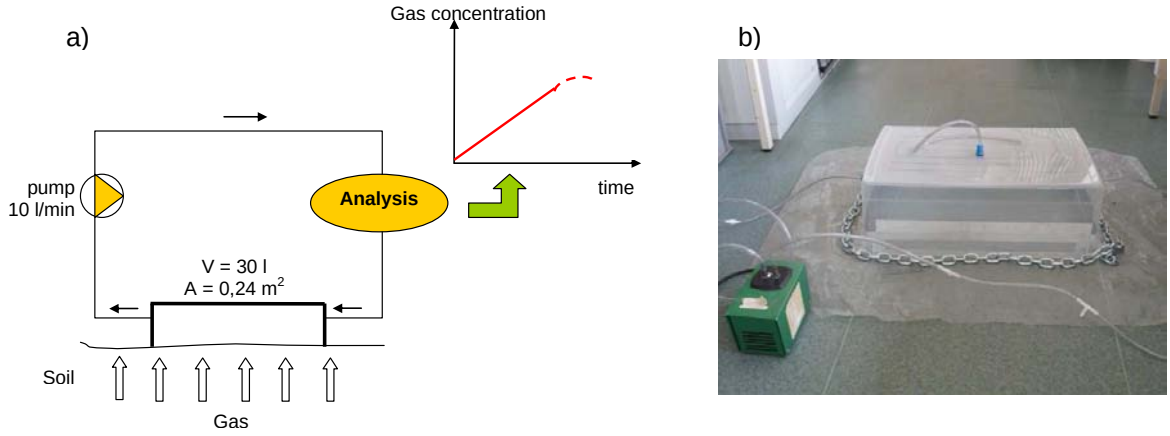


Figure 3.1: Schematic of the measurement system (a) and illustration of the developed prototype (b)

The prototype was a rectangular plastic box covering an area of 0.24 m² with an enclosed volume of 30 l. The internal atmosphere was renewed using an external pump with a flow rate of 10 l/min. The methane flux obtained outside the chamber was analysed using a portable Flame Ionisation Detector (FID) equipped with a 1 l/min internal pump.

The chamber had an attached plastic skirt held in place on the soil surface with a chain. In order to avoid physical disruption of the cover materials upon chamber installation, the use of an anchor or collar for sealing was avoided. Three holes, to accommodate fittings for vent tube and gas sampling ports for gas recirculation, were drilled in the chamber top and sidewalls, respectively.

First test runs were performed in the laboratory and in the field revealing the feasibility of the chamber. Given the high spatial variability in LFG fluxes, a small chamber (0.24 m²) may not give representative estimates of the gas fluxes emitted from the experimental biocovers in the landfill lysimeters (4 m²) unless several

chambers are used simultaneously or the number of replications is increased. Kaiser et al. (1996) reported higher variability from small boxes (0.0078 m²) than from larger boxes (5.76 m²). Therefore, it was decided to build a large chamber with the same operating principle covering almost the whole area of the lysimeter chamber. The challenge was to include the aeration pipe (D = 6 cm) of the lysimeter chamber into the design of the accumulation chamber.

Further development

The chamber was constructed by covering a square wooden base frame with a plastic tarp which had a hole in the middle reserved for the aeration pipe of the lysimeter chamber (see Figure 3.2). It had an internal length of 190 cm and therefore covered an area of 3.61 m². The chamber was equipped with one gas sampling port at the inlet and one at the outlet. The sampling port at the outlet was connected to a portable FID via an external pump with a flow rate of 55 l/min. The large chamber also had an attached plastic skirt.

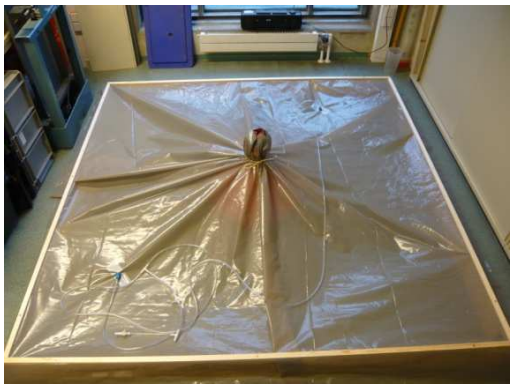


Figure 3.2: Illustration of the constructed accumulation chamber

Laboratory tests were conducted to study the effect of chamber behaviour on flux measurement with a constant emission source of pure CH_4, which was introduced directly into the chamber base via a hose. The CH_4 fluxes were regulated by a flow controller based on the rotameter principle. The rotameter was carefully calibrated by a bubble flow meter. For the calculation of the CH_4 flux (see equation 2.4) it is essential to know the exact chamber volume. In this case, the use of a plastic tarp was not suitable as the chamber volume could only be estimated.

Final chamber

Instead of the plastic tarp, plexiglas was used to cover the wooden base frame (see Figure 3.3). A hole in the middle of the chamber top (15 x 18 cm) was fitted with a plastic foil which could be mounted on the aeration pipe after installation of the chamber on the surface of the lysimeter. At the inlet and outlet side of the chamber four sampling ports were installed and connected by using tubes in cascade.

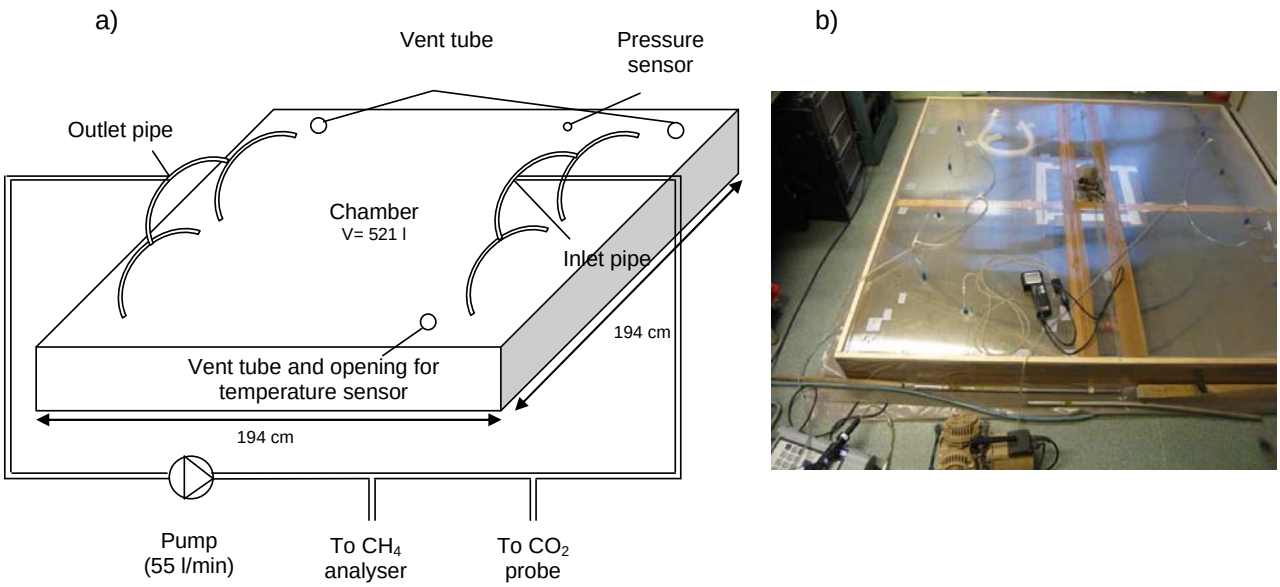


Figure 3.3: Schematic drawing (a) and illustration (b) of the developed chamber

The outlet tube of the chamber was connected to a portable FID (Thermo Scientific FID, TVA-100) and CO_2 detector (Vaisala CARBOCAP™ MI70 + GMP22) via an external pump (55 l/min) (see also Table 3.3). The FID was connected to a three-way-valve, sucking ambient air via an internal pump. Reading of CH_4 concentrations were taken every minute. 15 seconds before CH_4 concentrations were recorded, the open tube was closed and the valve was opened towards the chamber. The CO_2 detector is based on diffusive sampling. Therefore, the CO_2 probe was placed into a gas bag where CO_2 concentrations were measured simultaneously each minute.

To prevent pressure buildup, the chamber was equipped with a vent tube according to Hutchinson and Mosier (1981). The dimensions of the vent tube should allow a pressure loss not greater than 0.1 µbar. The internal tube volume should be five times greater than the volume of enclosed air displaced by the largest anticipated pressure wave. Table 3.1 gives the computed pressure wave amplitude for a range of windspeeds.

Table 3.1: Amplitudes of wind-caused air pressure waves at selected frequencies as a function of wind speed (Hutchinson and Mosier, 1981)

	Log Frequency (Hz)				
	-3.0	-2.0	-1.0	0.0	1.0
Wind speed (cm/s)	Pressure wave amplitude (µbar)				
100	3.3	1.1	0.33	0.11	0.03
200	8.9	2.8	0.89	0.28	0.09
400	59.0	19.0	5.9	1.9	0.59

Based on the values given in Table 3.1 the maximum rate of air displacement (v) from the cover caused by each pressure oscillation and the total volume of enclosed air displaced per pressure cycle (b) can be calculated using the following equations 3.1 and 3.2, respectively. Results are presented in Table 3.2 assuming an average barometric pressure of 1000 mbar and a chamber volume of 521.5 l.

$$v = \frac{2\pi f a V}{p_m} \cdot \qquad (3.1)$$

$$b = \frac{2aV}{p_m} \cdot \quad (3.2)$$

v	(µl/s)	volumetric air flow rate
b	(µl/cycle)	displaced volume
f	(Hz)	frequency
a	(µbar)	pressure wave amplitude
V	(µl)	chamber volume
p_m	(µbar)	average barometric pressure

Table 3.2: Maximum air displacement rate and total air volume displaced per cycle for a 521.5 l chamber volume and an average barometric pressure of 1000 mbar

	Log Frequency (Hz)				
	-3.0	-2.0	-1.0	0.0	1.0
Wind speed (cm/s)	Air displacement rate (µl/sec)				
100	10.85	36.18	108.53	361.78	986.68
200	29.27	92.09	292.72	920.90	2960.04
400	194.05	624.90	1940.47	6248.97	19404.71
	Air volume displaced (µl/cycle)				
100	3454.77	1151.59	345.48	115.16	31.41
200	9317.41	2931.32	931.74	293.13	94.22
400	61767.10	19891.10	6176.71	1989.11	617.67

Equations 3.3 and 3.4 predict the optimum vent tube dimensions based on the largest values of v and b from Table 3.2.

$$D = \frac{51200\mu rbv}{(\pi^2 g_c \Delta p)^{\frac{1}{16}}} \cdot \quad (3.3)$$

$$L = \frac{4rb}{\pi D^2} \cdot \quad (3.4)$$

D	(mm)	tube diameter
L	(mm)	tube length
µ	(Pa s)	air viscosity ($1.7*10^{-5}$)
r	(-)	the volume ratio 5:1
g_c	(m/s²)	gravity (9.81)
Δp	(µbar)	pressure drop (0.1)

The optimum vent tube dimensions of the final chamber are a diameter of 2.9 cm and a length of 48.3 cm. Therefore, the chamber was equipped with a 50 cm open tube (D = 3 cm), that connected the chamber volume with the outside atmosphere and thus allowed for a maximum diffusion loss of 1 %. However, the formulas provided by Hutchinson and Mosier (1981) are usually applied for small chambers up to 157 l (Bauer, 1996) and, thus, the applicability for large chambers is not confirmed.

To monitor the pressure in the chamber with respect to atmosphere, a fitting for a pressure sensor was drilled in the chamber top. The temperature inside the chamber was measured using a temperature probe inserted into the vent tube. Carrying straps were installed on the chamber for better handling.

In Table 3.3 the main characteristics of the measurement system used are summarised.

Table 3.3: Main characteristics of the used measurement system

Chamber internal length	190 cm
Chamber height	14.5 cm
Vent tube length	50 cm
Vent tube internal diameter	3 cm
Pumping flux	55 l/min
Dead volume	1,96 l
CO_2 detector and accuracy	Vaisala CARBOCAP™ MI70 + GMP222 ±1.5 % in the range of 0 to 10,000 ppm or ±2 % of reading
CH_4 detector and accuracy	Thermo Scientific FID, TVA-100 ±25 % of reading or ±2.5 ppm in the range of 1.0 to 10,000 ppm
Pressure sensor and accuracy	Testo 452 ± 10 Pa (0...20 hPa) and ±0.5 % of reading (20...100hPa)
Temperature probe	Testo 720, PT100 measuring range -50 to 240℃

Reliability testing

Laboratory tests were carried out to become familiar with the measurement system, to improve the measurement procedure and to quantify the accuracy of the system. In order to verify the performance and accuracy of the measurement system, different known LFG fluxes were introduced into the chamber under constant barometric pressure and temperature. It was also important to check the time course of concentration change in order to establish an appropriate sampling period.

The LFG fluxes were regulated by a flow control based on the rotameter principle. The rotameter was carefully calibrated by a bubble flow meter at the beginning of each test run.

The first laboratory tests using only the CH_4 analyser (portable FID) and a constant emission source of pure CH_4 have shown good accuracy and repeatability in the emissions measurement capability of the system. Figure 3.4 shows a comparison between actual methane emission rate and collected methane measured during the laboratory experiments. Methane emission rate was measured three times for each known gas flux rate of pure CH_4. Results deviate from imposed CH_4 flux values by 10 % or less. The dispersion coefficients of the results obtained under similar conditions were in the range of 1.3 – 8.5 %, thus the method does, under laboratory conditions, ensure a good measurement repeatability rate. However, when the applied CH_4 emission was increased (> 35 g/m²d), the measured CH_4 rates were all significantly higher than those applied, indicating that this measurement system overestimates higher fluxes and, thus, it is only able to detect small gas fluxes.

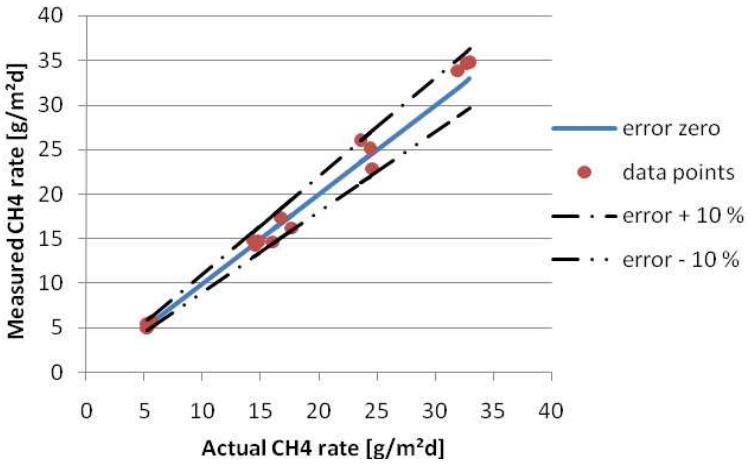


Figure 3.4: Laboratory results of method reliability test

Figure 3.5 shows methane concentrations changing with time measured during the first laboratory experiment. The CH_4 flux was calculated from the linear regression of the CH_4 concentrations versus time. It was found, that the appropriate time curve for gas sampling by this accumulation chamber should be about 6 – 9 min. The increase of CH_4 concentrations proved to be linear for all measurements. The coefficients of determination (R^2) ranged between 0.9506 – 0.9953.

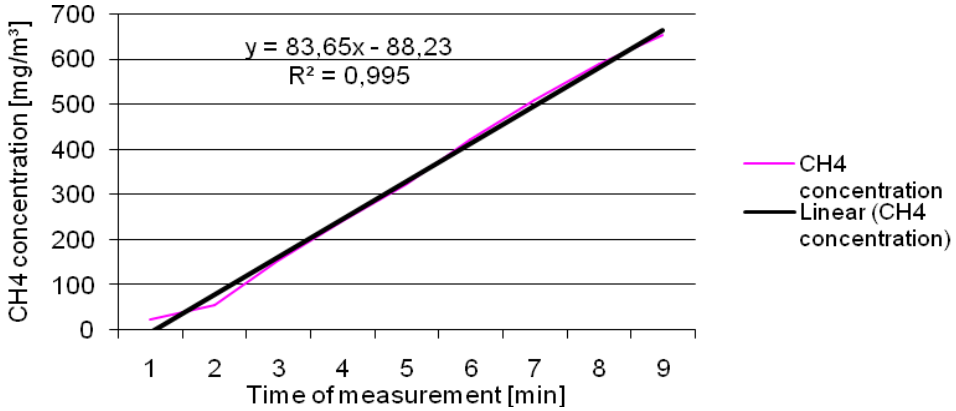


Figure 3.5: CH_4 enrichment in the chamber over 9 minutes

In the second series of laboratory experiments, a constant source of synthetic landfill gas (60 % CH_4, 40 % CO_2) was introduced into the chamber. CH_4 and CO_2 emission rates were measured simultaneously in the external loop using the CH_4 and CO_2 detector. Three to five replicates were performed for each LFG flux. It could be shown that the uncertainty margin of the modified measurement system was about ± 30 % (see Figure 3.6). This is a reasonable result considering the limitations of the experiment. The used flow controller is based on the rotameter principle, itself introducing an uncertainty of 3 – 10 % due to imprecise scaling.

The dispersion coefficients of the measured CH_4 and CO_2 fluxes ranged between 1 % and 16.5 % and 1.9 – 10.6 %, respectively. The coefficients of determination (R^2) were found in the range of 0.95 – 0.993. The chamber tended to underestimate CO_2 emission rate over the tested range of LFG fluxes, as can be seen in Figure 3.6. At higher fluxes the deviation from actual CO_2 and CH_4 values increased markedly and, thus, the measurement system gives only reliable results for small LFG fluxes.

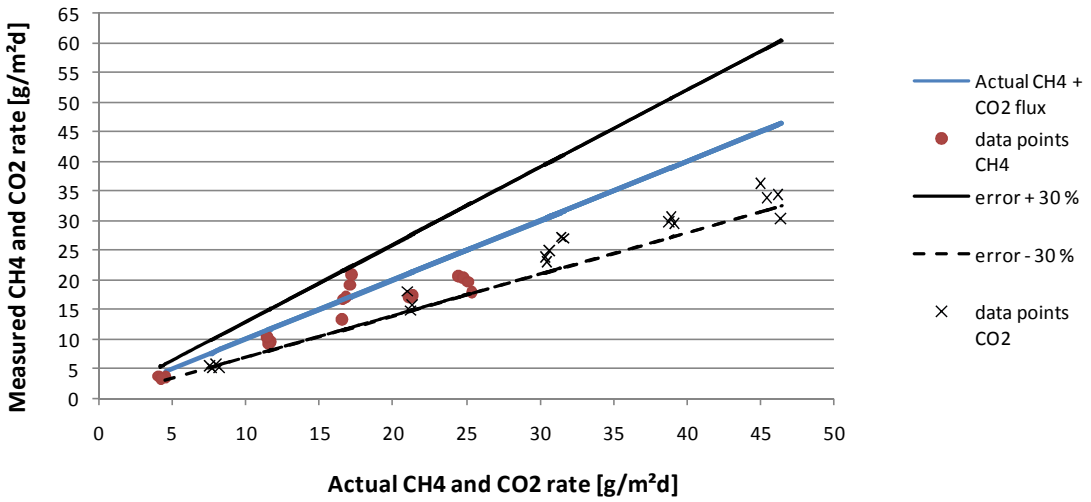


Figure 3.6: Laboratory results of second method reliability test

Following the laboratory validation phase, the chamber was fine tuned under natural conditions. Based on the measurements of the pressure difference, the installation of another two vent tubes at the accumulation chamber appeared necessary in order to compensate pressure differences during the measurements.

It can be concluded that reliable CH_4 and CO_2 emission rates are obtained in the range of 0.01 g CH_4/m^2d (lower detection limit) to 35 g CH_4/m^2d and 1 – 47 g CO_2/m^2d, respectively. Due to inhibition of fluxes through concentration and pressure build-up in closed chambers, the chamber can only be used for relative comparison of fluxes from the different biocovers and, thus, absolute data should be regarded with caution. Because of the size of the chamber and the need of power supply, portability is limited.

3.2 Suitability of substrates to enhance methane oxidation

3.2.1 Potential methane oxidation activity

Prior to biocover construction and implementation, laboratory experiments should be conducted for assessing potential CH_4 oxidation activity of possible cover materials. Common approaches are incubation experiments (batch tests) or flow-column set-ups (undisturbed or packed column experiments) under controlled laboratory conditions (Huber-Humer et al., 2009; Gerbert and Streese-Kleeberg, 2008). For batch tests, defined quantities (usually about 10 – 30 g) of cover material samples are placed in incubation jars. In the presence of methane and oxygen, the jars are incubated and the decrease in methane concentration is continuously measured over time which allows the determination of kinetic properties of methane oxidation and short-term responses to environmental factors (Scheutz et al., 2009; Maurice and Lagerkvist, 2004; Boeckx et al., 1996; Kightley et al., 1995). In general, batch tests are a fast method to characterise the methane oxidation potential of soil materials and to evaluate their suitability for biocover construction as batch conditions can be manipulated easily (Huber-Humer et al, 2009). However, due to small sample quantities, this approach may only be suitable for homogeneous materials whereas heterogeneous substances like composted waste materials are not considered to be representative, unless they are coupled with continuously charged column tests (Huber-Humer et al., 2009). Column experiments simulate a landfill cover material matrix through which gas is transported and therefore closely resemble the dynamic

behaviour in landfill settings (Scheutz et al., 2009). An undisturbed soil column, where a CH_4/CO_2-mixture is applied at known rate to the bottom, and the top is flushed with air, is considered to be the most realistic laboratory microcosm (Chanton et al., 2009). Due to considerable variability between undisturbed columns, some research is conducted with repacked columns which allow better control of soil composition (He et al., 2008). In general, column tests allow higher mass input, coarser particle size, and longer test runs in order to reveal some of the long-term phenomena that can occur, such as microbial exopolymers (EPS) formation influencing gas exchange and oxidation efficiency (Huber-Humer et al., 2009; Scheutz et al., 2009; Huber-Humer, 2004).

Before the lysimeter experiment was conducted, pre-investigations in the laboratory have been carried out to check material suitability and oxidation capacity. At the Institute of Waste Management in Vienna continuously charged soil columns (repacked) were used to test various cover materials, both mineral soils and compost materials, concerning their potential methane oxidation activity. Table 3.4 lists the substrates investigated in these experiments.

Table 3.4: Various substrates investigated in column experiments

Substrate	Abbr.	Source
Sewage sludge compost	SSC	Hollabrunn
Biowaste compost	BWC	Wien - Lobau
Sand 2/4	Sand-Fischa	Fischamend
Topsoil (silt)	TS-Fischa	Fischamend
Topsoil (loess)	TS-Zister	Zistersdorf
Subsoil (loess)	SS-Zister	Zistersdorf
Sand-compost mixture (60:40 vol. %)	SSC-Mix	Fischamend / Hollabrunn

Based on the column experiments (see Figure 3.7) the most suitable cover materials were selected for long-term observations in the landfill lysimeters under natural conditions.

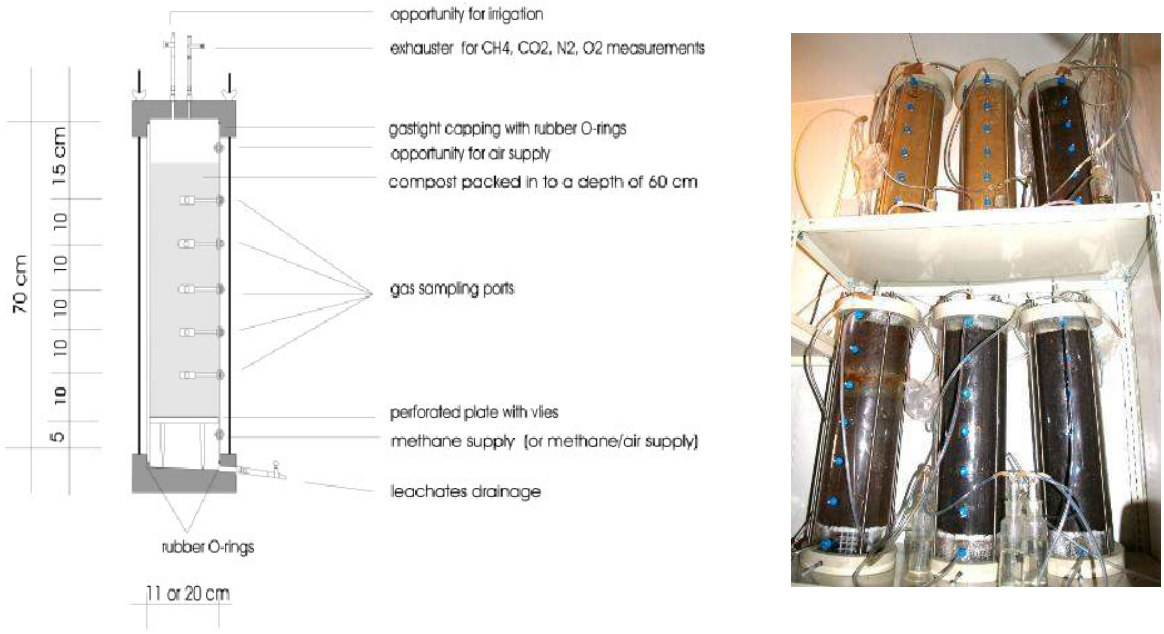


Figure 3.7: Laboratory test column to investigate methane oxidation capacity; continuous methane supply to the bottom; air supply to the top; lateral measuring holes to establish gas concentration and temperature profiles (Humer and Lechner, 1999a; Internal Report Nutzraum, 2008)

In Figure 3.8 the percents of oxidised methane load in all investigated substrates are pictured as time curves. While the mineral soils (TS-Fischa, TS-Zister, Sand-Fischa) showed only moderate methane oxidation potential (medial oxidation rate 38 %, 15 % and 20 %, respectively), the highest methane turnover rate was observed in mature sewage sludge compost (SSC). The mature compost material mitigated loadings up to 350 l CH_4/m²d indicating a constant CH_4 degradation rate of more than 95 %. The lowest methane oxidation potential was obserbed in the quite fresh biowaste compost (BWC, medial oxidation rate 0 %) and the mineral subsoil (SS-Zister, medial oxidation rate 4 %). Due to the low degree of maturity of the biowaste compost, other decomposition processes probably suppressed CH_4 oxidation. There was a high CO_2 production resulting from the endogenous respiration of the fresh compost material itself. Also the low organic content in the mineral material was not a beneficial property for methanotrophic activity.

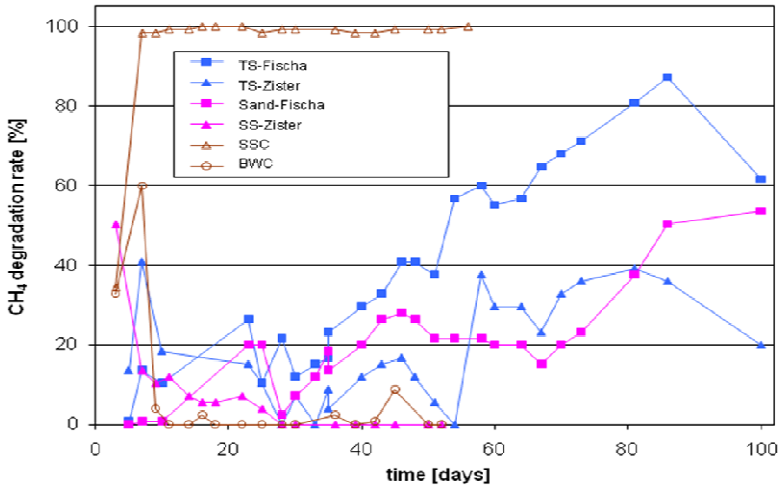


Figure 3.8: Methane degradation rates of tested substrates over 100 days under laboratory conditions (Gamperling et al., 2008)

3.2.2 Air Permeability

In general, the transport mechanism in porous media is controlled by both diffusive and advective forces (see section 2.2). In the context of landfills, the primary driving force for gas transport, especially through cover systems, is pressure gradients induced by natural fluctuations in barometric pressure and gas generation in waste and, thus, advective flux dominate diffusional flux (Stern et al., 2007, De Visscher et al., 2004; Czepiel et al., 2003). The air permeability of a soil, a measure of advective flux, is defined as its ability to conduct air by the mass flow of gas in the ease of response to a pressure gradient. The most important factor that controls the air permeability of soil is the water content and the pore structure of soil or substrate (Hamamoto et al., 2009). Since water content could vary significantly after the biocover system is constructed, it is necessary to determine the air permeability of possible cover materials at various water contents. The air permeability shows generally inverse correlation with water content. Lower water content leads to higher gas permeability whereas higher water content leads to lower gas permeability.

At the Austrian Institute of Technology in Seibersdorf a series of gas permeability tests were performed to investigate the variation of air permeability of the substrates listed in Table 3.4 at four different water contents (25 %, 50 %, 75 % and 100 % of WHC) by flowing air through a soil column. The air permeability test apparatus is shown in Figure 3.9.

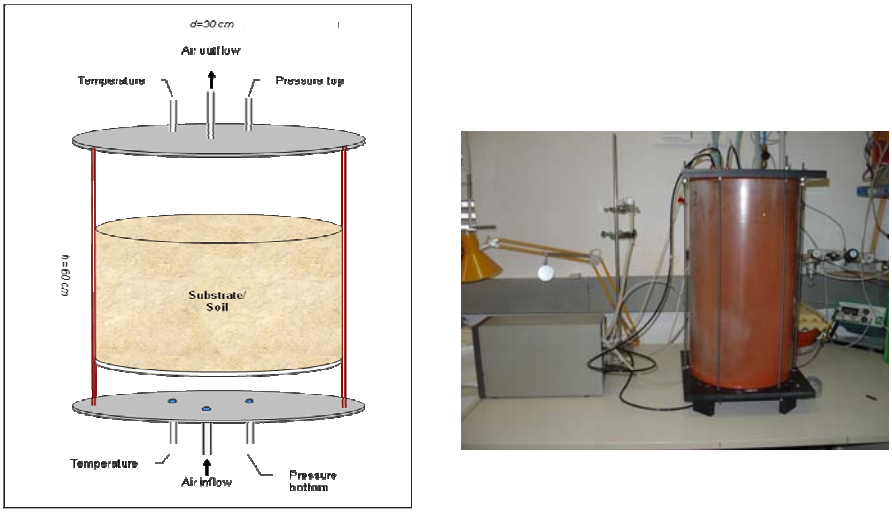


Figure 3.9: Schematic illustration of the air permeability test apparatus (Internal Report Nutzraum, 2008 and AIT Seibersdorf, 2008)

Different air flow rates (10 – 50 ml/min) were applied continuously to the bottom of the column and gas pressure and temperature at the column base and top were monitored during a test run. From this laboratory experiment the intrinsic permeability of the different substrate materials at various water contents was calculated using Darcy`s Law:

$$q_A = \frac{k_{int}}{\eta} * \frac{\Delta p}{\Delta z} \qquad (3.5)$$

q_A	(m³/m²s)	air mass flow
k_{int}	(m²)	intrinsic permeability of the porous media
η	(Pa s)	dynamic air viscosity
$\frac{\Delta p}{\Delta z}$	(Pa/m)	pressure gradient along a flow distance in the soil column

It is assumed that the intrinsic permeability is a function of the properties of the porous material, not the permeating fluid or gas. In general, the substrate used in a biocover should ensure sufficient air permeability even at 100 % WHC. Huber-Humer et al. (2008a) recommended that the intrinsic permeability of the material used as a landfill cover should not go below a value of $5*10^{-13}$ m² at 100 % WHC (equivalent to field capacity). This would mean an air flow rate of 10 l/m²h at a pressure gradient of 1 hPa at a soil depth of 1 m. Figure 3.10 shows the pressure at the bottom and top of the column of SSC-Mix at 34 % WHC with varying air flow rate.

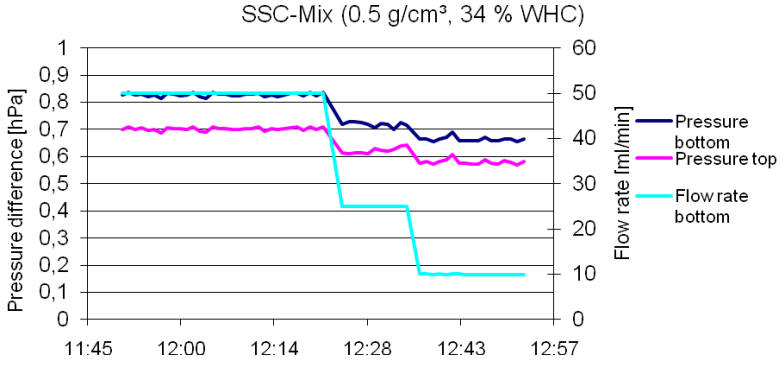


Figure 3.10: Pressure curve at the base and top of the column with varying flow rate (based on data from AIT Seibersdorf)

It could be shown that all tested substrates had sufficient intrinsic permeability up to 100 % WHC. Only sand (Sand-Fischa) showed limited air permeability (k_{int} = 3.5 *10^{-13} m²) at 75 % and 100 % WHC, with pressure differences up to 90 Pa between base and top of the soil column.

3.3 Lysimeter experiment

A lysimeter experiment was set up at the Austrian Institute of Technology in Seibersdorf, Austria in February 2008 to investigate three biocovers using different substrates as well as one control cover regarding water infiltration and methane emission mitigation in combination with in-situ aeration. However, this diploma thesis presents only data concerning the relative emission reduction effect of the passive biocovers. Therefore a monitoring program has been developed to verify biocover performances including

- monitoring of qualitative (gas and temperature profiles, surface FID-screening) and quantitative (stable isotope analysis) indications for in situ methane oxidation in the biocovers and
- measurement of remaining CH_4 and CO_2 surface emissions using a closed dynamic chamber in order to see the relative difference in landfill gas flux through the different ground surfaces.

The measurement campaign was conducted from August 2009 to April 2010.

3.3.1 Lysimeter setup

In a lysimeter facility four chambers (A, B, C and D), each measuring 2 m x 2 m x 3 m (width x length x depth), are filled with different substrates according to Table 3.5 and Figure 3.12. As can be seen in Figure 3.11, the lysimeter facility is a reinforced concrete structure with a basement level 1.2 m below the surroundings level and it is thermally shielded from the outside environment by a mound.

Figure 3.11: Lysimeter facility at the Austrian Institute of Technology in Seibersdorf, Austria

Chamber A, B and C included a 1 m thick substrate layer, whereas the control chamber (D) consisted of a 0.5 m layer. All substrate layers were underlain by a 0.2 m gas distribution layer consisting of 10/40 mm coarse gravel. A 0.2 m drainage layer was installed at the lowest point to evacuate infiltrating waters. Only the control cover included a 0.7 m drainage system to compensate for the smaller substrate layer.

Each of the lysimeter chambers were fed directly by biogas coming from fresh municipal solid waste, sieved with a 80 mm sieve. The municipal solid waste was

provided by a mechanical biological treatment plant ("Wiener Neustädter Stadtwerke und Kommunal Service GmbH") in Steinabrückl, Lower Austria, which was only mechanically pretreated. As a result, it was not possible to control the upward flux of biogas. Table 3.6 shows the solid matter and eluate analyses of the buried waste mass. A perforated pipe was installed in the middle of each chamber to aerate the waste body for rapid waste stabilisation (forced in-situ aeration) (see Figure 3.12 and Figure 3.17) as soon as the gas generation is stable and the evaluation of the passive biocovers is completed.

Table 3.5: Designs of the lysimeter chambers

Chamber A **"classic methane oxidation layer"**	**Chamber B** **"Sand-compost mixture"**
100 cm sewage sludge compost 20 cm coarse gravel (gas distribution layer) 150 cm municipal solid waste 20 cm coarse gravel (drainage layer)	2 cm topsoil Fischamend 48 cm sand 50 cm sand-compost mixture (60:40 % v/v) 20 cm coarse gravel (gas distribution layer) 150 cm municipal solid waste 20 cm coarse gravel (drainage layer)
Chamber C **"evapotranspiration cover"**	**Chamber D** **"control cover"**
50 cm topsoil Fischamend 50 cm subsoil Zisterdorf 20 cm coarse gravel (gas distribution layer) 150 cm municipal solid waste 20 cm coarse gravel (drainage layer)	10 cm topsoil Fischamend 40 cm subsoil Zisterdorf 20 cm coarse gravel (gas distribution layer) 150 cm municipal solid waste 70 cm coarse gravel (drainage layer)

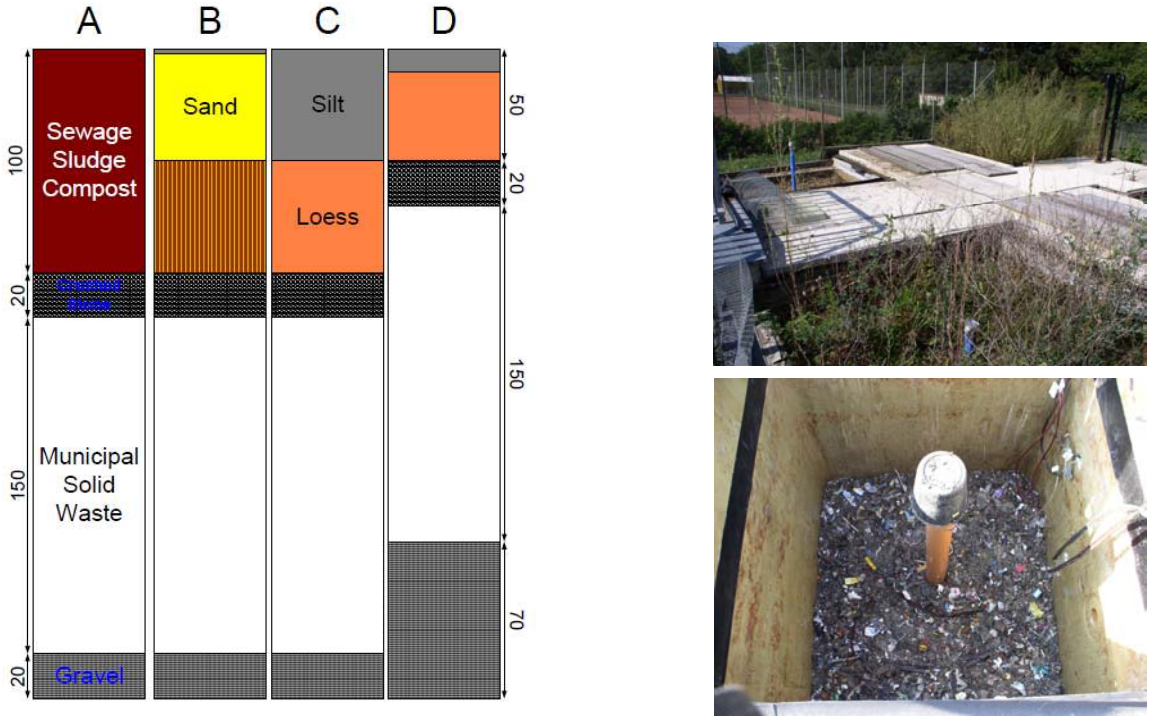


Figure 3.12: Lateral cross-section of the installed lysimeters (2 m x 2 m x 3 m) and view of the top of the lysimeter facility (AIT Seibersdorf 2008)

Table 3.6: Data of the municipal solid waste at the beginning of the lysimeter experiment (a – solid matter analyses, b – eluate analyses); RA = respiration activity; DM = dry matter; LOI = loss on ignition at 550 °C (organic content); TOC = total organic carbon; TC = total carbon; GS = potential of gas formation; COD = chemical oxygen demand; BOD = biochemical oxygen demand

a) Parameter	Unit	Charge 1	Charge 2	Method
LOI	% DM	48.1	48.5	ÖNORM EN 15169
TC	% DM	28.3	28.1	ÖNORM EN 13137
TOC	% DM	27.3	27.5	ÖNORM EN 13137
N_{total} (Dumas)	% DM	0.98	1.13	ÖNORM EN 13654
C/N	-	28	24	
TOC/LOI	-	0.57	0.57	
RA_4	mg O_2/g DM	48.7	68.8	ÖNORM S 2027-1
GS_{21}	Nl/kg DM	40.8	39.2	ÖNORM S 2027-2

b) Parameter	Unit	Charge 1 and 2	Method
pH	-	6.8	ÖNORM M 6244
Conductivity	mS/cm	6.7	ÖNORM EN 27888
NH_4^-N	mg/kg DM	429	ÖNORM EN ISO 11732
NO_3^-N	mg/kg DM	2	ÖNORM EN ISO 10304-2
COD	mg O_2/kg DM	61,071	ÖNORM M 6265
BOD	mg O_2/kg DM	30,400	ÖNORM EN 1899-2

3.3.2 Substrates used in the landfill lysimeters

In order to characterise the various substrates, standard physical and chemical parameters were analysed and soil grain analyses were performed.

The standard parameters were carried out in full agreement with the “Austrian Ordinance on Composting” (BGBl. II 292/2001). The water holding capacity was determined according to an institute specific technique described in Huber-Humer (2004).

Particle size distributions were analysed with two different methods. Sand and gravel fractions of the substrates were measured according to the sieving method of DIN ISO 3310 (1992), whereas the smaller fractions (sand, silt and clay) were measured by the conventional pipette method (ÖNORM L 1061). The sand fractions obtained by these methods are not directly comparable, as a dispersion agent was used only for the pipette method.

The following table presents some relevant substrate characteristics (see Table 3.7) of the cover materials placed in the landfill lysimeters (see section 3.3.1). Not surprisingly, the organic content (loss on ignition) of the compost materials was quite high compared to the natural soils. In addition, the sewage sludge compost was initially mature which was confirmed by the low respiration activity of 0.8 mg O_2/g DM. Importantly, low ammonium concentrations were present in all cover materials avoiding possible inhibition effects. The nitrate concentration was remarkably higher in the compost materials than in the mineral soil, revealing that the compost materials provide microorganisms with adequate nutrient supply. The mineral materials (TS-Fischa, Sand-Fischa, SS-Zister) exhibited a similar total pore volume (42.8 – 48.7 % v/v), whereas the sand-compost mixture (SSC-Mix) offers a higher porosity. Consequently, both compost materials (SSC and SSC-Mix) have a higher water-holding capacity.

Table 3.7: Data of the cover materials at the beginning of the lysimeter experiment; WHC = water holding capacity; RA = respiration activity; DM = dry matter; n.d. = not detectable; LOI = loss on ignition at 550 ℃ (organic content); TOC = total o rganic carbon, TC = total carbon

Parameter	Unit	SSC	SSC-Mix	TS-Fischa	Sand-Fischa	SS-Zister
Moisture content	% on a dry weight basis	64.7	21.9	21.8	7.7	18.5
WHC	% DM	117.1	43.2	41	28	37
Bulk density	kg/l	0.6	1.1	1.4	1.5	1.3
Total pore volume	% v/v	72.8	55.8	47.9	44.5	52.8
RA_4	mg O_2/g DM	0.7	0.1	-	-	-
LOI	% DM	24.9	4.0	5.2	1.1	2.9
TOC	% DM	11.8	1.5	1.5	0.1	0.4
TC	% DM	13.6	3.0	3.3	1.6	2.2
N_{total} (Dumas)	% DM	1.5	0.6	0.1	n.d.	n.d
$CaCO_3$	% DM	14.4	12.2	14.5	12.4	14.9
pH	-	6.9	7.1	8.0	8.0	8.3
Conductivity	mS/cm	2.8	0.5	0.1	0.1	0.1
NO_3^-N	mg/kg DM	1750	200	31	3	n.d.
NH_4^-N	mg/kg DM	n.d.	n.d	2	2	2

Regarding the particle size distribution, both soils, TS-Fischa and SS-Zister (Figure 3.14 and Figure 3.16), show quite similar curves with a higher content of finer fraction (< 2 mm), whereas the compost material SSC exhibits a balanced grain size distribution (approx. 50 % <2 mm and 50 % >2 mm) (Figure 3.13). Compared to the pure compost material SSC, the SSC-Mix exhibits a particle size distribution with a higher amount of sand-like fractions (< 2 mm) (Figure 3.13).

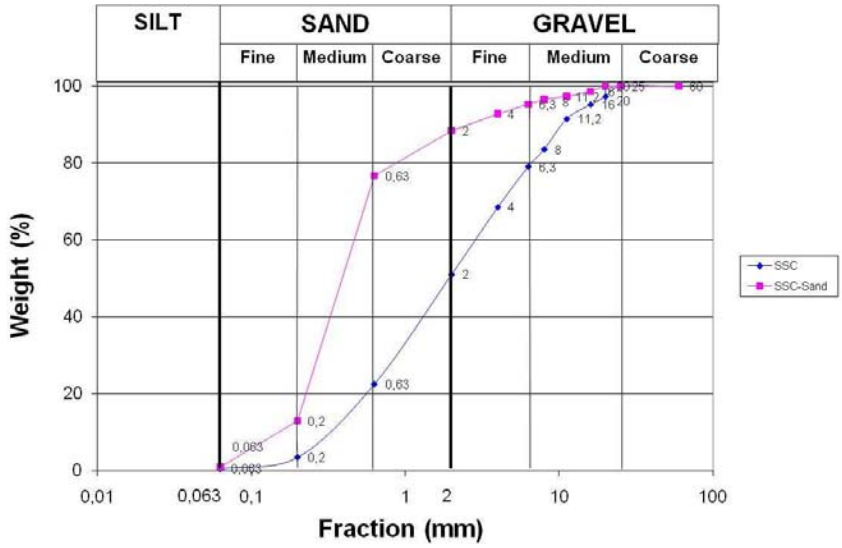


Figure 3.13: Particle size distribution of SSC and SSC-Mix according to DIN ISO 3310

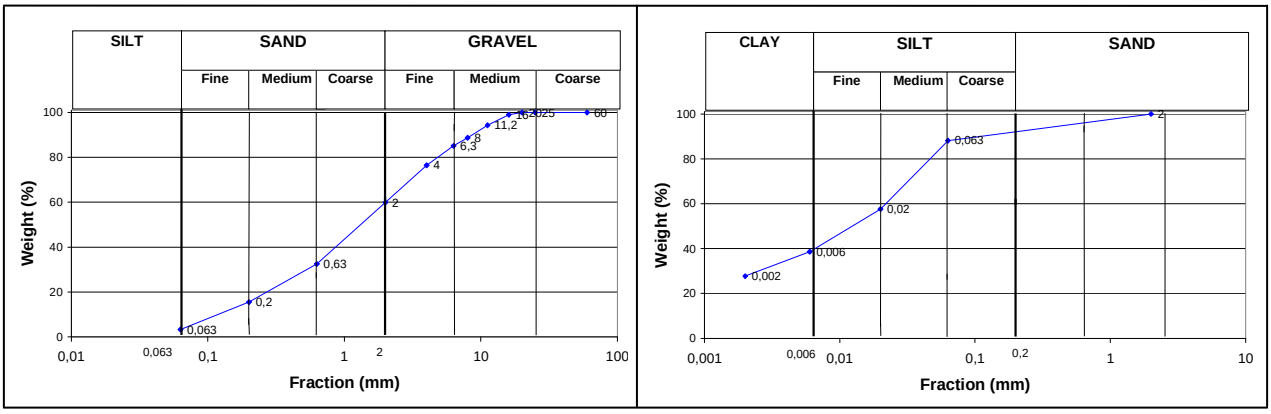


Figure 3.14: Particle size distributions of the topsoil Fischamend (TS-Fischa) according to DIN ISO 3310 (left side) and ÖNORM L 1061 (right side)

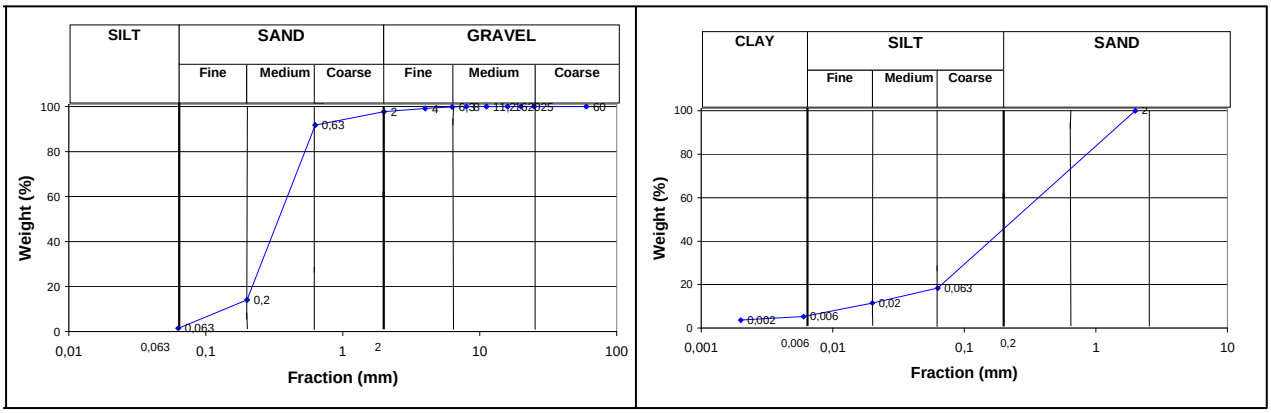


Figure 3.15: Particle size distributions of the sand Fischamend according to DIN ISO 3310 (left side) and ÖNORM L 1061 (right side)

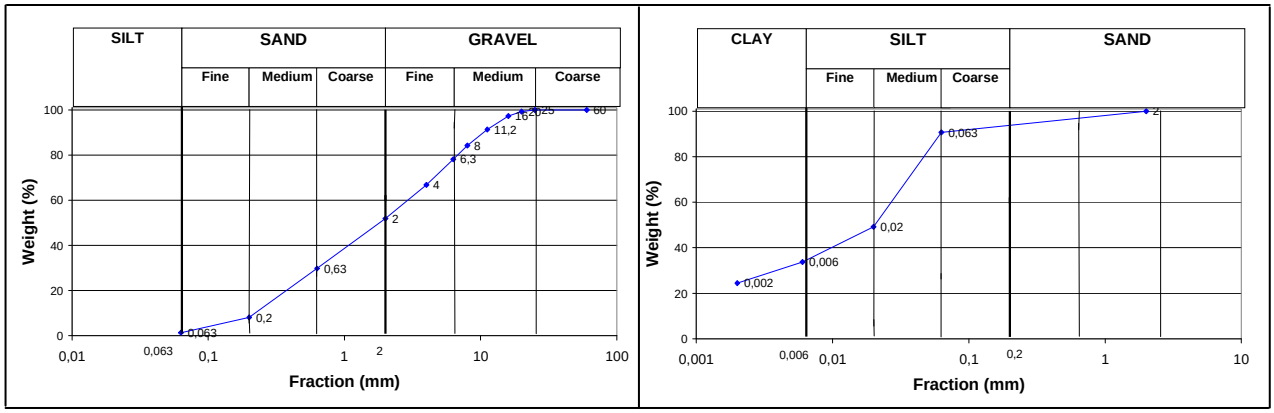


Figure 3.16: Particle size distributions of the subsoil Zistersdorf (SS-Zister) according to DIN ISO 3310 (left side) and ÖNORM L 1061 (right side)

3.3.3 Measurement of pore gas composition, temperature and water content

Gas probes, consisting of an open polyethylene (PE) pipe with an inner diameter of 5 mm, were installed permanently in six depths (25, 50, 75, 110, 160, 210 cm) of the lysimeter chambers A, B and C. The top end of the pipes was closed with rubber stoppers. Temperature sensors and water content sensors, connected to a data acquisition system and data loggers, were installed at four separate depths (25, 50, 75, 190 cm) and three depths (25, 75, 190 cm), respectively. Figure 3.17 depicts the measuring probes installed in the chambers A, B and C, making up for a total of 8 gas probes, 8 temperature sensors and 6 water content sensors for each chamber.

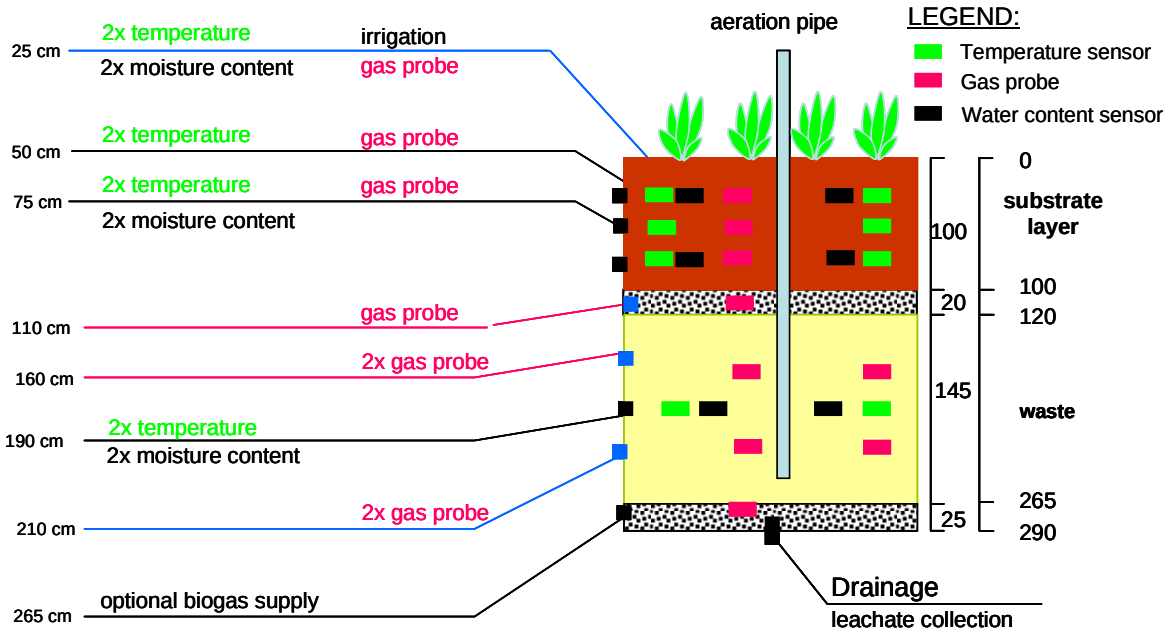


Figure 3.17: Design of landfill lysimeter and measuring probes installed in chambers A, B and C (Internal Report Nutzraum, 2008)

As lysimeter chamber D had a less thick substrate layer (0.5 m), the location of the measuring probes differed slightly from those installed in chambers A, B and C. As illustrated in Figure 3.18, in total 6 gas probes, 6 temperature sensors and 6 water content sensors were installed in chamber D.

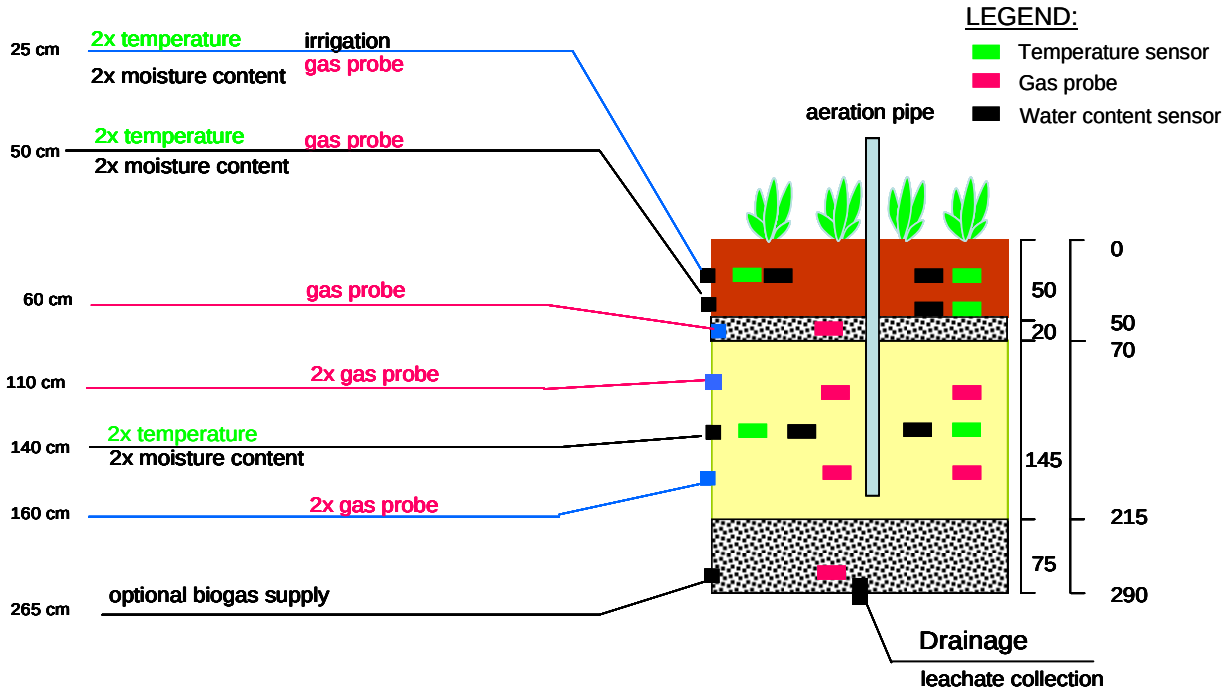


Figure 3.18: Design of landfill lysimeter and measuring probes installed at chamber D (Internal Report Nutzraum, 2008)

A sharp decline in O_2 and CH_4 concentrations with depth in the soil profile provides a qualitative indication of the start of microbial methane oxidation (Huber-Humer et al., 2009b). Also the temperature profile may be an indicator of the location of the

oxidation layer. From August 2009 to April 2010 measurements of gas concentration profiles were carried out on a weekly basis. The volumetric concentrations of CH_4, CO_2 and O_2 were measured using a mobile gas analyser (Multigas analyser LMSx) equipped with an infrared methane and carbon dioxide detector (precision of measurement: ±0.1 % in the range of 0–10 vol.%, and ±1 % in the range of 10–80 vol.%) and an electrochemical sensor able to detect O_2 (precision of measurement: ±0.5 % in the range of 0–25 vol. %). Nitrogen gas is a good indicator for the oxidation level as it is neither consumed by oxidation near the surface nor by soil respiration. In order to draw concentration profiles, N_2 was calculated as the difference between 100 % and the sum of the CH_4, CO_2 and O_2 concentrations for each depth within the soil profiles. Temperature and water contents within the chambers were measured by the employees of the Austrian Institute of Technology in Seibersdorf (Internal Report Nutzraum, 2008).

3.3.4 Measurement of gas fluxes

In order to assess biocover performance and functionality a series of surface emission measurements was made in December 2009 as well as in March and April 2010. Grass and other plants developed on the lysimeter chambers were cut away to ensure minimum disturbance. Measurement of the CH_4 and CO_2 concentrations inside the closed dynamic chamber (see Figure 3.19) was done using a portable Flame Ionisation Detector (Thermo Scientific FID, TVA-100; (precision of measurement: ±25 % of reading or ±2.5 ppm in the range of 1.0 to 10,000 ppm) and a portable carbon dioxide detector (Vaisala CARBOCAP™ GM70; precision of measurement: ±20 ppm + 2 % of reading), respectively. At the time of measurement, the chamber was placed on the ground of the lysimeter chamber, and the air from the chamber was pumped to the CH_4 and CO_2 analyser by the external pump. Gas composition in the chamber was analysed immediately after installation. The duration of each measurement was approximately 6 – 9 minutes, depending on the definiteness of the increase. As already described in chapter 3, the linear increase in CH_4 and CO_2 concentrations (R^2>0.9) in the chamber over time was used to calculate the LFG emission rates. Correlations less than R^2=0.7 were treated as zero fluxes. For each lysimeter chamber, three reliable measurements were made and then a mean flux rate and standard deviation were calculated. Before start of measurement, the Flame-Ionisation Detector was calibrated with a 1-point calibration using standard test gas (Linde, 500 ppm CH_4 in N_2) at the temperature and humidity of the calibration. In addition, weather conditions, in particular wind speed, barometric pressure and air temperature, were monitored.

Figure 3.19: Developed chamber placed on the lysimeter

3.3.5 Measurement of gas emissions (FID-measurement)

Repeated screenings of the surface methane concentrations represent a qualitative survey of the overall CH_4 emissions pattern from the different biocovers. Surface methane concentrations were scanned once a week (or every second week) using a portable FID (Thermo Scientific FID, TVA-100 Toxic Vapour Analyser) according to a predefined grid. These investigations provide a qualitative check of the biocover functionality under specific environmental and weather conditions. The FID was calibrated before start of measurement and weather conditions were monitored. As can be seen in Figure 3.20, a funnel-shaped probe was placed directly on the cover surface and the emitted methane was pumped to the FID via an internal pump.

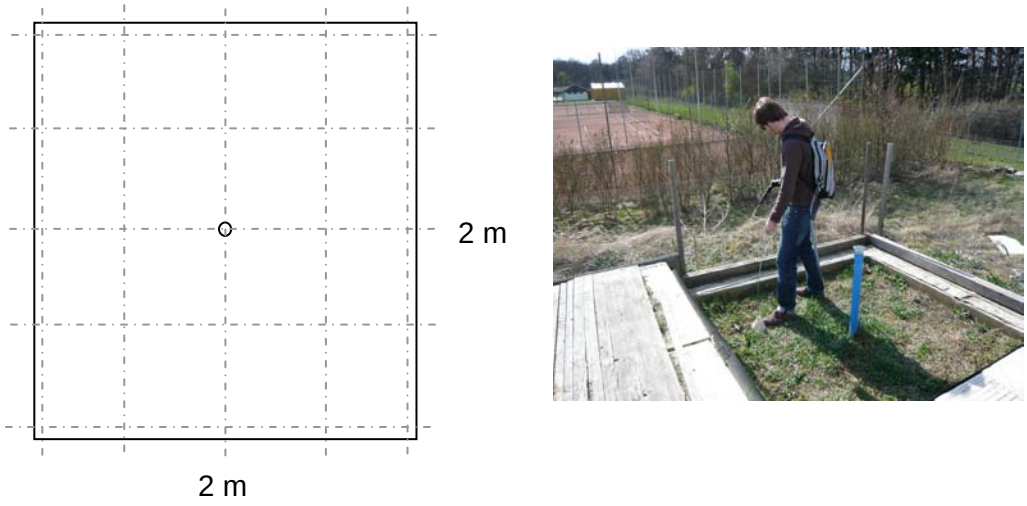


Figure 3.20: Screening of the surface emission concentration on a 50 x 50 cm grid

3.3.6 Stable Isotope Analyses

Background

The stable isotope technique is a promising field method for quantifying methane oxidation efficiency in landfill cover materials based on changes in the ratio of two stable carbon isotopes, namely ^{13}C and ^{12}C, which comprise 1 % and 99 % of carbon atoms, respectively. Methanotrophic bacteria preferentially oxidise the lighter isotope ^{12}C over ^{13}C in methane, leaving residual methane enriched in the heavier isotope ^{13}C (Barker and Fritz, 1981; De Visscher et al., 2004). As a result, a shift in the stable carbon isotope composition occur when methane is oxidised, changing the isotope ratio or fractionation. During the passage of landfill gas from anaerobic zones through zones of oxidation, methane becomes heavier and carbon dioxide becomes lighter. The carbon stable isotope composition is defined as follows:

$$\delta^{13}C_A‰ = \left(\frac{R_{sample}}{R_{st}} - 1\right) * 1000 \qquad (3.6)$$

with R_{sample} the $^{13}C/^{12}C$ ratio of the sample, and R_{st} the $^{13}C/^{12}C$ ratio of the reference standard VPDB (Vienna Peedee Belemnite; R_{st}=0.01124).

Typically, microbial CH_4 is produced at values between -50 ‰ and -60 ‰ while CH_4 actually emitted has ^{13}C-enriched values of -30 to -50 ‰ following oxidation (Chanton et al., 1999). Barker and Fritz (1981) found by experiment that CO_2 produced by methanotrophic bacteria was enriched in ^{12}C by 5.0–29.6 ‰, relative to the residual

methane. The negative δ value indicates a ^{13}C depletion relative to the carbonate standard whereas more positive values indicate a $\delta^{13}C$ enrichment.

Carbon isotope fractionation has been widely used to obtain in situ estimates of methane oxidation in cover materials by measuring the change in the isotope signature of methane produced inside the landfill and methane being emitted from the cover surface or in the upper part of soil profile (Chanton et al., 2008; De Visscher et al., 2004; Chanton and Liptay, 2000). Recently, isotope fractionation has been employed to quantify methane oxidation combined with flux chamber measurements (Abichou et al., 2006; Barlaz et al., 2004; Chanton and Liptay, 2000; Börjesson et al., 2001) but also with tracer techniques (comparison between $\delta^{13}C$ of atmospheric CH4 in an upwind transect to a downwind transect) (Börjesson et al., 2007; Chanton et al., 1999). The latter was used to measure the average methane oxidation rates of an entire landfill.

Knowledge of the $\delta^{13}C$ of emitted CH_4 compared to the $\delta^{13}C$ of unoxidised CH_4 in the anaerobic zone (determined from soil probe data) allows calculation of the fraction of methane that is oxidised during transport through the landfill cover materials. The percentage of oxidised CH4 (f_{ox}) can be determined according to two approaches. The first approach is based on the assumption that CH_4 moves as a closed system between source and sampling point and does not mix with other CH_4 ("simplified Rayleigh approach") using the following equation (Mahieu et al., 2006):

$$f_{oxc,\,z} = 1 - \left(\frac{\delta_z + 1000}{\delta_{anox} + 1000} \right)^{\frac{\alpha_{ox}}{1-\alpha_{ox}}} \tag{3.7}$$

where $f_{oxc,z}$ is the fraction oxidised at depth z, δ_z and δ_{anox} are standard $\delta^{13}C$ isotope ratios for sampling depth z or emitted CH_4 of the anoxic zone, and α_{ox} is the isotopic fractionation factor for microbial oxidation. The second approach assumes that CH4 in the porous media is well mixed ("open system approach"). The fraction oxidised ($f_{oxo,z}$) is calculated with the equation according to Chanton et al. (1999):

$$f_{oxo,\,z} = 1 - \left(\frac{\delta_z - \delta_{anox}}{1000(\alpha_{ox} - \alpha_{trans})} \right) \tag{3.8}$$

where α_{trans} is the isotope fraction factor from transport (α_{trans} = 1 for purely advective transport and α_{trans} = 1.014 where diffusion is important) (De Visscher et al., 2004). The fractionation factor for microbial oxidation (α_{ox}), an estimate of the preference of the methanotrophs for the lighter isotope, can be obtained empirically by controlled field or laboratory incubations of cover materials with known CH_4 headspace concentrations in order to derive site- and time-specific fractionation factors (De Visscher et al., 2004). Different values of α_{ox} have been reported in the literature, some of which are presented in Table 3.8.

However, to date the main disadvantage of the stable isotope analysis is the variations in methane stable isotope ratios due to an extremely specific fractionation influenced by the individual characteristics of methanotrophic bacteria and growth condition (Nozhevnikova et al., 2003 cited in Huber-Humer et al., 2009) and fractionation processes during gas transport (De Visscher et al., 2004). Consequently, the fractionation factor must be determined specifically for each landfill site, a process which is very laborious and costly and may constrain practical applicability (Huber-Humer et al., 2009). Stable isotope analysis is not applicable in situations where flux chambers yield negative fluxes, indicating that landfill covers

consume rather than emit methane (Scheutz et al., 2009). According to De Visscher et al. (2004) previous application of the conservative stable isotope analysis may have led to an underestimation of CH_4 oxidation. Also CH_4 flow through macro pores may contribute to underestimations as more emitted methane is sampled at the cover surface whereas completely oxidised methane is not represented (Huber-Humer and Lechner, 2007).

Present study

Gas samples for stable isotope analysis were taken at selected dates (21.10.2009 and 09.12.2009) to study methane oxidation efficiency of the biocovers. Samples were taken from each depth of the soil profile including waste body, gas distribution layer and substrate layer. Samples were extracted through a gas sampling tube (100 and 150 ml) and directly measured with a mobile gas analyser detecting CH_4, CO_2 and O_2. For the last date gas samples were also taken during surface flux measurements.

Isotope analyses were performed at the Austrian Institute of Technology in Seibersdorf. The gas chromatography combustion isotope ratio mass spectrometry system (GC/C/IRMS) consisted of a HP5859 Series II gas chromatograph (GC) coupled with a isotopic mass spectrometer (Finnigan MAT Delta S) via a combustion interface (Finnigan MAT combustion interface). The GC column was a CP PoraPlot Q fused silica column (25 m, 0.32 mm). The used gas standards had a precision and accuracy better than <0.05 ‰.

The isotopic fractionation factor for bacterial oxidation (α_{ox}) was determined according to literature values, as illustrated in Table 3.8. Only those values in the temperature range found during gas sampling dates were considered.

Table 3.8: Literature values of α_{ox} with associated soil temperature (adapted from Cabral et al., 2009)

α_{ox}	Soil temperature [℃]	Reference
1.0140	25	Liptay et al. 1998
1.0220	25	Liptay et al. 1998
1.0300	25	Liptay et al. 1998
1.0235	22	Chanton et al. (1999)
1.0288	9	Chanton et al. (1999)
1.0307	5	Chanton et al. (1999)
1.0311	4	Chanton et al. (1999)
1.0315	3	Chanton et al. (1999)
1.0304	6	Chanton et al. (1999)
1.0240	20	Chanton et al. (1999)
1.0266	15	Chanton et al. (1999)
1.0269	13	Chanton et al. (1999)
1.0240	21	Chanton et al. (1999)
1.0241	20	Chanton et al. (1999)
1.0316	3	Chanton et al. (1999)
1.0244	-	Chanton et al. (2008)
1.0250	35	Chanton and Liptay (2000)
1.0490	8	Chanton and Liptay (2000)

Previous studies have demonstrated that the fractionation factor depends on soil temperature modifying the calculated methane oxidation efficiency (Chanton and Liptay, 2000; Chanton et al. 1999). As a consequence, the fractionation factor was

corrected for temperature (see Table 4.3) based on Tyler et al.'s (1994; cited in Cabral et al., 2009) temperature dependence relationship:

$$\alpha_{ox} = \alpha_{ox}average + 0.00046(20 - T[°C]) \quad (3.9)$$

Due to the high variability of the fractionation factor (α_{ox}), a sensitivity analysis for the calculation of f_{oxc} and f_{oxo} was carried out relative to changes in α_{ox}. The oxidised fractions (f_{oxc} and f_{oxo}) were calculated for closed and open systems using equations 3.7 and 3.8, respectively. The isotope fraction factor from transport (α_{trans}) was set to be 1 for purely advective systems (see section 4.4).

3.3.7 Measurement of meteorological data

Meteorological data, including air temperature, precipitation, atmospheric pressure and wind speed were continuously recorded by a weather station of the Central Institute for Meteorology and Geodynamics installed near the lysimeter facility.

4 RESULTS AND DISCUSSIONS

Each of the lysimeters was fed directly by biogas coming from fresh municipal solid waste (see section 3.3.1) and, thus, the magnitude of the CH_4 loading could not be controlled. Methane production in the respective solid waste layers did not start until summer 2009, although the lysimeter experiment was set up in February 2008. In order to activate and intensify the LFG production, the waste layers were irrigated additionally in summer and autumn 2009. Following the irrigation, the CH_4 concentration increased slightly in all waste layers and reached a peak in lysimeters A, B and D in autumn 2009. Though all the four lysimeters contained the same type of waste, methane development in the solid waste layers of lysimeters A and B did not indicate the same range as other two. The gas composition by volume in the waste layer of lysimeters C and D was close to or above 50 vol.% for CO_2 and CH_4 except for lysimeters A and B (30 vol.% and 8 vol.% CH_4). In winter 2009/2010 CH_4 concentration started to decrease in all waste layers, most probably due to the low temperatures. However, CH_4 concentration increased again in spring 2010. As can be seen in Figure 4.1, in spring 2010 CH_4 concentration in the waste layers of lysimeters C and D ranged between 40 vol.% and 50 vol.% whereas it was still low in the lysimeters A and B (15 vol.% and 2 vol.% CH_4) indicating a low LFG production rate.

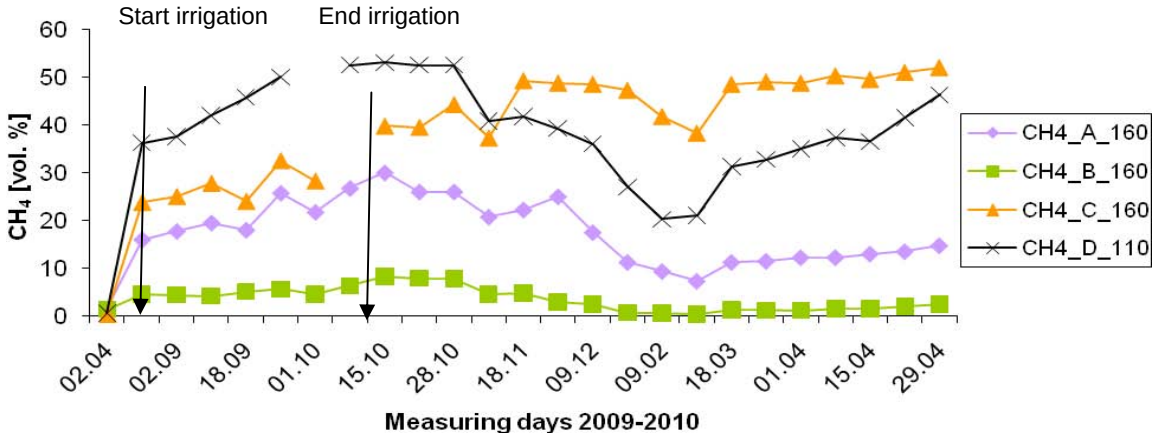


Figure 4.1: Methane development in the solid waste layers of the lysimeters A - C (at 160 cm depth) and D (at 110 cm depth)

4.1 Pore gas composition, temperature and water content

The gas concentration profiles of the lysimeters were determined as described above. During the monitoring period none of the solid waste layers reached a stable methane production phase, except for lysimeter C. Because of these circumstances, a systematic comparison of the different biocovers concerning their relative emission reduction effect was challenging. In general, the interpretation of gas concentration profiles in biocovers is quite difficult as gas composition in the landfill covers is controlled by numerous microbial processes (e.g. uptake and release of gases), bio-chemical processes (e.g. CO_2 fixation) and physical conditions as well as meteorological processes (see section 2.2). Although gas concentration profiles represent a snapshot of a single measuring event, they can also give a comprehensive insight into the zoning of processes taking place in the cover layers, particularly when combined with other data. As a first step those gas concentration

profiles were compared, where the concentration of CH_4 in the waste layers showed its highest value. The highest concentration of CH_4 in the waste body of lysimeters A, B, and D was observed in the middle of October 2009, while the CH_4 level in the waste layer of lysimeter C reached a peak at the end of April 2010. Figure 4.2 presents the gas composition (CH_4, CO_2, O_2 and N_2) and the temperature along the solid waste layers and the top covers of all lysimeters at selected dates. As already mentioned, the design of lysimetesr C and D differed only in the thickness of the substrate layer. Although the gas and temperature profiles of lysimeters C and D are shown at different dates, the similar LFG composition at these dates allows a better comparison. The volumetric moisture contents in the upper part of the substrate layers were also in the same range (17 – 21 vol.% based on dry weight). The location of the methane oxidation zone inside the four biocovers was mainly indicated by a sharp decline in methane and oxygen.

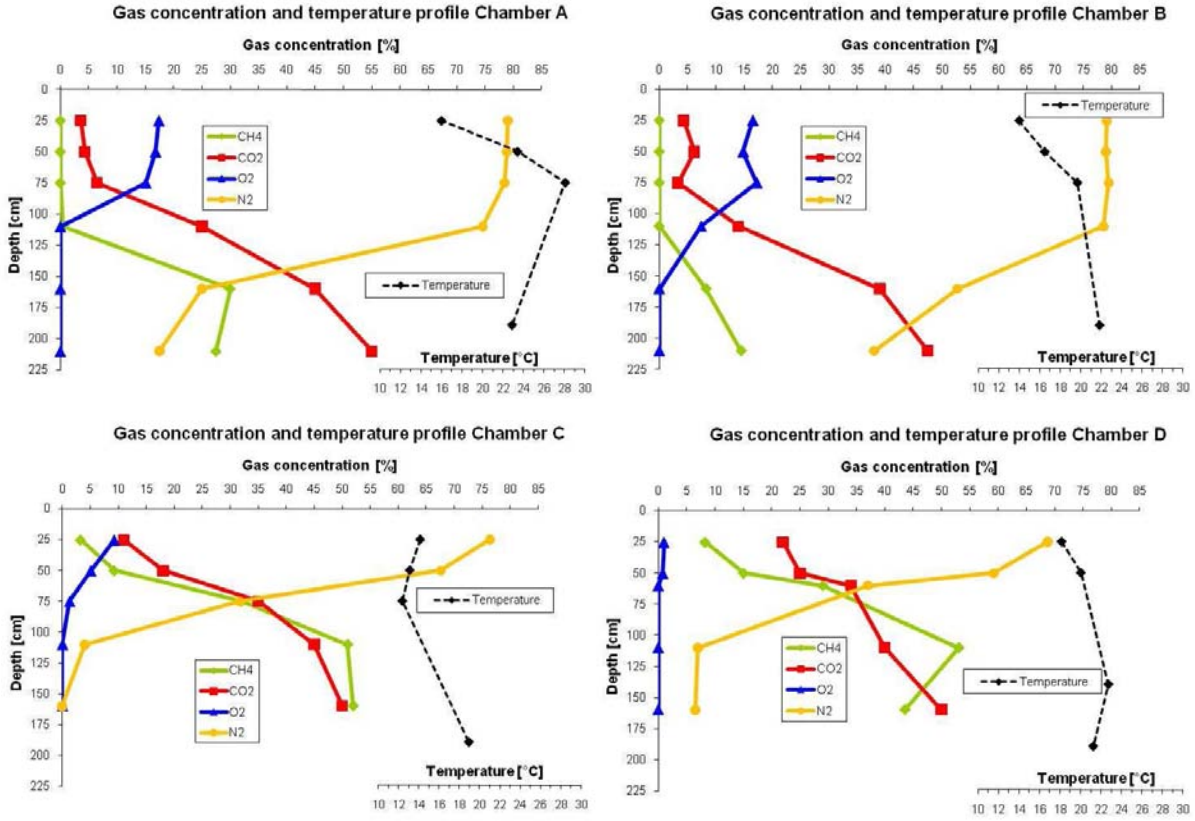


Figure 4.2: Typical soil gas concentration and temperature profiles of all lysimeters (A, B, D: 15.10.2009, C: 29.04.2010)

Lysimeter A

The gas composition in the solid waste layer of lysimeter A contained approx. 30 % CH_4, 55 % CO_2 and 15 % N_2 (v/v). O_2 has never been detected in the waste layer. The shift in CH_4 and CO_2 concentrations between 110 and 160 cm indicates that all the methane was reduced in the waste layer and the gas distribution layer beneath the compost layer. This finding was supported by the stable isotope analyses (see section 4.4). In addition, the peak in the temperature profile also indicates CH_4 uptake as well as the location of the oxidation zone (note that between 75 and 190 cm no temperature sensors were installed). There was no CH_4 detectable in the substrate layer as well as on the surface of the lysimeter (see appendix A.2). Similar observations have been made by Cabral et al. (2009) and Scheutz et al. (2009b), where some oxidation occurred within the gas distribution layer. A biofilter

experiment by Gebert and Gröngröft (2006b) also showed high CH_4 oxidation rates in coarse, purely mineral material. In the present case, it is assumed that nutrients of the sewage sludge compost might have been washed into the gas distribution layer by precipitation, promoting the methane oxidation processes in this zone. The high temperature insulation capacity of the compost material (see Figure 4.3) might have additionally supported the activity of methane oxidising bacteria within the gas distribution layer. In addition, the coarse compost substrate enabled a good aeration down to the gas distribution layer as indicated by the uniformly high N_2 concentration through the base of the cover. In contrast to oxygen, N_2 might be a better indicator of air penetration as it is neither used for CH_4 oxidation near the surface nor by soil respiration. However, also the CH_4 loading might have been quite low allowing for a large diffusive intrusion of atmospheric air.

Lysimeter B

The gas profile of lysimeter B revealed a similar pattern, but with a lower level of CH_4 concentration in the solid waste layer (14.5 vol.% CH_4, 47.5 vol.% CO_2, 38 vol.% N_2). The methane depletion also occurred at a depth of 110 to 160 cm, where the CH_4 concentration decreased to zero (see also stable isotope analyses, section 4.4). There was no CH_4 detectable in the substrate layer up to 110 cm. The nearly vertical N_2 concentration profile clearly indicates that the substrate (SSC-Mix) was well aerated throughout the 1 m cover depth. The O_2 peak at a depth of 75 cm may have resulted from a leakage of the gas probe. In the SSC-Mix layer, O_2 penetrated deeper into the cover layer than in the SSC, most probably due to the lower CH_4 loading.

Lysimeter C

The gas concentrations in the solid waste layer of lysimeter C showed a typical landfill gas composition during the methanogenic phase (52 vol.% CH_4, 45 vol.% CO_2). A steady decrease in CH_4 concentration was observed throughout the substrate cover (TS-Fischa, SS-Zister). The concentration of CH_4 at a depth of 25 cm was quite low (<1 %). However, dilution of the pore gas also contributes to the decrease in CH_4 concentration. CH_4 concentrations indicate that the main part of the oxidation process has taken place at a depth between 75 cm and 50 cm from the surface since about half of the inlet concentration is reduced in this layer. The concentrations of O_2 and N_2 confirm that the soil was well aerated. Oxygen could be detected till a depth of 50 cm and deeper and N_2 reached down to the gas distribution layer. On the basis of the emission measurements (FID-surface screenings and surface flux measurement), the substrate cover was not efficient in oxidising the entire CH_4 loading. Although, the substrate cover was well aerated throughout the upper part of the cover, preferential flows along the edges of the lysimeter chamber have been observed (see section 4.2). Despite the observed CH_4 emissions, a significant portion of CH_4 was oxidised according to stable isotope analysis (see section 4.4). When the LFG in the waste layer showed a similar composition to those in lysimeter A (approx. 30 vol.% CH_4, 55 vol.% CO_2 and 15 vol.% N_2) in August/Sept. 2009, no CH_4 emissions could be detected at the surface by FID screening (see appendix A.2). These results indicate that the biocover system does have an ability to reduce the methane emission to a certain extent. It is currently discussed in the literature that, in addition to climate and soil type, the magnitude of methane flux into the cover material has a significant influence on oxidation. For design purposes it is not only useful to quantify the maximum methane oxidation capacity of a cover

system under specific climatic conditions but also to know the flux at which the cover can oxidise 100 % of the methane coming from below (Abichou et al., 2010).

Lysimeter D

The LFG composition in the waste layer of lysimeter D consisted of approx. 53 % CH_4 and 45 % CO_2 (v/v). The gas profile shows a sharp decline in CH_4 almost throughout the substrate (TS-Fischa, SS-Zister), although the concentrations of CH_4 at a depth of 25 cm were still not negligible (approx. 8 vol.%). It is possible, that further oxidation may have taken place above a depth of 25 cm. However, low oxygen concentration as well as a sharp decrease in N_2 near the surface indicates restriction of aeration. In contrast, from December 2009 onwards higher O_2 concentration was measured in the upper part of the substrate cover.

According to the emission measurements (FID-surface screenings, see appendix A.2A.1A.2), the control cover showed a poor CH_4 oxidation efficiency, as high CH_4 concentrations (> 1000 ppm) could be detected along the edges of the lysimeter. The substrate cover was not even efficient in oxidising the entire CH_4 loading, when the gas composition was similar to those in lysimeter A (approx. 37 vol.% CH_4, 55 vol.% CO_2 and 8 vol.% N_2) (see appendix A.2 on 27.08.2009). On the other hand, a considerable percentage of CH_4 was oxidised based on the stable isotope analyses (see section 4.4). However, the percentage that is emitted in a more homogeneous way determines methane oxidation (Oonk, 2010). Since a significant portion of the methane load was most probably emitted through preferential flows along the edges of the lysimeter (see FID screenings, section 4.2), the oxidation based on stable isotope analyses may have been overestimated. The improvement of quantification of methane oxidation considering both homogeneous and preferential flows is currently discussed in the literature (Oonk, 2010).

Temperature profiles

Figure 4.3 compares the seasonal temperature profiles measured in all lysimeters. The temperature profiles exhibit variations with depth and time and clearly show the different temperature performances and insulation capacities of the used substrates. The heat produced during waste degradation (and to a much lesser extent during microbial CH_4 oxidation) could be better maintained in the coarse sewage sludge compost (lysimeter A) than in the finer soil and compost materials (lysimeters B-D) throughout the whole monitoring period. As already mentioned, the CH_4 loading was always higher in lysimeters C and D compared to the others assuming a greater heat release. Nevertheless, the higher porosity (and thus higher amount of air-filled space) of the sewage sludge compost makes CH_4 oxidation more independent of low or varying external temperatures due to the lower thermal conductivity of the coarse substrate (Scheffer and Schachtschabel, 1992). As already described in section 2.4.5, the optimum temperature for CH_4 oxidation appears to be in the range of 20 – 38 °C (Visvanathan et al., 1999; Börjesson and Svensson, 1997; Figueroa 1993; Whalen et al., 1990). In summer and autumn 2009, the temperatures within the SSC-cover were about 10 to 15°C higher than in the other biocovers and remained in the vicinity of 20 – 25°C at a depth of 50 cm downwards. The shape of the temperature profile of lysimeter A, therefore, indicates more favourable conditions for CH_4 uptake during this period compared to the other substrate covers. During winter 2009/2010 the temperatures within all cover materials were relatively low (< 10°C) which might have reduced CH_4 oxidation. However, also waste decomposition in all lysimeters, except for lysimeter C, decreased during this time as indicated by decreasing CH_4 concentration levels in the waste layers (see Figure 4.1).

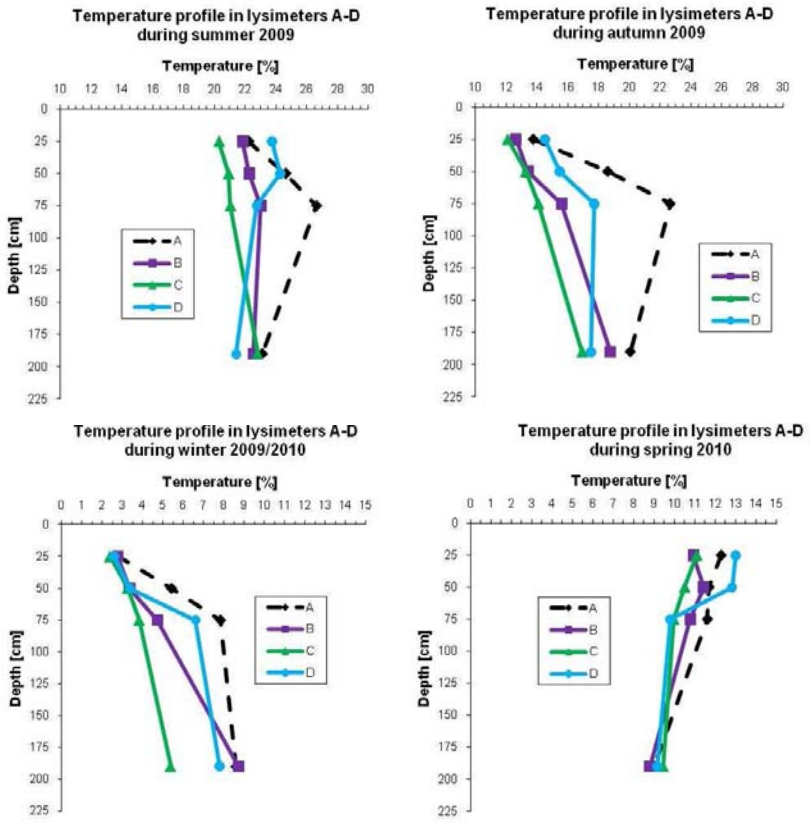


Figure 4.3: Comparison of seasonal, average temperature profiles in all lysimeters during the monitoring period from August 2009 to April 2010 (SD in summer: 0.4 – 1.7; SD in autumn: 3.2 – 5.3; SD in winter: 1.5 – 3.3; SD in spring: 1.4 – 5.9)

Moisture content profiles

As discussed previously, the moisture content is a crucial factor influencing CH_4 oxidation. In Figure 4.4 the seasonal moisture contents measured at different depths of the four lysimeters are depicted. The water content depended mainly on precipitation and temperature as well as on the capacity of the substrates to retain or release moisture. The average moisture content in all waste layers remained stable at approx. 50 vol.% throughout the entire monitoring period. In general, the volumetric moisture content was lower in the upper layers than in the lower ones, except for lysimeter B. Moisture content in the soil layers of lysimeters C and D (TS Fischa, SS Zister) was not significantly influenced by the season. During the investigation period the moisture content in the top layer of lysimeter C (25 cm) remained almost stable at about 20 % (based on dry weight) which correspond to 50 % WHC in that material (TS-Fischa). This is in the optimum range for methane oxidation (Boeckx et al., 1996; Figueroa, 1993). In the lower layer (75 cm) of the substrate (SS-Zister) higher moisture contents were found (approx. 40 vol.%) corresponding to more than 100 % WHC. In the top layer of lysimeter D (25 cm) the actual moisture content ranged between 24 – 32 vol.% (60 – 80 % WHC), while in the lower layer (50 cm) higher moisture contents were detected varying from 31 – 41 vol.% (85 – 100 % WHC). The high water levels in the lower layers of lysimeter C and D may have blocked gas transport causing bypass flows along the edges of the lysimeters (see section 4.2) and/or a reduction of methane consumption. Near the surface of lysimeter B (sand) the greatest fluctuations occurred with the lowest moisture content in summer (6.4 vol.%, 25 % WHC) and the highest in winter (22 vol.%, 80 % WHC). Lower moisture levels were found in the deeper layer (SSC-

Mix) varying from 6 – 13 vol.% (15 – 30 % WHC) due to the higher water holding capacity of SSC-Mix. In the top layer of lysimeter A (SSC) the moisture content was close to the lower range (< 10 %) while in deeper regions higher moisture content occurred at approx. 36 vol.% (30 % WHC). According to Huber-Humer (2004), the threshold value for microbial activity in compost materials was found at a moisture content of about 25 % WHC under laboratory conditions. As can be seen in Figure 4.4, the moisture content in the lower layer of lysimeter A never fell below that threshold while the moisture level in the deeper region of lysimeters C and D may have caused a decline in methane oxidation rates.

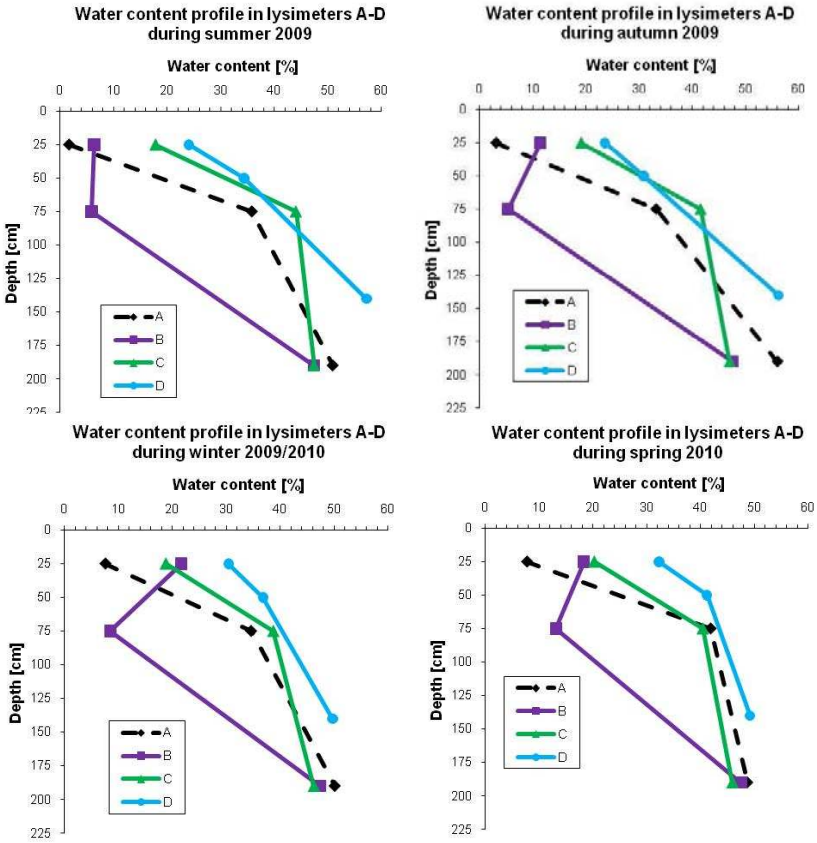


Figure 4.4: Comparison of seasonal, average water content profiles in all lysimeters during the monitoring period from August 2009 to April 2010 (SD in summer: 0.1 – 2.9; SD in autumn: 0.2 – 6.5; SD in winter: 0.1 – 3.0; SD in spring: 0.1 – 3.3)

4.2 Screening of methane surface concentration (FID-measurements)

To qualitatively survey the overall CH_4 emission pattern from the different biocovers and the expected escape routes, repeated FID surface screenings were performed. Figure 4.5 shows examples of surface CH_4 concentration patterns of lysimeter C and D at selected dates. During the monitoring period from August 2009 to April 2010 no methane emissions above background level were detected on the biocover surfaces of lysimeters A and B (see appendix A.2). The data gathered from FID screenings for CH_4 surface concentrations indicates that CH_4 emissions varied significantly between the different measuring events and suggests that landfill gas was distributed unevenly to the substrate layers.

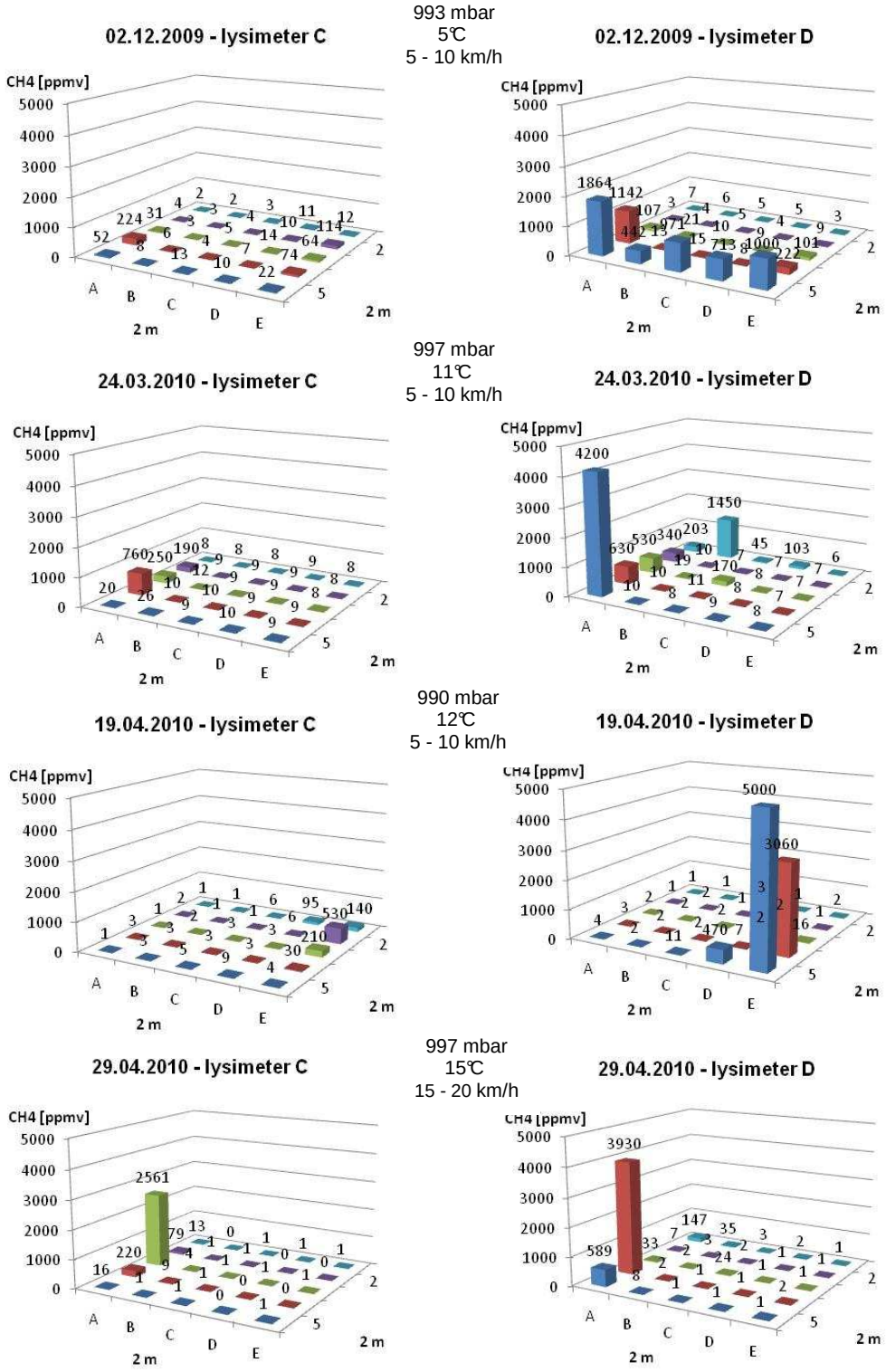


Figure 4.5: Spatial and temporal variability of surface CH_4 concentrations within a 50 cm grid on lysimeters C and D for selected measuring events (02.12.2009, 24.03.2010, 19.04.2010, 29.04.2010)

In general, higher surface CH_4 concentrations were observed along the edges of the lysimeters C and D. This indicates that a main part of the load was coming from the sides of the lysimeters through the gas distribution layer, although the walls of the lysimeter chambers were sealed with framed rubber lips at the upper part of the substrate layers. It can be hypothesised that the high moisture content in the soil layers of lysimeters C and D may have blocked the gas transport. However, the preferential flows may also be an artefact of the lysimeter facility itself. In contrast,

stable isotope analyses show that a significant portion of CH_4 could be oxidised in the substrate covers (see section 4.4). As already mentioned the preferential flows along the edges of the lysimeters had a significant influence on the actual oxidation. In general, loamy soils tend to develop crack formations additionally supporting the preferential channels along the edges.

Methane emission through the substrate layer of lysimeter C seemed to be relatively low as also indicated by the surface flux measurements (0 – 1.1 g/m²d, see section 4.3). The surface screening showed ambient CH_4 concentrations ranging from background concentrations up to 760 ppmv. Only on 29.04.2010 were CH_4 concentrations higher than 2,500 ppmv detected. The general picture was that methane escaped preferentially from the eastern and western edges of lysimeter C.

Even though the CH_4 loading was always higher in lysimeter C than in lysimeter D (as indicated by higher CH_4 concentration in the LFG, see Figure 4.1), higher CH_4 concentrations were found at the cover surface of lysimeter D (control cover). Highest CH_4 emissions escaped along the edges, suggesting that the retention time might have been too low for CH_4 oxidation to occur. The CH_4/CO_2 ratio curve of lysimeter D further indicates the occurrence of preferential flows (see Figure 4.19). Following the surface flux measurements (see section 4.3), CH_4 emissions from lysimeter D (0.9 – 8.2 g/m²d) were significantly greater than fluxes from lysimeter C (0 – 1.1 g/m²d).

4.3 Flux measurements

Table 4.1 lists the average CH_4 and CO_2 emissions from the 6 – 8 measurement campaigns conducted on the four lysimeters, together with the temperature and wind speed of the measuring day as well as the pressure change 24h prior to the beginning of the measuring day (9am to 9am). In addition, the pressure change over the measuring day (9am to 5pm) is presented.

As can be seen in Figure 4.6 and Table 4.1, all measurement campaigns were done during stable pressure conditions (during measuring day: -0.4 to 0.3 hPa/h; 24 h priors to the measurement campaigns: -0.3 to 0.3 hPa/h) and wind speeds < 4.2 m/s. Consequently, no indication was obtained that atmospheric pressure influences landfill gas emissions. Only the measurement campaigns on 10.12.2009 and 29.04.2010 were done during decreasing pressure ($\Delta p > 0.6$ hPa/h). Between the middle of December and end of March no surface emission measurements were performed.

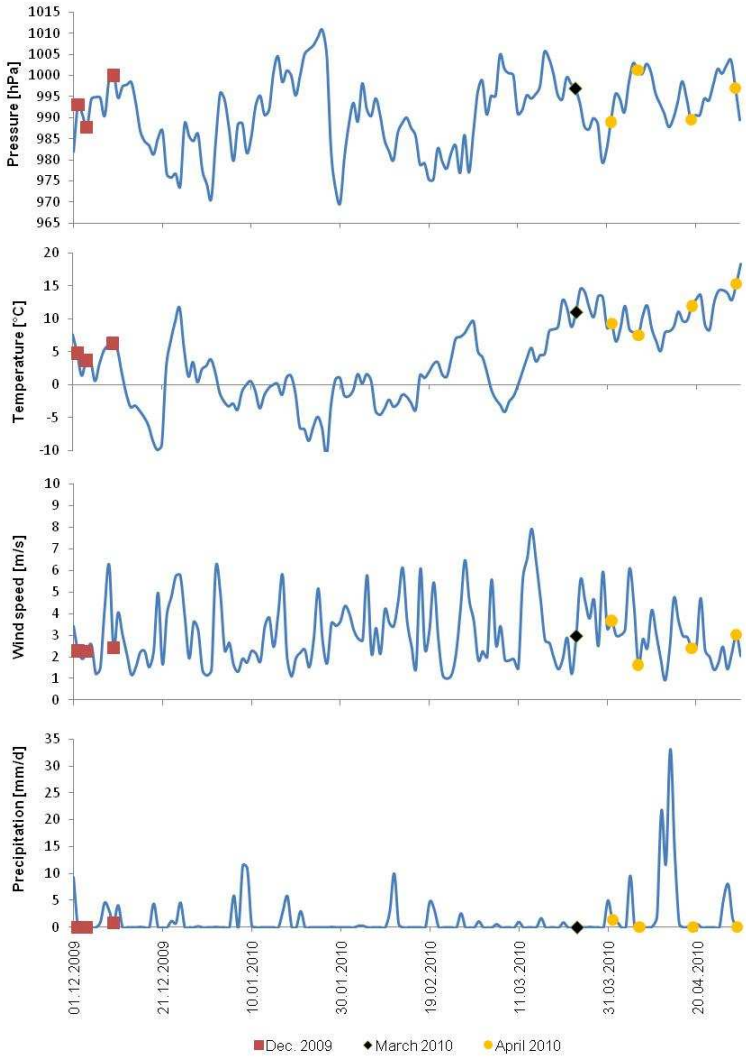


Figure 4.6: Weather conditions under measurement campaigns: atmospheric pressure, temperature, wind speed and precipitation

The average methane emissions from lysimeter D ranged from 0.9 – 8.2 g/m²d whereas lysimeter C had almost zero CH_4 emissions (0 – 1.1 g/m²d) during the measurement campaigns. In contrast, no CH_4 emissions were detected from the cover surface of lysimeters A and B. Carbon dioxide emissions were 2.9 – 21.9 g/m²d from lysimeter A, -4.7 (uptake) to 16 g/m²d from lysimeter B, 5.8 – 9 g/m²d from lysimeter C and 6.1 – 22 g/m²d from lysimeter D. It should be noted that the ability to compare emission data between lysimeters is limited due to the varying methane development in the solid waste layers of the lysimeters (see Figure 4.1).

Table 4.1: Results of the measurement campaigns

Lys.	Date	Δp^a [hPa/h]	Δp^b [hPa/h]	T [℃]	Wind speed [m/s]	Emissions [g/m²d]		Std. error	
						CH_4	CO_2	CH_4	CO_2
D	02.12.09	0.6	0.0	7.2	1.6	2.0	10.0	0.6	2.6
D	04.12.09	-0.26	0.34	3.5	1.5	8.2	22.0	1.0	2.4
A	10.12.09	0.22	-0.71	7.4	1.8	0.0	21.9	-	2.3
B						0.0	16.0	-	1.9
C						1.1	5.8	0.1	0.3
D						2.3	9.0	2.1	4.5
A	24.03.10	-0.06	-0.35	16.1	4.2	0.0	8.8	-	0.3
B						0.0	5.5	-	1.2
C						0.5	6.9	0.1	1.4
D						2.9	11.1	0.2	0.8
A	01.04.10	0.27	-0.19	15.3	3.8	0.0	2.9	-	3.0
B						0.0	5.5	-	0.8
C						0.0	7.5	-	0.6
D						0.5	6.1	0.2	2.5
A	07.04.10	-0.09	-0.33	12.8	1.8	0.0	13.5	-	1.8
B						0.0	6.5	-	0.5
C						1.0	8.2	0.3	1.0
D						0.9	7.8	0.1	0.7
A	19.04.10	-0.31	-0.10	18.2	3.7	0.0	10.0	-	0.5
B						0.0	-4.7	-	0.9
C						0.7	7.7	0.2	0.4
D						1.5	9.0	1.1	2.5
A	29.04.10	-0.26	-0.69	20.8	4.1	0.0	11.3	-	5.9
B						0.0	8.3	-	0.3
C						0.6	9.0	0.1	0.4
D						1.4	11.4	0.5	1.8

[a] The pressure gradient 24 h prior to the measurement campaign; [b] The pressure gradient is the pressure difference over the measuring day from 9 am to 5 pm; Temperature (T) and wind speed are the average between 9 am and 5 pm

In general, methane emission fluxes from landfill show high spatial and temporal variability. Table 4.2 gives a summary of methane emissions released from different landfills and large field trials. The measured surface emissions correspond with the results reported in Barlaz et al. (2004) and Maurice and Lagerkvist (2003). However, due to the use of different measuring methods, surface emission rates are hardly comparable.

Table 4.2: Summary of methane emissions rates from landfills and large field trials (adapted from Scheutz et al., 2009)

Reference	Country	Cover material	GES	Approach	CH_4 emission rate [g/m²d]		
					Average (n)	Min.	Max.
Maurice & Lagerkvist (2003)	Sweden	120 cm silty soil	no	SCT	0.35 (3-12)	<0.04	2
Scheutz et al. (2008)	France	30 cm top soil on 70 cm clay	yes	SCT	-0.001 (12)	-2.5	29
Scheutz et al. (2003)	France	80 cm loam on 40 cm coarse sand	yes	SCT	2 (23)	-0.01	10
		40 cm coarse sand	yes	SCT	37.8		49.9
Börjesson and Svensson (1997)	Sweden	10-80 cm sandy loam	-	SCT	0.01-7.7 (72)	-0.3	18.4
Barlaz et al. (2004)	USA	100 cm clay soil cover	no	SCT			>15
		100 cm yard waste compost underlain by 15 cm gravel and 15 cm clay	no	SCT		-1.73	1.33
Jugnia et al. (2008)	Canada	80 cm sand/compost	no	CDC	2.5 – 30		210
Humer & Lechner (2001a,c)	Austria	90 cm compost underlain by 30 cm gravel	yes	DFT	0-0.3 (27)	-0.5	2.1
		90 cm compost underlain by 30 cm gravel	yes	DFT	0-1.9 (27)	-3.3	5.6
		40 cm compost cover	yes	DFT	0-248.9 (20)	0	706.8
Stern et al. (2007)	USA	50 cm compost underlain by 15 cm crushed glass and 65-75 cm clay cover	-	SCT	2 (44)		
		65-75 cm (control cover)	-	SCT	10.6 (45)		
		30 cm compost underlain by 15 cm crushed glass and 15 cm clay	-	SCT	300 - 700		
		60 cm compost underlain by 15 cm crushed glass and 15 cm clay	-	SCT	30 - 70		

SCT, static chamber method; DFT, dynamic flux tunnel; CDC; closed dynamic chamber; GES, gas extraction system

In Figure 4.7, a graphical presentation of the emission data in Table 4.1 can be seen.

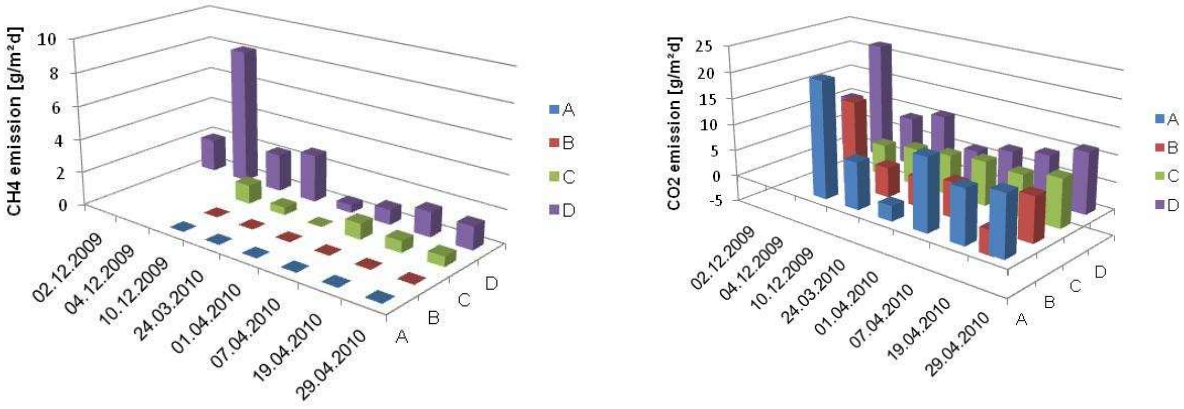


Figure 4.7: Graphical presentation of the emission data in Table 4.1

The individual emission data of the lysimeters are addressed particularly in the following.

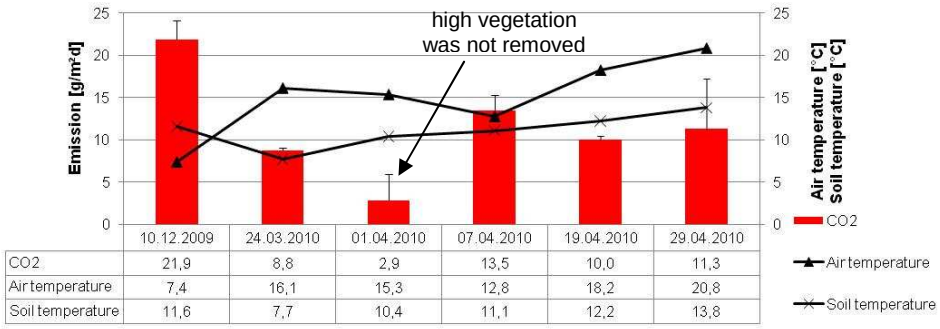
Lysimeter A

In Figure 4.8 the development of CO_2 emissions of lysimeter A over six measuring campaigns are depicted together with the air and soil temperatures of the respective measuring days. In general, results of surface emission measurements showed high CO_2 emissions and no CH_4 emissions indicating significant methane oxidation in lysimeter A. However, respiration of compost may also contribute to the overall CO_2 emissions, though the compost material placed in lysimeter A had a very low oxygen consumption (AT_4 = 0.7 mg O_2/g DM) under laboratory conditions. Experiments by Scheutz et al. (2009b, c) showed that respiration of compost (placed in a biowindow) generated significant CO_2 emissions even though the compost has been originally characterised as mature and stable. With the exception of the CH_4 and CO_2 concentrations measured on 10.12.2009, the CH_4 and CO_2 concentrations at the bottom of the waste layer remained at approx. 17 vol.% and 35.5 vol.%, respectively, as can be seen in Figure 4.9. On 10.12.2009, CH_4 and CO_2 concentrations were slightly higher (22 vol.% CH_4, 45 vol.% CO_2) explaining the high observed CO_2 emission rate on this day.

Compared to the other lysimeters, almost higher CO_2 emissions were observed at the biocover surface of lysimeter A but with lower CO_2 concentrations in the upper part of the substrate cover. This indicates a higher dilution in the porous sewage sludge than in the finer soil and compost materials placed in lysimeters B, C and D. In general, materials with a high air-filled pore space such as composts enable higher diffusive or advective intrusion of atmospheric air (Mor et al., 2006). It is also hypothesised that CO_2 emission rates (and also CH_4 emissions) were underestimated with lysimeter C and D since high volumetric moisture contents in the upper part of the soil layers may have blocked gas transport causing bypass flows along the edges of the lysimeters (see the discussions below).

Several studies have shown that CH_4 uptake by methanotrophs increases with rising temperatures (Gebert et al., 2003; Boeckx et al., 1996; Figueroa, 1993; Whalen et al., 1990) when the CH_4 concentration, among other factors, is not limiting (Einola et al., 2007). Although LFG concentration remained almost stable during spring measurement campaigns, no significant relation between increasing air temperature

and CO_2 emissions has been found. It is hypothesised that either methane was a limiting factor or vegetation had a great influence on CO_2 emission measurement.



	10.12.2009	24.03.2010	01.04.2010	07.04.2010	19.04.2010	29.04.2010
CO2	21,9	8,8	2,9	13,5	10,0	11,3
Air temperature	7,4	16,1	15,3	12,8	18,2	20,8
Soil temperature	11,6	7,7	10,4	11,1	12,2	13,8

Figure 4.8: Development of CO_2 emissions of lysimeter A and the corresponding soil temperature at 50 cm depth and air temperature. The chamber data are shown as the mean of 3 measurements with error bar representing the positive standard deviation.

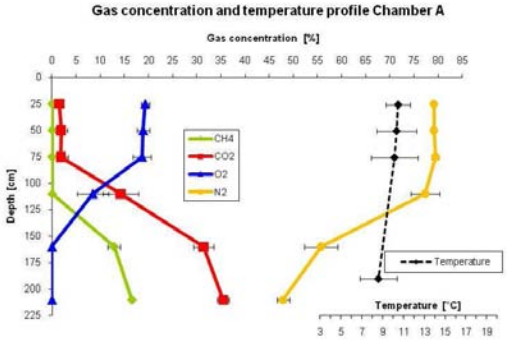


Figure 4.9: Average soil gas concentration and temperature profile of lysimeter A during spring 2010.

Except for 10.12.2009, all surface emission measurements were conducted during the growing season. In the presence of landfill gas, the vegetation developed differently on lysimeters A-D, as can be seen in Figure 4.10. Although specific plant species were sown (see Internal Report Nutzraum 2008), spontaneous vegetation developed on the surfaces of the lysimeters. The vegetation coverage was about 30 - 50 % at lysimeters C and D whereas lysimeter A was fully covered. At the end of April the vegetation coverage of lysimeter B was close to 100 %. It seems that the compost material placed in lysimeter A provided a better nutrient supply for vegetation than mineral soils.

Figure 4.10: Vegetation condition of the lysimeters A, B, C and D (from left to right) on 01.04.2010 (top) and 29.04.2010 (bottom)

Prior to surface flux measurements, grass and small plants were moved from the cover soil to avoid plant respiration and photosynthesis. However, due to the small chamber height (14.5 cm), the vegetation´s condition had an effect on the CO_2 emission rate even when the vegetation height was low. As can be seen in Figure 4.8, the CO_2 emission rate on 01.04.2010 was relatively low compared to the other measurement campaigns. The vegetation was not cut on this day, as the main part of the vegetation was short (< 5 cm) and only some plant species were higher than 10 cm. To investigate the influence of vegetation on the CO_2 emission measurement in the chamber, surface emission measurements were performed before and after vegetation was cut. As shown in Figure 4.11, CO_2 emission rate increased 4.5-fold after high vegetation was removed indicating the photosynthetic use of CO_2 by vegetation. On 29.04.2010 even a higher increase (7.5-fold) was observed. However, no plant physiological investigations have been conducted so far in order to verify these observations. But several studies have demonstrated that the net photosynthetic rate of a wide range of plant species is stimulated when plants are exposed to an elevated CO_2 environment (Lewis et al., 2002; Ziska and Bunce, 1997; Arp, 1991). In order to improve the CO_2 emission measurement, the chamber could be equipped with both a transparent and an opaque cover. Measuring the CO_2 flux under both light and dark conditions will result in estimates of the assimilation by the vegetation.

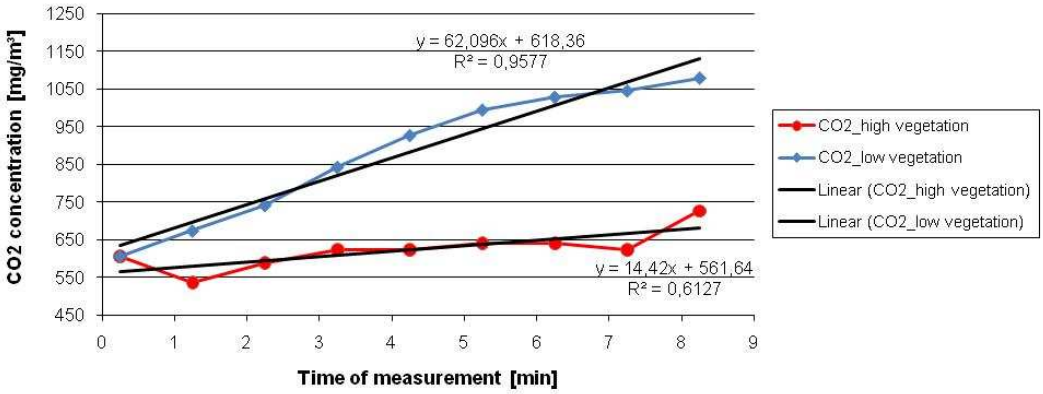
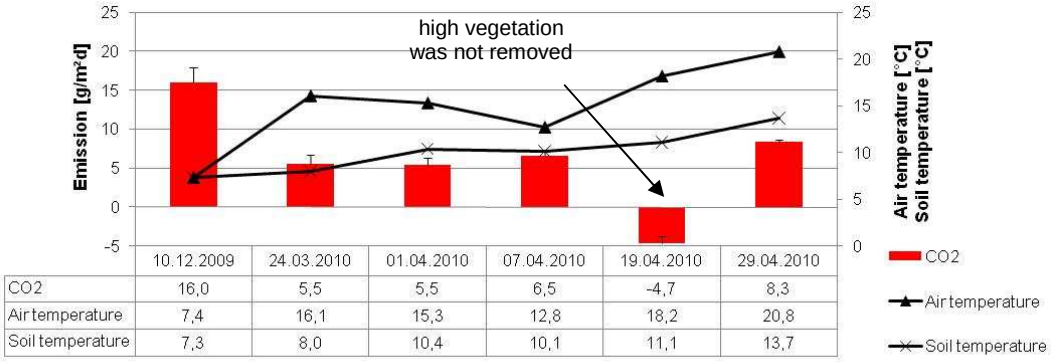


Figure 4.11: Influence of vegetation height on CO_2 emission measurement (07.04.2010)

Lysimeter B

The surface emissions of lysimeter B revealed a similar pattern, but with a lower load of LFG in the solid waste layer (4 vol. % CH_4, 30 vol. % CO_2, see Figure 4.13). Elevated CO_2 emissions and no CH_4 emissions were detected during surface emission measurements (see Figure 4.12). The higher CO_2 emission rate observed on 10.12.2009 might be a result of the higher LFG concentration measured at the bottom of the waste layer (9 vol. % CH_4, 45 vol. % CO_2). Similar to lysimeter A, no significant relation between increasing air temperature and CO_2 emissions has been found. It seems that methane was a limiting factor due to low concentrations observed in the waste layer. Besides, vegetation also influenced CO_2 emission measurements. On 19.04.2010 CO_2 emission was negative indicating that much of the carbon dioxide emitted from the biocover surface can be rapidly assimilated by plant photosynthetic processes. The main height of the vegetations was < 5 cm, only some plant species were higher than 10 cm. CO_2 uptake has also been reported by Börjesson and Svensson (1997) and Chen et al. (2008).



	10.12.2009	24.03.2010	01.04.2010	07.04.2010	19.04.2010	29.04.2010
CO2	16,0	5,5	5,5	6,5	-4,7	8,3
Air temperature	7,4	16,1	15,3	12,8	18,2	20,8
Soil temperature	7,3	8,0	10,4	10,1	11,1	13,7

Figure 4.12: Development of CO_2 emissions of lysimeter B and the corresponding soil temperature at 50 cm depth and air temperature. The chamber data are shown as the mean of 3 measurements with error bar representing the positive standard deviation.

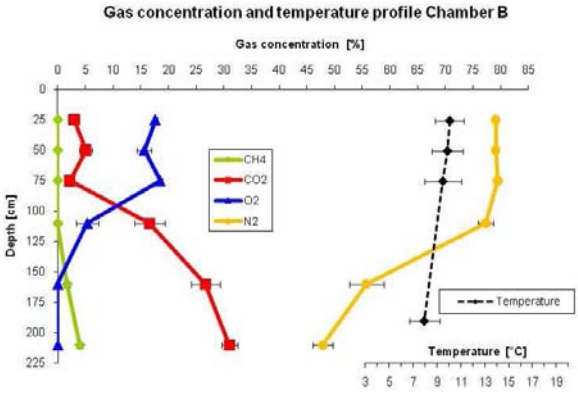
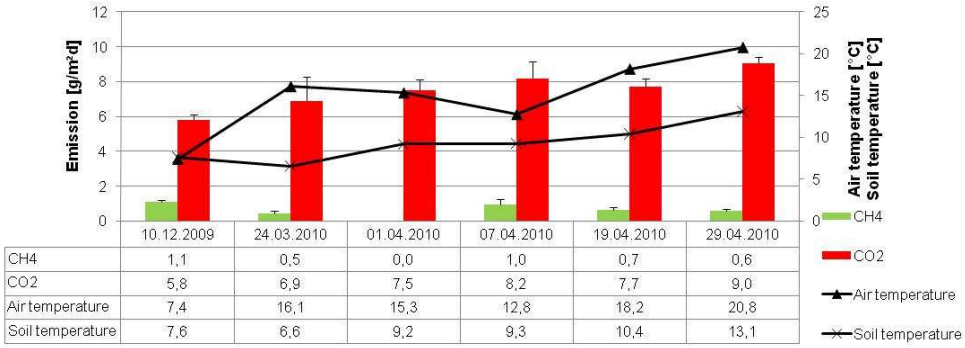


Figure 4.13: Average soil gas concentration and temperature profile of lysimeter B during spring 2010

The influence of vegetation was also investigated during surface emission measurement at lysimeter B on 29.04.2010. Similar to the investigations at lysimeter A (see Figure 4.11) CO_2 emissions increased approx. 4-fold after higher vegetation (> 10 cm) was removed. The measured CO_2 emission rates should be interpreted with caution, as they represent only snapshots of a single measuring event. In general, CO_2 emissions are low in the daytime due to photosynthesis and increase at night. In addition, CO_2 emissions can vary during a day. Park and Shin (2001) reported maximum CO_2 fluxes when the temperature was at its peak. Although CO_2 emissions were corrected for the temperature measured inside the enclosed chamber, humidity also influences CO_2 emissions. However, relative humidity inside the chamber was not monitored during the sampling period. In order to crosscheck the disturbance effect of relative humidity change on CO_2 concentration, a background measurement and alignment should be done in future studies using the closed dynamic chamber.

Lysimeter C

The concentrations of CH_4 and CO_2 in the biogas sampled at the base of the waste layer were around 50 vol.% during all surface flux measurements (see Figure 4.1). This indicates that the methane supply was not a limiting factor during this period. As can be seen in Figure 4.14, methane emissions remained low during these measurement campaigns, while carbon dioxide emissions were increasing over time as the temperature rose.



	10.12.2009	24.03.2010	01.04.2010	07.04.2010	19.04.2010	29.04.2010
CH4	1,1	0,5	0,0	1,0	0,7	0,6
CO2	5,8	6,9	7,5	8,2	7,7	9,0
Air temperature	7,4	16,1	15,3	12,8	18,2	20,8
Soil temperature	7,6	6,6	9,2	9,3	10,4	13,1

Figure 4.14: Development of CH_4 and CO_2 emissions of lysimeter C and the corresponding soil temperature at 50 cm depth and air temperature. The chamber data are shown as the mean of 3 measurements with error bars representing the positive standard deviation.

Figure 4.15 presents the average soil gas concentration and temperature profile during spring 2010.

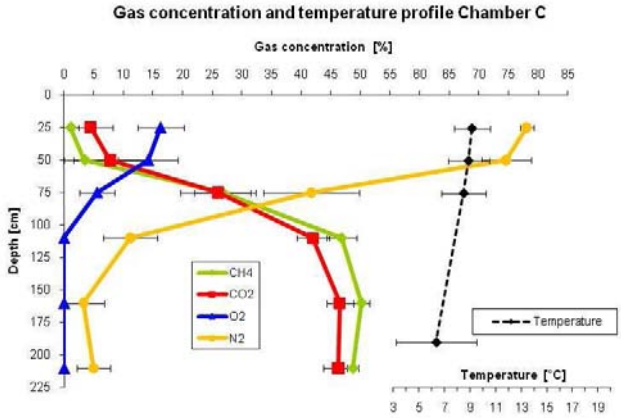


Figure 4.15: Average soil gas concentration and temperature profile of lysimeter C during spring 2010

In Figure 4.16 CO_2 emissions from lysimeter C related to atmospheric pressure and air temperature are presented. The trend line indicates a significant relation between change of air temperature and CO_2 emissions (R^2=0.79, n=18). Fluxes of CO_2 were also positively correlated with soil temperatures at 25 cm and 50 cm depth (R^2=0.67 and R^2=0.52). In contrast, CO_2 emissions were only weakly correlated with soil temperature at 75 cm depth (R^2=0.33). No correlations were obtained between CO_2 emissions and atmospheric pressure change (R^2 = 0.02). Similar observations have been made by Scheutz et al. (2009b), where CO_2 emission depended more on temperature (R^2=0.62) than on barometric pressure (R^2=0.03). By excluding the day 01.04.2010, CH_4 emissions showed a negative correlation with air temperature (R^2=0.57, n=15). Since CH_4 concentrations were not limiting, these observations strongly suggest that the increasing CO_2 emissions and decreasing CH_4 emissions over time were most likely due to the higher temperatures resulting in higher methane oxidation rates.

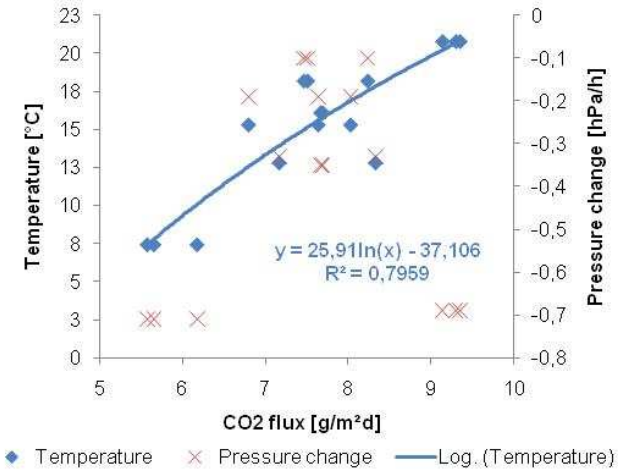


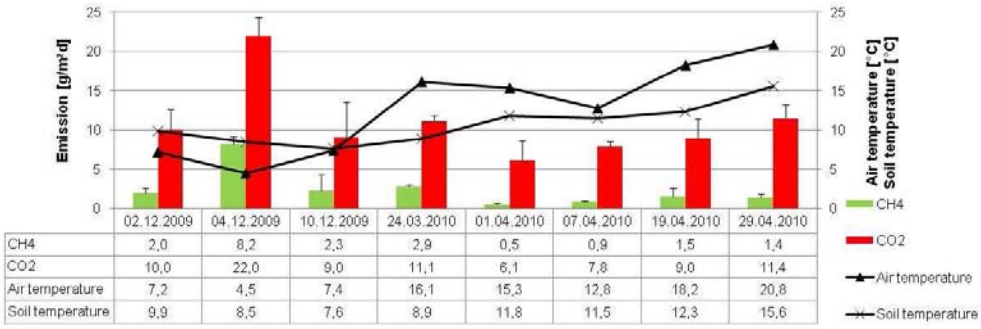
Figure 4.16: CO_2 emissions from lysimeter chamber C related to temperature and atmospheric pressure change (9am – 5 pm on the measuring day)

However, results obtained by the surface flux measurements should be interpreted with caution, not only for CH_4 but also CO_2. As can be seen in Figure 4.4, the high volumetric moisture content in lysimeter C, especially in the upper part of the substrate cover may have blocked gas transport causing bypass flows along the edges of the lysimeters. In contrast, gas permeability tests conducted at the Austrian Institute of Technology in Seibersdorf revealed that all substrates used in this study should ensure sufficient air permeability even at 100 % WHC. Despite the use of a plastic skirt attached on the closed chamber for sealing purposes, a great portion of the methane and carbon dioxide flux may have bypassed leading to an underestimation of CH_4 and CO_2 emissions.

Lysimeter D

The development of CH_4 and CO_2 emissions of lysimeter D over eight measuring campaigns are presented in Figure 4.17 together with the air and soil temperatures of the respective measuring days. As can be seen in Figure 4.1, methane concentration in the biogas sampled at the base of the waste layer started to decrease after the end of October 2009. During the surface emissions measurement, CH_4 concentrations were around 40 vol.% in the waste layer (see Figure 4.18). Clear differences in CH_4 emissions were observed between both the growing and the winter season. Compared to lysimeter C, CH_4 and CO_2 fluxes were weakly, negatively correlated with temperature (R^2=0.30 and R^2=0.36, respectively). There are no specific reasons for the relatively high CH_4 emission rate recorded on 04.12.2009. However, several studies have demonstrated diurnal variation in the release of CH_4 from landfills (Börjesson and Svensson, 1997). There was almost no CH_4 detectable in the substrate layer up to 50 cm. Although a significant portion of CH_4 could be oxidised (see stable isotope analyses, section 4.4), the main part of the CH_4 load most probably came from the side of the lysimeter through the gas distribution layer beneath the substrate layer (see Figure 4.5). The high volumetric moisture content throughout the soil profile (see Figure 4.4) remained high during surface flux measurements (approx. 32 vol.% based on dry weight, 80 – 100 % WHC) and probably caused a blockage for gas transport. As a consequence, the absolute data should be interpreted with caution. As already discussed, preferential flows decrease the overall oxidation efficiency of the biocover. Consequently, the oxidation based on stable isotope analyses may have been overestimated since only

the methane load that is emitted in a more homogeneous way determines methane oxidation.



	02.12.2009	04.12.2009	10.12.2009	24.03.2010	01.04.2010	07.04.2010	19.04.2010	29.04.2010
CH4	2,0	8,2	2,3	2,9	0,5	0,9	1,5	1,4
CO2	10,0	22,0	9,0	11,1	6,1	7,8	9,0	11,4
Air temperature	7,2	4,5	7,4	16,1	15,3	12,8	18,2	20,8
Soil temperature	9,9	8,5	7,6	8,9	11,8	11,5	12,3	15,6

Figure 4.17: Development of CO_2 emissions of lysimeter D and the corresponding soil temperature at 50 cm depth and air temperature. The chamber data are shown as the mean of 3 measurements with error bars representing the positive standard deviation.

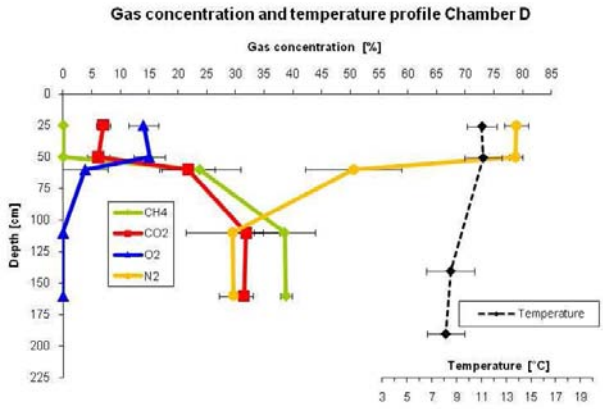


Figure 4.18: Average soil gas concentration and temperature profile of lysimeter D during spring 2010

In general, CH_4 emissions from lysimeter D (0.9 – 8.2 g/m²d) were significantly higher than fluxes from lysimeter C (0 – 1.1 g/m²d). This result indicates that low-dimensioned covers do not achieve as high CH_4 oxidation capacities as properly designed biocovers.

Figure 4.19 presents typical CH_4/CO_2 ratio curves for all lysimeters. In general, a decreasing ratio is a strong indication for methane consumption. The shape of the curves further confirms the location of the oxidation zone, which is indicated by the sharpest decline in the curves.

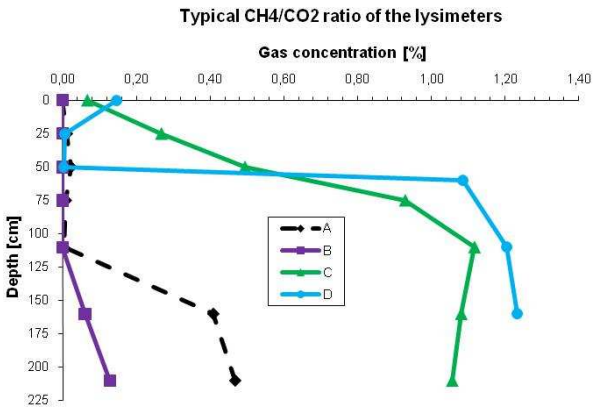


Figure 4.19: Typical CH_4/CO_2 ratio of the lysimeters during the study period

For lysimeters C and D the ratios changed from about 1.1 - 1.2 to very low ratios, even approaching to zero. However, for lysimeter D the CH_4/CO_2 ratio of the emitted landfill gas captured within the closed chamber (depth 0 cm) was higher than in the subsurface. This finding is a further indication for preferential flows along the edges of lysimeter D.

4.4 Isotope fractionation

Two sets of samples (21.10.2009 and 09.12.2009) from each depth of the cover profile were subjected to an analysis for stable isotopes. Stable isotope analyses were used as a first indication to what extent the biogas had been oxidised within the different biocovers. Table 4.3 presents the $\delta^{13}C$ values of both sampling dates as well as the percentage of oxidised CH_4 for open and closed systems (equation 3.7 and 3.8, see section 3.3.6) and the fractionation factor for bacterial oxidation, α_{ox}, corrected for temperature using Eq. 3.9 (see section 3.3.6). The value used for average α_{ox} (1.027) was determined according to literature values (Table 3.8) in the temperature range of 6 – 26℃. This approach was also conducted by Cabral et al. (2009). Samples from the deepest waste layers had the lowest CH_4-$\delta^{13}C$ values representing landfill gas before oxidation. These values ranged from -54.8 to -44.1 ‰ which are higher than those found in the literature (-50 to – 60 ‰; Chanton et al., 1999).

Table 4.3: Summary of stable isotope analyses and estimated fraction of CH_4 oxidised by closed system (f_{oxc}) and open system (f_{oxo})

lysimeter and depth	**material**	**$\delta^{13}C$ [‰]** 21.10.		**$\delta^{13}C$ [‰]** 09.12.		**f_{oxc} [%]**	**f_{oxo} [%]**	**f_{oxc} [%]**	**f_{oxo} [%]**	**α_{ox}[a]** (corr. f. temp)[c]	
[cm]		CH_4	CO_2	CH_4	CO_2	21.10.		09.12.		21.10	09.12
A											
25	SSC		-24.1		-24.5						
50	SSC		-25.0		-24.1						
75	SSC		-24.1		-24.9						
110	GDL	-35.2	-24.4		-24.1	58.3	81.6			1.024	
160	waste	-50.4	-12.5	-52.3	-13.8	17.0	17.2	0.7	0.7	1.026	1.030
210	waste	-54.8	-13.5	-52.5	-13.5	0	0	0	0	1.026	1.030
B											
25	sand		-20.2		-21.4						
50	SSC-Mix		-21.3		-21.8						
75	SSC-Mix		-20.4		-21.8						
110	GDL		-21.9		-21.8						
160	waste	-51.3	-13.6	-45.1	-14.6	1.2	1.1	3.2	3.0	1.026	1.030
210	waste	-51.6	-11.3	-46.0	-10.8	0	0	0	0	1.026	1.030
C											
25	TS-Fischa	-28.4	-21.1		-18.7	59.7	84.5			1.030	
50	TS-Fischa	-27.4	-20.4		-16.6	61.9	89.9			1.030	
75	SS-Fischa	-40.8	-15.2	-42.3	-14.3	38.9	45.6	19.3	19.8	1.029	1.032
110	GDL	-49.2	-10.4	-45.7	-9.9	16.6	16.7	9.6	9.3	1.029	1.032
160	waste	-51.8	-11.2	-45.1	-11.9	8.5	8.2	11.7	11.5	1.028	1.031
210	waste	-54.1	-10.2	-48.7	-11.1	0	0	0	0	1.028	1.031

lysimeter and depth	material	$\delta^{13}C$ [‰] 21.10.		$\delta^{13}C$ [‰] 09.12.		f_{oxc} [%]	f_{oxo} [%]	f_{oxc} [%]	f_{oxo} [%]	α_{ox}[a] (corr. f. temp)[c]	
[cm]		CH_4	CO_2	CH_4	CO_2	21.10.		09.12.		21.10	09.12
D											
25	TS-Fischa	-22.4	-23.7	-26.2	-30.4	54.7	74.4	45.3	56.4	1.029	1.033
50	SS-Fischa	-26.3	-22.2	-31.5	-30.4	48.6	62.4	35.4	40.7	1.029	1.033
60	GDL	-37.8	-13.4	-43.3	-13.2	21.1	22.1	4.8	4.6	1.029	1.033
110	waste	-42.7	-11.0	-43.2	-9.2	5.5	5.3	5.5	5.2	1.027	1.031
160	waste	-44.1	-12.7	-44-8	-8.6	0	0	0	0	1.027	1.030

[a] the average α_{ox} (1.027) was adopted from literature values (Table 3.8) and corrected for temperature (see eq. 3.9 in section 3.3.6)

The CH_4-$\delta^{13}C$ values usually increased at more shallow depths indicating a ^{13}C enrichment whereas CO_2-$\delta^{13}C$ values decreased. The ^{12}C enrichment of CO_2 and the corresponding ^{12}C depletion of CH_4 at more shallow depths is strong evidence that methane oxidation is microbially mediated (Figure 4.20).

On 09.12.2009 gas samples for isotope analysis were also taken during surface flux measurement. However, the isotope ratios obtained during the chamber measurements are not comparable to the isotope composition measured in the gas profiles of the lysimeters due to the different volume injection for the GC. While injection volumes of gas sampled at the different depths of the lysimeters ranged between 3 and 30 µl, a large sample volume (5,000 µl) of gas sampled during chamber measurement was injected in order to obtain appropriate $\delta^{13}C$ values. Therefore, the $\delta^{13}C$ values of CH_4 and CO_2 obtained in the closed chamber were rejected.

The individual stable isotope analyses are addressed in the following.

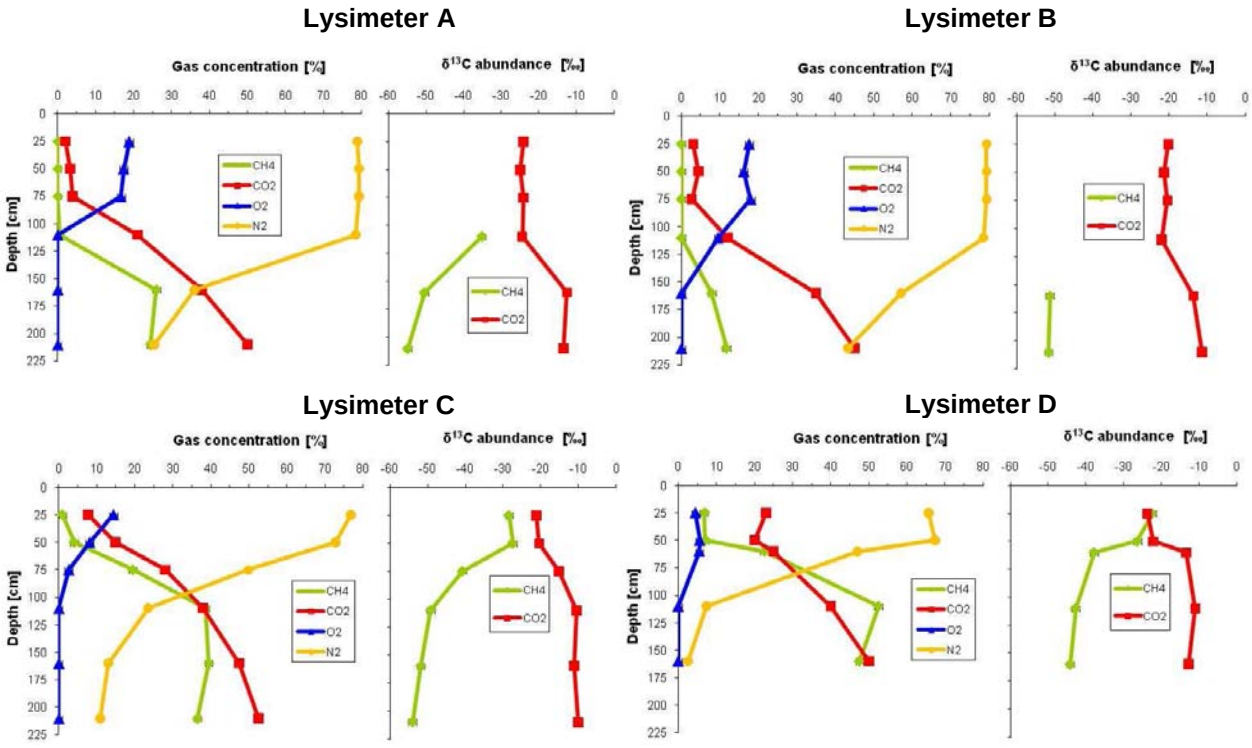


Figure 4.20: Gas concentration profiles and stable isotope results for all lysimeters on 21.10.2009

Lysimeter A

As can be seen in Table 4.3, the October analyses (21.10.2009) revealed that methane in lysimeter A had already been oxidised by an average of. 58 % (closed system interpretation, eq. 3.7) or 82 % (open system interpretation, eq. 3.8) before reaching the substrate layer. Between 25 cm and 75 cm no methane could be detected and CO_2-$\delta^{13}C$ remained at approx. -24 ‰ indicating no further ^{12}C enrichment (see Figure 4.20). This supports the assumption that CH_4 oxidation was going on deeper in the gas distribution layer and in the waste itself (see section 4.1).

In contrast, on 09.12.2009 no indication for CH_4 oxidation in the deeper layers has been found. However, there was also no methane detectable in the substrate layer and CO_2-$\delta^{13}C$ value decreased slightly from 160 cm to 110 cm indicating a possible CH_4 oxidation within the gas distribution layer. Although the CH_4 concentration decreased slightly in the waste layers during both sampling dates, the very low fraction of oxidised methane may have resulted from a measurement error.

Lysimeter B

The LFG production in lysimeter B was probably too low in order to see any evidence for microbial CH_4 via stable isotope analysis. Only gas samples of the waste layer contained methane. The $\delta^{13}C$ of CH_4 ranged from -51.6 to 45.1 ‰ between both sampling dates. A slight decrease of CO_2-$\delta^{13}C$ value from 160 cm to 110 cm indicate that CH_4 oxidation was possibly occurring within the gas distribution layer and the waste body itself.

Lysimeter C

Based on the stable isotope analyses conducted in October 2009, the oxidation zone in lysimeter C seems to be located between 50 cm and 25 cm from the surface. In contrast, gas composition and temperature profiles (see section 4.1) indicate that the main part of the oxidation process took place in a depth between 75 cm and 50 cm from the surface. The f_{ox} values obtained at 25 cm and 50 cm ranged between an average of 61 % (closed system interpretation) or 87 % (open system interpretation). These estimated fractions of oxidised CH_4 correspond well with the results obtained by column experiment under laboratory conditions (see Figure 3.8). Oxidation activity was already detectable in the gas distribution layer, where $f_{oxc,o}$=16.6 %. As can be seen in Figure 4.20, the CH_4 concentration decreased steadily throughout the substrate layer, with a quite low concentration of CH_4 (1 %) at the uppermost sampling point (25 cm).

The analysis conducted in December 2009 revealed different results. Although the CH_4 concentration in the solid waste layer increased from 40 vol.% to 49 vol.% between October and December 2009, no methane was measured in the upper part of the substrate layer (25 – 50 cm). As a consequence, oxidation was only detectable at the base of the 1 m thick soil cover (at 75 cm), where $f_{oxc,o}$=19 %. It is assumed that the high moisture content in the lower layer may have blocked gas transport causing preferential flows along the edges of the lysimeter. In addition, the temperature within the profile of lysimeter C was relatively low (< 10℃) which might have reduced CH_4 oxidation.

Lysimeter D

On 21.10.2009 the most enrichment in CH_4-$\delta^{13}C$ (-44.1 to 22.4 ‰) was observed at 25 cm depth indicating that the oxidation zone is located between 50 cm and 25 cm from the surface. Compared to lysimeter C, the oxidation efficiency was slightly lower

and reached 55 % (f_{oxc}) and 74 % (f_{oxo}) at 25 cm. This is also in the range of the potential CH_4 degradation capacity measured in column experiments under laboratory conditions (see Figure 3.8). However, the concentrations of CH_4 near the surface were still not negligible (approx. 7 vol.%). On 09.12.2009, stable isotope analysis revealed slightly lower f_{ox} values (f_{oxc}= 45 % and f_{oxo}= 56 %) most probably due to the lower temperatures within the soil profile (<10 ℃) and lower CH_4 concentration in the solid waste layer compared to the October analysis.

Considerations concerning α_{trans}

The fraction of methane oxidised in an open system (f_{oxo}) is calculated with the transport fractionation factor (α_{trans}) according to Eq. 3.8 (see section 3.3.6). In general, α_{trans} is difficult to assess as it depends on the relative importance of diffusion relative to advection in the transport of CH_4. In the present case, the transport fractionation factor was set to be 1 for purely advective systems. Based on the potential of gas formation (GS_{21}) of the waste bodies (potential CH_4 load = 130 l/m²h), it was determined that gas transport is dominated by advection with the diffusive flux representing 5 – 15 % of this potential methane load. The diffusive flux was determined by using Fick´s first law:

$$J = -D\frac{dC}{dz} \qquad (4.1)$$

J	(mol/m²s)	diffusion flux
D	(m²/s)	diffusion coefficient
dC/dz	(mol/m^4)	concentration gradient

The calculated diffusive flux is based on a typical concentration gradient between the waste layer (160 cm) and the top of the substrate layer (25 cm) of lysimeters C and D (dC/dz = 15 – 22 mol/m^4), the diffusion coefficient of methane in air (0.16 cm²/s) and a substrate porosity of 20 %. However, the assumption of α_{trans}=1 may have lead to an underestimation of the oxidation efficiency of the lysimeters (assuming an open system, f_{oxo}) since the calculations are based on potential methane loading (LFG loading date were not available).

Sensitivity Analysis

Because the isotopic fraction factor for microbial oxidation (α_{ox}) was calculated based on previous laboratory and field studies (Table 3.8), a sensitivity analysis of f_{oxc} (closed system) and f_{oxo} (open system) (presented in Table 4.3) to changes in α_{ox} ± standard deviation (σ) has been carried out. The data given in Table 4.4 show that f_{oxc} decreases by an average of 19 % when adopting $\alpha_{ox}+\sigma$ while it increases by an average of 31 % with $\alpha_{ox}-\sigma$. The sensitivity analysis for f_{oxo} showed slightly greater variations. When $\alpha_{ox}+\sigma$ is adopted, f_{oxo} decreased by 23 % and increased by 36 % with $\alpha_{ox}-\sigma$. In the case of samples of the shallow subsurface of lysimeters C and D as well as the gas distribution layer of lysimeter A, the adoption of $\alpha_{ox}-\sigma$ resulted in oxidation efficiencies higher than 100 %, suggesting that the $\alpha_{0x}-\sigma$ value (1.019) is too low. In general, the closed system interpretation (f_{ox}) was almost always less than the open system interpretation (f_{oxo}) with the highest deviation of 45 % (see Table 4.3 and Table 4.4). It can be concluded that the closed system approach yields a lower limit appraisal of methane oxidation (Chanton et al., 2009).

Although $\alpha_{ox}\pm\sigma$ represents a slight variation in the adopted α_{ox} (0.8 %), it has a significant influence on the oxidation efficiency of the different biocovers. This result suggests a certain degree of uncertainty using isotope analyses to quantify microbial

CH_4 oxidation of landfill covers. Therefore, oxidation efficiency analyses should be interpreted with caution. Another source of concern is the natural variability in α_{ox}. In the literature the isotopic fractionation factor varies considerably (Templeton et al., 2006; Chanton and Liptay, 2000; Chanton et al., 1999) depending on soil temperature and other environmental factors. Consequently, α_{ox} should be determined at each landfill site and within each soil type at a site to minimise this source of uncertainty (Chanton et al., 2008b).

Table 4.4: Results of sensitivity analysis of f_{oxc} (closed system) and f_{oxo} (open system) to changes in α_{ox} for the sampling date 21.10.2009

	closed system					**open system**				
lysimeter and depth [cm]	mean f_{oxc} [%]	f_{oxc} [%] (with $\alpha_{0x}+\sigma$)	% difference in relation to f_{oxc}	f_{oxc} [%] (with $\alpha_{0x}-\sigma$)	% difference in relation to f_{oxc}	mean f_{oxo} [%]	f_{oxo} [%] (with $\alpha_{0x}+\sigma$)	% difference in relation to f_{oxo}	f_{oxo} [%] (with $\alpha_{0x}-\sigma$)	% difference in relation to f_{oxo}
lysimeter A										
110	58.3	48.2	17.4	73.2	-25.6	81.6	60.8	25.5	100 (123.9)	-22.6
160	17.0	13.3	22.0	23.8	-40.2	17.2	13	24.3	25.4	-47.3
210	0	0		0		0	0		0	
lysimeter B										
160	1.2	0.9	23.1	1.8	-43.9	1.1	0.9	23.8	1.7	-45.5
210	0	0		0		0	0		0	
lysimeter C										
25	59.7	51.4	13.9	70.9	-18.8	84.5	66.5	21.3	100 (115.8)	-18.4
50	61.9	53.3	13.9	73.4	-18.5	89.9	70.4	21.7	100 (124.2)	-11.3
75	38.9	32.1	17.5	49.4	-27.0	45.6	35.6	22.0	63.5	-39.3
110	16.6	13.3	19.9	22.1	-33.7	16.7	13.0	22.0	23.3	-39.3
160	8.5	6.7	21.6	11.8	-38.8	8.2	6.3	23.0	11.6	-42.4
210	0	0		0		0	0		0	
lysimeter D										
25	54.7	46.4	15.2	66.5	-21.5	74.4	58.1	22.0	100 (103.5)	-34.4
50	48.6	40.6	16.5	60.4	-24.3	62.4	48.5	22.3	87.6	-40.4
60	21.1	16.9	19.8	28.1	-33.2	22.1	17.2	22.3	31.0	-40.4
110	5.5	4.3	22.5	7.8	-41.9	5.3	4.0	23.6	7.6	-44.8
160	0	0		0		0	0		0	

Average α_{0x}=1.027;($\alpha_{0x}+\sigma$)=1.035; ($\alpha_{0x}-\sigma$)=1.019

5 CONCLUSIONS

A main part of the present thesis was to develop a measuring system to quantify remaining CH_4 and CO_2 surface fluxes from the different biocovers. It was decided to build an accumulation chamber based on the closed dynamic chamber technique covering almost the whole area of the lysimeter (4 m²). The challenge was to include the aeration pipe (D = 6 cm) of the lysimeter into the design of the accumulation chamber without disturbing the measurement procedure. In contrast, chambers with a basal sampling area of < 1 m² are the most common tools. In order to verify the performance and accuracy of the measuring system, laboratory tests were conducted under different known CH_4 and CO_2 fluxes. Following the laboratory validation phase, it can be concluded that reliable CH_4 and CO_2 emission rates can be obtained in the range of 0.01 g CH_4/m²d (lower detection limit) to 35 g CH_4/m²d and 1 – 47 g CO_2/m²d, respectively. However, absolute emission data should be regarded with caution due to inhibition of fluxes through concentration and pressure build-up in closed chambers. In the present case, the measurement of surface fluxes both under open and closed vent tubes revealed high sensitivity to small pressure gradients between the chamber headspace and the external environment. In addition, altering emission rates have been observed in response to wind events. Therefore, measurement campaigns were done during periods with wind speed < 4 m/s.

Among the four biocovers studied, CH_4 emission fluxes could only be detected from the biocovers with mineral soil covers (lysimeters C and D), whereas no CH_4 emissions were measured on the biocovers with mature sewage sludge compost and sand/compost mixture, respectively (lysimeters A and B). It should be noted that methane production developed differently in the solid waste layers and methane load of lysimeters A and B did not indicate the same range as in lysimeters C and D. According to stable isotope analyses and gas composition profiles of lysimeters A and B, most of the methane oxidation processes were going on below the profile in the gas distribution layer and in the waste. There was no methane detectable in the substrate layer as well as on the surface of lysimeters A and B. However, the mature sewage sludge compost (SSC) placed in lysimeter A showed, in principle very promising results regarding the optimal ambient conditions for methanotrophic bacteria. Beside the nutritional factors, the physical properties such as high porosity and water-holding capacity as well as low thermal conductivity of the compost material provided an adequate methane oxidation milieu. High nitrogen and oxygen concentrations suggest that the substrate cover was well aerated. Moreover, the SSC-cover was capable of retaining the moisture content and the temperature profiles at the optimum level as suggested by other research studies. Compared to the other substrates investigated in this study, sewage sludge compost provided also better conditions to foster the growth of plants, especially during spring.

Following the surface flux measurements, methane emissions from lysimeter D (0.9 – 8.2 g/m²d) were significantly greater than fluxes from lysimeter C (0 – 1.1 g/m²d), although the methane load was probably lower (as indicated by lower CH_4 concentrations in the solid waste layer). This result indicates that low-dimensioned biocovers do not achieve as high CH_4 oxidation capacities as properly designed biocovers. However, a relative comparison of the emission data is limited due to the varying methane development in the solid waste layers of the lysimeters. In addition, results obtained by the surface flux measurements should be interpreted with caution, not only for CH_4 but also CO_2. From FID screenings it is known that higher

surface CH_4 concentrations are seen along the edges of the lysimeters C and D. Although the closed dynamic chamber had an attached plastic skirt for sealing purposes, a great portion of methane and carbon dioxide fluxes may have bypassed leading to an underestimation of CH_4 and CO_2 emissions. In general, it is assumed that the chamber area of 3.61 m² allowed small point sources, such as cracks and fissures to be addressed more reliably.

It is hypothesised that the high moisture content in the mineral soil layers may have blocked gas transport causing preferential flows along the edges of the lysimeters. In contrast, gas permeability tests conducted at the Austrian Institute of Technology in Seibersdorf revealed that all substrates used in this study should ensure sufficient air permeability even at 100 % WHC. However, the preferential flows may also be an artefact of the lysimeter facility itself. In general, loamy soils tend to crack formation, which may have additionally supported the preferential channels.

The study revealed that the magnitude of methane flux into the cover material is a main controlling factor on methane oxidation. When the landfill gas composition in the solid waste layer of lysimeter C was similar to those in lysimeter A, no CH_4 emissions could be measured. For design purposes, it is not only useful to quantify the maximum methane oxidation capacity of a cover system under specific climatic conditions but also to know the flux at which the cover can oxidise 100 % of the methane coming from below (Abichou et al., 2010).

Since already a small vegetation height had an influence on CO_2 emissions measurements, in future the chamber height should be adaptable on vegetation's condition. In order to compensate the resulting increased chamber volume, four fans could be spaced uniformly near the bottom of the enclosed chamber, with air flow directed diagonally upward. In addition, the chamber could be equipped with both a transparent and an opaque cover to investigate the influence of vegetation on CO_2 emission measurement. The use of a combined dark-light measurement enables an estimation of the CO_2 assimilation of the vegetation enclosed in the chamber. Moreover, a background measurement and alignment of relative humidity change on CO_2 concentration in the closed dynamic chamber should be done in order to crosscheck the disturbance.

Stable isotope analyses performed at the four lysimeters indicate that almost all biocover systems were able to reduce CH_4 emissions. The LFG production in lysimeter B was probably too low in order to see an evidence for microbial methane oxidation via stable isotope analysis. The estimated fractions of oxidised CH_4 correspond well with the results obtained by column experiment under laboratory conditions. A considerable percentage of CH_4 was oxidised in the mineral soil covers placed in lysimeters C and D. However, the percentage that is emitted in a more homogeneous way determines methane oxidation (Oonk, 2010). Since a significant portion of the methane load was most probably emitted through preferential flows along the edges of the lysimeters, the oxidation based on stable isotope analyses may was overestimated. Due to the probably low methane load in lysimeter A, methane has already been reduced at a great portion in the gas distribution layer before reaching the SSC-cover. However, under laboratory conditions, the mature compost material mitigated loadings up to 350 l CH_4/m^2d indicating a CH_4 degradation rate of more than 95 %. Because the actual fractionation factor (α_{ox}) for microbial oxidation was not determined specifically for this study, a sensitivity analysis of f_{oxc} (closed system interpretation) and f_{oxo} (open system interpretation) to changes in $\alpha_{ox} \pm$ standard deviation (σ) has been carried. Since small differences in

the adopted α_{ox} had measurable effects on the oxidation efficiency of the different biocovers, results should be interpreted with caution, however, not only in this study.

It can be concluded that a mature, well-structured compost material underlain by a coarse gas distribution layer represent an attractive alternative of the reduction of methane coming out from landfills, particularly for mechanically and biologically preteated (MBT) waste or old and small landfills with low CH_4 production potential.

The next challenge will be to explore the efficiencies of the different biocovers regarding water infiltration and methane mitigation in combination with in-situ aeration.

Future studies should also focus on the complex interaction of methanotrophic bacteria with plants species obtaining aerenchyma in anoxic soils (e.g. Typhaceae) in order to investigate their relevance for CH_4 oxidation.

6 INDEXES

6.1 Literature

Abichou T., Powelson D., Chanton J. (2004): Bio-Reactive Cover Systems, Florida Center for Solid and Hazardous Waste Management, University of Florida, Report #04-0232006

Abichou T., Chanton J., Powelson D., Fleiger J., Escoriaza Y. L., Stern J. (2006): Methane flux and oxidation at two types of intermediate landfill covers, Waste Management 26, p. 1305–1312

Abichou T., Johnson T., Mahieu K., Chanton J., Romdhane M., Mansouri I. (2010): Developing a Design Approach to Reduce Methane Emissions from California Landfills, GeoFlorida 2010: Advances in Analysis, Modelling & Design, Proceedings of the GeoFlorida 2010 Conference

Abushammala M. F. M., Basri N. E. A., Kadhum A. A. H. (2009): Review on Landfill Gas Emission to the Atmosphere, European Journal of Scientific Research 30, No. 3, pp. 427-436

Ait-Benichou S., Jugnia L.-B., Greer C., Cabral A. R. (2009): Methanotrophs and methanotrophic activity in engineered landfill biocovers, Waste Management 29, p. 2509–2517

AIT Seibersdorf (2008): Internal Report

Albanna M., Fernandes L, Warith M. (2007): Methane oxidation in landfill cover soils; the combined effects of moisture content, nutrient addition, and cover thickness, J. Environ. Eng. Sci. 6, p. 191 – 200

Angel R. and Conrad R. (2009): In situ measurement of methane fluxes and analysis of transcribed particulate methane monooxygenase in desert soils, Environmental Microbiology 11(10), p. 2598–2610

Arp J. W. (1991): Effects of source-sink relations on photosynthetic acclimation to elevated CO_2, Plant, Cell and Environment 14, p. 869 – 875

Asman, W.A.H.; Andreae, M.O.; Conrad, R.; Denmead, O.T.; Ganzeveld, L.N.; Helder, W.; Kaminski, T.; Sofiev, M.A.;Trumbore, S. (1999): How can fluxes of trace faces be validated between different scales?, published in Bouwman, A.F.(ed.): Approaches to Scaling of Trace Gas Fluxes in Ecosystems, Developments in Atmospheric Science 24

Bain W. G., Hutyra L., Patterson D. C., Bright A. V., Daube B. C., Munger J. W., Wofsy S. C. (2005): Wind-induced error in the measurement of soil respiration using closed dynamic chambers, Agricultural and Forest Meteorology 131, p. 225–232

Bauer S. (1997): Messung der Methanemissionen von Deponien, Diploma Thesis at the Insitut of Waste Management, University of Natural Resources and Applied Life Sciences, Vienna

Barker J. and Fritz P. (1981): Carbon isotope fractionation during microbial methane oxidation, Nature 293, p. 289–291.

Barlaz M., Green R., Chanton J., Goldsmith C., Hater G. (2004): Evaluation of a biologically active cover for mitigation of landfill gas emissions, Environmental Science and Technology 38, p. 4891–4899.

Bedard C. and Knowles R. (1989): Physiology, biochemistry and specific inhibitors of CH4, NH4 and CO oxidation by methylotrophs and nitrifiers, Microbiol. Rev. 53, p. 58–84

Bergamaschi P., Lubina C., Konigstedt R., Fischer H., Veltkamp A. C., Zwaagstra O. (1998): Stable isotopic signatures (delta C-13, delta D) of methane from European landfill sites, Journal of Geophysical Research-Atmospheres 103, p. 8251 – 8263

Berger J., Fornés L. V., Ott C., Jager J., Wawra B., Zanke U. (2005): Methane oxidation in a landfill cover with capillary barrier, Waste Management 25, p. 369–373

Boeckx P. and van Cleemput O. (1996): Methane oxidation in a neutral cover soil: influence of moisture content, temperature, and nitrogen-turnover, J Environ Qual 25, p. 178–183

Boeckx P., van Cleemput O., Villaralvo I. (1996): Methane emission from a landfill and the methane oxidation capacity of its covering soil, Soil Biology and Biochemistry 28, No. 10/11, p. 1397-1405

Boeckx P., van Cleemput O., Villaralvo I. (1997): Methane oxidation in soils with different textures and land use, Nutrient Cycling in Agroecosystems 49, p. 91–95

Boeckx P. and van Cleemput O. (2000): Methane oxidation in landfill cover soils, in Singh S. N. (ed): Trace Gas Emissions and Plants, Kluwer Academic Publishers, Dordrecht, p. 197–213

Bodelier P.L.E., Laanbroek H.J. (2004): Nitrogen as a regulatory factor of methane oxidation in soils and sediments. FEMS Microbiology Ecology 47, p. 265–277

Bogner J., Spokas K., Burton R., Sweeney R., Corona V. (1995): Landfills as atmospheric methane sources and sinks, Chemosphere, Vol. 31, No. 9, pp. 4119-4130,

Bogner J., Meadows M., Czepiel P. (1997a): Fluxes of methane between landfills and the atmosphere, Natural and engineered controls, Soil Use Manage. 13, p. 268-277

Bogner J., Spokas K. A., Burton E. A. (1997b): Kinetics of Methane Oxidation in a Landfill Cover Soil: Temporal Variations, a Whole-Landfill Oxidation Experiment, and Modelling of Net CH4 Emissions

Bogner J., Spokas K., Chanton J., Powelson D., Fleiger J., Abichou T. (2005): Modelling landfill methane emissions from biocovers: a combined theoretical-empirical approach, Proceedings Sardinia 2005, Tenth International Waste Management and Landfill Symposium 3 - 7 October 2005, Italy, CISA Publisher

Bogner J., Chanton J. P., Blake D., Abichou T., Powelson D. (2010): Effectiveness of a Florida Landfill Biocover for Reduction of CH4 and NMHC Emissions, Environ. Sci. Technol. 44, Vol. 4, p. 1197–1203

Bohn S. and Jager J. (2009): Microbial methane oxidation in landfill top covers – process study on an MBT landfill, Proceedings Sardinia 2009, Twelfth International Waste Management and Landfill Symposium 5 - 9 October 2009, Italy, CISA Publisher

Börjesson G. and Svensson B. H (1997): Seasonal and diurnal methane emissions from a landfill and their regulation by methane oxidation, Waste Manage. Res. 15, p. 33-54

Börjesson G. Sundh I., Tunlid A., Frostegard A., Svensson B. (1998): Microbial oxidation of CH4 at high partial pressures in an organic landfill cover soil under different moisture regimes, FEMS, Microbiol. Ecol. 26, p. 207 – 217

Börjesson G., Chanton J., Svensson B. (2001): Methane oxidation in two Swedish landfill cover soils determined with the use of $_{13}C/_{12}C$ isotope ratios, Journal of Environmental Quality 30, p. 369–376

Börjesson G., Sundh I., Svensson B. (2004): Microbial oxidation of CH4 at different temperatures in landfill cover soils. FEMS Microbiol Ecol 48, p. 305–312

Börjesson G., Samuelsson J., Chanton J. (2007): Methane Oxidation in Swedish Landfills Quantified with the Stable Carbon Isotope Technique in Combination with an Optical Method for Emitted Methane, Environ. Sci. Technol. 2007, 41, p. 6684-6690

Bosse U. And Frenzel P. (1997): Activity and Distribution of Methane-Oxidizing Bacteria in Flooded Rico Soil Microcosms and in Rice Plants (Oryza sativa), Applied and Environmental Microbiology, Vol. 63, No. 4, p. 1199 – 1207

Bradford M. A., Ineson P., Wookey P. A., Lappin-Scott H. M. (2001): Role of CH4 oxidation, production and transport in forest soil CH4 flux, Soil Biology & Biochemistry 33, p. 1625-1631

Butterbach-Bahl K., Kock M., Willibald G., Hewett B., Buhagiar S., Papen H., Kiese R. (2004): Temporal variations of fluxes of NO, NO2, N2O, CO2, and CH4 in a tropical rain forest ecosystem, Global Biogeochemical Cycles 18 (3), pp. GB3012 1-11

Cabral A. R., Arteaga K., Rannaud D., Ait-Benichou A., Pouet M.-F., Allaire S., Jugnia L. B., Greer C. (2007): Analysis of methane oxidation and dynamics of methanotrophes within a passive methane oxidation barrier, Proceedings Sardinia 2007, Eleventh International Waste Management and Landfill Symposium 1 - 5 October 2007, Italy, CISA Publisher

Cabral A. R., Capanema M. A., Gebert J., Moreira J. F., Jugnia L. B. (2009): Quantifying Microbial Methane Oxidation Efficiencies in Two Experimental Landfill Biocovers Using Stable Isotopes, Water Air Soil Pollution, Springer Science

Camarda M., Gurrieri S., Valenza M. (2009): Effects of soil gas permeability and recirculation flux on soil CO2 flux measurements performed using a closed dynamic accumulation chamber, Chemical Geology 265, p. 387–393

Cardellini C., Chiodini G., Frondini F., Granieri D., Lewicki J., Peruzzi L. (2003): Accumulation chamber measurements of methane fluxes: application to volcanic-geothermal areas and landfills, Applied Geochemistry 18, p. 45–54

Castro M. S., Mellilo J. M., Steudler P. A., Chapman J. W. (1994): Soil moisture as a predictor of methane uptake by temperate forest soils, Canadian Journal of Forest Research 24, p. 1805 – 1810

Chan Y. S. G., Chu L. M., Wong M. H. (1997): Influence of landfill factors on plants and soil fauna – an ecological perspective, Environmental Pollution 97, No. 1-2, p. 39-44

Chanton J. P. (2005): The effect of gas transport on the isotope signature of methane in wetlands, Organic Geochemistry 36, p. 753–768

Chanton J. P., Rutkowski C. M., Mosher B. (1999): Quantifying methane oxidation from landfills using stable isotope analysis of downwind plumes, Environmental Science and Technology 33(21), p. 3755–3760.

Chanton, J. and K. Liptay. (2000): Seasonal variation in methane oxidation in landfill cover soils as determined by an in situ stable isotope technique, Global Biogeochem. Cycles 14, p. 51–60

Chanton J. P., Powelson D. K., Abichou T., Hater G. (2008): Improved Field Methods to Quantify Methane Oxidation in Landfill Cover Materials Using Stable Carbon Isotopes, American Chemical Society, Environmental Science and Technology 42, No. 3, p. 665-670

Chanton J. P., Powelson D. K., Abichou T., Fields D., Green R. (2008b): Effect of Temperature and Oxidation Rate on Carbon-isotope Fractionation during Methane Oxidation by Landill Cover Materials, Environmental Science and Technology 42, p. 7818 – 7823

Chanton J. P., Powelson D. K., Green R B. (2009): Methane Oxidation in Landfill Cover Soils, is a 10% Default Value Reasonable?, J. Environ. Qual. 38, p. 654–663

Chen I-C., Hegde U., Chang C-H., Yang Sh-Sh. (2008): Methane and carbon dioxide emissions from closed landfill in Taiwan, Chemosphere 70, p. 1484–1491

Chiodini G., Cioni R., Guidi M., Raco B., Marini L. (1998): Soil CO2 flux measurements in volcanic and geothermal areas, Applied Geochemistry 13, No. 5, p. 543-552

Christophersen M., Linderod L., Jensen P. E., Kjeldsen, P. (2000): Methane oxidation at low temperatures in soil exposed to landfill gas, J. Environ. Qual., 29, p. 1989-1997

Christophersen M. and Kjeldsen P. (2001): Lateral gas transport in soil adjacent to an old landfill: factors governing gas migration, Waste Manage. Res. 19, p. 144 – 159

Christophersen M., Kjeldsen P., Holst H., Chanton J. (2001): Lateral gas transport in soil adjacent to an old landfill: factors governing emissions and methane oxidation, Waste Manage. Res, 19, p. 126 – 143

Cossu R., Raga R., Zane, M. (2003): Methane Oxidation and Attenuation of Sulphurated Compounds in Landfill Top Cover Systems: Lab-Scale Test, Proceedings Sardinia, Ninth International Waste Management and Landfill Symposium, S. Margherita di Pula, Cagliari, 6–10 October, Italy, CISA Publisher

Czepiel P.M., Shorter J.H., Mosher B., Allwine E., McManus J.B., Harriss R.C., Kolb C.E., Lamb B.K. (2003): The influence of atmospheric pressure on landfill methane emissions, Waste Management 23, p. 593 – 598

Damman B., Streese J.,Stegmann, R. (1999): Microbial oxidation of methane from landfills in biofilters, In Proceedings Sardinia 1999, Seventh International Waste Management and Landfill Symposium, vol. 2, Cagliari, Italy, 4B8 October 1999, p. 517–524.

Denmead O. T. (1979): Chamber systems for measuring nitrous oxide fluxes in the field, Soil Sci Soc Am J 43, p. 89–95

Denmead O. T. (2008): Approaches to measuring fluxes of methane and nitrous oxide between landscapes and the atmosphere, Plant Soil 309, p. 5–24

De Visscher A. D. and van Cleemput (2003): Induction of enhanced CH4 oxidation in soils: NH4 inhibition patterns, Soil Biology & Biochemistry 35, p. 907–913

De Visscher A., De Poureq I., Chanton, J. (2004): Isotope fractionation effects by diffusion and methane oxidation in landfill cover soils, Journal of Geophysical Research, 109 (D18), 8

De Visscher A., Schippers M., Van Cleemput O. (2001): Short-term kinetic response of enhanced methane oxidation in landfill cover soils to environmental factors, Biology and Fertility of Soils 33, p. 231–237

De Visscher A., Thomas D., Boecks P., Van Cleemput O. (1999): Methane oxidation in simulated landfill cover soil environments, Environmental Science and Technology 33, p. 1854-1859

Dunfield P. F., Liesack W., Henckel T., Knowles R., Conrad R. (1999): High-Affinity Methane Oxidation by a Soil Enrichment Culture Containing a Type II Methanotroph, Applied and Environmental Microbiology 65, No. 3, p. 1009–1014

Einola J.-K. M., Kettunen R. H., Rintala J. A. (2007): Responses of methane oxidation to temperature and water content in cover soil of a boreal landfill, Soil Biology & Biochemistry 39, p. 1156–1164

Einola J.-K. M., Karhu A. E., Rintala J. A. (2008): Mechanically–biologically treated municipal solid waste as a support medium for microbial methane oxidation to mitigate landfill greenhouse emissions, Waste Management 28, p. 97–111

Fang C. and Moncrieff J. B. (1998): An open-top chamber for measuring soil respiration and the influence of pressure difference on CO2 efflux measurement, Functional Ecology 12, p. 319–325

Figueroa R. (1993): Methane Oxidation in landfill top soils, Proceedings Sardinia 1993, Fourth International Waste Management and Landfill Symposium 11 - 15 October 1993, Italy, CISA Publisher

Fowler, D. (1999): Experimental designs appropriate for flux determination in terrestrial and aquatic ecosystems, published in Bouwman, A.F.(ed.): Approaches to Scaling of Trace Gas Fluxes in Ecosystems, Developments in Atmospheric Science 24

Galle B., Klemendtson L., Griffith D. W. (1994): Application of an FTIR system for measurement of N2O fluxes using micrometeorological methods, an ultralarge chamber system and conventional field chambers, Journal of Geophysical Research 99, p. 16575-16583

Gamperling O., Huber-Humer M., Wimmer B. (2008): Alternative Top-Cover System for Treating Weak Dwindling Gas Emissions from in-situ Aerated MSW-landfills, In Barlaz M., Lagerkvist A., Matsuto T.: The Fifth Intercontinental Landfill Research Symposium, Copper Mountain Conference Center 10. – 12. September 2008, Colorado USA

Gao F., Yates S. R., Yates M. V., Gan J., Ernst F. F. (1997): Design, Fabrication, and Application of a Dynamic Chamber for Measuring Gas Emissions from Soil, Environmental Science & Technology 31, No. 1, p. 148-153

Gebert J., Gröngröft A., Miehlich G. (2003): Kinetics of microbial landfill methane oxidation in biofilters, Waste Management 23, p. 609–619

Gebert J. and Gröngröft A. (2006): Passive landfill gas emission – influence of atmospheric pressure and implications for the operation of methane oxidising-biofilters, Waste Management 26, p. 245-251

Gebert J. and Gröngröft A. (2006b): Performance of a passively vented field-scale biofilter for the microbial oxidation of landfill methane, Waste Management 26 (4), p. 399 – 407

Gebert J., and Gröngröft A. (2009): Role of soil gas diffusivity for the microbial oxidation of methane in landfill covers. In: Proceedings Sardinia 2009, Twelfth International Waste Management and Landfill Symposium. S. Margherita di Pula, Cagliari, Italy; 5 – 9 October 2009, CISA, Environmental Sanitary Engineering Centre, Italy

Gebert J. and Streese-Kleeberg J. (2008): Methanoxidation an der Deponieoberfläche, Deponietechnik, Hamburger Berichte 30

Gerzabek M. H. And Reichenauer T. G. (2006): Innovative in-situ Methoden zur Sicherung und Sanierung von Altablagerungen und Altstandorten, Facultas Universitätsverlag, Wien

Hamamoto S., Moldrup P., Kawamoto K., Komatsu T., Rolston D. E. (2009): Unified measurement system for the gas dispersion coefficient, air permeability, and gas diffusion coefficient in variably saturated soil, SSSAJ 73, No. 6, p. 1921 – 1930

Hanson R. S. and Hanson T. E. (1996): Methanotrophic Bacteria, Mirobio. Rev. 60, No. 2, p. 439-471

Hartless R. (1995): Measuring pressures and flow of landfill gas, In: Regulation environmental impact and aftercare, Proceedings Sardinia 1995, Fifth International Waste Management and Landfill Symposium 2 - 6 October 1995, Italy, CISA Publisher, p. 517–531

Haubrichs R. (2007): Entwicklung eines verteil-geregelt belüfteten Filtersystems zur biologischen Behandlung von methanhaltigen Deponieschwachgasen, Forum Siedlungswasserwirtschaft und Abfallwirtschaft Universität Duisburg – Essen, Heft 31, Shaker Verlag

Haubrichs R. and Widmann R. (2006): Evaluation of aerated biofilter systems for microbial methane oxidation of poor landfill gas, Waste Management 26, p. 408 - 416

He R., Ruan A., Jiang C., Shen D-S. (2008): Responses of oxidation rate and microbial communities to methane in simulated landfill cover soil microcosms, Bioresource Technology 99, p. 7192–7199

Hilger H., Wollum A., Barlaz M. (2000): Landfill Methane Oxidation Response to Vegetation, Fertilization, and Liming, J. Environ. Qual. 29, p. 324 – 334

Hilger H. and Humer M. (2003): Biotic landfill cover treatment for mitigation methane emissions, Environmental Monitoring and Assessment 84, Kluwer Academic Publishers, the Netherlands, p. 71–84

Huber-Humer M. (2004): Abatement of landfill methane emissions by microbial oxidation in biocovers made of compost, Doctoral thesis, University of Natural Resources and Applied Life Sciences Vienna, Institute of Waste Management

Huber-Humer M. and Lechner P. (2003): Effect of methane oxidation on the water balance of the landfill cover and the vegetation layer. Proceedings Sardinia 2003, Ninth International Waste Management and Landfill Symposium 2 - 6 October 2003, Italy, CISA Publisher, p. 517–531

Huber-Humer M. and Lechner P. (2007): Scientific Report: ESF Exploratory Workshop on "Mitigation of Methane Emissions through microbial oxidation on landfills – evaluation and quantification approaches, European Science Foundation – Setting Science Agendas for Europe

Huber-Humer M., Amann A., Bogolte T., Dos Santos M., Hagenauer I., Pauliny W., Reichenauer T., Watzinger A. and Wimmer B. (2008a): Technischer Leitfaden – Methanoxidationsschichten, Erstellt im Rahmen der ÖVA-Arbeitsgruppe „Leitfaden Methanoxidationsschichten"

Huber-Humer M., Gebert J., Hilger H. (2008b): Biotic systems to mitigate landfill methane emissions, Waste Management & Research 26, p. 33–46

Huber-Humer M. and Lechner P. (2009): Biocover construction and monitoring – implementation criteria and processes, Proceedings Sardinia 2009, Twelfth International Waste Management and Landfill Symposium 5 - 9 October 2009, Italy, CISA Publisher

Huber-Humer M., Röder S., Lechner P. (2009): Approaches to assess biocover performance on landfills, Waste Management 29, p. 2092–2104

Humer M. and Lechner P. (1999a): Alternative approach to the elimination of greenhouse gases from old landfills, Waste Management and Research 17, p, 443–452

Humer M. and Lechner P. (1999b): Methane Oxidation in Compost Cover Layers on Landfills, Proceedings of the Seventh International Waste Management and Landfill Symposium, 4–8 October, S. Margherita di Pula, Cagliari, Sardinia, Italy

Humer M. and Lechner P. (2001a): Design of a landfill cover layer to enhance methane oxidation – results of a two year field investigation, Proceedings Sardinia 2001, Eighth International Waste Management and Landfill Symposium 1 - 5 October 2001, Italy, CISA Publisher

Humer M. and Lechner P. (2001b): Microoganisms against the greenhouse effect – suitable cover layers for the elimination of methane emissions from landfills, In: Proc. Of the Solid Waste Association of North America, 6th Annual Landfill Symposium, June 18-22, San Diego, CA

Humer M. and Lechner P. (2001c): Microbial methane oxidation for the reduction of landfill gas emissions, Journal of Solid Waste Technology and Management 27, p. 146-151

Hutchinson G. L. and Mosier A. R. (1981): Improved Soil Cover Method for Field Measurement of Nitrous Oxide Fluxes, Soil. Sci. Soc. Am. J. 45, p. 311 – 316

Internal Report Nutzraum (2008): Innovative in-situ Methoden zur Sanierung von Altablagerungen und kontaminierten Standorten – 1. Zwischenbericht, coordinated by Reichenauer T., Data from PP1 Huber-Humer M., Huber P., Gamperling O., Wimmer B., Bogolte T., Mellendorf M., Kinner P.

IPCC (2007): Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on

Climate Change [Solomon S., Qin D., Manning M., Chen Z., Marquis M., Averyt K.B., Tignor M., Miller, H.L. (eds.)] Cambridge University Press, Cambridge, UK and New York, NY, USA

IPCC (2007b): Climate change 2007: Synthesis report, Cambridge University Press, Cambridge

Islam T., Jensen S., Reigstad L. J., Larsen O., Birkeland N.-K. (2007): Methane oxidation at 55oC and pH 2 by a thermoacidophilic bacterium belonging to the Verrucomicrobia phylum, PNAS, Vol. 105, No. 1, p. 300 – 304

Jeris J. and McCarty P. (1965): The Biochemistry of Methane Fermentation Using C14 Traces, Journal Water Poll. Control Fed., 37(2), p. 178-192

Jones H. A. and Nedwell D. B. (1993): Methane emission and methane oxidation in landfill cover soil, FEMS Microbiol. Ecol. 102, 185 – 195

Jugnia L.-B., Cabral A.R., Greer C.W. (2008) Biotic methane oxidation within an instrumented experimental landfill cover, Ecological Engineering 33, p. 102–109

Kaiser E.-A., Munch J. C., Heinemeyer O. (1996): Importance of soil cover box area for the determination of N20 emissions from arable soils, Plant and Soil 181, p. 185 – 192

Kettunen R.H., Einola J.-K.M., Rintala J.A. (2006) Landfill methane oxidation in engineered soil columns at low temperature. Water, Air and Soil Pollution 177, p. 313–334

Kightley D., Nedwell D. B., Cooper M. (1995): Capacity for methane oxidation in landfill cover soils measured in laboratory-scale soil microcosms, Applied and Environmental Microbiology, Vol. 61 (2), p. 592 – 601

King G. M. and Schnell S. (1994): Effect of increasing at methane concentration on ammonium inhibition of soil methane consumption, Nature 370, p. 282 – 284

Kjeldsen P. And Fischer E.V. (1995): Landfill gas migration – field investigation at Skellingsted landfill, Denmark, Waste Manage. Res. 13, p. 467 – 484

Kjeldsen P. (1996): Landfill gas migration in soil, In: Christensen T.H., Cossu R., Stegmann R. (eds.): Landfilling of Waste: Biogas, E. & FN Spon. London, UK

Kusa K., Sawamoto T., Hu R., Hatano R. (2008): Comparison of the closed-chamber and gas concentrations gradient methods for measurement of CO2 and N2O fluxes in two upland field soils, Soil Science and Plant Nutrition 54, p. 777 – 785

Latham B. and Young A. (1993): Modellization of the effects of barometric pressure on migrating landfill gas, Proceedings Sardinia 1993, Fourth International Waste Management and Landfill Symposium 11 – 15 October, Italy, CISA Publisher

Lee S.-W., Im J., DiSpirito A. A., Bodrossy L., Barcelona M. J, Semrau J. D. (2009): Effect of nutrient and selective inhibitor amendments on methane oxidation, nitrous oxide production, and key gene presence and expression in landfill cover soils: characterization of the role of methanotrophs, nitrifiers, and denitrifiers, Appl Microbiol Biotechnol 85, p. 389–403

Le Mer J. and Roger P. (2001): Production, oxidation, emission and consumption of methane by soils: A review, Eur. J. Soil Biol. 37, p. 25−50

Lengeler J. W., Drews G., Schlegel H. G. (1999): Biology oft he Prokaryontes, Georg Thieme Verlag, Stuttgart, Germany

Lewis J. D., Wang X. Z., Griffin K. L., Tissue D. T. (2002): Effects of age and ontogeny on photosynthetic responses of a determinate annual plant to elevated CO_2 concentrations, Plant, Cell and Environment 25, p. 359 – 368

Liebner S., Rublack K., Stuehrmann T., Wagner D. (2009): Diversity of Aerobic Methanotrophic Bacteria in a Permafrost Active Layer Soil of the Lena Delta, Siberia, Microb Ecol 57, p. 25–35

Livingston G. P. and Hutchinson G. L.. (1995): Enclosure-based measurement of trace gas exchange: Applications and sources of error, In: Matson P. A. and Harriss R. C. (ed.): Methods in Ecology, Biogenic trace gases: Measuring emissions from soil and water, Blackwell Science, Malden, p. 14–51

Mahieu K., De Visscher A., Vanrolleghem P. A., Van Cleemput O. (2006): Carbon and hydrogen isotope fractionation by microbial methane oxidation: Improved determination, Waste Management 26, p. 389–398

Mainiero R. and Kazda M. (2004): Effects of Carex rostrata on soil oxygen in relation to soil moisture, Plant and Soil 270, p. 311 – 320

Mandernack K.W., Kinney C.A., Coleman D., Huang Y.S., Freeman K.H. & Bogner, J. (2000): The biogeochemical controls of N2O production and emission in landfill cover soils: the role of methanotrophs in the nitrogen cycle, Environmental Microbiology 2, p. 298–309

Maurice C. and Lagerkvist A. (2004): Assessment of the methane oxidation capacity of soil, Waste Management & Research 22, No. 1, p. 42-48

Mor S., De Visscher A., Ravindra K., Dahiya R. P., Chandra A., Van Cleemput O. (2006): Induction of enhanced methane oxidation in compost: Temperature and moisture response, Waste Management 26, p. 381–388

Mosher B. W., Czepiel P. M., Harris R. C., Shorter J. H., Kolb C. E., McManus J. B., Allwine E., Lamb B. K. (1999):.Methane emissions at nine landfill sites in the northeastern United States, Environ. Sci. Technol. 33, p. 2088-2094.

Müller J., Eschenröder A., Diepenbrock (2009): Through-flow chamber CO2/H2O canopy gas exchange system—Construction, microclimate, errors, and measurements in a barley (Hordeum vulgare L.) field, Agricultural and Forest Meteorology 149, p. 214 - 229

Nagendran R., Selvam A., Joseph K., Chiemchaisri C. (2006): Phytoremediation and rehabilitation of municipal solid waste landfills and dumpsites: A brief review, Waste Management 26, p. 1357–1369

Nastev M., Therrien R. Lefebvre R., Gelinas P. (2001): Gas production and migration in landfills and geological materials, Journal of Contaminant Hydrology 52, p. 187–211

Nesbit S. P. and Breitenbeck G. A. (1992): A laboratory study of factors influencing methane uptake by soils." Agric. Ecosystem Environ 41, p. 39-54

Oonk H. (2010): Literature review: Methane from landfills - Methods to quantify generation, oxidation and emission, OonKAY!, Innovations in Environmental Technology, Online under URL: http://www.sustainablelandfillfoundation.eu/documenten/100428%20Final%20report%20-%20review%20landfill%20methane%20SLF.pdf, 06/2010

Pawlowsky M., Stepniewski W., Czerwinski J. (2003): The effect of texture on methane oxidation capacity in a sand layer – a model laboratory study, Environmental Engineering Studies, Polish Research on the Way to the EU, Kluwer, Academic/Plenum Publishers, p. 339 – 354

Perera M. D. N., Hettiaratchi J. P. A., Achari G. (2002): A mathematical modeling approach to improve the point estimation of landfill gas surface emissions using the flux chamber technique, J. Environ. Eng. Sci. 1, p. 451–463

Pokryszka Z., Tauziede C., Cassini P. (1995): Development and validation of a metod for measuring biogas emissions Rusing a dynamic chamber, Proceedings Sardinia 1995, Fifth International Waste Management and Landfill Symposium 2 - 6 October 1995, Italy, CISA Publisher

Poulsen T. G. (2005): Impact of wind turbulences on landfill gas emissions, Proceedings Sardinia 2005, Tenth International Waste Management and Landfill Symposium 3 - 7 October 2005, Italy, CISA Publisher

Reay D. S. and Nedwell D. B. (2004): Methane oxidation in temperate soils: effects of inorganic N, Soil Biology & Biochemistry 36, p. 2059–2065

Reay D. S., Nedwell, D. B., McNamara N., Ineson P. (2005): Effect of tree species on methane and ammonium oxidation capacity in forest soils, Soil Biology & Biochemistry 37, p. 719–730

Reay D. S., Smith K. A., Hewitt C. N. (2007): Methane: importance, source and sinks. In: Reay D. S., Hewitt C. N., Grace J. (eds): Greenhouse gas sinks, Wallingford, Oxfordshire, p. 143–151

Reichmann R. and Rolston D. E. (2002): Atmospheric Pollutants and Trace Gases; J. Environ. Qual. 31: p. 1774 – 1781

Röder S., Huber-Humer M., Lechner P. (2004): Efficiency and Monitoring of Methane Oxidation in Bio-Covers on Landfills, In: Verstraete W. (ed.): Proceedings of the European Symposium on Environmental Biotechnology – ESEB 2004, 25-28 April 2004, Oostende, Belgium, p. 147 – 150

Savanne D., Cassini P., Pokryszka Z., Tauziede C., Tregoures A., Berne P., Sabroux J. C., Cellier P., Laville P. (1995): A comparison of methods for estimating CH4 emission from MSW landfills, Proceedings Sardinia 1995, Fifth International Waste Management and Landfill Symposium 2 - 6 October 1995, Italy, CISA Publisher

Scheffer P. and Schachtschabel P. (1992): Lehrbuch der Bodenkunde, 13. Auflage, Ferdinand Enke Verlag, Suttgart

Scheutz C. and Kjeldsen P. (2004): Environmental Factors Influencing Attenuation of Methane and Hydrochlorofluorocarbons in Landfill Cover Soils, J. Environ. Qual. 33, p. 72–79

Scheutz C., Kjeldsen P., De Visscher A., Gebert J., Hilger H. A., Huber-Humer M., Spokas K. (2009): Microbial methane oxidation processes and technologies for mitigation of landfill gas emissions, Waste Management & Research 27, Issue 5, p. 409-455

Scheutz C., Fredenslund A. M., Pedersen G. B., Pedicone A., Kjeldsen P. (2009b): Biocover – Evaluation of Methane Oxidation Efficiency of Biocover System, Task 6 of the project: Reduction of Greenhouse Gas Emissions from Landfills by use of Engineered Biocovers, Department of Environmental Engineering, Technical

University of Denmark, URL: http://www2.er.dtu.dk/publications/fulltext/2009/ENV2009-078.pdf, 03/2010

Scheutz C., Pedicone A., Kjeldsen P. (2009c): Evaluation of respiration based CO2 emissions from compost landfill covers, Proceedings Sardinia 2009, Twelfth International Waste Management and Landfill Symposium, 5-9 October 2009, Sardinia, Italy

Sitaula B., Bakken L., Abrahamsen G. (1995): CH4 uptake by temperate forest soil: effect of N input and soil acidification. Soil. Biol. Biochem. 27, Vol. 7, p. 871–880

Smith K. A., Clayton H., Arah J. R. M., Christensen S., Ambus P., Fowler D., Hargreaves K. J., Skiba U., Harris G. W., Wienhold F. G., Klemedtsson L., Galle B. (1994): Micrometeorological and chamber methods for measurement of nitrous oxide fluxes between soils and the atmosphere: Overview and conclusion, Journal Geophysical Research 99, p. 16541-16548

Sparks D.L. (1995): Environmental Soil Chemistry, Academic Press, San Diego

Spokas K. A. and Bogner J. E. (2010): Limits and dynamics of methane oxidation in landfill cover soils, Waste management, in press

Stein V. B. and Hettiaratchi J. P. A. (2001): Methane oxidation in three Alberta soils: influence of soil parameters and methane flux rates, Environmental Technology 22, p. 101 - 111

Steinlechner E., Berghold H., Cate F. M., Jungmeier G., Spitzer J., Wutzl Ch. (1994): Möglichkeiten der Vermeidung und Nutzung anthropogener Methanemissionen, Joanneum Research, Institut für Umweltgeologie und Ökosystemforschung

Stern J. C., Chanton J., Abichou T., Powelson D., Yuan L., Escoriza S, Bogner J. (2007): Use of a biologically active cover to reduce landfill methane emissions and enhance methane oxidation, Waste Management 27, p. 1248–1258

Stralis-Pavese N. Sessitsch A., Weilharter A., Reichenauer T., Riesing J., Csontos J., Murrell J. C., Bodrossy L. (2004): Optimization of diagnostic microarray for application in analysing landfill methanotroph communities under different plant covers, Environmental Microbiology 6 (4), p. 347–363

Streese J. and Stegmann R. (2003): Microbial oxidation of methane from old landfills in biofilters, Waste Management 23, p. 573–580

Striegl R. G., Mc Connaughey T. A., Thorstenson D. C., Weeks E. P., Woodward J. C. (1992): Consumption of Atmospheric Methane by Desert Soils, Nature 357, p. 145-147

Tecle D., Lee J., Hasan S. (2009): Quantitative analysis of physical and geotechnical factors affecting methane emission in municipal solid waste landfill, Environ. Geol. 56, p. 1135 - 1143

Templeton A. S., Chu K.-H., Alvarez-Cohen L., Conrad M. E. (2006): Variable carbon isotope fractionation expressed by aerobic CH_4-oxidizing bacteria, Geochim. Cosmochim. Acta 70, p. 1739 - 1752

Thomas H. R. and Ferguson W. J. (1999): A fully coupled heat and mass transfer model incorporating contaminant gas transfer in an unsaturated porous medium, Computers and Geotechnics 24, p. 65-87

Theissen A. R. and Murrell J. C. (2005): Facultative Methanotrophs Revisited, Journal of Bacteriology 187, No. 13, p. 4303-4305

Topp E. And Hanson R. S. (1991): Metabolism of Radiatively Important Trace Gases by Methane-Oxidizing Bacteria, p. 71 – 89, in J. E. Rogers and W. B. Whitman (eds.): Microbial Production and Consumption of Greenhouse Gases – Methane, Nitrogen Oxides, and Halomethanes, American Society for Microbiology, Washington D.C.

Tregoures A., Beneito A., Berne P., Gonze M. A., Sabroux J .C., Savanne D., Pokryszka Z., Tauziede C., Cellier P., Laville P., Milward R., Arnaud A., Levy F., Burkhalter R. (1999): Comparison of seven methods for measuring methane flux at a municipal solid waste landfill site, Waste Manage. & Res. 17, pp. 453-458

Van Huissteden J., Maximov T. C., Kononov A. V., Dolman A. J. (2008): Summer soil CH4 emission and uptake in taiga forest near Yakutsk, Eastern Siberia, Agricultural and Forest Meteorology 148, p. 2006–2012

Vishwakarma P., Dumont M. G., Bodrossy L., Stralis-Pavese N., Murrell J. C., Dubey S. K. (2009): Ecological and molecular analyses of the rhizospheric methanotroph community in tropical rice soil: effect of crop phenology and land-use history, Current Science 96, NO. 8, p. 1082-1089

Visvanathan C., Pokhrel D., Cheimchaisri W., Hettiaratchi J. P. A., Wu J. S. (1999): Methanotrophic activities in tropical landfill cover soils: effect of temperature, moisture content and methane concentration, Waste Management and Research 17, p. 313 – 323

Visvanathan C., Tubtimthia O., Kuruparan P. (2004): Influence of landfill top cover design on methane oxidation: pilot scale lysimeter experiments under tropical conditions, APLAS Kitakyushu 2004, Third Asian-Pacific Landfill Symposium, October 27-29 Kitakyushu, Japan

Von Arnold K., Nilsson M., Hanell B., Weslien P., Klemedtsson L. (2005): Fluxes of CO2, CH4 and N2O from drained organic soils in deciduous forests, Soil Biology & Biochemistry 37, p. 1059–1071

Wang Y., Wu W., Ding Y., Liu W., Perera A., Chen Y., Devare M. (2008): Methane oxidation activity and bacterial community composition in a simulated landfill cover soil is influenced by the growth of Chenopodium album L., Soil Biology & Biochemistry 40, p. 2452–2459

Watzinger A., Reichenauer T.G., Blum W.E.H., Gerzabek M.H., Zechmeister-Boltenstern S. (2005): The effect of landfill leachate irrigation on soil gas composition: methane oxidation and nitrous oxide formation. Water, Air, and Soil Pollution, 164, p. 295–313

Whalen S.C., Reeburgh W.S., Sandbeck K.A. (1990): Rapid methane oxidation in a landfill cover soil, Applied and Environmental Microbiology 56, p. 3405–3411

Whalen, S.C. and Reeburgh, W.S. (1996): Moisture and temperature sensitivity of CH_4 oxidation in boreal soils. Soil Biology & Biochemistry 28, p. 1271–1281

Williams P.T. (2005): Waste Treatment and Disposal, 2nd ed. John Wiley & Sons Ltd, England, pp. 171-244

Wilshusen J. H., Hettiaratchi J. P. A., Stein V. B. (2004): Long-term behaviour of passively aerated compost methanotrophic biofilter columns, Waste Management 24, p. 643 – 653

Wolf H.J. and Hanson R.S. (1980): Isolation and characterization of methane-oxidizing yeasts, Journal of General Microbiology, 114, p. 187–194

Xiaoli C., Ziyang L, Shimaoka T., Nakayama H., Ying Z., Xiaoyan C., Komiya T., Ishizaki T., Youcai Z. (2010): Characteristics of environmental factors and their effects on CH4 and CO2 emissions from a closed landfill: An ecological case study of Shanghai, Waste Management 30, p. 446–451

Yuan L. (2006): Methane emission and oxidation through landfill covers, Doctoral thesis, Florida State University, Department of Civil and Environmental Engineering, USA

Zak M. (2008): Sauerstoffumsatz und mikrobieller Besatz der Rhizome und Wurzeln von Typha, Diplomarbeit an der Universität Ulm, Institut für Systematische Botanik und Ökologie

Zeiss C.A. (2006): Accelerated methane oxidation cover system to reduce greenhouse gas emissions from MSW Landfills in cold, semi, arid regions, Water, Air, and Soil Pollution 176, p. 285–306

Zhang H., He P., Shao L. (2009): N2O emissions at municipal solid waste landfill sites: Effects of CH4 emissions and cover soil, Atmospheric Environment 43, p. 2623–2631

Ziska L. H. And Bunce J. A. (1997): The role of temperature in determining the stimulation of CO_2 assimilation at elevated carbon dioxide concentration in soybean seedlings, Physiologia Plantarum 100, p. 126 – 132

6.2 Figures

6.3 Tables

6.4 Formula Symbols, Indices, Abbreviations

6.4.1 Glossary

Some letters were multiple allocated due to the various literatures.

A	m^2	Surface area covered by the chamber
a	µbar	Pressure wave amplitude
b	µl/cycle	Displaced volume
D	mm	Diameter
D	m^2/s	Diffusion coefficient
dC	ppm, mg/m^3	Change in gas concentration
dt	min, sec	time
dz	cm	Change in sampling depth
F	g/m^2d	Gas emission flux
f	Hz	Frequency
f	%	Oxidised fraction
g	m/s^2	gravity
J	mol/m^2s	Diffusive flux
L	mm	Tube length
p	µbar	Barometric pressure
Q	m^3/h, l/s	Volume flow rate
r	-	Volume ratio
T	℃, K	Temperature
V	m^3	Head space, chamber volume
v	µl/s	Volumetric air flow rate

6.4.2 Greek Symbols

Δ	-	Delta, difference
µ	Pa s	Air viscosity
α	-	Isotopic fractionation factor
δ	-	Isotope ratio
σ	-	Standard deviation

6.4.3 Indices

anox	anoxic
c	Closed system approach
int	intrinsic
m	mean
o	Open system approach
ox	oxidised
trans	transport
z	Sampling deth

6.4.4 Abbreviations

a ..... annual
ALSAG ..... Altlastensanierungsgestz
AIT ..... Austrian Institute of Technology
BGBl ..... Bundesgesetzblatt
BOD ..... biological oxygen demand
BWC ..... biowase compost
C ..... carbon
CDC ..... closed dynamic chamber
CH_4 ..... methane
C:N ..... carbon/nitrogen ratio
CO_2 ..... carbon dioxide
COD ..... chemical oxygen demand
d ..... day
DFT ..... dynamic flux tunnel
DM ..... dry matter
e.g. ..... exempli gratia (= for example)
EPS ..... Exopolimeric substances
FID ..... Flame-Ionisation-Detector
GC ..... gas chromatography
GES ..... gas extraction system
GHG ..... greenhouse gas
h ..... hour
H_2 ..... hydrogen
INERSIS ..... Institut National de l'Environnement et des Risques
IPCC ..... Intergovernmental Panel on Climate Change
kCal ..... kilocalorie
Kg ..... kilogram
l ..... litre
LFG ..... landfill gas
LOI ..... loss on ignition
n.d. ..... not detectable
m ..... meter
MBT ..... mechanical-biological pretreatment
MMO ..... Methane Mono-oxygenase
n ..... number of samples
N_2 ..... nitrogen
O_2 ..... oxygen
ppm ..... parts per million
pMMO ..... particulate Methane Mono-oxygenase

RA ..... respiration activity
SCT ..... static dynamic chamber
SD ..... standard deviation
sMMO ..... soluble Methane Mono-oxygenase
SS ..... subsoil
SSC ..... Sewage sludge compost
Tg ..... Tera gram (=10^{12} g)
TOC ..... total organic carbon
TS ..... topsoil
VOC ..... volatile organic compounds
v/v ..... volume per volume
w/w ..... weight per weight
WHC ..... water holding capacity

APPENDIX

A.1 Gas composition and temperature profiles

02.04.2008

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0,1	4,8	16,3	25	0,00
50	0	9,2	13,2	50	0,00
75	0	14	9,1	75	0,00
110	0	0	0	190	0,00
160	1,5	65	0		
210	2,1	82,5	0		
Kammer B					
25	0	6,8	16,7	25	0,00
50	0,1	10	14	50	0,00
75	0	17	9,9	75	0,00
110	0,1	30	3,8	190	0,00
160	1,4	70	0		
210	1,8	85	0,05		
Kammer C					
25	0	6,5	16,7	25	0,00
50	0	12	12,6	50	0,00
75	0	17,5	9,4	75	0,00
110	0,1	31	2	190	
160	0,3	60	0		
210	0,4	75	0		
Kammer D					
25	0	9,8	14,3	25	0,00
50	0	15	11,2	50	0,00
60	0,2	38	3	140	0,00
110	0,55	62,5	0	190	0,00
160	0,65	75	0		

27.08.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2	19,1	25	23,96
50	0	2,6	18,4	50	25,26
75	0	4,3	16,7	75	27,23
110	0	17	5,2	190	26,60
160	16	45	0		
210	11,75	55	0		
Kammer B					
25	0	3,8	17	25	
50	0	5,9	15,2	50	23,51
75	0	1,8	19,1	75	23,86
110	0	15	7,3	190	22,78
160	4,55	45	0		
210	4,15	55	0		
Kammer C					
25	0	5,4	16,8	25	22,08
50	0	14	10,9	50	22,51
75	11	39	3,1	75	22,37
110	28	55	0	190	
160	23,75	62,5	0		
210	22	65	0		
Kammer D					
25	0	15	7,9	25	24,33
50	0,2	15	8,1	50	24,66
60	22	40	0	140	23,28
110	36,25	55	0	190	21,80
160	31,75	55	0		

02.09.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,2	18,9	25	22,70
50	0	2,8	18,2	50	24,77
75	0	4,4	16,6	75	27,05
110	0	19	4,3	190	24,34
160	17,75	50	0		
210	13,75	55	0		
Kammer B					
25	0	4,1	16,8	25	23,76
50	0	5,8	15,2	50	22,65
75	0	1,5	19,3	75	23,45
110	0	16	6,1	190	22,45
160	4,4	45	0		
210	5,6	55	0		
Kammer C					
25	0	4,3	18	25	20,75
50	0	11	13,4	50	21,43
75	8	32	5,5	75	21,60
110	29,5	55	0	190	
160	25	60	0		
210	23	65	0		
Kammer D					
25	0	15	7,5	25	23,63
50	0,2	15	7,7	50	24,34
60	21	40	0	140	23,12
110	37,5	50	0	190	21,67
160	32,5	55	0		

09.09.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,1	19,2	25	20,42
50	0	2,8	18,4	50	23,72
75	0	4,6	16,7	75	26,49
110	0	19	4,2	190	22,62
160	19,5	50	0		
210	16,25	60	0		
Kammer B					
25	0	4,2	16,7	25	20,62
50	0	5,8	15,2	50	21,20
75	0	16	19,2	75	22,48
110	0	16	5,6	190	22,20
160	4,2	42,5	0		
210	6,75	55	0		
Kammer C					
25	0	4,8	17,1	25	18,89
50	0	14	11	50	19,95
75	12	37	3,9	75	20,37
110	32,5	55	0	190	
160	27,75	60	0		
210	26	65	0		
Kammer D					
25	0	16	7	25	22,30
50	0	15	7,4	50	23,38
60	22	39	0	140	22,77
110	42	50	0	190	21,43
160	36,75	55	0		

18.09.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,3	18,7	25	22,04
50	0	2,1	18,8	50	24,55
75	0	3,6	17,2	75	26,30
110	0	16	5,3	190	23,29
160	18	37	2,6		
210	14	45	2,95		
Kammer B					
25	0	2,7	17,6	25	21,56
50	0	3,6	16,8	50	21,90
75	0	1,6	19,1	75	22,63
110	0	11	9,8	190	23,18
160	5,1	38	0		
210	7,05	50	0		
Kammer C					
25	0	5,9	15,7	25	20,03
50	0	13	9,3	50	20,53
75	8,9	27	4,1	75	20,79
110	29,5	40	1,8	190	
160	24	45	3,7		
210	22	45	4,25		
Kammer D					
25	0	11	9,1	25	24,26
50	0	12	8,7	50	24,71
60	31	37	0	140	22,26
110	45,75	45	0	190	21,08
160	44,25	47,5	0		

24.09.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	3	18	25	22,21
50	0	3,8	17,2	50	24,59
75	0	7	14,8	75	26,29
110	0	23	1,9	190	22,46
160	25,75	50	0		
210	21	60	0		
Kammer B					
25	0	4,1	16,7	25	21,64
50	0	5,3	15,5	50	21,81
75	0	2,4	18,5	75	22,39
110	0	15	6,5	190	22,12
160	5,6	40	0		
210	9,1	50	0		
Kammer C					
25	0	4,6	17,3	25	19,65
50	0	13	11,3	50	20,16
75	16	36	3,1	75	20,28
110	36,5	55	0	190	
160	32,5	55	0		
210	29,5	60	0		
Kammer D					
25	3,8	20	4,5	25	24,64
50	11	25	2,4	50	24,54
60	34,5	37	0	140	22,38
110	50	42,5	0	190	21,08
160	44,5	50	0		

01.10.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,6	18,5	25	21,4
50	0	2,9	18	50	24,3
75	0	4,7	16,2	75	27,1
110	0,1	25	0	190	26,6
160	21,75	38,5	1,6		
210	18	45	1,8		
Kammer B					
25	0	2,7	17,8	25	23,2
50	0	3,9	16,6	50	21,1
75	0	1,6	19,3	75	22,4
110	0	11	9,7	190	23,1
160	4,55	31	2		
210	6,7	39	1,95		
Kammer C					
25	0	4,3	16,8	25	18,7
50	0	11	11	50	19,6
75	7,6	20	5,6	75	20,2
110	29,5	36	2	190	29,3
160	28,25	45	2,1		
210	28,25	52,5	1,05		
Kammer D					
25	was irrigated				
50					
60					
110					
160					

07.10.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,2	19,2	25	21,3
50	0	2,5	18,7	50	24,1
75	0	3,4	17,6	75	27,6
110	0,4	24	0	190	25,0
160	26,75	42,5	0		
210	22,25	50	0		
Kammer B					
25	0	3,3	17,3	25	20,7
50	0	4,7	15,8	50	20,2
75	0	2,8	17,7	75	21,9
110	0	14	7,6	190	23,5
160	6,3	35	0		
210	9	45	0		
Kammer C					
25	was irrigated				
50					
75					
110					
160					
210					
Kammer D					
25	6,5	22	1	25	23,8
50	14,5	24	1,2	50	23,8
60	35	34	0	140	22,6
110	52,5	40	0	190	21,2
160	44,25	50	0		

15.10.2009

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature
[cm]	%	%	%	%	[cm]	°C
Kammer A						
25	0	3,6	17,4	79	25	16,0
50	0	4,3	16,8	78,9	50	23,4
75	0	6,5	15,1	78,4	75	28,1
110	0,4	25	0	74,6	190	22,9
160	30	45	0	25		
210	27,5	55	0	17,5		
Kammer B						
25	0	4,3	16,5	79,2	25	14,0
50	0	6,2	14,8	79	50	16,5
75	0	3,3	17,2	79,5	75	19,7
110	0	14	7,4	78,6	190	21,9
160	8,3	39	0	52,7		
210	14,5	47,5	0	38		
Kammer C						
25	0,3	8	13,9	77,8	25	14,9
50	1,6	18	6,5	73,9	50	17,1
75	13	29	3,7	54,3	75	18,4
110	43,5	50	0	6,5	190	20,4
160	39,75	52,5	0	7,75		
210	35,75	55	0	9,25		
Kammer D						
25	8,2	22	1	68,8	25	18,2
50	15	25	0,8	59,2	50	20,1
60	29	34	0	37	140	22,7
110	53	40	0	7	190	21,3
160	43,5	50	0	6,5		

21./22.10.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,1	18,9	25	14,0
50	0	3,3	17,4	50	20,8
75	0	4	16,7	75	26,3
110	0,5	21	0	190	23,0
160	26	38	0		
210	24,5	50	0		
Kammer B					
25	0	3,1	17,6	25	11,6
50	0	4,5	16,2	50	13,6
75	0	2,7	18	75	17,1
110	0	12	9,5	190	21,5
160	7,9	35	0		
210	11,8	45	0		
Kammer C					
25	1	7,8	14,4	25	12,6
50	4,1	15	8,1	50	14,1
75	19,5	28	2,6	75	15,4
110	38,5	38	0	190	18,7
160	39,5	47,5	0		
210	36,5	52,5	0		
Kammer D					
25	6,8	23	4,5	25	15,2
50	7	20	5,6	50	16,6
60	22,5	25	5,5	140	20,9
110	52,5	40	0,1	190	20,1
160	47,5	50	0		

28.10.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,8	18,1	25	15,2
50	0	2,8	18,2	50	19,7
75	0	5,7	15,6	75	24,2
110	0,1	24	0	190	20,3
160	26	40	0		
210	27,5	50	0		
Kammer B					
25	0	3	17,7	25	13,1
50	0	4,4	16,3	50	13,9
75	0	2,5	18,2	75	15,6
110	0	11	9,6	190	19,6
160	7,75	36,5	0		
210	15,75	45	0		
Kammer C					
25	0,5	7,7	15	25	13,0
50	2,6	15	9,1	50	13,7
75	19	30	4,1	75	14,4
110	46,5	45	0	190	16,9
160	44,25	50	0		
210	39	55	0		
Kammer D					
25	0,9	15	6,3	25	16,1
50	2,1	17	5,3	50	16,5
60	32,5	35	0	140	18,6
110	52,5	40	0	190	18,4
160	49,5	45	0		

10.11.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25	0	2,2	17,8	25	11,0
50	0	2,4	17,4	50	15,9
75	0	5	14,5	75	20,8
110	0,1	17	2,6	190	18,5
160	20,75	29,5	3,15		
210	21,75	37,5	2,8		
Kammer B					
25	0	2,8	17,4	25	9,4
50	0	3,9	16,3	50	10,3
75	0	1,4	19,3	75	12,8
110	0	9	11,2	190	17,8
160	4,55	25,5	2,5		
210	8,85	34,5	2,7		
Kammer C					
25	0	4,6	17,4	25	9,3
50	1,6	11	13,3	50	10,3
75	10	19	9,9	75	11,1
110	38,5	37	2,7	190	14,1
160	37,25	39	2,9		
210	33,75	40	3,15		
Kammer D					
25	5,1	15	5,5	25	10,5
50	7,4	16	6,2	50	11,3
60	30	28	2,6	140	15,6
110	40,75	32,5	2,5	190	16,0
160	38,25	33	3,8		

18.11.2009

Mittelwert	CH4	CO2	O2	Mittelwert	Temperature
[cm]	%	%	%	[cm]	°C
Kammer A					
25				25	11,7
50				50	15,0
75				75	18,9
110	0	24	0,6	190	16,6
160	22,25	38,5	0		
210	26,5	50	0		
Kammer B					
25				25	10,4
50				50	10,5
75				75	12,0
110	0	13	9	190	16,0
160	4,8	33,5	0		
210	12,5	45	0		
Kammer C					
25				25	9,8
50				50	10,1
75				75	10,5
110	49	45	0	190	12,9
160	49,25	50	0		
210	45,75	52,5	0		
Kammer D					
25				25	10,9
50				50	11,1
60	18,5	29	0	140	13,9
110	41,75	37	0	190	14,6
160	36,5	38	0		

02.12.2009

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							
25	0	2,2	17,6	80,2	25	9,5	0,0
50	0	2,8	16,7	80,5	50	13,2	0,0
75	0	4,8	14,2	81	75	16,4	0,0
110	0	18	13	69	190	14,7	0,0
160	25	37,5	0	37,5			0,7
210	27,25	45	0	27,75			0,6
Kammer B							
25	0	3,8	16,4	79,8	25	8,4	0,0
50	0	5,6	14,6	79,8	50	9,2	0,0
75	0	2,7	17,4	79,9	75	10,9	0,0
110	0	12	8,1	79,9	190	14,3	0,0
160	2,95	29,5	0	67,55			0,1
210	9,45	40	0	50,55			0,2
Kammer C							
25	0	5	16,9	78,1	25	8,2	0,0
50	0,4	9,5	13,8	76,3	50	9,1	0,0
75	12	22	7	59	75	9,7	0,5
110	47	45	0	8	190	11,5	1,0
160	48,75	45	0	6,25			1,1
210	46	50	0	4			0,9
Kammer D							0,19
25	0	11	6,9	82,1	25	8,9	0,0
50	0,5	11	6,2	82,3	50	9,9	0,0
60	22	24	0	54	140	12,5	0,9
110	39,25	33,5	0	27,25	190	13,2	1,2
160	39,5	34,5	0	26			1,1

09.12.2009

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							0
25	0	3,4	17	79,6	25	7,6	0,0
50	0	4,2	16,3	79,5	50	11,6	0,0
75	0	7,1	13,5	79,4	75	15,6	0,0
110	0	20	1,1	78,9	190	14,0	0,0
160	17,5	33	0	49,5			0,5
210	21,75	40	0	38,25			0,5
Kammer B							0,00
25	0	4,5	15,6	79,9	25	6,8	0,0
50	0	6,1	14,1	79,8	50	7,3	0,0
75	0	2,1	18,4	79,5	75	9,4	0,0
110	0	12	7,8	80,2	190	13,6	0,0
160	2,4	28,5	0	69,1			0,1
210	8,5	45	0	46,5			0,2
Kammer C							0,20
25	0	3	19,6	77,4	25	6,7	0,0
50	0	5,6	18,4	76	50	7,6	0,0
75	2	7,2	16,9	73,9	75	8,3	0,3
110	42	40	0	18	190	10,8	1,1
160	48,5	45	0	6,5			1,1
210	46,5	50	0	3,5			0,9
Kammer D							0,22
25	0,5	14	16,3	69,2	25	6,8	0,0
50	1,5	13	6,2	79,3	50	7,6	0,1
60	27,5	30	0	42,5	140	11,8	0,9
110	36	33	0	31	190	12,6	1,1
160	42	34,5	0	23,5			1,2

26.01.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature
[cm]	%	%	%	%	[cm]	°C
Kammer A						
25	0	0	20	80	25	1,73
50	0	0	19,7	80,3	50	4,43
75	0	0	19,8	80,2	75	6,70
110	0	4,8	12,3	82,9	190	8,17
160	11,25	25	0	63,75		
210	14,75	32	0	53,25		
Kammer B						
25	0	2,4	17,6	80	25	1,61
50	0	5,8	13,5	80,7	50	2,27
75	0	2,6	17,6	79,8	75	3,62
110	0	15	3,4	81,6	190	8,38
160	0,7	22,5	0	76,8		
210	3,15	30	0	66,85		
Kammer C						
25	0,3	2,2	19,5	78	25	1,58
50	0,6	3,9	18,7	76,8	50	2,44
75	5,1	13	14,4	67,5	75	3,06
110	31,5	34	0,3	34,2	190	5,27
160	47,25	40	0	12,75		
210	47,75	45	0	7,25		
Kammer D						
25	0	0,7	20,2	79,1	25	1,49
50	0	0,9	19,6	79,5	50	2,30
60	5,8	9,6	9	75,6	140	5,93
110	27	23	0	50	190	7,31
160	38,25	28	0	33,75		

09.02.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature
[cm]	%	%	%	%	[cm]	°C
Kammer A						
25	0	0,2	20,8	79	25	0,78
50	0	0,3	20,6	79,1	50	3,26
75	0	0,2	20,7	79,1	75	5,24
110	0	4,8	17,8	77,4	190	6,72
160	9,35	25	0	65,65		
210	14,25	30,5	0	55,25		
Kammer B						
25	0	4,3	15,5	80,2	25	1,06
50	0	7,4	12,3	80,3	50	1,73
75	0	4,5	15,1	80,4	75	2,96
110	0	13	4,5	82,5	190	7,13
160	0,6	21	0	78,4		
210	2,4	28	0	69,6		
Kammer C						
25	1,5	2,2	18,5	77,8	25	0,81
50	2,6	4,9	17	75,5	50	1,66
75	13	16	10,7	60,3	75	2,19
110	35	35	0	30	190	4,19
160	41,75	39	0	19,25		
210	42,5	40	0	17,5		
Kammer D						
25	0,1	2,6	17,6	79,7	25	0,81
50	0	2,4	18,5	79,1	50	1,60
60	1,5	6,3	12,9	79,3	140	4,84
110	20,25	21	0	58,75	190	6,15
160	22,25	22	0	55,75		

24.02.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature
[cm]	%	%	%	%	[cm]	°C
Kammer A						
25	0	1,8	17,8	80,4	25	0,70
50	0	2,1	17,2	80,7	50	2,68
75	0	2,3	17,2	80,5	75	4,28
110	0	8,8	8,7	82,5	190	5,69
160	7,3	23,5	0	69,2		
210	11,75	29	0	59,25		
Kammer B						
25	0	2,5	17,3	80,2	25	1,56
50	0	6,9	11,7	81,4	50	1,88
75	0	1,5	18,7	79,8	75	2,83
110	0	14	3,4	82,6	190	6,26
160	0,35	20	0	79,65		
210	1,7	26	0	72,3		
Kammer C						
25	0,4	2	19	78,6	25	0,78
50	0,7	3,5	18,2	77,6	50	1,49
75	8,3	12	12,9	66,8	75	1,99
110	34	33	0	33	190	3,22
160	38,25	38	0	23,75		
210	39	40	0	21		
Kammer D						
25	0,3	4	14,1	81,6	25	1,04
50	0	3,5	16,8	79,7	50	1,61
60	13	18	1,3	67,7	140	4,25
110	21	21,5	0	57,5	190	5,43
160	26,5	23	0	50,5		

18.03.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature
[cm]	%	%	%	%	[cm]	°C
Kammer A						
25	0	0,9	19,1	80	25	5,39
50	0	1,2	18,9	79,9	50	5,34
75	0	2,2	18,3	79,5	75	5,92
110	0	12	9,5	78,5	190	6,31
160	11,25	28	0	60,75		
210	16,25	34	0	49,75		
Kammer B						
25	0	2,3	18,2	79,5	25	5,08
50	0	4,2	16,2	79,6	50	4,90
75	0	1,2	19	79,8	75	4,94
110	0	14	7,1	78,9	190	6,37
160	1,25	24,5	0	74,25		
210	3,65	31	0	65,35		
Kammer C						
25	0,6	1,8	18,8	78,8	25	4,14
50	1,1	3,1	18,2	77,6	50	3,99
75	12,5	12	13,7	61,8	75	3,90
110	43	40	0	17	190	3,78
160	48,5	45	0	6,5		
210	48	45	0	7		
Kammer D						
25	0	8,2	15,8	76	25	5,46
50	0	7,5	16	76,5	50	5,38
60	21	24	0	55	140	5,48
110	31,75	27,5	0	40,75	190	6,15
160	38,5	28,5	0	33		

24.03.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							0
25	0	0,9	20,1	79	25	8,99	0,0
50	0	0,9	20	79,1	50	7,67	0,0
75	0	0,8	20,2	79	75	7,09	0,0
110	0	10	13	77	190	6,67	0,0
160	11,5	29	0	59,5			0,4
210	16,25	34	0	49,75			0,5
Kammer B							0,00
25	0	2,6	18,1	79,3	25	8,60	0,0
50	0	4,4	16,4	79,2	50	7,98	0,0
75	0	2	18,7	79,3	75	7,19	0,0
110	0	14	7,2	78,8	190	6,51	0,0
160	1,2	24,5	0	74,3			0,0
210	3,65	30	0	66,35			0,1
Kammer C							0,07
25	0,7	2,6	18,3	78,4	25	7,36	0,3
50	3,3	4,7	16,7	75,3	50	6,59	0,7
75	29,5	26	6	38,5	75	6,03	1,1
110	46	40	0	14	190	3,74	1,2
160	49	45	0	6			1,1
210	48	45	0	7			1,1
Kammer D							0,26
25	0	6,3	15,7	78	25	9,37	0,0
50	0	3,8	18,5	77,7	50	8,89	0,0
60	16	16	8,6	59,4	140	6,05	1,0
110	32,75	28,5	0	38,75	190	6,40	1,1
160	38	30	0	32			1,3

01.04.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							0
25	0	0,5	19,8	79,7	25	10,61	0,0
50	0	0,4	19,9	79,7	50	10,44	0,0
75	0	0,7	19,1	80,2	75	9,88	0,0
110	0	11	7,2	81,8	190	7,99	0,0
160	12,25	30	0	57,75			0,4
210	16,5	35	0	48,5			0,5
Kammer B							0,00
25	0	2,7	17,9	79,4	25	10,24	0,0
50	0	4	16,6	79,4	50	10,36	0,0
75	0	1,9	18,8	79,3	75	10,03	0,0
110	0	15	7	78	190	7,41	0,0
160	1,15	25	0	73,85			0,0
210	3,65	30	0	66,35			0,1
Kammer C							0,00
25	0,3	2,5	18,1	79,1	25	9,35	0,1
50	2	4,3	16,2	77,5	50	9,22	0,5
75	22	22	7,6	48,4	75	8,83	1,0
110	45,5	45	0	9,5	190	4,86	1,0
160	48,75	45	0	6,25			1,1
210	48	45	0	7			1,1
Kammer D							0,08
25	0,1	7,9	9,4	82,6	25	11,32	0,0
50	0,1	7,6	11,5	80,8	50	11,78	0,0
60	17,5	20	3,7	58,8	140	8,05	0,9
110	35	31	0	34	190	7,57	1,1
160	38,5	31,5	0	30			1,2

07.04.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							0
25	0,1	1,8	19,2	78,9	25	10,47	0,1
50	0,1	2,5	18,4	79	50	11,05	0,0
75	0,1	2,4	18,7	78,8	75	11,25	0,0
110	0,1	15	7,3	77,6	190	8,74	0,0
160	12,25	31	0	56,75			0,4
210	17	35	0	48			0,5
Kammer B							0,00
25	0	3,3	17,2	79,5	25	9,87	0,0
50	0	4,6	15,8	79,6	50	10,06	0,0
75	0	2,2	18,2	79,6	75	10,10	0,0
110	0	15	5,9	79,1	190	8,30	0,0
160	1,5	26	0	72,5			0,1
210	3,85	30	0	66,15			0,1
Kammer C							0,11
25	1,1	1,8	18	79,1	25	9,10	0,6
50	2,2	3	16,8	78	50	9,31	0,7
75	23,5	21	7,3	48,2	75	9,10	1,1
110	44,5	40	0	15,5	190	5,93	1,1
160	50,25	45	0	4,75			1,1
210	50	45	0	5			1,1
Kammer D							0,11
25	0,1	5,3	15,7	78,9	25	11,10	0,0
50	0	5,4	15,7	78,9	50	11,54	0,0
60	25,5	26	0	48,5	140	9,36	1,0
110	37,25	31,5	0	31,25	190	8,59	1,2
160	40	33	0	27			1,2

15.04.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							
25	0,1	2,8	17,5	79,6	25	10,88	0,0
50	0	3,5	16,6	79,9	50	11,55	0,0
75	0	3,6	16,3	80,1	75	12,20	0,0
110	0,1	18	2,8	79,1	190	9,89	0,0
160	13	31,5	0	55,5			0,4
210	16,25	35,5	0	48,25			0,5
Kammer B							
25	0	2,7	18	79,3	25	10,09	0,0
50	0	6,8	12	81,2	50	10,14	0,0
75	0	1,9	18,4	79,7	75	10,28	0,0
110	0	18	2,6	79,4	190	9,18	0,0
160	1,55	27	0	71,45			0,1
210	3,9	31	0	65,1			0,1
Kammer C							
25	3,1	5,3	15,6	76	25	9,67	0,6
50	7,2	9,5	13,3	70	50	9,55	0,8
75	22	19	8,3	50,7	75	9,41	1,2
110	51	40	0	9	190	6,61	1,3
160	49,5	50	0	0,5			1,0
210	49,5	50	0	0,5			1,0
Kammer D							
25	0,2	8,2	8,6	83	25	10,92	0,0
50	0,5	6,2	11,9	81,4	50	11,01	0,1
60	27,5	26	0	46,5	140	10,49	1,1
110	36,5	31	0	32,5	190	9,60	1,2
160							

19.04.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							0
25	0	2,5	18	79,5	25	11,88	0,0
50	0,1	3,3	17	79,6	50	12,18	0,0
75	0	4,2	15,5	80,3	75	12,32	0,0
110	0,1	19	4,4	76,5	190	10,90	0,0
160	13,5	32,5	0	54			0,4
210	16,75	36,5	0	46,75			0,5
Kammer B							0,00
25	0	2,8	17,5	79,7	25	11,54	0,0
50	0	7	13,4	79,6	50	11,06	0,0
75	0	1,8	18,8	79,4	75	10,61	0,0
110	0	19	2,3	78,7	190	9,49	0,0
160	1,95	27	0	71,05			0,1
210	4	32	0	64			0,1
Kammer C							0,09
25	0,2	4,7	17,7	77,4	25	10,77	0,0
50	0,5	8,8	16,1	74,6	50	10,41	0,1
75	6,2	13	13,9	66,9	75	10,02	0,5
110	47	40	0	13	190	10,84	1,2
160	51	47,5	0	1,5			1,1
210	49	50	0	1			1,0
Kammer D							0,15
25	0	8,3	14,2	77,5	25	12,30	0,0
50	0	8,6	13,2	78,2	50	12,27	0,0
60	33	27	0	40	140	10,61	1,2
110	41,5	32	0	26,5	190	9,94	1,3
160							

29.04.2010

Mittelwert	CH4	CO2	O2	N2	Mittelwert	Temperature	CH4/CO2
[cm]	%	%	%	%	[cm]	°C	
Kammer A							0
25	0	1,9	19,6	78,5	25	14,0	0,0
50	0,1	2,4	19,2	78,3	50	13,8	0,0
75	0	1,2	19,9	78,9	75	13,7	0,0
110	0	16	10,3	73,7	190	11,0	0,0
160	14,75	34,5	0	50,75			0,4
210	16,5	37	0	46,5			0,4
Kammer B							0,00
25	0	3,4	17,4	79,2	25	14,0	0,0
50	0	5,3	16	78,7	50	13,7	0,0
75	0	3	18	79	75	12,7	0,0
110	0	20	4,2	75,8	190	10,1	0,0
160	2,45	31	0	66,55			0,1
210	4,7	33	0	62,3			0,1
Kammer C							0,07
25	3,3	11	9,3	76,4	25	14,1	0,3
50	9,3	18	5,1	67,6	50	13,1	0,5
75	32	35	1,3	31,7	75	12,4	0,9
110	51	45	0	4	190	18,9	1,1
160	52	50	0	0			1,0
210	49	50	0	0			
Kammer D							0,13
25	0	7,2	15,1	77,7	25	16,1	0,0
50	0	5,2	16,4	78,4	50	15,6	0,0
60	27	20	6,9	46,1	140	11,7	1,4
110	46,25	36,5	0	17,25	190	10,7	1,3
160							

A.2 FID-screenings

FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 29.05.08, 11h

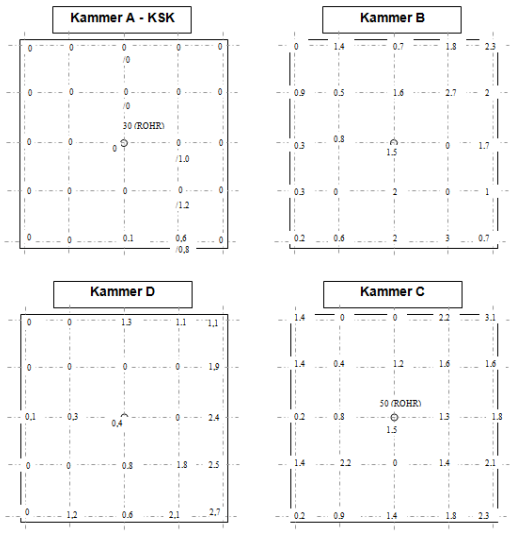
Ausführende: OG, PPH, JT

Werte in ppm CH_4
Gerät: PE-Micro-FID

Witterung (Temp, Wind, Luftdruck, etc.):
Sonnig, 25-26 °C
Wind: 6,5 km/h in Bodennähe, 15 km/h in 1,5m Höhe
Luftdruck 995 mbar
Lichtlot: kein Wasser in Belüftungsrohr

Beschreibung der Oberfläche (trocken/feucht, etc.):
trocken mit Pflanzenbewuchs
Kammer A, B: Teil der Pflanzen kurz zuvor entfernt, tw. offener Boden
Kammer C: Teil der Pflanzen kurz zuvor entfernt, aber noch immer geschlossene Pflanzendecke
Kammer D: geschlossene Pflanzendecke mit hochwüchsigen Ackerenf

Hintergrund: 1 ppm in Umgebungsluft!



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 27.08.09

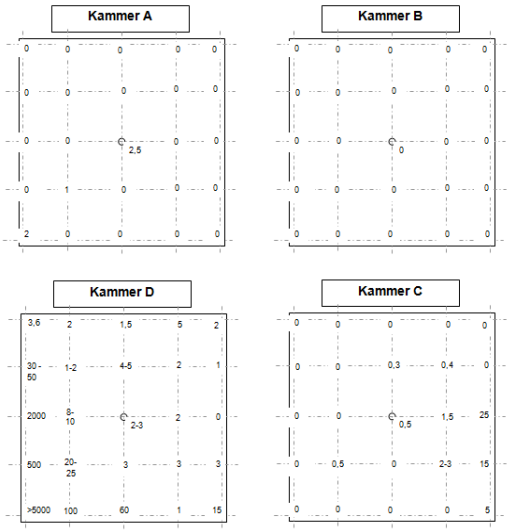
Ausführende: Hrad, Huber-Humber

Werte in ppm CH4, Umgebung: -0,1 – 0,7 ppm
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 12:40 0 – 2,1 km/h, sonnig, wolkenlos
Bodennähe: 0 km/h
Temp.: 32°C
Ende: 13:10 0 – 2,8 km/h
Bodennähe: 0 km/h
Temp.: 31,6°C

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, starker Bewuchs (bis 1,5 m)
Kammer B: trocken, starker Bewuchs (bis 2 m)
Kammer C: trocken, geschlossen Pflanzendecke (bis ½ m)
Kammer D: trocken, starker Bewuchs



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 02.09.09

Ausführende: Hrad

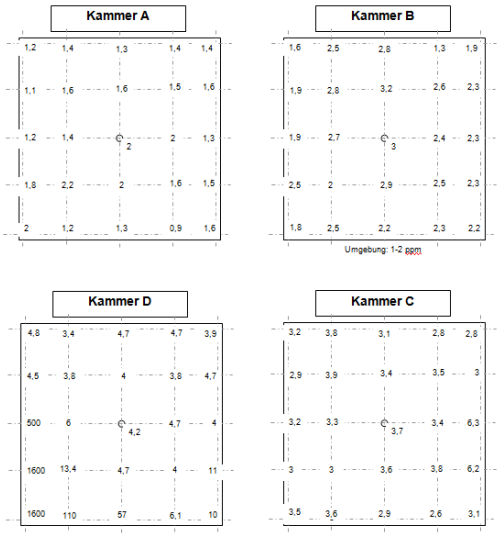
Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 14:10 0 – 2,8 km/h, bewölkt
Bodennähe: 0 km/h
Temp.: 30,8°C
Umgebung: 0,2 ppm
Ende: 15:15 0 km/h
Bodennähe: 0 km/h
Temp.: 30,8°C

Luftdruck: 985 mbar

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, starker Bewuchs (bis 1,5 m)
Kammer B: trocken, starker Bewuchs (bis 2 m)
Kammer C: trocken, geschlossen Pflanzendecke (bis ½ m)
Kammer D: trocken, starker Bewuchs



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 09.09.09

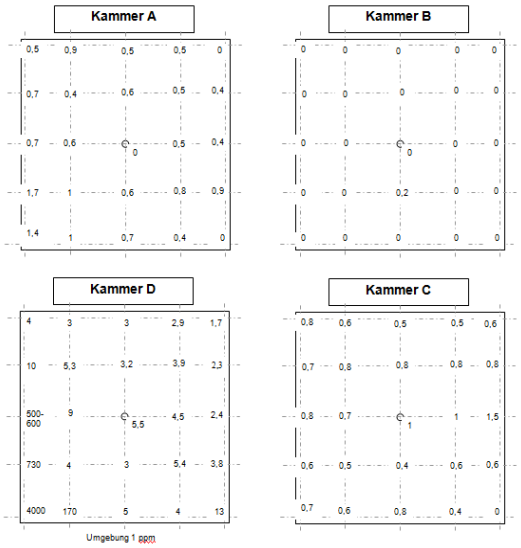
Ausführende: Hrad

Werte in ppm CH4, Umgebung 0,08 ppm
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 15:40 0 – 1 km/h, sonnig, wolkenlos
Bodennähe: 0 km/h
Temp.: 25,8°C
Luftdruck: 992 mbar
Ende: 16:40 0 km/h
Bodennähe: 0 km/h
Temp.: 26,1°C

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, frisch gemäht
Kammer B: trocken, frisch gemäht
Kammer C: trocken, geschlossen Pflanzendecke (bis ½ m)
Kammer D: trocken, starker Bewuchs (bis 1,5 m)



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 18.09.09

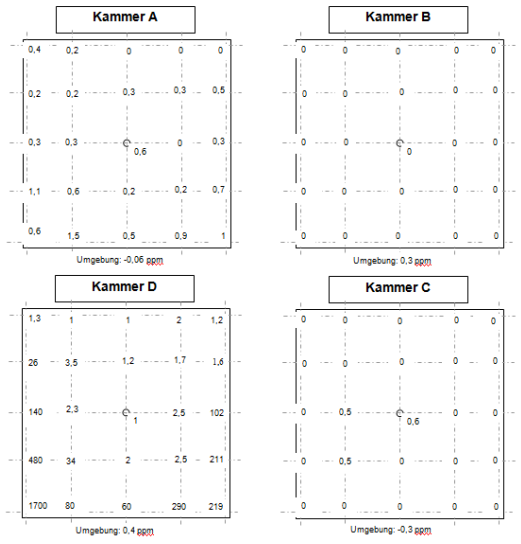
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 14:20 0 – 1 km/h, bewölkt
Bodennähe: 0 km/h
Temp.: 24,6°C
Luftdruck: 988 mbar
Ende: 15:00 0 – 1km/h
Bodennähe: 0 km/h

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, gemäht, geschlossene Pflanzendecke
Kammer B: trocken, gemäht, geschlossene Pflanzendecke
Kammer C: trocken, gemäht, geschlossene Pflanzendecke
Kammer D: trocken, gemäht, geschlossene Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 24.09.09

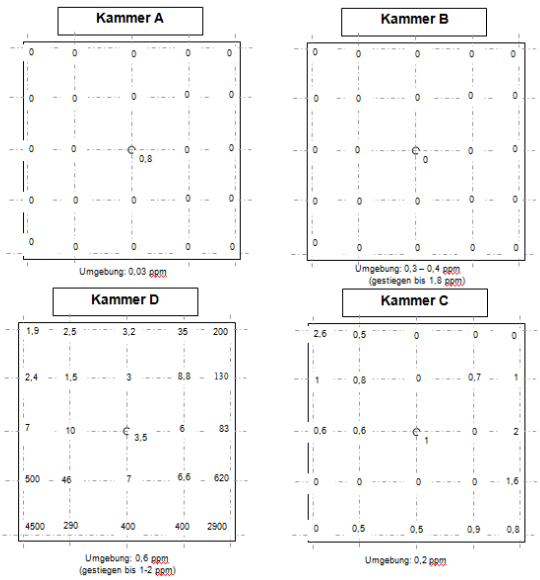
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 14:00 0 – 2,5 km/h, sonnig
Bodennähe: 0 km/h
Temp.: 26,2°C
Luftdruck: 991 mbar
Ende: 15:00 0 – 1,8 km/h
Bodennähe: 0 km/h
Temp.: 26,8°C

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: sehr trocken, geschlossene Pflanzendecke
Kammer B: sehr trocken, geschlossene Pflanzendecke
Kammer C: sehr trocken, geschlossene Pflanzendecke
Kammer D: sehr trocken, geschlossene Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 01.10.09

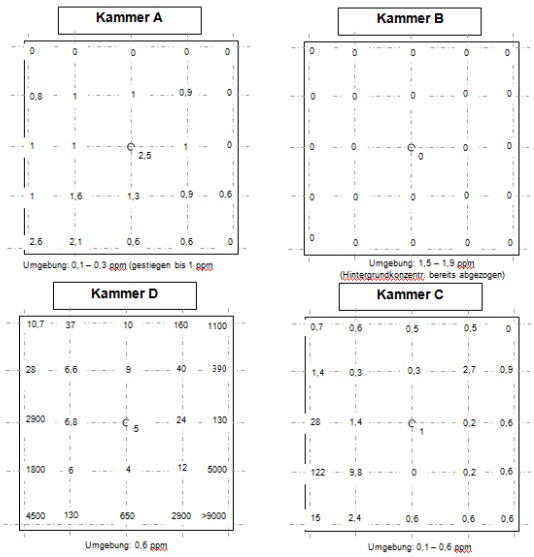
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 13:45 0 – 2,1 km/h, sonnig
Bodennähe: 0 km/h
Temp.: 26,6°C
Luftdruck: 981 mbar
Ende: 14:40 0 – 1,8 km/h
Bodennähe: 0 km/h

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: sehr trocken, geschlossene Pflanzendecke
Kammer B: sehr trocken, geschlossene Pflanzendecke
Kammer C: sehr trocken, geschlossene Pflanzendecke
Kammer D: sehr trocken, geschlossene Pflanzendecke
(wurde bewässert)



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 07.10.09

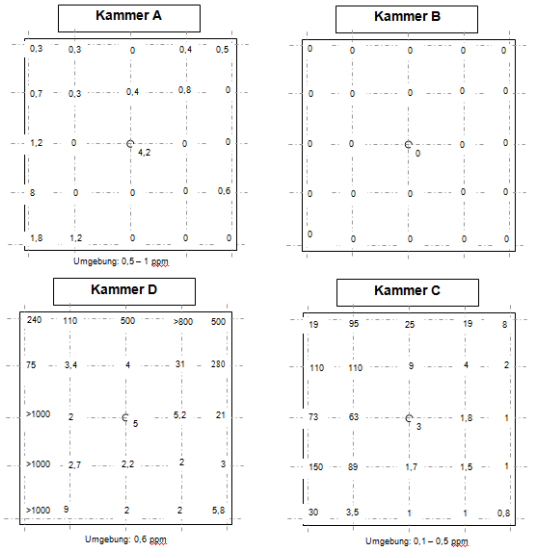
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 15:40 0 – 5 km/h (Spitzen bis 6,4 km/h), sonnig
Bodennähe: 0 km/h
Temp.: 29,5°C
Luftdruck: 987 mbar
Ende: 17:00 0 – 5 km/h (Spitzen bis 6,1 km/h)
Bodennähe: 0 km/h

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, geschlossene Pflanzendecke
Kammer B: trocken, geschlossene Pflanzendecke
Kammer C: etwas feucht, geschlossene Pflanzendecke
(wurde bewässert)
Kammer D: trocken, geschlossene Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 21.10.09

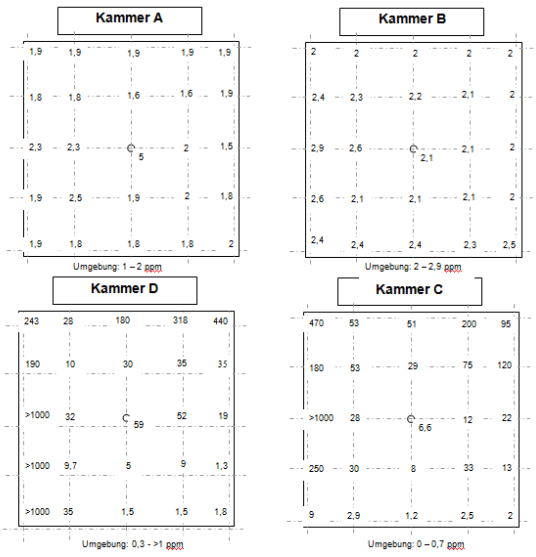
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 14:05 0 – 5 km/h (Spitzen bis 6,4 km/h), sonnig
Bodennähe: 0 – 2,1 km/h
Temp.: 9,3 °C
Ende: 15:05

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: feucht, geschlossene Pflanzendecke
Kammer B: etwas feucht, geschlossene Pflanzendecke
Kammer C: feucht, geschlossene Pflanzendecke
(wurde bewässert)
Kammer D: feucht, geschlossene Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 28.10.09

Ausführende: Hrad

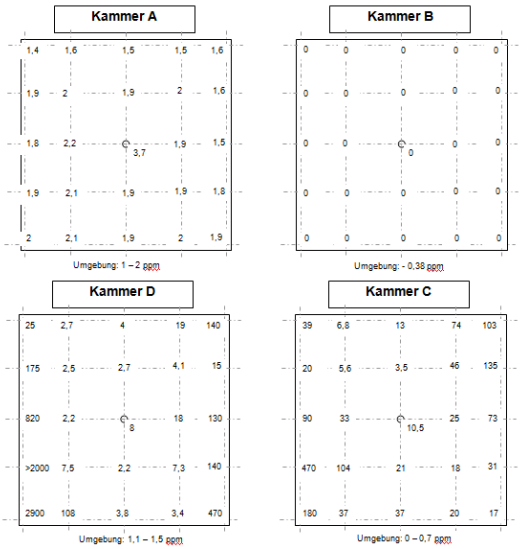
Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 16:00 0 – 0,4 km/h, bewölkt
Bodennähe: 0 km/h

Temp.: 13,1 °C
Ende: 17:05

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: feucht, geschlossene Pflanzendecke
Kammer B: feucht, geschlossene Pflanzendecke
Kammer C: feucht, geschlossene Pflanzendecke
Kammer D: feucht, geschlossene Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 18.11.09

Ausführende: Hrad

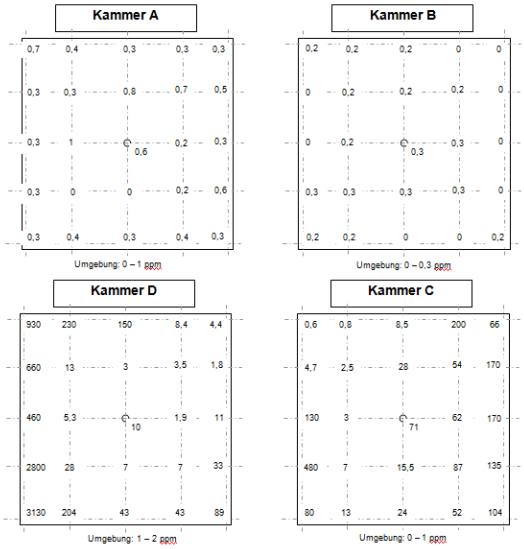
Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 14:05 0 – 3,2 km/h, bewölkt
Bodennähe: 0 km/h

Temp.: 13,2 °C
Ende: 15:10

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: feucht, geschlossene Pflanzendecke
Kammer B: feucht, geschlossene Pflanzendecke
Kammer C: feucht, geschlossene Pflanzendecke
Kammer D: feucht, teilw. kahle Stellen



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 02.12.09

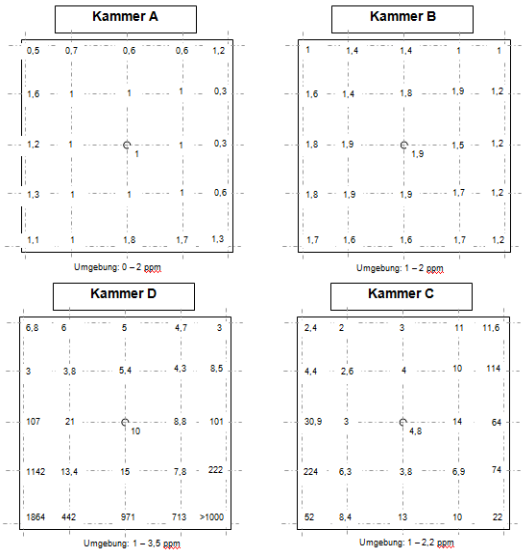
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 10:15 0 – 2,1 km/h, sonnig
Bodennähe: 0 km/h
Luftdruck 984 mbar
Temp.: 8 °C
Ende: 11:10

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: feucht, geschlossene Pflanzendecke
Kammer B: feucht, geschlossene Pflanzendecke
Kammer C: feucht, geschlossene Pflanzendecke
Kammer D: feucht, teilw. kahle Stellen



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 26.01.2010

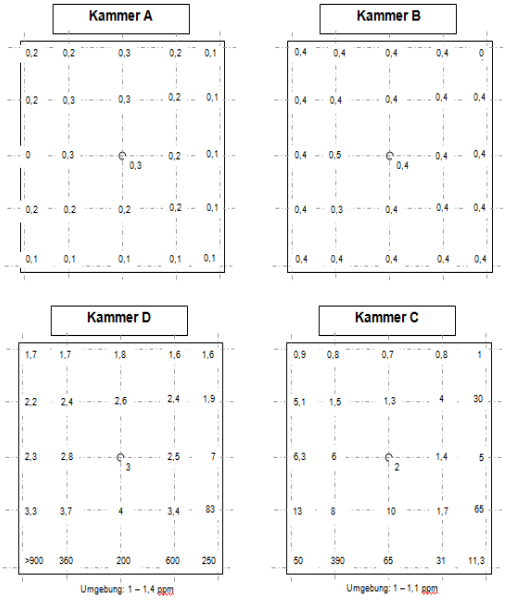
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 13:00 0 – 5 km/h, bedeckt, teilw. sonnig
Bodennähe: 0 – 1 km/h
Luftdruck 1000 mbar
Temp.: - 0,1 °C
Ende: 13:50

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: Schnee bedeckt (2-3 cm)
Kammer B: Schnee bedeckt (2-3 cm)
Kammer C: Schnee bedeckt (2-3 cm)
Kammer D: Schnee bedeckt (2-3 cm)



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 09.02.2010

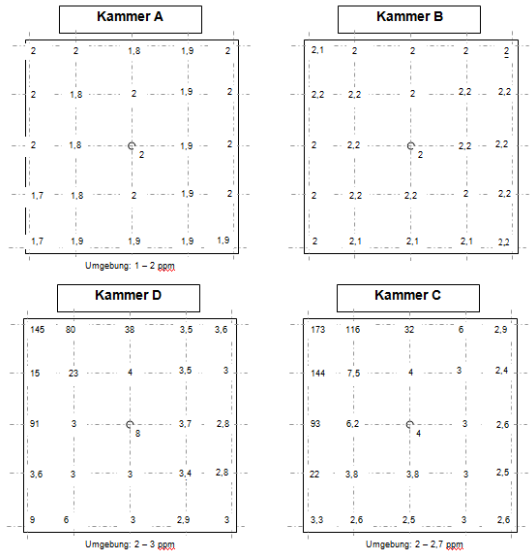
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 14:10 0 – 3,9 km/h, Spitzen 8 km/h, bewölkt
Bodennähe: 0 – 2,8 km/h
Luftdruck 974 mbar
Temp.: -0,5 °C
Ende: 15:10

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: teilw. mit Schnee bedeckt
Kammer B: teilw. mit Schnee bedeckt
Kammer C: teilw. mit Schnee bedeckt
Kammer D: teilw. mit Schnee bedeckt



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 18.03.2010

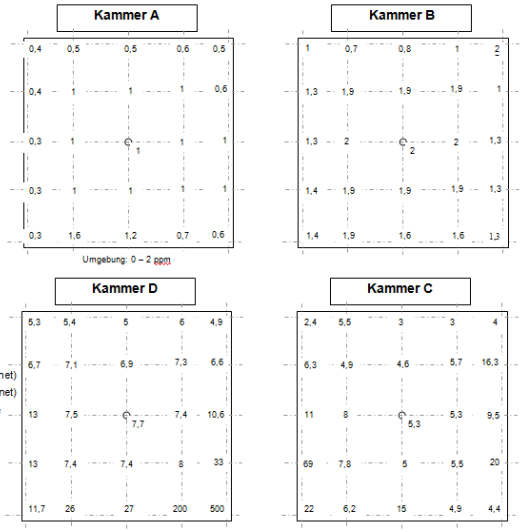
Ausführende: Hrad

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 13:50 sonnig, windstill
Temp.: 17,7 °C
Ende: 14:40

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, geschlossene Pflanzendecke
Kammer B: trocken, geschlossene Pflanzendecke (vertrocknet)
Kammer C: trocken, geschlossene Pflanzendecke (vertrocknet)
Kammer D: trocken, teilweise geschlossene Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 24.03.2010

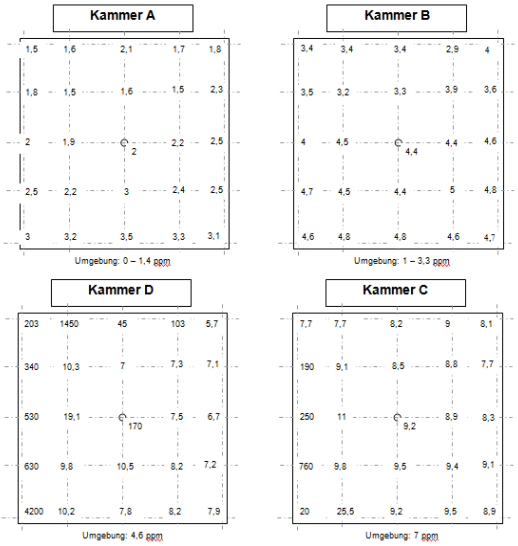
Ausführende: Egle

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 11:20 0 – 5,7 km/h, Spitzen bis 10 km/h
Bodennähe: 2,8 km/h
Temp.: 17,8°C
Luftdruck: 986 mbar
Ende: 12:30

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, geschlossene Pflanzendecke
Kammer B: trocken, geschlossene Pflanzendecke
Kammer C: trocken, geschlossene Pflanzendecke
Kammer D: trocken, teilweise geschlossene Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 01.04.2010

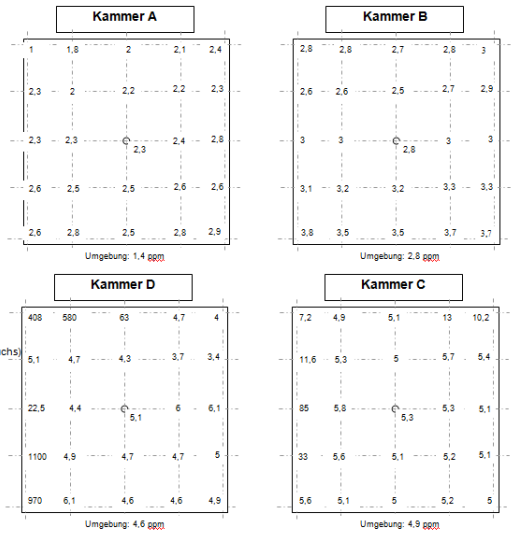
Ausführende: Egle

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 10:45 0 – 2,5 km/h, sonnig
Bodennähe: 0 km/h
Temp.: 21,2 °C
Luftdruck: 980 mbar
Ende: 11:40

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: feucht, geschloss. Pflanzendecke (starker Bewuchs)
Kammer B: trockener, fast geschloss. Pflanzendecke
Kammer C: trockener, fast geschloss. Pflanzendecke
Kammer D: trocken, 50 % geschloss. Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 07.04.2010

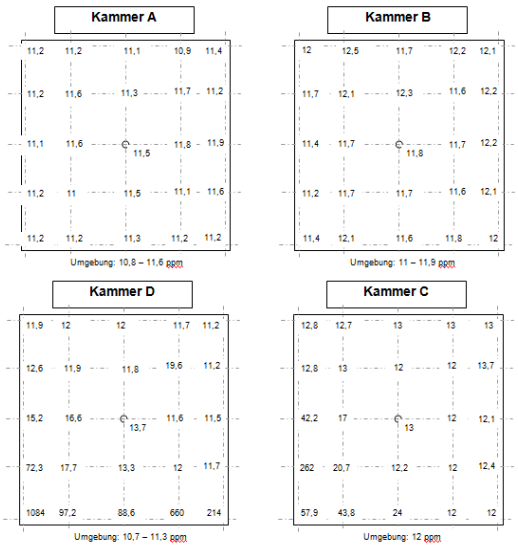
Ausführende: Andreas

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 15:40 0 – 0,7 km/h, sonnig
Bodennähe: 0 km/h
Temp.: 25,4 °C
Luftdruck: 990 mbar
Ende: 17:10

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, geschloss. Pflanzendecke (gemäht)
Kammer B: trocken, fast geschloss. Pflanzendecke
Kammer C: trocken, fast geschloss. Pflanzendecke (80 %)
Kammer D: trocken, 50 % geschloss. Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 19.04.2010

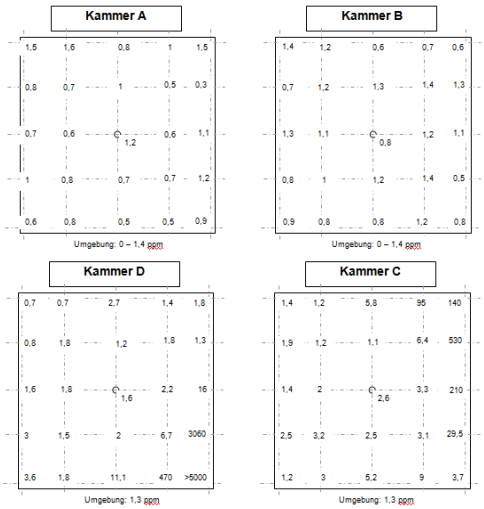
Ausführende: LE

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 14:10 0 – 3,4 km/h, sonnig
Bodennähe: 0 – 2,1 km/h
Temp.: 24,3 °C
Luftdruck: 980 mbar
Ende: 15:15

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: feucht, geschloss. Pflanzendecke
Kammer B: feucht, fast geschloss. Pflanzendecke
Kammer C: feucht, fast geschloss. Pflanzendecke (80 %)
Kammer D: feucht,. 50 % geschloss. Pflanzendecke



FID-Kartierung - Lys Seibersdorf

(Aufnahme Messrasterpunkt und visuelle Auffälligkeiten)

Datum: 29.04.2010

Ausführende: AS

Werte in ppm CH4
Gerät: Thermo FID, TVA-100

Witterung (Temp, Wind, Luftdruck, etc.):
Start: 16:45 0 – 11 km/h, sonnig
Bodennähe: 0 – 2,5 km/h
Temp.: 26 °C
Luftdruck: 986 mbar
Ende: 17:50

Beschreibung der Oberfläche (trocken/feucht, etc.):

Kammer A: trocken, geschloss. Pflanzendecke (geschnitten
Kammer B: trocken, fast geschloss. Pflanzendecke (geschnitten
Kammer C: trocken, geschloss. Pflanzendecke (80 %)
Kammer D: trocken,. 60 % geschloss. Pflanzendecke