Session schedule

	Sunday June 24th	Monday June 25th	Tuesday June 26	Wednesday June 27th
08.30 – 09.00		Opening session		
09.00-11.00		Session A1 and B1	Session A4 and B4	Session A7 and B7
11.00 – 13.00		Lunch and poster session	Lunch and poster session	Lunch and closure session
13.00-15.00		Session A2 and B2	Session A5 and B5	
15.30-17.30		Session A3 and B3	Session A6 and B6	Optional visit to industry landfill And steel mill.
18.00 -	Reception and buffée	Evening program on site & dinner	Boat tour w dinner	

Session list

	A	B
1	Mistakes in landfill research and practise	Landfill Mining
2	Landfill covers – materials and design	Status quo and future perspectives of landfillaeration
3	The myth or control of even gas distribution in methane oxidation systems	Anaerobic Decomposition of Forest Products in Landfills
4	Methane Oxidation Systems – are they still in their infancy?	Modeling Landfill Gas Generation in Landfills Receiving Biodegradable Waste
5	The use of stable isotopes for quantification of methane oxidation in landfill covers - critical factors, limitations, and future developments	Disaster Waste
6	Landfill Aftercare Completion: Landfill aftercare completion I: from active to passive care	The Fate of Emerging Compounds in Landfills Part I
7	Landfill Aftercare Completion: Landfill aftercare completion II: decision making	The Fate of Emerging Compounds in Landfills Part II

Session plans follows in the order above, A1, B1, A2, B2 et c.

Table of contents

A1. Mistakes session

Disasters or smaller mishaps in landfill research

Lale Andreas, Ole Hjelmar

Did you ever experience that the running time of a project was over before you got the answers you were looking for? Or got results that you willingly would like to hide in a drawer of your desk? Or found looking at the final report, that the lay-out of the project was poor and the outcome most likely would have been much better with a different approach?
…
Things that happen, things that many of us probably have experienced at least once during
their research carrier and that we naturally avoid talking about.
However, this time we do!
Such experiences can (and should!) at least serve as bad examples, and why not also for others. We should use them to learn something. Often it is our mistakes that we get new insights from.
The funnier of those experiences we may tell to friends sitting around the regulars' table with a glass of beer and having a good laugh.
This session will do without the beer. Laughing is allowed but, in the first instance, we want to have a critical look at some miscarried projects or parts of bigger projects. We want to discuss questions like:

- What were the reasons for the failure?
- How are similar mistakes avoidable?
- Could we take corrective action in good time and turn things to a better end?
- Could (or can) the results be used anyhow, maybe in another connection?
- Were there, in spite of everything, positive things/surprises?
- Lessons learned – recommendation for others?

B1 Landfill mining

Session Coordinator: Dr. Ir. Daneel Geysen1

Enhanced Landfill Mining Consortium (ELFM)

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Introduction:
Resource efficiency is a worldwide top priority. Developing new mining opportunities, developing new mining technologies and strengthening recycling are the main pillars of a sustainable resource policy. New products and processes should of course comply with sustainable materials management including clean and environmentally safe production and efficient use of land.
Recycling was less developed in the past. Landfilling was the main waste management technique for several decades. Still now, not all waste streams can be recycled and landfilling is still the main waste management technique in many countries. The current landfilling principle is eternal containment of pollutants and eternal aftercare. An approach which does not sound cost effective and cannot completely exclude the risks for the next generations.
Old as well as new landfills constitute an interesting mine for recycling of materials or for recovering energy and land. The in-situ extraction of methane from landfills is a well known technology but not yet always applied. Less common is the ex-situ mining of landfills for the valorization of resources and land. However, the amount of land, metals and combustibles such as wood and plastics in landfills is huge. The combination of several interesting business drivers can provide the economic conditions that allow the full valorization of a landfill including the recovery of resources. Such drivers can be:

- Providing land for new industrial activities, residential area or eco service systems;
- Free space for new landfill capacity;
- Value of materials in the landfill (material or energy);
- Methane extraction (green energy) - GHG emission reduction;
- Groundwater pollution prevention;
- Soil/groundwater sanitation.

New technologies need to be developed for the efficient mining of old landfills.
Such new technologies should improve the following area's of technology:
• Landfill exploration to better estimate the value of materials in landfills allowing business planning;
• Conditioning prior to the mining activity to overcome loss of gases, odor hindrance, instability, high quality material (e.g. low moisture content of combustibles), …;
• Excavation of materials from landfills;
• Separation of materials for valorisation;
• Efficient transformation in to energy and valuable materials;
• The development of tools that can be used for policy and for the

selection of the best scenario/technology.
Landfill mining offers a role for new and operational landfills in the Sustainable Materials Management perspective. It is currently not possible to recycle all materials at the highest efficiency. In the current waste management approach (Lansinks ladder), priority is given to prevention and recycling followed by incineration with energy recovery. Landfilling is considered to be the final way out if no other options are available. However, not all wastes are combustible. And current combustion technology does not address materials valorization. More materials can be recycled if new and specific technologies can be developed; if sufficient amounts are available and if it is possible to select the ideal moment in time for recycling based on market conditions. Temporary storage until the technology is available or until the market is ready seems an interesting option. Landfills will be transformed from a threat in to a resource recovery opportunity. The landfill-for-resource framework including the concept of the temporary storage can reshape the future of landfills towards a high tech logistic/recycling centre.

Temporary storage in view of the valorization of materials and land requires a different landfilling approach than currently practiced. The landfilling practice should not result in high recycling costs later on. This is for example the case if wastes are mixed in such way that separation becomes difficult or that one waste pollutes the other. Instead, the time of storage can be used to improve the quality of the material. Controlled leaching and collection can for example remove pollutants. One of the main issues for recycling of materials from landfills is the content of fines. Another issue is the technical difficulty to separate the highly mixed waste content.

A consortium of academic partners, SME's and companies developed the ENV- FP7 research proposal 'Elmire'. Elmire addresses both in-situ and ex-situ mining of resources. The business opportunities are clear. Only for Europe, it is estimated that the 150.000 to 500.000 landfills might provide new jobs for 240.000 - 800.000 people, provide up to 5% of the current DMC/year, provide more than 3% of the EU-27 renewable energy target for 2020 and avoid up to 1 trillion € on landfill remediation costs.

Several new fields of research related to landfills and landfilling have to be developed. This session will focus on two topics. The first topic is related to old landfills and one of the first stages in the mining process being the conditioning
phase. How can we prevent the emission of CH4 and odor hindrance upon opening the landfill? How can we remove excess water while maintaining the stability during excavations?
The second topic is related to a new approach on landfilling. The session will focus on the development of the temporary storage and the possibilities for separate storage of the different wastes in order to decrease separation costs later. Also controlled leaching needs to be developed further to stabilize the contaminants in materials or to remove pollutants during the storage of wastes (e.g. polluted soils, ashes, …).
The first topic is close the other sessions that take place at the ICLRS meeting. The second topic is different and might be new to the audience. However, the concept of the temporary storage is not only applicable to new landfills. Not all the materials that become available upon mining of old landfills can be recycled directly. Part of it needs to be stored again following the temporary storage concept.

By combining these two sessions we hope to stimulate researchers to extent their activities towards the landfill-for-resource approach.
First part:
Session Chair: Daneel Geysen Session length: 60 min.

Focus: (cfr status of landfill aeration worldwide)

This part of the session is dedicated to the conditioning phase prior to the direct excavation. This session is closely related to other sessions covering aeration, methane extraction and stabilization.

Objectives:
Define and list possible processes that take place during the opening of landfills and that can result in a problem. List possible actions and research questions that can be resolved if proper conditioning is applied.

Session Schedule

Time (min)	Suggested presentation topics and discussion issues	First Author
0 - 10	Welcome + Introduction to landfill mining and different phases.	Daneel Geysen
10 - 20	Introduction to conditioning	Daneel Geysen
20 - 60	Discussion and developing a list of: 1. Processes 2. Potential problems 3. Actions/techniques 4. Research questions	All (moderation: Daneel Geysen)

Second part:
Session Chair: Daneel Geysen Session length: 60 min.

Focus:
This session is dedicated to the implementation of the temporary storage and more specifically to the potential of improving the quality of stored materials. Contaminants hamper recycling. Detoxification is a main issue in the development of closed loop processes.

Objectives:
Define and list possible processes that improve the quality of the materials present in landfills and make the recycling later on easier.

Session Schedule:

Time (min)	Suggested presentation topics and discussion issues	First Author
0 - 15	Description of the temporary storage as a concept.	Daneel Geysen
15 - 60	Discussion and developing a list of: 1. Potential processes that can improve the quality of materials (remove pollutants, breakdown of pollutants, stabilization, …) 2. Potential techniques that can be combined with the concept of the temporary storage (use of green energy for percolate circulation, …) 3. Potential processes that make separation later on easier 4. Research questions	All (moderation: Daneel Geysen)

A2 LONG-TERM BEHAVIOUR OF LANDFILL COVERS

Session Coordinator
Lale Andreas

Luleå University of Technology

Description

Landfill covers are a genuine part of landfills and are required to last for very long periods of time. The overall function is to shield the surrounding environment from the landfill while more specific purposes can vary and focus on other functions such as the minimization of gas emissions, the reduction of water infiltration, the improvement of the aesthetic appeal of the site and so on.

The session will focus on cover designs and materials that are helpful to achieve the required durability, but also on processes occurring in the materials the understanding of which being crucial for the ability to make reliable predictions of the long-term performance of the materials Moreover, methods to evaluate material properties, alterations over time and the long-term performance of cover constructions will be discussed.

Special attention will be paid to:

- Experiences from the use of secondary materials in landfill covers;
- Interactions between the mineralogical composition of a material, the surrounding environmental conditions and the leaching/immobilization processes related to these (e.g. solid solutions);
- The environmental impact connected to the mass flow due to leaching from secondary materials if used above the barrier layer.

Session Outputs

The aim of the session is to increase the awareness of possibilities to design landfill covers but also other geotechnical constructions using beneficial properties of waste/secondary materials and a better understanding of the opportunities of combing materials in order to achieve a desired function. Ideally, a group is formed that will work with a review paper about recent developments in the use of secondary materials in landfill covers addressing the long-term stability and technical and environmental performance.

Session Organization

As in all ICLRS sessions, the emphasis is put on the discussion. The session chairs will focus the discussion on a series of questions. Some of the discussion time might be spent in small groups generating ideas and reporting back to the overall group; other time will be for open discussion with the whole audience. All attendees are encouraged to contribute to the discussion with queries and ideas using media such as paper copies, transparencies, slides, etc. In that case, the material is limited to one-two pages per attendee.

Schedule

0:00 - 0:10 **Introduction & Overview**
Lale Andreas (Luleå University of Technology)

0:10 - 0:25 **Why trace elements often are immobilized in ashes and slags. On the role of solid solution in iron (hydr)oxides**
Rolf Sjöblom, Tekedo AB, Sweden

0:25 - 0:40 **Leaching behaviour of ashes in a landfill cover construction**
Lale Andreas (Luleå University of Technology)

0:40 - 0:55 ***Utilization of Chemically Stabilized Soil in a Landfill Top Cover***
Jurate Kumpiene (Luleå University of Technology)

0:55 - 1:35 **Additional contributions and discussions**

1:35 - 02:00 **Discussion & Conclusions**

B2. Status quo and future perspectives of landfill aeration

Session Coordinators: Dr.-Ing. Marco Ritzkowski1 and Prof. Dr.-Ing. Rainer Stegmann2
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Introduction:

The creation of sustainable landfills is a fundamental goal in waste management worldwide. In this respect landfill aeration is considered a tool for the accelerated, controlled and sustainable conversion of conventional anaerobic landfills into a biological stabilized state associated with a minimized emission potential. The technology has been successfully applied to landfills in Europe, North America and Asia, following different strategies depending on the geographical region, the specific legislation and the available financial resources. Furthermore methodologies for the incorporation of landfill aeration into the carbon trade mechanisms have been developed in recent years.

Against this background IWWG has set up a Task Group on landfill aeration and comprising
experts from different countries. One of the essential outcomes of this group is the monograph "Landfill Aeration", edited by Rainer Stegmann and Marco Ritzkowski. Currently the group is focusing on the compilation of an overview on the status of landfill aeration worldwide,
definitions of landfill aeration methods, stabilization and quality criteria for landfill aeration as well as the set-up of a database. These issues shall be further developed until and during the proposed session as well as in the framework of a task group meeting during the symposium.

The session is going to focus on recent examples of landfill aeration projects including challenges regarding operation, monitoring and completion. Research findings regarding specific processes and emissions during aeration will be addressed and an overview on existing methodologies for landfill aeration as CDM projects will be provided.

The workshop will be chaired by the IWWG Task Group on Landfill Aeration Chairman Marco Ritzkowski, from the Hamburg University of Technology, Germany.

First part: Status of landfill aeration worldwide

Session chair: Marco Ritzkowski

Session length: 60
min.

Focus

This part of the session is intended to provide a classification of landfill aeration in the framework of landfill aftercare and completion of the latter. Through this classification, a major European approach in connection with landfill aeration is described. Furthermore, several other intentions of landfill aeration exist and are applied on a worldwide level. The intention of the session is to identify suitable aeration concept under varying boundary conditions and to discuss potential synergies among the different approaches. Furthermore, expectations and limitations in connection with aeration projects considered for CDM and JI should be discussed.

- **Methodologies**: Advantages and disadvantages as well as applicability conditions of different aeration concepts.
- **Climate projects:** What are the (realistic) potentials of landfill aeration projects with regard to GHG emissions reduction, based on experiences so far?

Objectives

The session is intended to come up with an overview on different projects as well as their specific and general performance.

Session schedule

Time (min)	Suggested presentation topics	First Author
0 - 5	Welcome and introduction	Session chair
5 - 20	Landfill aeration in the framework of landfill aftercare and its completion.	Rainer Stegmann
20 - 35	Questions and discussion	Auditorium
35 - 45	Overview on landfill aeration projects worldwide	Marco Ritzkowski

45 - 60	Questions and discussion	Auditorium

***Second* part: Processes and emissions during landfill aeration**
Session chair: Rainer Stegmann
Session length: 60 min.

Focus
The second part of the session will focus on processes and emissions to occur during landfill
aeration. Major aspects for discussion are the leachate quality under the impact of aeration and mass balances as an approach for the determination of waste stabilization.

- **Leachate quality**: Impact of aeration on leachate quality with a focus on nitrogen compounds (role of temperatures, humic matter formation, biomass incorporation etc.)
- **Mass balances:** What are suitable approaches to follow the progress of bio-stabilisation?
  Is there a need for standardizations in order to avoid "virtual" stabilization?

Objectives
The session is intended to come up with an enhanced understanding of the processes to occur
during landfill aeration and in particular with an overview on issues demanding further research. In the framework of the IWWG Task Group "Landfill aeration" it has to be decided if a common paper on the issue of "leachate quality under the impact of aeration" should be prepared.

Session schedule

Time (min)	Suggested presentation topics	First Author
0 - 10	Long term leachate management based on anaerobic/aerobic landfill simulator studies	Markku Pelkonen
10 - 15	Direct questions	Auditorium
15 - 25	Leachate quality under the impact of aeration	Marco Ritzkowski
25 - 30	Direct questions	Auditorium
30 - 40	Enhanced biodegradation at the Landgraaf test-cell	Hans Woelders
40 - 60	Questions and discussion	Auditorium

A3 The myth or control of even gas distribution in methane oxidation systems

Session chair:

Peter Kjeldsen

The Technical University of Denmark, Department of Environmental Engineering

SESSION LENGTH: 2 hours

GOAL:

The methane oxidation in the active compost or soil layer of a biocover system is controlled by the retention time in the zone of oxygen and methane counter gradients. If the methane load into the active layer is unevenly distributed, "hot spots" may occur with lower retention times which lead to low methane oxidation efficiencies resulting in an inadequate overall efficiency of the biocover system. It is therefore crucial to avoid the formation of such hot spots. The reason can be several factors, such as heterogeneous material properties, presence of fissures, cracks, or other macro pores. Heterogeneous water distribution in the active layer may lead to locally reduced permeability and diffusivity resulting in blockage of the gas transport. Therefore it is recommended to construct engineered biocover systems with a homogeneous active layer and a coarse gas distribution layer (GDL) beneath.

Overall gas transport in a biocover system is in most cases governed by a combination of advective and diffusive processes. In coarse materials even very small pressure differences can make the advective process dominating to the diffusion process. Both processes are strongly affected by the water content. This means that capillary effects introduced by the differences of the inherent permeability of the active layer and the GDL (the GDL in most cases having the highest permeability) may form regions of high water content at the interface between the two layers. Depending of the slope of this interface these effects may be more or less dominating, thus potential leading to hot spot formation. Identification of scenarios were unwanted effects from capillary processes have been avoided, are therefore highly needed. Overall, the challenge of securing an even gas distribution to biocover systems is significant, and still a lot of open questions exist.

The purpose of this session is (1) to provide an overview of the experiences of existing large-scale biocover systems in respect to gas distribution ability, (2) methods and approaches to select proper combinations of materials and constructional details to avoid hot spot formation and (3) to identify future research directions to develop biocover systems where the problems of uneven gas distribution has been solved.

SESSION APPROACH: An atmosphere of collegial exchange will be encouraged during the session proceedings. Presentations will be 10 minutes, with 10 minutes for direct questions to authors after each thematic block of presentations. Presentation times will be strictly enforced, so that the remainder of the session is available for group discussion, possibly in small subgroups. These discussions are an important part of the session content, because they should allow presenters and attenders to more fully explore and synthesize the work presented. The session will end with report-backs from the groups and a summary statement

by the session moderator. Together, the presentations and discussion should allow us to better assess the current state of knowledge and identify future research directions.

QUESTIONS FOR DISCUSSION: Discussion questions will be provided at the time of the session. Besides, some may evolve during the presentations.

Session schedule

Total time (min)	Presentation	Corresponding Author
0-10	Welcome by Session Chair	P. Kjeldsen
10-20	Short introduction on challenges for optimized gas distribution in biocover systems	P. Kjeldsen
20-30	Requirements on gas permeability of gas distribution and methane oxidation layers in biological methane oxidation systems	J. Gebert
30-40	Heterogeneous gas distribution within a biocover designed for methane oxidation	C. Geck
40-50	Capillary barriers and uniform biogas distribution in biocovers	A. Cabral
50-60	Direct questions to presentations	
60-100	Group discussion	
100-120	Report of discussion groups and wrap-up	

B3 Anaerobic Decomposition in Landfills: carbon dynamics

Fabiano Ximenes

New South Wales Department of Primary Industries - Australia

Morton Barlaz

North Carolina State University, USA

Background

The threat of climate change has sparked a renewed level of interest in the dynamics of carbon from a variety of sources in landfills. Forest products in the form of wood and paper have historically been significant inputs of biogenic organic carbon to landfills. It is important to understand the extent and rate of decomposition organic materials, and more specifically wood and paper products as they vary significantly in their chemistry and susceptibility to decay. Decomposition data are required to inform estimates of carbon storage and methane generation from landfills. There have been several studies of the behavior of forest products in landfills over the past five years, including work at both field-scale and laboratory-scale across at least two continents (North America and Australia).

Session Objective

The session aims to foster discussion on the latest findings in the area of forest products decomposition in landfills, and its wider application to the development of carbon balances and greenhouse gas emission estimates from landfills.

Session Outline

Session schedule

Time (min)	Presentation topics
0 – 5	Welcome and introduction; review of session objective
5 - 10	Overview of current studies on forest products decomposition. This time will be used to provide all session participants with a common level of background information. It will include summaries of ongoing research in the U.S. and Australia.
10 - 20	Enhanced Abiotic Humification of Organics and its Implication on Carbon Sequestration in Landfills (Dongbei Yue)
20 - 30	Decomposition and Carbon Storage in Forest Products in Laboratory- and Field-Scale Landfills (Barlaz)
30 - 40	Enhanced biodegradation at the Landgraaf test-cell (Hans Oonk or Hans

	Woelders)
40 - 50	The Use of Copper Oxide Alkaline Hydrolysis to Quantify Lignin in Municipal Waste Components (Florentino dela Cruz)
50 – 70	Time for session participants to review posters with detailed information 0that is pertinent to this session
70 - 110	Discussion of questions outlined below
110 - 120	Summary and discussion of logical next steps.

The following topics illustrate the types of issues that will be discussed.

- Analytical issues associated with the chemical analysis of wood, paper and other organic materials
- How do we scale laboratory results to field-scale landfills?
- Given that there are differences in the behavior of specific paper pulps, how do we use data on paper decomposition to inform estimates of carbon storage and methane production? Is that still the case – I thought the eucalyptus reactors had eventually caught up with the Acacia ones in terms of methane production?
- How do we use the results of field-scale excavations?
- Should different decomposition factors be applied to different landfill types (C&D, C&I, MSW and bioreactor) for the purposes of greenhouse gas estimations?

Session Outputs

We hope that this session will serve as a motivating force for the preparation of a review article on wood decomposition in landfills. The session will also serve as a forum to explore opportunities for collaborative research.

A4.Methane Oxidation Systems – are they still in their infancy?

Session organized by CLEAR

Session chairs

Marion Huber-Humer and Alexandre Cabral

BOKU-University Vienna, Institute of Waste Management and

Département de génie civil, Faculté de génie, Université de Sherbrooke

Duration 2 hours

Background: We have been organizing specialized sessions and workshops on methane oxidation in optimized landfill covers, biocover, bio-windows or biofilters for more than 10 years now. Despite frequent meetings to discuss about this issue and about the results of lab-based investigations, field trials, pilot-scale and demonstration projects all over the world, these systems are not yet even considered in most countries. What are the reasons? Are we still far away from practically applicable methane oxidation systems? What are the concerns?

Since biocovers, biofiltres and bio-windows are part of a capping system, they must not disturb any of the several different functions of the capping system, i.e.: control leachate generation; minimize methane and odor emissions; provide a suitable basis for vegetation or for other intended after-use options; protect against erosion; etc.

Can we engineer optimized landfill capping systems to oxidize methane given these constraints?

In order to answer to this crucial question, we have to put all different pieces of this puzzle together to get an overall look on how such systems will act over the long-term, while considering the conflicting needs embedded in the design of methane oxidizing structures. For example, how to reconcile water infiltration control with methane oxidation? Indeed, sufficient infiltration is necessary during the early life of MSW-landfills to trigger the degradation processes. However, over the longterm, leachate minimization is desired.

Goal: The purpose of this session is to consider and discuss results, experiences and observations from different field trials and demonstration projects on methane oxidation including other aspects of cover functions, for example leachate regulation/minimization. The goal is to develop a strategy that may help to place methane oxidation in the main stream of landfill cap design.

Session Introduction: The session will be introduced by the session co-chairs, who will welcome the attendees, review the goals of the session, and describe the format that will be followed.

Session Approach: An atmosphere of collegial exchange will be encouraged during the session proceedings. Presentations will be limited to 15 minutes, with some extra time allocated for direct questions to the presenter. Small discussion groups will then be formed and will have 20 min to discuss about key questions prepared by the co-chairs (listed below). The main questions/ideas/statements/conclusions will be summarized by one individual. A brief whole-audience discussion will ensue.

The questions listed below are provided to guide the discussions.

Questions for Discussion:

- How many and which experiences from demonstration projects or practical applications are available that address both gas and water infiltration issues?
- Is there emerging clarity about the environmental and material factors that have the most potent effects on methane oxidation and water infiltration control? How can these factors be combined in the best way?
- What is the role of vegetation? What do we really know about it?
- How close are we to being able to reliably assess the long-term behavior and functionality of biocovers and/or evapotranspiration caps? How close do we need to be?
- Can both functionalities (methane oxidation enhancement and water regulation/minimization) actually be combined in one cover? Which criteria are the most relevant for such an all-in-one approach? Is an all-in-one approach really appropriate?
- Are there drawbacks to such systems that need to be addressed? What kind of operation/maintenance/regeneration are they likely to require?
- How do we sell the importance of methane oxidation?

Session Output: Clarification of when and in which form of capping systems are viable. Identification of most important factors addressing the main aspects discussed. Definition of the role of vegetation and its maintenance. Development of a long-term strategy with respect to all relevant aspects.

B4 Modeling Landfill Gas Generation in Landfills Receiving Biodegradable Waste

Morton Barlaz

North Carolina State University, USA

Debra Reinhart

University of Central Florida, USA

Background

The reliable prediction of landfill gas generation remains an elusive goal. There are many landfill gas generation models in use around the world (e.g., LandGem in the U.S., GasSim in the U.K., IPCC model). All of the models take an empirical approach to landfill gas generation and require two input parameters, though the form of these parameters varies amongst models; (1) the rate of decay, and (2) the methane potential of the material that is undergoing decomposition. Some models divide the waste into multiple fractions (e.g., GasSim), while others calculate the methane potential based on the mass of degradable organic carbon (e.g., IPCC). When these models are applied to real landfills, there can be considerable differences between predicted gas generation and the volumes that are actually recovered, a problem that is exacerbated by the fact that gas collection efficiencies vary with time and are uncertain. This problem is further confounded by the fact that waste compositions are changing and less biodegradable material is entering landfills as programs to divert food waste and fiber grow.

More reliable predictions of landfill gas generation will be useful for several reasons including (1) improved emissions estimates, (2) more reliable estimates of the feasibility of a landfill's potential for energy recovery, and (3) guidance to owners on the appropriate design capacity for gas to energy conversion equipment, and how this capacity requirement will change over time.

Session Objective

The objective of this session is to provide a forum for a discussion of recent research on landfill gas generation modeling in different parts of the world.

Session Schedule

Time (min)	Presentation topics
0 – 5	Welcome and introduction; review of session objective
5 - 20	Background on landfill gas modelling and a brief review of models in use in different countries
20 - 50	Presentation of analysis of field data from US Landfills
50 - 60	Questions on presentations
60 - 110	Discussion of session questions
110 – 120	Summary

The following topics will be discussed.

- What factors should be considered (temperature, waste composition, etc) and how?
- Can we use field data for model validation and can it be shared among researchers?
- How is uncertainty in modeling best expressed?

Session Outputs

This session will inform researchers from various perspectives on new approaches to gas modeling and will hopefully lead to sharing of data across countries to evaluate different approaches to landfill gas modeling.

A5 The use of stable isotopes for quantification of methane oxidation in landfill covers – critical factors, limitations, and future developments

Session chairs:

Charlotte Scheutz

The Technical University of Denmark, Department of Environmental Engineering

Julia Gebert

University of Hamburg, Institute of Soil Science, Germany

SESSION LENGTH: 2 hours

GOAL:

Stable carbon isotope analysis provides a field approach for the in situ quantification of fractional CH_4 oxidation, that is, the percentage of CH_4 that is oxidized during transport through the landfill cover materials, based on the difference between the $\delta^{13}C$ of emitted CH_4 compared to the $\delta^{13}C$ of unoxidized CH_4 in the anaerobic zone. The method relies on the preference of methanotrophs for the stable carbon isotope of smaller mass, ^{12}C rather than ^{13}C, expressed by the fractionation factor α_{ox}. The lighter isotopologue also has a higher diffusion coefficient, causing a fractionation due to diffusive gas transport, expressed by the fractionation factor $\alpha_{trans.}$ The latter is dependent on soil properties, and environmental variables such as soil moisture, as well as the relationship between diffusive and advective flux. In general, the isotopic methods are deployed in three ways: (1) at ground level, using static chambers and comparing the $\delta^{13}C$ of CH_4 in the refuse (by sampling at gas recovery wells, gas collection headers, or deep gas probes) to the emitted CH_4 collected in the chamber; (2) in the lower atmosphere, relying on a comparison between the $\delta^{13}C$ of atmospheric CH_4 in an upwind transect to a downwind transect; and (3) below ground level, relying on soil gas profiles for CH_4 and $\delta^{13}C$. These approaches can also be used in combination to derive CH_4 oxidation at various temporal and spatial scales.

The stable isotopic method is more and more frequently used and oxidation results reported without discussion of the associated uncertainty or methodological limitations. The method is associated with uncertainty related to determination of the fractionation factors (oxidation, transportation), physical settings (spatial and temporal variability), and other limitations (atmospheric uptake/negative fluxes, detections limits in relation to very dilute samples).

The purpose of this session is three fold (1) to provide an overview of the methodology as currently applied, (2) to point to limitations and uncertainty and (3) to identify future research directions to develop the use of stable isotopes further.

SESSION APPROACH: An atmosphere of collegial exchange will be encouraged during the session proceedings. Presentations will be 10 minutes, with 10 minutes for direct questions to authors after each thematic block of presentations. Presentation times will be strictly enforced, so that the remainder of the session is available for group discussion, possibly in small subgroups. These discussions are an important part of the session content, because they should allow presenters and attenders to more fully explore and synthesize the work presented. The session will end with report-backs from the groups and a summary statement by the session moderators. Together, the presentations and discussion should allow us to better assess the current state of knowledge and identify future research directions.

The session chairs will circulate suggestions for discussion questions to session presenters before the conference. Authors will be asked to adapt their presentations to the thematic focus of the session.

QUESTIONS FOR DISCUSSION: Discussion questions will be provided at the time of the session, but some sample topics include:

- What are the basic assumptions when using the Raleigh equation?
- Which isotope fractionation factor due to oxidation, α_{ox} should be used - can we rely on literature data?
- What is the isotope fractionation factor due to transportation, $\alpha_{trans,}$ can we assume that it is 1 and if not what is the associated uncertainty?
- What is the influence of temperature and moisture on the fractionation?
- What are the capabilities, limitations, availability and costs of stable isotopic analysis?
- How much accuracy can we expect and it this acceptable for landfills operators, policy makers, researchers, etc.?
- What is the next step in developing a more robust method based on isotopes for quantifying methane oxidation?

Session schedule

Total time (min)	Presentation	Corresponding Author
0-10	Welcome by Session Chairs	C. Scheutz, J. Gebert
10-20	Short introduction on challenges in using stable carbon isotopes for quantification of methane oxidation in biocovers	C. Scheutz
20-30	Drawbacks in the Evaluation of Methane Oxidation Efficiencies Using Carbon Stable Isotopes	M. Capanema
30-40	Stable isotope fractionation during microbial methane oxidation – new insights from gas push-pull tests	J. Streese-Kleeberg
40-50	Effect of diffusive gas transport on the fractionation of stable isotopes in landfill covers	J. Gebert
50-60	Methane Oxidation at a Landfill Scale	J. Chanton
60-70	Direct questions to presentations	
70-100	Group discussion	
100-120	Report of discussion groups and wrap-up	

B5 LANDFILLS AND DISASTER WASTE MANAGEMENT

Session Coordinator
Toshihiko Matsuto
Hokkaido University, Japan

Description

Disasters waste is now a common category in solid waste management due to various big disasters occurred all over the world recently, which includes earthquakes, tsunamis, hurricanes, floods. Disaster waste has clear distinction from ordinary waste such as MSW and industrial waste in the following points.

1. Due to tremendous amount of waste production, existing waste management system is not sufficient at all. Overall planning of waste management, starting from removal of waste, collection, transport, temporary storage, separation, pre-treatment recycle, and landfill should be set up.
2. Characteristics of generated waste depend on a disaster mechanism. Waste generated by earthquake is very similar to construction demolition waste basically, but tsunami waste is composed of the mixture of everything: MSW, ELV, C&D waste etc. and even hazardous material is included. Waste management strategy should cover all these different waste and material together.
3. Before transferred to waste management facilities, disaster waste is piled on an open area which is accessible by people, so various problems happen. Sanitation, odor, spontaneous fire are among those problems, which are not usually in solid waste management, and experience and knowledge is lacking.
4. In addition to the unfamiliarity for waste management people, every disaster is significantly site-specific each other by geographical condition. Therefore it is very difficult to discuss about disaster waste in a universal manner.

Despite of the limitation and difficulty listed above, it is quite useful to share knowledge and experience in academic meeting. “Disaster will happen when least expected”. This session was organized to give participants

Schedule

0:00 - 0:10 **Introduction: Disaster Waste Management and Landfills**
Toshihiko Matsuto (Hokkaido University)

0:10 – 0:25 Deposition and Landfilling of Disaster Waste

Per EO Berg(Swedish Civil Contingencies Agency)

0:25 – 0:40 Establishing excreta discharging facility at Truitier dumpsite in Port au Prince 2010. Conditions and constrains.

Sofia Billvik(Swedish Civil Contingencies Agency)

0:40 – 1:00 **Audience Questions and Discussion**

1:00 –1:15 Inexperienced Challenge on Tsunami Disaster Waste in Japan

Toshihiko Matsuto (Hokkaido Univerisity)

1:15 – 1:30 Characteristics of temperatures and gas components in debris storage sites in East Japan Earthquake disaster areas

Hideki Yoshida (Muroran Institute of Technology)

1:30 – 2:00 **Audience Questions and Discussion**

A6 AND A7 Landfill Aftercare Completion

Introduction

There are not many examples of existing regulations that provide clear guidance for landfill aftercare completion. Those regulations that do address it in general require that aftercare can only be completed when active care has ended. In addition most experts agree that during aftercare a progressive approach to reduce aftercare and monitoring efforts makes sense and is likely to be followed by most operators and authorities. This should be based on periodic assessments and will result in a shift from active to more passive measures. But when exactly can a measure be considered passive? Some monitoring may have to be continued when active aftercare is ended. When the after-use generates some income or benefits for society, some remaining maintenance and monitoring could very well be acceptable. It is not entirely clear what level of monitoring and maintenance can be considered acceptable. Depending on materials used, engineered containment can have a finite functional life expectancy. Continued functionality of a surface sealing requires maintenance and replacement. A cover is always an engineered construction. Again it is not entirely clear what covers can be approached with active and with passive care. It has been proposed that passive care means hydraulic equilibrium and absence of both active management and engineered containment. In reality, containment will gradually decline. While containment still functions, water might build-up inside the landfill and become a liability. It is obvious there is a need for a more precise distinction between active and passive care and an outlook on how to deal with containment in the long term. The objective of Session I is to address the distinction between active and passive care.

Aftercare completion is often defined in terms of ‘no remaining threat to human health and the environment’. Regulations that date back to the late 1980’s often try to achieve this by means of the ‘dry tomb approach’. Complete isolation should result in a ‘zero emission’. There are two problems with this approach. It is technically impossible to construct hectares of bottom liner or surface sealing that stop all water transport. Some technical guidelines even explicitly mention ‘an unavoidable emission in the technical sense’. Zero emission is wishful thinking. It is necessary that regulations consider acceptable risk. Moreover stabilisation processes are inhibited due to limited water transport. The emission potential of the waste is therefore handed over to future generations. It has been proposed a decision to release a landfill from aftercare can be made when the potential release of pollutants in the absence of active control is below attenuation capacity of the environment. In that respect it is important to consider the remaining emission potential of the waste mass. Session II will address the question how we can determine the remaining emission potential of a waste body and how this can be incorporated in a decision-making framework.

Landfill aftercare completion I: from active to passive care

Session chair: Heijo Scharff

Session length: 2 hours

Focus

The session will focus on containment, active and passive aftercare. There are some proposals (United States, Austria) of a gradual progress from active to passive aftercare. The active and passive measures mentioned in these proposals could be made more precise by providing concrete examples and assign these to a specific category of phase in aftercare. Authors will

be selected for presentation in this session based on providing such concrete examples. Questions that could be addressed are:

- Can concrete examples of aftercare measures be assigned with consensus to the categories active and passive care?
- What required level of continued monitoring and maintenance can be considered acceptable after ending aftercare?
- What kind of engineered containment can be considered passive care?
- What kind of engineered containment should be considered active care?
- Can expected long term problems with containment be solved?

Suggestions for discussion purposes

	Aftercare period			Monitoring period
	Active aftercare	Partially active aftercare	Passive (self sustaining) aftercare	Custodial care
Responsible	Landfill owner/operator	Landfill owner/operator	Landfill owner/operator	Society/public authority
Activities				
Contact prevention	Inspection and maintenance of recultivation layer ..	Inspection and maintenance of recultivation layer ..	Inspection and maintenance of recultivation layer ..	Site surveillance and after-use concept ..
Leachate control	Leachate collection by means of pumping Leachate treatment on-site Offsite trucking or discharge of leachate to sewer ..	Leachate collection by windmill pumping Leachate treatment onsite in constructed wetlands Treated leachate discharge to sewer, surface water, or infiltration into subsurface ..	Leachate collection by means of gravitation No treatment required Treated leachate discharge to sewer, surface water, or infiltration into subsurface ..	Leachate collection by means of gravitation No treatment required Treated leachate discharge to sewer, surface water, or infiltration into subsurface ..
Gas control	Active gas collection Landfill gas utilisation (engine, turbine, boiler, upgrading, ..) Landfill gas flaring ..	Passive gas collection Ignition supported flaring Methane oxidation cover, window or filter (active or passive) ..	Passive gas collection Methane oxidation cover, window or filter (passive only) ..	Passive gas collection Methane oxidation cover, window or filter (passive only) ..
Emission potential reduction	Flushing, leachate recirculation In-situ aeration ..	Windmill operated recirculation Windmill operated in-situ aeration ..	..	..
Monitoring	Cap inspection and emission monitoring Monitoring of ambient concentrations (e.g. groundwater wells, perimeter gas monitoring wells, gas emissions, surface water) ..	Cap inspection and emission monitoring Monitoring of ambient concentrations (e.g. groundwater wells, perimeter gas monitoring wells, gas emissions, surface water) ..	Cap inspection and emission monitoring Monitoring of ambient concentrations (e.g. groundwater wells, perimeter gas monitoring wells, gas emissions, surface water) ..	Cap inspection Monitoring of ambient concentrations is not required. Monitoring natural attenuation can be considered ..

Approach

An atmosphere of collegial exchange will be encouraged during the session. Presentations will be short and focused. Direct questions should concern the focus of the session. This will allow maximum opportunity for constructive discussion. The audience will be asked to actively participate in assigning concrete examples to care categories in a schematic of progressive reduction of aftercare effort.

Goal

The optimal result of the session would be consensus on a more precise distinction between active and passive aftercare measures. The intended output of the session is a schematic of active and passive care measures and a list of unresolved questions.

Session schedule

Time (min)	Suggested presentation topics	First Author
0 – 5	Welcome and introduction	chair
6 – 18	Existing guidance on active / passive control (EPCC, ÖWAV,..)	David Laner
19 – 31	Long term problems with / solutions for containment	Ole Hjelmar
32 – 44	Direct questions to presentations	Auditorium
45 – 120	Discussion	Auditorium

Landfill aftercare completion II: decision making

Session chair: Heijo Scharff

Session length: 2 hours

Focus

The session will assume that release from aftercare is desirable to minimise a potential intergenerational conflict. To achieve that, it is necessary to be able to assess that a landfill no longer poses a threat to human health and the environment. Based on that assessment a decision can be made. The session will focus on how to determine the remaining emission potential of a waste body and how this can be incorporated in a decision-making framework. Session II will 'park' the issue of atmospheric emission and will focus on leachate control. The emission potential to soil, groundwater and/or surface water is the result of the leaching characteristics of the waste and the actual infiltration rate. The infiltration rate depends on type of cover and climatic conditions. Questions that could be addressed are:

- Can a decision-making framework with universal applicability be designed?
- Can such a framework accommodate both risk assessment and limit value approaches?

- What kind of information should be considered in a decision-making framework?
- To what extent do leachate quality data reflect the conditions in the waste body?
- Is it conceivable that we can rely on leachate quantity and quality dynamics?
- Do we need waste samples and leaching tests?

Approach

An atmosphere of collegial exchange will be encouraged during the session proceedings. Presentations will be short and focused. Direct questions should concern the focus of the session. This will allow maximum opportunity for constructive discussion.

Goal

A good result of the session would be a shared notion on what is relevant or necessary for a decision-making framework and recommendations on aspects that need further attention and development. The intended output of the session is a 'sketch' of a decision-making framework including a list of agreed aspects and unresolved issues. Publication of the output of both sessions is considered.

Session schedule

Time (min)	Suggested presentation topics	First Author
0 – 4	Welcome and introduction	chair
5 – 14	Leachate decision framework	Marion Crest
15 – 24	Dynamics in leachate quantity and quality	Johann Fellner
25 – 34	Feasibility of leachate quality assessment	Timo Heimovaara
45 – 54	Feasibility of waste sampling and analysis	Morton Barlaz
55 – 64	Direct questions to presentations	Auditorium
65 – 120	Discussion	Auditorium

B6 AND B7 The Fate of Compounds of Emerging Concern in Landfills

Session Coordinators

Nicole Berge, PhD, University of South Carolina

Debra R. Reinhart, Ph.D., P.E., University of Central Florida

Topic Description:

As production and subsequent disposal of endocrine disrupting compounds (EDCs), pharmaceuticals, personal care products (PPCPs), nanomaterials, and other chemical and biological contaminants of emerging concern increases, so does the importance of understanding their behavior during waste management. Products containing these materials are abundant in MSW landfills. For example, shampoo, detergents, carpets and textiles all contain compounds of concern (e.g., 4-nonylphenol, fluorochemicals) and are routinely discarded in landfills. In addition, engineered nanomaterials including titanium oxides, silver oxides, C_{60} fullerenes, carbon nanotubes, and silica have been incorporated in several commercially-available and commonly discarded products, including plastics, ink jet printer paper, textiles, cosmetics, sunscreens, and sporting goods. EDCs have been found in trace levels in soil and groundwater contaminated by landfill leachate, signifying the importance of understanding the fate of such compounds. Our abilities to detect these contaminants are improving and new materials are emerging, creating the need for additional research.

This session has two parts: Part 1 focuses on chemical and biological contaminants of emerging concern and Part 2 focuses on the fate of engineered nanomaterials. The goals of each part of the session are: (1) to assess our current knowledge of the fate and transport of contaminants of emerging concern, (2) discuss challenges associated with the study of the fate and transport of these contaminants, and (3) propose innovative approaches to such study. The session will involve audience participation and explore the potential for future collaborations to further research on the topic. A product of this session could be an international collaborative proposal such as US National Science Foundation (NSF) Partnerships for International Research and Education (PIRE) or Science Across Virtual Institutes (SAVI) and development of an emerging compound and/or nanowaste IWWG working group. Specific items of discussion include:

Key Questions:

- The best approach to determine the fate of these materials as they are collected, treated, and disposed
  - What is the magnitude of these materials in the waste stream?
  - What is the potential for contaminant emissions during waste management and where does this occur?
- Chemical and biological contaminants of concern in landfills
  - Will these compounds be biodegraded within the landfill and if so what byproducts will be produced?

- How does waste age influence compound mobility?
- How does the presence of organic matter influence the fate and transport of these contaminants?
- What leachate characteristics control compound mobility?
- Will these compounds migrate through landfill bottom liners?

- Fate of engineered nanomaterials in landfills
  - Given the large number of nano materials and their functionalization, how can these wastes best be regulated?
  - How will leachate characteristics (particularly organic matter) and waste age influence nanomaterial transport?
  - Will nanomaterials influence microbial processes?
  - Will nanomaterials penetrate landfill bottom liners?
  - How will nanomaterials influence leachate treatment?
  - What controls nanomaterial release from commercial products into leachate?

Session Organization

Part 1: Chemical and biological contaminants of emerging concern (2 hrs):

0-0:15 **Introduction – challenges and opportunities**

Debbie Reinhart (NSF)

0:15-0:35 **Fluorochemical signatures in municipal waste and landfill leachate**

Mort Barlaz (NC State)

0:35-0:50 **Gas production and water quality impacts of land disposal of cattle carcasses**

Shannon Bartelt-Hunt (UNL)

0:50 -0:55 Break

0:55 - 1:20 Large group discussion of session questions

1:20 – 1:50 Audience break-out groups to explore what is needed next and the potential for collaboration

1:50 – 2:00 Reconvene and report

Part 2: Fate of engineered nanomaterials (2 hrs):

0-0:15 **Introduction – challenges and opportunities**

Nicole Berge (USC)

0:15-0:30 **Fate of nanoparticles in municipal solid waste landfills**

Stephanie Bolyard (UCF)

0:30-0:45 **Role of mobile sorbent nanoparticles in transport of toxic metals in clay mineral barriers**

Anne Stringfellow (University of South Hampton)

0:45-1:00 **Behavior of single walled carbon nanotubes in mature leachate**

Nicole Berge (USC)

1:00 – 1:05 Break

1:05 - 1:25 Large group discussion of session questions

1:25 – 1:35 **US National Nanotechnology Initiative**

Debbie Reinhart (NSF)

1:35 – 1:50 Audience break-out groups to explore what is needed next and the potential for collaboration

1:50 – 2:00 Reconvene and report

List of abstracts

First author	Abstract title	Session
Aljaradin	Potential of groundwater contamination due to uncontrolled landfilling A case study form Jordan	
Andreas	Leaching behaviour of ashes in a landfill cover construction	Landfill covers
Andreas	Implementing a lab-developed liner recipe in a full scale cover construction – challenges, setbacks and success	Mistakes
Barlaz	Feasibility of Waste Sampling and Analysis	Aftercare completion
Bartelt-Hunt	Gas production and water quality impacts of land disposal of cattle carcasses.	Emerging compounds
Baziene	The influence of precipitation on leachate parameters in new EU standards compliant landfill	
Berg	Deposition and Landfilling of Disaster Waste	Disaster waste
Berge	Behavior of Single Walled Carbon Nanotubes in Mature Landfill Leachate	Emerging compounds 2
Billvik	Establishing excreta discharging facility at Truitier dumpsite in Port au Prince 2010. Conditions and constrains.	Disaster waste
Bolyard	Fate of Nanoparticles in Municipal Solid Waste Landfills	Emerging compounds 2
Brännvall	Characterisation of waste material mixtures for landfill top cover application	Landfill covers
Bun	Integrated modeling and up-scaling of landfill processes and heterogeneity using stochastic approach	
Cabral	Capillary barriers and uniform biogas distribution in biocovers	
Capanema	Drawbacks in the Evaluation of Methane Oxidation Efficiencies Using Carbon Stable Isotopes	Stable isotope
Chanton	Methane Oxidation at a Landfill Scale.	Stable isotope
Crest	A Decision-Making Framework for Long-Term Leachate	Aftercare

	Managemen	completion
Dela Cruz	Determination of Sources of Organic Matter in Landfill by Analysis of CuO Oxidation Products of Lignin	Forest products
DelaCruz	Hydrogen Sulfide Production from Different Sulfur-Containing Materials under Anaerobic Conditions	
Fellner	Dynamics in leachate quantity and quality at MSW landfills	Aftercare completion
Gebert	Effect of diffusive gas transport on the fractionation of stable isotopes in landfill covers	Stable isotope
Gebert	Requirements on gas permeability of gas distribution and methane oxidation layers in biological methane oxidation systems	
Geck	Heterogeneous gas distribution within a biocover designed for methane oxidation	
Hao	Acetate oxidizing syntrophs functioning as methanogenic initiation center from acid crisis in anaerobic degradation	
Heimovaara	Is it possible to quantify emission potential from high resolution monitoring of leachate dynamics?	Aftercare completion
Huber-Humer	Experiences from biocover implementations in Austria regarding leachate minimization and enhancement of methane oxidation	
Imhoff	Biocover Properties Affecting Spatial Variability of Methane Oxidation in Landfill Covers	
Kjeld	Reducing methane emissions from Icelandic landfills by use of passive oxidizing biocovers	
Konstantaki	Geophysical monitoring of in situ redox processes	
Kumpiene	Arsenic leaching in landfilled soil	Landfill cover
Lagerkvist	Some simple and frequent errors is landfill research	Mistakes
Laner	How to consider long-term technical barrier performance in an aftercare completion framework?	Aftercare completion
Lang	Fluorochemical Signatures in Municipal Waste and Landfill Leachate	Emerging compounds

Lu	Landfill Leachate Treatment by Plant-Cover Soil Irrigation System: Field Monitoring on Ecophysiological Response of Nerium indicum. Mill	
Masood	Status of landfills in developing countries: A case study on Pakistan	
Matsuto	Inexperienced Challenge on Tsunami Disaster Waste in Japan	Disaster waste
Ndanga	Influence of vegetation type on microbial oxidation of methane in landfill covers.	
Nordmark	Mobility of redox sensitive elements due to organic matter in contaminated soil; bottom ash and residual waste fraction	
Oonk	Enhanced biodegradation at the Landgraaf test-cell	Forest Products
Pelkonen	Long term leachate management based on anaerobic/aerobic landfill simulator studies	Landfill aeration
Reinhart	Modeling Landfill Gas Generation in Landfills Receiving Biodegradable Waste	Gas Modeling
Ritzkowski	How does landfill aeration impact on leachate composition?	Landfill aeration
Riviere	Improvement of waste moisture content during filling phase: consequences on biogas and leachate production	
Röwer	Heterogeneous emission from a biocover designed for methane oxidation	
Scharff	Mistakes in construction of a controlled field trial - and how they were fixed	
Sjöblom	Why trace elements are often immobilized in ashes and slags. On the role of solid solution in iron (hydr)oxides.	Landfill covers
Stegmann	Landfill aeration worldwide: Concepts, indications and findings	Landfill aeration
Streese-Kleeberg	Stable isotope fractionation during microbial methane oxidation – new insights from gas push-pull tests	Stable isotope
Stringfellow	Role of mobile sorbent nanoparticles in transport of toxic metals in clay mineral barriers	Emerging compounds 2

Thabet, Tolaymat.	Environmental Characterization of "Salt Cake": A Secondary Aluminum Processing Waste	
van Turnhout	Quantification of (bio)geochemical heterogeneous activity in full-scale landfills	
Wang	Modernizing Methane Generation Models and Data from U.S. Landfills	Gas Modeling
Wang	Decomposition and Carbon Storage in Forest Products in Laboratory- and Field-Scale Landfills	Forest products
Wikström	Fluorescent in situ hybridization technique in anaerobic process studies	
Woelders	Sustainable emission reduction of Dutch landfills in the aftercare period	
Ximenes	Decomposition of composite wood products and paper products under controlled anaerobic conditions	Forest products
Yoshida	Characteristics of temperatures and gas components in debris storage sites in East Japan Earthquake disaster areas	Disaster waste
Yue	Enhanced Abiotic Humification of Organics and its Implication on Carbon Sequestration in Landfills	Forest products
Zdanevitch	Measurement of persistent organic pollutants in landfill leachates.	Emerging compounds
Zhou	Three-dimensional Modeling of Tracer Gas Emissions over a Landfill	

Potential of groundwater contamination due to uncontrolled landfilling A case study form Jordan

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Landfills have been the most common and cheapest methods of waste disposal in many countries around the world, so also in developing countries. It is considered to be cost effective and reliable method if the land is available. However the inadequate management and operation of these landfills could pose serious negative environmental impacts, moreover, lack of sufficient knowledge of climatic topography, hydrologic and geologic conditions, in combination with limited legal requirements on proper landfilling, and limited environmental control of the solid waste management can make landfills a source of pollution to the surroundings.
Landfills have been identified as one of the major threats to groundwater recourses where, the placed waste in landfills is subjected to infiltration form precipitation or from co-disposal of liquid and sludge with MSW.

Jordan has seen a large increase in population during the past five decades as a result of population growth and forced migrations, and also, accompanied with this increase, a cultural and economical development that has improved the standard of living and changing consumer habits in the community, resulting in a clear increase in the volume of waste (Aljaradin and Persson 2011). Recent studies showed that the rate of production of solid waste in Jordan is estimated at about (1.46 million tons per year), and is expected to reach (2.5 million tons) by year 2015 (e.g., Abu Qdais, 2007). Figure 1 illustrates landfilling practices in the country.

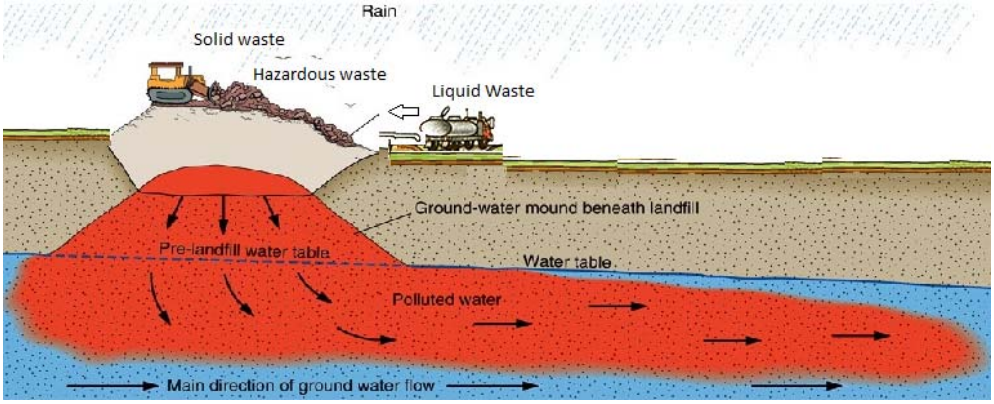

Fig. 1. Landfilling practice at Jordan.

Generation of leachate from municipal solid waste landfill in arid regions was neglected on the assumption that minimal leachate could be formed in the absence of precipitation. Many studies, on the other hand, have identified potential of contamination due to uncotrolled landfilling (e.g., Mor et al., 2006; Christensen, 2004; Ali, 2006; Abu-Rukah, 2001; 2005; Mutewekil et al., 2008; Awawdeh, 2009). Often, liquid waste is landfilled together with solid waste. Therefore, Leachate quality data were collected from one of the major landfills where co-disposal of MSW and other solid and liquid wastes is practiced. The analysis of data

suggests that leachates from both landfills are severely contaminated with organics, salts and heavy metals. Furthermore, the leachate analysis shows that contamination of the soil under the landfill occurs.

Groundwater from various wells were found to be unsuitable for domestic water use. The most alarming problems for drinking purposes were found in three wells that were classified as having too high fluorine concentration while one of them having too high mercury concentration and the other having high content of both chlorine and sodium ions giving a salty taste to the water and thus making it unsuitable for drinking purposes. Nearly all wells had very hard waters making them unsuitable for domestic uses due to scale deposit problems, with concentrations exceeding the guideline values. Therefore, this groundwater may cause severe health damages to humans if indigested.

References

Abu-Rukah (2005). Study of colloidal content and associated heavy metals in landfill leachate: a case study of El-Akader landfill site: Jordan. Geneva, SUISSE, Inderscience Enterprises.

Abu-Rukah, Y. A.-K., O. (2001). "The assessment of the effect of landfill leachate on ground-water quality-a case study. El-Akader landfill site-north Jordan." Journal of Arid Environments 49(3): 615-630.

Abu Qdais, H. A. (2007). "Techno-economic assessment of municipal solid waste management in Jordan." Waste Management 27(11): 1666-1672.

Aljaradin, M. and K. M. Persson (2011). "Current situation of municipal solid waste management in Jordan." Waste Management 31(8): 1897-1898.

Ali El-Naqa, N. H., Mustafa Kuisi (2006). "GIS-based evaluation of groundwater vulnerability in the Russeifa area, Jordan." Revista Mexicana de Ciencias Geológicas 23(3): 10.

Awawdeh, M., Jaradat, Rasheed (2009). "Evaluation of aquifers vulnerability to contamination in the Yarmouk River basin, Jordan, based on DRASTIC method." Arabian Journal of Geosciences 3(3): 273-282.

Christensen, D. L. B. a. T. H. (2004). "Speciation of Heavy Metals in Landfill Leachate: A Review." Waste Management & Research 22(3).

Mor, S., K. Ravindra, et al. (2006). "Leachate Characterization and Assessment of Groundwater Pollution Near Municipal Solid Waste Landfill Site." Environmental Monitoring and Assessment 118(1): 435-456.

Mutewekil M. Obeidat, N. A. H. Fayez Y. Ahmad, et al. (2008). "Assessment of nitrate contamination of karst springs, Bani Kanana, northern Jordan." Revista Mexicana de Ciencias Geológicas 25(3): 426-437.

LEACHING BEHAVIOUR OF ASHES IN A LANDFILL COVER CONSTRUCTION

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In 2000 Telge Återvinning – a waste management recycling company – started investigating ashes from the incineration of biofuel and industrial waste in collaboration with the Division of Waste Science and Technology at Luleå University of Technology. The company was given a permit from the Swedish Environmental Court to cover four hectares of their municipal waste landfill.

In 2006 the company received an unlimited permit to cover the remaining part of the landfill using secondary materials and following the tested design.

Residuals from household and industrial waste were tested. Initially, residuals from biofuel incineration were subject to testing. Later the material palette was extended to treated MSWI ashes, waste water sludge, contaminated soils, and compost. Several different sub-fractions of ashes were included in the investigation e.g. fly ash, aged bottom ash, and bottom ash products after up-grading including dewatering, separation and sifting.

Six test areas were built in order to give a good representation for cover construction in flat and steep areas with different compositions of liner materials.

The results show that in all areas the hydraulic conductivity construction yields less then 50 liters per square meters and years and can be less the than 5 liters in a repository for hazardous waste if required. The leachate below the liner varies depending on the used materials in the different areas. In general, it contains high concentrations of chloride, TOC and nitrogen but low concentrations of heavy metals and other trace elements.

In accordance with literature data the field observations show the liner material constructed only by ash material under certain conditions can form a monolithic structure due to very slow processes thus indicating small pore volumes that unable water – air to interact with other media.

Implementing a lab-developed liner recipe in a full scale cover construction – challenges, setbacks and success

Lale Andreas, * *Waste Science & Technology, Luleå University of Technology, Luleå, Sweden, laan@ltu.se*

Metallurgical slags from steel industry are valuable materials with beneficial properties for use within different types of constructions. Depending on their properties and the specific requirements of the construction, different slags suit different applications.

Many landfills have to be covered in Sweden and Europe within the next years. For the municipal landfill of Hagfors, the steel company Uddeholms AB initiated a joint research project together with the municipality of Hagfors and Luleå University of Technology in 2003. The work showed that several slags from scrap metal based steel making can be used in the construction. As a result of the project, Hagfors municipality signed a contract with the steel company about using slags instead of natural raw materials in the landfill cover which contributes to a sustainable use of limited natural resources.

The design of the top cover was developed by the research groups of Waste Science and Technology and Process metallurgy at the department of Civil, Environmental and Natural Resources Engineering at Luleå University of Technology in close cooperation with Uddeholms AB and Hagfors municipality.

Electric arc furnace slag (EAFS) and ladle slag from Uddeholms AB were tested in the lab and in full scale during the covering of the Hagfors landfill. The slags were used in the foundation layer, the liner and the drainage layer. Five test areas were built between 2005 and 2011 and samples were taken twice per year. The results show that the technical and environmental demands with regard to permeability and leaching can be fulfilled.

This presentation focuses on experiences from the project related to the transfer of the results from the lab to the full scale construction work in the field.

Feasibility of Waste Sampling and Analysis

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The objective of this presentation is to provide a framework for a discussion on the need for sampling solids as part of an evaluation of post-closure care at a landfill. Solids that are buried in a landfill have at least two characteristics that influence the potential threat of a landfill to human health and the environment; (1) the extent of biodegradation, and (2) the leaching potential. The extent of biodegradation can contribute to an assessment of the potential for additional gas production and settlement, while the leaching potential can contribute to an assessment of leachate quality. Thus, it could be argued that characterization of the buried solids is critical for a mechanistic assessment of landfill behavior. Alternately, it could be argued that trends in methane production, settlement and leachate generation provide suitable surrogates for solids characterization and solids sampling is not necessary.

A discussion of the utility of solids sampling must consider a number of issues. First is the fundamental issue of whether results from individual samples are meaningful. A unit volume of waste in a landfill does not exist in isolation. Contaminants leached from the unit volume will pass through a waste column prior to reaching the bottom of a landfill and contaminants may be diluted or attenuated. Similarly, biodegradation potential is most relevant in the context of the configuration of the final cover and the potential for methane oxidation in the cover. If the answer to the first question is yes, then the practical issue of whether solids sampling is feasible and cost-effective must be addressed. Specifically, how many samples are required to represent a large, heterogeneous waste mass? Representative sampling may be feasible at shallow, small landfills but regional landfills greater than 100 ha in area and 50 m in depth are not uncommon, particularly in the U.S. In the case of large landfills, representative sampling of the solids appears infeasible.

Gas production and water quality impacts of land disposal of cattle carcasses.

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Introduction

Cattle and calf production is a significant industry in the United States, with over 94 million animals in production in 2009 and a total value of over $82 billion dollars (USDA NASS, 2009). Although reported routine mortality rates for cattle production facilities are relatively low (approximately 1.3%) (Loneragan et al. 2001), surveys from the United States Department of Agriculture indicated, on average, over 2.2 million deaths per year occur in the U.S. at cattle and calf production facilities (USDA NASS, 2010). In addition to routine mortalities, mass mortality events may occur due to weather-related stress or outbreaks of infectious disease.

Carcass management methods include on-site burial, composting, landfilling, rendering, and incineration, and these management strategies have been applied to both routine and catastrophic animal mortalities. A lack of available incineration capacity in the United States, coupled with economic and technical limitations (Scudamore et al. 2002) make burial or composting attractive for cattle carcass disposal. Previously, outbreaks of infectious disease have required acute disposal of large numbers of carcasses. The bovine spongiform encephalopathy (BSE or 'mad cow') outbreak in the United Kingdom has generated 180,000 confirmed and over 2 million suspected BSE cases since 1985 (Smith and Bradley, 2003), while over 6.5 million animal mortalities were produced in 2001 during a foot and mouth disease (FMD) outbreak, with the majority being disposed of via landfills or land burial (Scudamore et al. 2002).

There have been a limited number of studies evaluating the environmental impacts of various animal mortality disposal options. Soil and groundwater contamination attributable to cattle carcass composting or poultry and cattle carcass burial were reported in previous studies. There have been very few studies evaluating air quality impacts or the presence of emerging contaminants in leachate generated from animal mortality burial. Because mortality management is typically conducted on-site with limited regulatory oversight, quantifying the potential environmental impacts of these activities is necessary to assess the risk to environmental health and to develop appropriate strategies to minimize emissions.

In this study, the air quality impacts of land burial of cattle carcasses were investigated using laboratory-scale anaerobic decomposition reactors. A field-scale burial experiment was performed to evaluate leachate quality, including the presence of both conventional and emerging contaminants.

Results and Discussion

Methane and CO_2 production rates observed in laboratory reactors is presented in Figure 1.

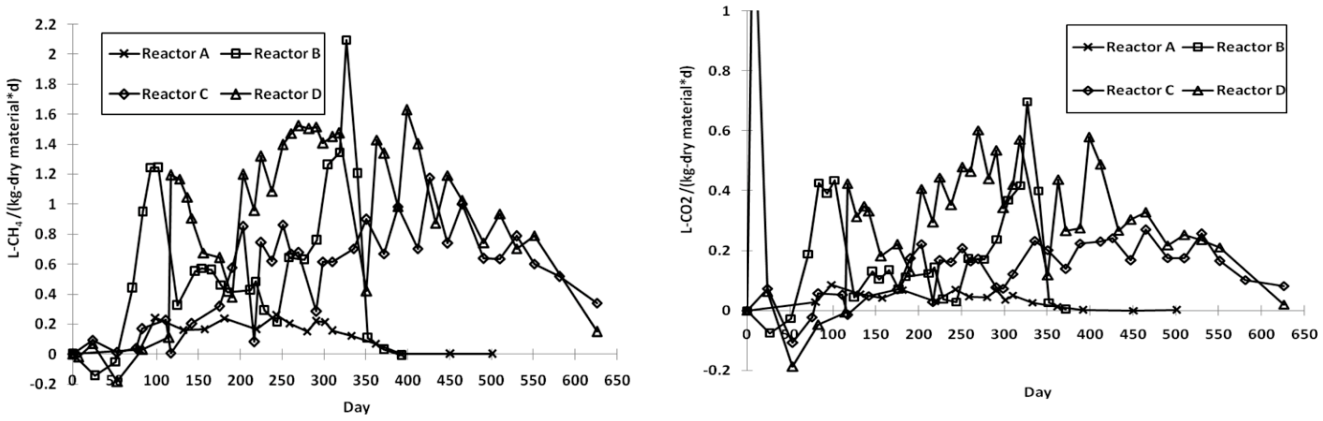

Figure 1. CH_4 (left panel) and CO_2 (right panel) production rates observed in laboratory reactors (Yuan et al. in press).

In field-scale burial experiments, compounds including 17α-estradiol, 17β-estradiol, estrone were detected at concentrations up to 2.7 µg/L. Other antimicrobials including monensin was also routinely detected in leachate.

Conclusions

Using the results of this laboratory study, an estimated annual production of carbon dioxide emissions contributed by land disposal of 2.2 million mortalities would be approximately 1.6 Tg CO_2 equivalents. In 2009, the US EPA reported emissions of 419.3 Tg CO_2 equivalents from the agricultural sector (EPA, 2011), therefore emissions from cattle carcass burial accounts for less than 1% of total emissions due to agricultural operations. Although the emissions from cattle carcass disposal is small relative to other sources of emissions from the agricultural sector, greenhouse gas emissions from animal carcass burial may be considerable if other types of animal production (poultry, swine) are also considered.

Data from field-scale burial experiments indicates that leachate can contain concentrations of steroid hormones and antimicrobials that are higher than that routinely detected in wastewater treatment effluent or runoff from fields amended with animal manure.

References

Loneragan, G.H., Dargatz, D.A., Morley, P.S., Smith, M.A., 2001. Trends in mortality ratios among cattle in US feedlots. J. Am. Vet.Med. Assoc. 219, 1122-1127.

Scudamore, J.M., Trevelyan, G.M., Tas, M.V., Varley, E.M., Hickman, G.A.W., 2002. Carcass disposal: lessons from Great Britain following the foot and mouth disease outbreaks of 2001. Rev. Sci. Tech. Off. Int. Epiz. 21, 775-787.

Smith, P.G., Bradley, R., 2003. Bovine spongiform encephalopathy (BSE) and its epidemiology. Brit. Med. Bull. 66, 185-198.

U.S. Department of Agriculture/ National Agricultural Statistics Service, 2009. Agricultural statistics 2009. U.S. Government Printing Office: Washington, DC, 2009.

U.S. Department of Agriguture/ National Agricultural Statistics Service Home Page. http://www.nass.usda.gov/Statistics_by_Subject/index.php (accessed Oct 25, 2010).

Yuan, Q., Saunders, S.E., Bartelt-Hunt, S.L. Methane and carbon dioxide production from land disposal of cattle carcasses. Waste Manage. In press.

The influence of precipitation on leachate parameters in new EU standards compliant landfill

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Introduction

Landfill leachate, which forms during different stages of stabilisation of municipal solid waste, may have a negative impact on the environment. The composition of the leachate depends on the type of landfilled waste, technical parameters of the landfill, the age of accumulated waste, processes of decomposition and meteorological conditions (Gasiūnas et al. 2002). Waste composition and landfill technical parameters are the factors which can be directly impacted by human activity, but meteorological conditions cannot be changed. The main aims of this research:

- identification of the main characteristics of leachate collected from a new post Landfill Directive landfill and comparison with leachate forming in other landfill, and
- determination, through seasonal qualitative investigations of leachate, of the dependence of leachate characteristics on precipitation.

Materials and methods

A regional landfill meeting EU standards was created in Kazokiškės, a subdistrict in Vilnius county, close to Vilnius, the capital city of Lithuania. . Vilnius is one of the largest cities of the country with a population of over 500 000. Kazokiškės landfill accepts waste from the entire county of Vilnius, which, on the average, amounts to 20 000 tonnes per month.

Over the period of 24 months more than 20 samples were taken and analysed, taking account of the season. The following parameters were measured: pH, COD, BOD, suspended solids, nitrite, nitrate, ammonia, chloride, sulphate, phosphate, carbonate and metals (Fe, Ni, Pb, Ca, Mg, Mn, Cr, Zn, Cu). The quantity of metals was determined by the method of atomic absorption spectrophotometry. As composition of leachate is impacted by several factors (waste composition, moisture content, etc.), the impact of the precipitation on concentration of different leachate components was analysed on the basis of statistical analysis using the Pearson correlation coefficient. The closer is the value of the coefficient (r) to 1 the stronger is the liner correlation (Ahlgren et. al., 2003).

Results and discussion

The concentration of different parameters in the leachate from the new landfill was compared to that of the old landfill; for example, BOD was on average three times greater (range 0.9 to 8.7). Average chloride, Zn, Ni, and Cu concentrations were lower in the new landfill, Pb and Cr concentrations were similar in the two landfills, but Mn concentration in the samples from the new landfill were much greater than in the old landfill.

Figures 1 and 2 show the influence of precipitation on leachate parameters in the old and new landfills. Waste in the new landfill is not covered and therefore is exposed to much more

precipitation which leaches out larger amounts of different substances. The inverse dependence of some metals shows that as precipitation increases, the concentrations of elements in leachate decreases. In the old landfill, Zn, Cr, and Cu also show a negative correlation, but Pb demonstrates a weak positive correlation, perhaps indicating that leaching of Pb is less influenced by precipitation, particularly in an aged, covered landfill.

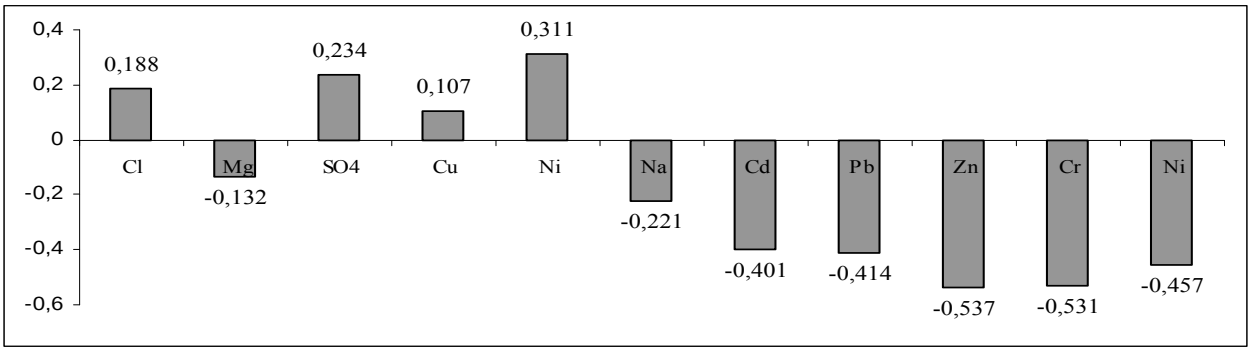

Fig. 1. Correlative dependence of dissociated elements on precipitation in new landfill

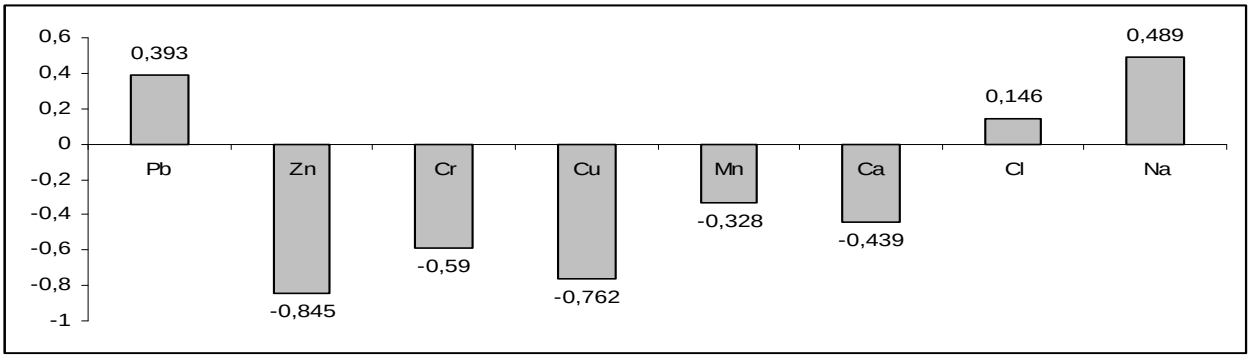

Fig. 2. Correlative dependence of dissociated elements on precipitation in old landfill

Conclusion

The most frequent information found in literature is about inter-correlation between leachate parameters without taking account of or without analysing meteorological conditions (Tatsi and Zouboulis, 2002). The investigations covered only a comparison of the quantities of leachate parameters showing that a difference in composition may be determined by a number of factors, such as waste composition, waste age, engineering solutions and meteorological conditions.

The performed analysis reveals the influence of meteorological conditions (correlation coefficients vary in a range from 0.311 to -0.537 in the new landfill and 0.489 to -0.845 in the old landfill) although this is applicable only to certain parameters. Most of the measured values have a very weak correlation, which shows a minor dependence on precipitation. The obtained results may allow predictions about possible changes in the quantities of leachate parameter concentrations under rainfall conditions. Such information can be helpful in applying new methods and innovative technologies which can minimise the influence of atmospheric phenomena on landfilled waste.

References

El-Fadel M., Bou-Zei, E., Chahina W., Alayli B., (2002), Temporal variation of leachate quality from pre – sorted and baled municipal solid waste with high organic and moisture content, *Waste Management*, **22**, 269 – 282.

Ettler V., Mihaljevic M., Matura M., (2008), Termporal variation of trace elements in waters polluted by municipal solid waste landfill leachate, *Bull Environ Contam Toxico*, **80**, 274-279.

Gasiūnas V., Strusevičius Z., Steponavičius A., (2002), Landfill leachate and environmental impact, [Sąvartynų filtrato tyrimai ir poveikis aplinkai], *Water Management Engineering*, **21**, 34-37.

Maximova A., Koumanov, B., (2006), Study on te content of chemicals in landfill leachate, Chemicals as Intentional and Accidental Global Environmental Threats*, Springe*, 345–356.

Per Ahlgren P., Jarneving B., Rousseau, R., (2003), Requirements for a cocitation similarity measure, with special reference to Pearson's correlation coefficient, *Journal of American Sosiety for Information Science ant Technology*, **54**, 550-560.

Tatsi A. A., Zouboulis A. I., (2002), A field investigation of quantity and quality of leachate from a municipal solid waste landfill in a Mediterranean climate (Thessaloniki, Greece), *Advances in Environmental Research,* **6**, 207 – 219.

Vasarevičius S., Čegariova J., Sližyte D., (2005), Ivertigation and evaluation of land fill leachate permeability in the soil, *Journal of Engineering and Landscape Management*, **13**, 108-115.

Deposition and Landfilling of Disaster Waste

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During 2011 Joint UNEP/OCHA Environment Unit (JEU) and Swedish Civil Contingencies Agency (MSB) published the "Disaster Waste Management Guidelines" Guidelines is a tool for all

Disaster waste is a well-recognized threat to health, safety and the environment, and can

also be a major impediment to post-disaster rescue operations. Experience shows that disaster waste is often managed in an ad hoc manner, however, and that substantial improvements can be made in future response efforts. In collaboration between by the Swedish Civil Contingencies Agency –

or MSB for short – and the Joint UNEP/OCHA Environment Unit have developed guidelines aiming to be a part of the improvements (http://ochanet.unocha.org/p/Documents/DWMG.pdf). They represent much of the best current knowledge and lessons learned on disaster waste management, and provide national authorities and international relief experts alike with sound and practical advice to help them manage disaster waste. They were developed following a request by governments at the international Advisory Group on Environment Emergencies, and are based on extensive consultations with national and international stakeholders.

A typical disaster is followed by an intensive period of waste removal in order to clear streets and roads for relief actions and to open up for reconstruction of buildings and infra structure. Beside that, an acceptable hygienic situation in IDP camps must be established as well as health care waste management of an acceptable standard among remaining hospitals and clinics, emergency field hospitals and other health care establishments. There is seldom functioning treatment plants available for the disaster waste, why landfilling and temporary storage is the main treatment process.

Where contingency planning is done, there shall be areas reserved and prepared for this purpose, but that is seldom the case and temporary disposal sites have to be improvised. The guidelines provide a structure for the establishment of temporary disposal sites as well as a guide for the closure.

Writing the guidelines we had to avoid the existence of a best solution standing in between and block the use of an appropriate one. Therefore there is a potential for development of the guidelines Annex IV Developing temporary disposal sites and Annex VII Guidelines for closure of temporary disposal sites. I would like to open for a discussion on that subject.

Behavior of Single Walled Carbon Nanotubes in Mature Landfill Leachate

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Introduction

Escalating production and subsequent incorporation of engineered nanomaterials in consumer products increases the likelihood of nanomaterials being discarded in landfills. Many commonly discarded products have been reported to contain nanomaterials. Plastic materials, for example, have been documented to contain titanium oxides (TiO_2), silver (nano-Ag), and carbon nanotubes (CNTs). Although direct measurement of particle disposal has not yet occurred, life cycle assessments suggest that over 50% of nanomaterials produced (worldwide yearly production of 350, 500, and 5000 tonnes/yr for nano-Ag, CNTs, and TiO_2, respectively, in 2007/2008 (Mueller and Nowack 2008)) will eventually reside in landfills. Despite the large mass of particles projected to be deposited in landfills, little research has been directed towards understanding the fate and transport of such materials in solid waste environments. It is likely nanomaterial mobility in landfills will be dictated by leachate composition. Recent studies have demonstrated that solution chemistry, such as the organic and electrolyte species and concentration, significantly influence nanomaterial aggregation and thus subsequent mobility (Li et al. 2008; Wang et al. 2008). It is important to understand how high electrolyte concentrations coupled with high concentrations of complex organic matter typically found in landfill leachate influence particle behavior.

The purpose of this work is to conduct an experimental study to understand how the combination of high organic concentration and ionic strength typical in older leachate influences single walled carbon nanotube (SWNT) behavior in waste environments. Laboratory-scale experiments were conducted to evaluate how organics and ionic strength typical of older leachates influence carbon nanotube surface charge and mobility through representative solid waste environments. Humic acid was used to simulate the complex organics found in old leachate and sodium chloride (NaCl) was used as the electrolyte species. Nanotube electrophoretic mobility (EPM) was measured in batch experiments over a range of organic (200 – 800 mg/L) and salt concentrations (50 – 400 mM via NaCl). EPMs were converted to zeta potential using the Helmholtz-Smoluchowski relationship (note this relationship assumes the materials are spherical). To assess potential particle mobility, a suspension of carbon nanotubes, humic acid, and NaCl was passed through a saturated 1-D column filled with representative municipal solid waste (MSW).

Results

Results from EPM measurements conducted on the SWNTs at different solution chemistries suggest that electrolyte concentration is a dominant factor influencing changes in surface charge. An increase in the salt concentration results in a decrease in absolute value of the ZP for all humic acid concentrations evaluated. The observed decrease is likely a result of the compression of the diffusive double layer, consistent with data reported in the literature for other nanomaterials, including carbon nanotubes and fullerene nanomaterials.

The effluent SWNT breakthrough curve for the column experiment is shown in Figure 1. Breakthrough of the SWNT dispersion occurs slightly before that of the conservative, nonreactive tracer. This phenomenon has been observed in other SWNT transport experiments and likely results from a size exclusion effect (Jaisi and Elimelech 2009). Approximately 60% of the SWNTs were recovered in the effluent.

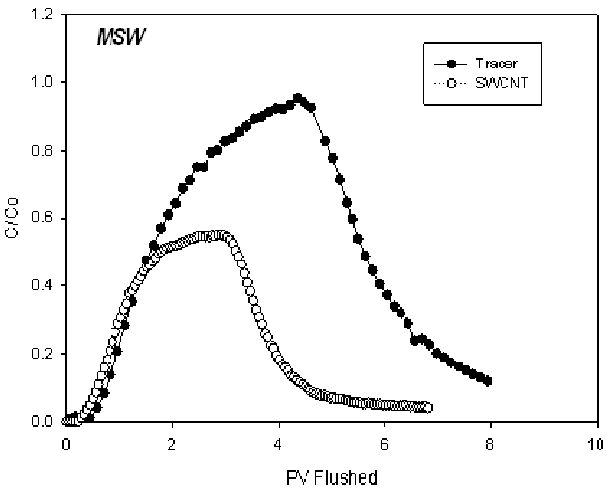

Figure 1. SWNT breakthrough curve.

Conclusion

Results from EPM measurements conducted on the SWCNTs at different solution chemistries suggest that the electrolyte concentration is a dominant factor influencing changes in surface charge. Even at high ionic strengths, the humic acid creates a steric barrier to material aggregation/agglomeration. This steric barrier likely aids the transport of the materials through the waste. Results from the column experiment indicate the carbon nanotubes may be mobile through solid waste, suggesting particle placement within landfills needs to be examined more closely.

References

Jaisi, D., and M. Elimelech (2009). "Single-walled carbon nanotubes exhibit limited transport in soil columns." Environmental Science & Technology **43** (24) 9161–9166.

Li, Y., Y. Wang, K. D. Pennell and L. M. Abriola (2008). "Investigation of the Transport and Deposition of Fullerene (C60) Nanoparticles in Quartz Sands under Varying Flow Conditions." Environ. Sci. Technol.: ASAP Article, 10.1021/es801305y.

Mueller, N. C. and B. Nowack (2008). "Exposure modeling of engineered nanoparticles in the environment." Environmental Science & Technology **42**(12): 4447-4453.

Wang, Y. G., Y. S. Li, J. D. Fortner, J. B. Hughes, L. M. Abriola and K. D. Pennell (2008). "Transport and retention of nanoscale C-60 aggregates in water-saturated porous media." *Environ. Sci. Technol.* **42**(10): 3588-3594.

Establishing excreta discharging facility at Truitier dumpsite in Port au Prince 2010. Conditions and constrains.

Sofia Billvik, M.Sc.

One month after the earthquake 2010 among the 2.5 million people living in Port au Prince (PaP) about 50 % had access to latrines, 10 % to water toilets with septic tanks and approximately 40 % of the population has no access to any form of facilities for defecation. In these areas open defecation and "flying toilets" (defecation done in plastic bags, food packages, etc) was the common practice.
There were a number of private actors working with desludging of latrines in PaP. However, there was no official disposal site for the sludge trucks to empty the excreta. The trucks were emptied at open ground, in wetlands and open watercourses causing health risk and environmental damages.
After the earthquake a large number of desludgable latrines and portable toilets were installed in the IDP camps and a fleet of about 40 desludging trucks was assembled through donations. The need of establishing organized sites for excreta disposal and treatment was higher than ever.
DINEPA (Direction Nationale de l'Eau Potable et Assainissement) was given the responsibility for sanitation in Haiti but had not, at the time, built up any organization for working with sanitation.
The MSB Haiti Disaster Waste and Excreta Management Team seconded to UNICEF supported DINEPA with technical expertise in designing the facility for excreta disposal and in the process of identifying new sites for long term excreta disposal.

The first task was to carry out a baseline assessment at Truitier landfill in order to find out if a facility for safe excreta disposal can be established temporary within the landfill. Considered within the baseline assessment are historical and recent deposited material, geology and geotechnical preconditions, hydro geological aspects and limitations, areal capacity and space limitations, logistical weaknesses, working conditions for workers at the site, etc. This will be used as background information for looking into the possibility of designing a sludge management system at Truitier.
The second task was to do the technical design and identify location for the sludge disposal facility within Truitier landfill. Design includes dimensioning, infrastructure within the site, emptying procedure, logistics of sludge trucks within the dumpsite, fencing.
The Truiter excreta disposal site was planned as a temporary facility to be used during 6 months while identifying and establish a number of new sludge treatment facilities in the Grand Port au Prince region.

Conditions and constrains will be discussed in the paper.

Fate of Nanoparticles in Municipal Solid Waste Landfills

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Introduction

Over the past decade engineered particles with nanoscale dimensions have been key to advancement in drug delivery and pharmaceuticals, cosmetics, foods, environmental remediation, nanotechnology, biomaterials, and energy production (Bhatt and Tripathi, 2011; Musee, 2011; Colvin 2003; Dreher 2004; Nyberg, Turco et al. 2008; Bittnar, Bartos et al. 2009; Sattler, 2011). Nanoparticles (NPs) are defined as having three dimensions between 1 and 100 nm (Standards 2007; Linkov and Steevens, 2009; Sattler, 2011), while nanomaterials (NMs) have at least one dimension between 1 nm and 100 nm (Roco, 2003). In 2010 it was reported that 1,317 consumer products that contained nanomaterials were available on the market (Nanotech-Project, 2011). Of these products, just over 55% were health and fitness related including electronic components, cosmetics, antibacterial agents, polishing and binding agents, solar cells and UV-absorbers in sunscreen lotion, among many others (Linkov and Steevens, 2009).

Given the increase in NM use and the large fraction of waste placed in landfills worldwide, the probability of these products reaching municipal solid waste (MSW) landfills at the end of their useful life is high. Since nanotechnology use is still in its early stages, there are currently no regulations pertaining to the disposal of NMs and their fate in MSW landfills is still unknown. Therefore, there is a need to study the fate and transport of NMs within waste environments and determine whether these products can potentially affect human health and the environment.

Methodology

This research seeks to understand the fate of coated zinc oxide (ZnO), titanium dioxide (TiO_2), and nano silver (Ag) within waste environments by focusing on the interactions between the NMs and landfill leachate components. Studying the fate of these NMs within waste environments was accomplished by conducting four main tasks: collection and characterization of landfill leachate, evaluating the effects of NMs on landfill biological processes, size fractionation of leachate exposed to coated ZnO, TiO_2, and Ag, and the chemical speciation of Zn, Ag, and Ti modeled using Visual MINTEQ.

Middle-aged (BOD/COD: 0.34-0.54) and mature (BOD/COD: 0.07-0.11) leachate samples were collected and characterized from MSW landfills in Florida, USA. All leachate samples were analyzed for pH, biochemical and chemical oxidation demand, ammonia-nitrogen, conductivity, and heavy metals. Laboratory methods were conducted following the procedures outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 2005).

Leachate was exposed to varying concentrations of coated ZnO, TiO_2, and Ag, individually, to observe any concentration-dependent effects on biological processes, solids aggregation, and dissociation. ZnO, TiO_2, and Ag were added to reactors containing 4 L of leachate for a final concentration of 100 µg/L, 1.0 mg/L, and 100 mg/L, while continuously stirred at room

temperature (24ºC) for 60-days. Results were compared to a control reactor treated in the identical manner without the addition of NMs.

The effect of NMs and their byproducts on both aerobic and anaerobic biological landfill processes was evaluated by performing BOD_5 and Standard Test Method for Determining Anaerobic Biodegradation Potential of Organic Chemicals under Methanogenic Conditions (ASTM E2170), commonly referred to as BMP, tests on leachate. Aliquots of approximately 40 mL were removed biweekly from each reactor for BMP and BOD_5 tests. Results were compared to tests on a leachate control which was treated in identical manner without the addition of NMs. Any decreases in BOD_5 and BMP relative to control samples suggest inhibitory effects.

Size fractionation was used to determine the effect of time and stirring on solids aggregation and to quantify the mass change in each fraction. Any changes in weight among the fractions were attributed to the interactions of leachate constituents and constant stirring. Aliquots of approximately 150 ml were taken from each reactor at 7, 30, and 60 days and fractionated, following which the total solids content was measured. The aliquots were filtered in series using a 1,500 nm (glass), 450 nm (nylon), 200 nm (nylon), and 1.0 nm filter (1,000 NWML regenerated cellulose). The concentration of Zn, Ag, and Ti in each size fractions was quantified using Inductively Coupled Plasma-Optical Emission Spectrum (ICP-OES). The interaction of leachate constituents with coated ZnO, TiO_2, and Ag could promote conditions where Zn, Ag, and Ti could dissociate from ZnO, TiO_2, and Ag nanoparticles. The background concentration of Zn, Ag, and Ti in leachate, as received, were also determined before the addition of coated ZnO, TiO_2, and Ag.

The chemical speciation of Zn, Ag, and Ti in leachate is instrumental to understanding the toxicity and bioavailability of these metals in the environment. Visual MINTEQ, an equilibrium speciation model, was used to estimate the chemical species and equilibrium mass distribution of dissolved species found in leachate. The concentration of Zn, Ag, and Ti in the fraction less than 1.0 nm, determined from metals analysis, was used as input to the chemical speciation model.

Overall understanding of the fate of NMs and behavior in leachate will directly benefit landfill operations and leachate management. Research data will also aide regulatory agencies in addressing potential nanotoxicological issues and develop a knowledge base to enhance the understanding of the fate and transport of NMs in landfills.

Preliminary Results

Results to date suggest that ZnO and TiO_2 did not have an inhibitory effect on anaerobic or aerobic processes when exposed to mature or middle aged leachate. BOD_5 results after exposure to ZnO and TiO_2 were analyzed and the rate of disappearance of biodegradable matter did not vary significantly between the control and reactors exposed to NMs. Additionally the BMP test did not vary substantially over a 90-day period. This is also supported by HRTEM images that show the crystalline structure of both ZnO and TiO_2 still intact which conclude that both NM coatings were stable during the 60-day exposure time. Dispersion of hydrophobic NMs was observed, presumably due to interaction between metal NMs and high concentrations of humic acid in leachates.

References

APHA (2005). Standard Methods for the Examination of Water & Wastewater. Washington, DC, American Public Health Association.
Bhatt, I., Tripathi, B.N., Interaction of engineered nanoparticles with various components of the environment and possible strategies for their risk assessment, Chemosphere, Volume 82, Issue 3, January 2011, Pages 308-317, ISSN 0045-6535, 10.1016/j.chemosphere.2010.10.011.
Bittnar, Z., P. J. M. Bartos, et al. (2009). Potential Environmental and Human Health Impacts of Nanomaterials Used in the Construction Industry. Nanotechnology in Construction 3, Springer Berlin Heidelberg: 1-14.
British Standards. (2007). British Standards (BSI), 2007. Terminology for Nanomaterials, Public Available Specification No. 1362007. London: British Standards Institution: 16.
Colvin, V. L. (2003). "The potential environmental impact of engineered nanomaterials." Nat Biotech 21(10): 1166-1170
Dreher, K. L. (2004). "Health and Environmental Impact of Nanotechnology: Toxicological Assessment of Manufactured Nanoparticles." Toxicological Sciences 77(1): 3-5
Farré, M., K. Gajda-Schrantz, et al. (2009). "Ecotoxicity and analysis of nanomaterials in the aquatic environment." Analytical and Bioanalytical Chemistry 393(1): 81-95.
Linkov, I., and Jeffery Steevens (2009). Nanomaterials: Risks and Benefits, Springer.
Musee, N., (2011), "Nanowastes and the environment: Potential new waste management paradigm," Environment International, 37(1): 112-128.
Nanotech-Project, (2011). Analysis, Project on Emerging Nanotechnologies, Woodrow Wilson International Center for Scholars, March. 2011.
<http://www.nanotechproject.org/inventories/consumer/analysis_draft/>
Nyberg, L., R. F. Turco, et al. (2008). "Assessing the Impact of Nanomaterials on Anaerobic Microbial Communities." Environmental Science & Technology 42(6): 1938-1943.
Roco, M.C., (2003). Nanotechnology: convergence with modern biology and medicine. Current Opinion in Biotechnology 14, 337–346.
Sattler, Klaus D. *Handbook of Nanophysics.* Boca Raton: Taylor & Francis, 2011. Print.

CHARACTERISATION OF WASTE MATERIAL MIXTURES FOR LANDFILL TOP COVER APPLICATION

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Introduction

The aim of this study is to investigate the behaviour of various waste mixtures in a landfill top cover construction. Waste materials like fly ash, sewage sludge, peat waste and gypsum were characterised and leaching behaviour of the mixtures was evaluated. The changes in distribution of trace elements in various fractions with an aid of the modified sequential extraction procedure were also investigated. Mixing various materials and using the mixtures as secondary construction materials, might lead to changes of materials behaviour that can cause leaching or immobilisation of hazardous compounds as well as the enhancement or deterioration of the mechanical stability of the construction. Therefore, it is important to understand the leaching behaviour of waste materials and their mixtures.

Material and methods

Two different ashes, sewage sludge, peat waste and gypsum waste samples were collected in the different places in Sweden. A standard one-stage compliance batch leaching test (SS-EN 12457-4) at a liquid to solid ratio of 10 l/kg (L/S 10) was applied to estimate leachable concentrations of chemicals in materials. Modified sequential extraction was performed and following fractions were obtained: *Fraction (I): Exchangeable metals. Fraction (II): Bound to Carbonates. Fraction (III): bound to Fe(III) oxyhydroxides: Fraction (IV): Bound to Fe-Mn oxides. Fraction (V): Bound to Organic matter and secondary sulphides. Fraction (VI): Residual fraction*.

Results

The following mixtures were analysed and presented in this paper: Mixture 1: MSWI fly ash 23.3%, Peat residues 53,7% and sewage sludge 23.0%; Mixture 2: MSWI fly ash 37.5%, gypsum waste 26.4% and sewage sludge 36.1%. For example mixing materials resulted in decreased leaching of Pb (Figure 1) and distribution of Pb in various fractions depends on the materials mixed (Figure 2). Further evaluation of results is on-going and conclusions will be drawn.

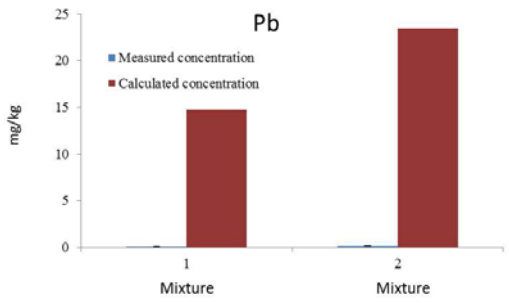

Figure 1. Measured and calculated leaching of Pb

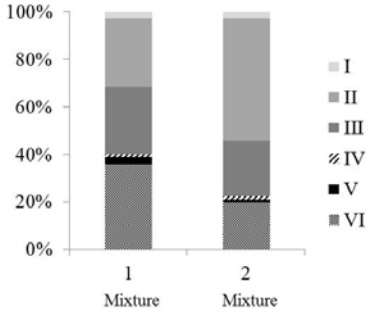

Figure 2. Distribution of Pb in various fractions

Integrated modeling and up-scaling of landfill processes and heterogeneity using stochastic approach

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Introduction

Municipal solid waste landfills are a very complex and heterogeneous systems. The waste in a landfill body is a heterogeneous mixture of a wide range of materials containing high levels of organic matter, high amounts of salts and a wide range of different organic and inorganic substances, such as heavy metals and organic solvents.
A range of processes of different nature occur within landfills. Bio-geochemical processes in a landfill body lead to the development of landfill gases, a mixture of predominantly methane (CH4) and carbon dioxide (CO2) and smaller amounts of trace gases. Water flow through the landfill is the main driving force for biodegradation and leachate generation. This induces loss of matter via leachate and gas generation hence a new pores become available for water flow and settlements take place.
An estimation of the remaining emission/contaminating potential of the landfill under different landfill management scenarios is the main goal of our research. This can give valuable information on when it is "safe" to release the landfill from active aftercare and what actions need to be done in order to reach this threshold earliest.

Methodology

The main idea is to apply stochastic approach and transfer functions to model water/leachate residence time in the landfill body. This time may then be used together with the amounts of water infiltration through the landfill as an input to the biodegradation model coupled with the settlement model. Biodegradation and settlement will in turn affect water/leachate flow through the waste.
The objective of research requires integrating models of water flow; bio-geochemical reactions; reactive transport and settlement on the landfill body (see Fig. 1). Understanding of processes at a small scales occurring within a landfill (biodegradation of organic matter) is needed in order to model large scale processes (landfill gas and leachate generation as well as settlements). However high heterogeneity cannot be modeled precisely using deterministic approaches. Therefore stochastic approach to handling heterogeneity is applied.
We assume that a landfill is divided into a large number of columns which may be considered as flow paths. Each column has its own random hydraulic properties and chemical composition, although is considered to be a homogeneous itself.
All water coming into the landfill body is split into small amounts each with a random travel time defined by a travel function. A flow model is being developed which will allow defining the probability distribution for "travel times" of water through the landfill. This model considers the presence of impermeable layers (such as plastic sheets) which induces large rates of preferential flow. The model will be calibrated and validated using LeachXS database and results obtained from measurements at a column scale and full scale landfills.

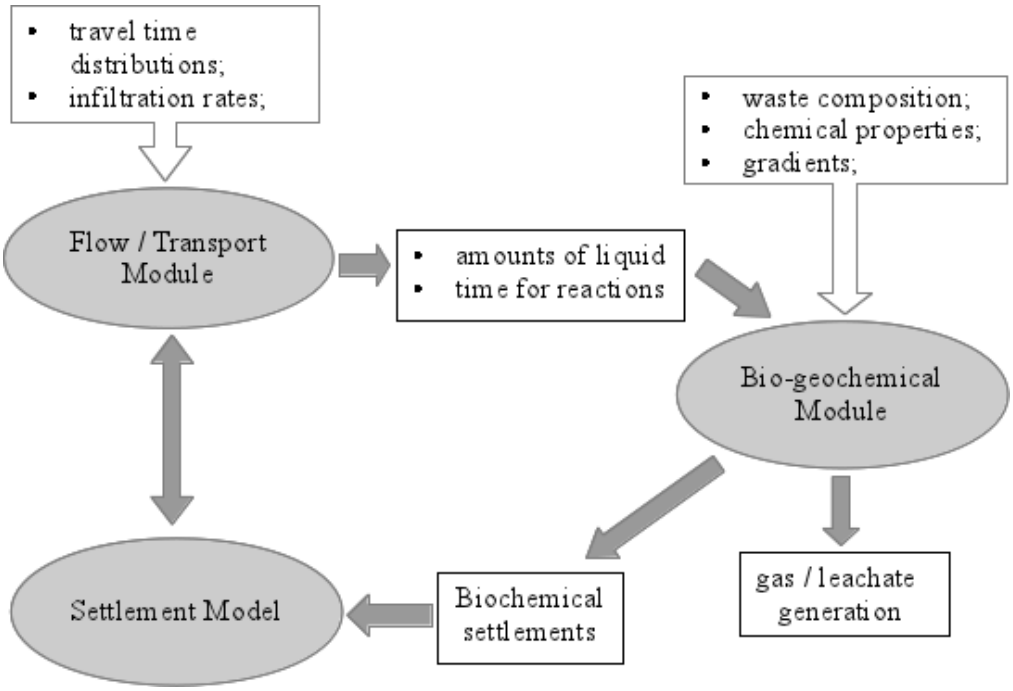

Figure 1. Multiphysics modelling framework for landfills

We assume that during its lifetime water will react with the matter within the landfill body. Each water particle enters one of the available pathways with individual composition and chemical properties. These properties and waste composition are also considered to be random however homogeneous for each column/pathway. A chemical equilibrium model designed within ORCHESTRA framework and biodegradation model use previously modeled amounts of water input into the landfill and its reaction time. Other chemical parameters and their gradients/distributions will be obtained from geophysical measurements. The bio-geochemical module computes amounts of landfill gases and leachate produced as well as the amounts of compounds remaining in the landfill, thus allowing estimates of the remaining emission potential of the landfill.
Bio-geochemical processes occurring within the landfill will cause settlements which will affect water flow. Therefore the usage of the settlement model may serve an additional way to validate results. Another important effect of settlements is their effect on the water flow. This however is very difficult if not impossible to quantify.
The results modeled for each water particle/pathway are summed up will represent overall trends for the entire landfill.

Conclusions

Integrating water flow/transport, bio-geochemical and settlement models as well as geophysical measurements will provide us with the valuable information on what gas/leachate emission potential remains in the landfill while application of the stochastic approach appears to be a feasible way to handle high heterogeneity of solid waste landfills. This framework allows quantitative assessment of when the emission potential from the landfill is low enough and hence it is safe to stop aftercare.

Capillary barriers and uniform biogas distribution in biocovers

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The concept of incorporating a gas distribution layer (GDL) below the methane oxidizing layer (MOL) was proposed in the late 1990's and was eventually implemented in most large-scale field projects. The idea was to foster uniform distribution of biogas; or – better - the avoidance of concentrated gas flow (hotspots), which might lead to a less performing biocover. However good the idea of installing a GDL, it has been seriously challenged recently due to field evidence of hotspots appearances upslope at several field sites where the biocover was equipped with GDL. This challenge has taken the form of a session in this symposium, which shall deal with the eventual myth of uniform gas distribution. In this paper, the Authors address this issue from the perspective of capillary barrier design principles and monitoring data from field plots. It is shown that without proper design, the capillary barrier formed between the GDL and the MOL simply directs biogas flow to regions with greater air-field porosity. Documentation of such occurrences are presented and discussed. It is anticipated that the material presented will be of practical use to biocover developers and that the issue of capillary barrier formation in biocovers will receive greater attention of researchers in the future.

Drawbacks in the Evaluation of Methane Oxidation Efficiencies Using Carbon Stable Isotopes

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As part as a broad study about biocovers design for landfills, oxidation efficiencies were evaluated via the mass balance method using CH_4 loading and outflux data from two field experimental plots (Cabral et al., 2010a; Roncato and Cabral, 2012). In addition, oxidation efficiencies were evaluated for the same plots using the carbon stable isotope method (Cabral et al. 2010b).

The goal of this particular paper is to discuss some important limitations related to the carbon stable isotope method. In this study, a straightforward comparison between mass balance and stable isotope results could not be made because samples taken inside flux chambers (thus at the surface) were quite diluted (volumetric concentrations of CH_4 usually below 0.05%) and with the methodology adopted and the equipment available, it was not possible to obtain reliable isotopic compositions ($\delta^{13}C$ values) for them. Comparisons between oxidation efficiencies were thus made using isotopic data for samples taken at a depth of 0.10 m.

35-ml gas samples were collected and transferred to evacuated glass vials, which were stored at 4°C until isotopic analyses were performed. Calculation of the isotopic fractionation factor for bacterial oxidation, α_{ox}, was performed by means of an incubation test at 20°C. For the purpose of calculating oxidation efficiencies, α_{ox} had to be corrected to the actual soil temperature. The isotopic fractionation factor associated with gas transport in the biocover (α_{trans}) was assumed to be equal to 1.0, thereby considering advection as the main gas transport within the soil. This is one first source of uncertainty related to the calculation of oxidation efficiencies performed in this study. Indeed, calculations ignoring diffusive flux within biocovers and biofilters may lead to an important underestimation of oxidation efficiencies calculated using the carbon stable isotopes method.

The results show that efficiencies are usually much lower when adopting the stable isotope method. The discrepancies may result from several causes. An important one is possibly related to the fact that the most active zone for CH_4 oxidation in biocovers is located very near the surface, where O_2 availability is greater. As a consequence, efficiencies calculated using stable isotope data from samples taken at 0.10 m (thus not at the surface) can – at most – provide an idea of the total efficiency of the system.

Another important cause for the discrepancies between mass balance and stable isotopes results are associated with potential errors in the estimation of α_{ox}. Our results show that a mere 0.5% change in the α_{ox} value resulted in an important dispersion in oxidation efficiencies. For example, the oxidation efficiency of 68.9% obtained for one sampling date might well have been 55.2% or 92.9%, which is quite a wide dispersion. In addition, for several samples, the higher end of oxidation efficiency values were > 100% following reduction of only 0.5% to the temperature-adjusted α_{ox}. In real life, i.e. in most cases involving laboratory and field test results, standard deviations greater than 0.5% are expected. As a consequence, greater dispersions than those mentioned above may be found.

A third and important cause for the discrepancies in efficiencies result from the possibility that $\alpha_{trans} > 1.0$. Following the same procedure adopted by Chanton et al. (2011) to recalculate α_{trans} values, the efficiencies obtained in the present study were recalculated and a 1.6- to 3.3-fold increase was obtained.

References

Cabral, A. R.; Moreira, J. F.; Jugnia, L. B. (2010a). Biocover Performance of Landfill Methane Oxidation: Experimental Results. *J. Environ. Eng. (ASCE)* 138 (8): 785-793.

Cabral, A. R.; Capanema, M. A.; Gebert, J.; Moreira, J. F.; Jugnia, L. B. (2010b). Quantifying microbial methane oxidation efficiencies in two experimental landfill biocovers using stable isotopes. *Water, Air, and Soil Pollution* 209: 157-172.

Chanton, J.; Abichou, T.; Langford, C.; Hater, G.; Green, R.; Goldsmith, D.; Swan, N. (2011). Landfill methane oxidation across climate types in the U.S. *Environmental Science and Technology* 45 (1): 313-319.

Roncato, C.D.L., and Cabral, A.R. (2012; *in press*). Evaluation of methane oxidation efficiency of two biocovers: field and laboratory results. *J. Env. Eng. (ASCE)* DOI: 10.1061/(ASCE)EE.1943-7870.0000475\.

Methane Oxidation at a Landfill Scale.

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Because samples of methane for isotopic analysis from chambers and soils do not factor in methane released from landfill cover secondary porosity nor piping and gas well infrastructure, we conducted isotope measurements on whole landfill emitted methane. This was done by two approaches, (1) by collecting methane upwind (background) and downwind of the landfill, and (2) by collecting methane from representative areas of the landfill with different covers just prior to dawn on still windless nights when atmospheric inversions cause the air over the landfill to stagnate and accumulate methane. Sampling upwind/downwind was guided and performed in accompaniment with a Picarro cavity ring down laser system which can give real time measurements of atmospheric methane. In this manner we were able to capture the isotopic signature of all of the methane emitted from the landfill. Percent oxidation was determined from the isotope data. Upwind measurements serve as background for both the downwind and nighttime samples. Anoxic methane was determined by sampling gas wells across the landfill. Samples were collected and results are shown from 11 different landfill over 17 sampling occasions.

Table Caption. Means and variability for each landfill sampling date. Means are calculated as the mean between fraction oxidized with and without allowing for diffusion. These values should be multiplied by 100 to obtain % oxidation. Variability is range of the measurements divided by two.

	avg. frac oxidized	variability
California October 20 -23, 2009	0.22	0.10
Milwaukee, July 2010	0.52	0.24
Midwest 1 November 9, 2010	0.40	0.19
Kentucky 1 July 30 2010	0.30	0.14
Kentucky 1 Aug 10, 2010	0.21	0.10
Kentucky 1 December 2, 2010	0.14	0.07
Georgia Sept 14, 2010	0.18	0.08
Midwest 2, November 3, 2010	0.41	0.19
Midwest 3, November 4, 2010	0.13	0.06
Midwest 4 September 23, 2010	0.18	0.08
Midwest 4 October 19, 2010	0.30	0.14
Midwest 5 August 25, 2010	0.27	0.11
Midwest 5 Sept 22, 2010	0.14	0.06
Midwest 6 Nov 20, 2010	0.00	0.00
Midwest 6 Aug 11, 2010	0.10	0.05
Midwest 7 July 23, 2010	0.32	0.15
Midwest 7 Nov 12, 2010	0.03	0.02
Overall mean and standard error	**0.23**	**0.03**

The results indicate that the mean fraction oxidized across all the landfills was 23±3%. These results are over twice the accepted default value for methane oxidation. However as will be discussed below, they are still considerable overestimates as will be discussed in our presentation.

A Decision-Making Framework for Long-Term Leachate Management

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This presentation provides an example of a decision-making framework developed to assess modification or termination of leachate management at closed MSW landfills based on evaluation of leachate data. The framework was developed as the Leachate Module within the Evaluation of Post-Closure Care (EPCC) Methodology, which was initially released in 2006 and has been continually updated and improved through ongoing research programs in the USA and Europe since that time. It is hoped that review of an existing methodology and the challenges faced in its development and application (rather than starting with a "blank slate") will facilitate and focus the discussion on what is relevant or necessary for a decision-making framework, and help session participants identify aspects that may need further attention and development. The primary approach of the Leachate Module is to make performance-based decisions regarding the potential for leachate emissions to pose an environmental threat. This approach is based on the concept of 'functional stability,' which is a term used to define a closed landfill that does not present an unacceptable threat in the absence of aftercare, at which point aftercare can be ended although some *de minimus* level of control would typically still be provided to protect against disturbance of buffer zones and/or passive barriers, mainly the cover.

The Leachate Module comprises three main steps, the first of which is a statistical demonstration that historical concentration trends for selected indicator parameter(s) are decreasing or steady, such that there is a high level of confidence that leachate quality will not worsen in the future under given landfill conditions. Second, a strategy for long-term leachate management and discharge to receiving waters (if necessary) is developed. Third, using a step-down approach based on decreasing distance between the landfill source and the potential environmental point of exposure, the acceptability of the proposed strategy is determined based on evaluation of potential leachate impacts. This is done through statistical comparison of leachate quality to applicable groundwater and surface water standards for regulated analytes. Passing the evaluation in full should allow implementation of the new leachate management strategy, followed by an appropriate confirmation monitoring program to verify that leachate data trends and landfill behavior are as predicted. A key position taken during framework development was that solids sampling is not useful and long-term leachate management can be better assessed and defined based on emissions data coupled with a long-term strategy for the site.

Determination of Sources of Organic Matter in Landfill by Analysis of CuO Oxidation Products of Lignin

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Abstract

The study of lignin degradation is essential in understanding carbon turnover in anaerobic environments such as landfills. Lignocellulosic materials constitute the major fraction of the organic matter in municipal solid waste (MSW) and are made up of three basic biopolymers: cellulose, hemicellulose and lignin (50%, 12% and 15%, respectively). While both cellulose and hemicellulose are readily converted to methane and carbon dioxide in landfills, lignin is generally considered preserved (Colberg, 1988) and constitutes a significant fraction of the biogenic carbon that is stored in landfills. The ratio of cellulose plus hemicellulose to lignin (C+H)/L is used as a monitoring tool to evaluate the rate and extent decomposition in landfills (Barlaz, 2006; Wang, Byrd, & Barlaz, 1994). The use of (C+H)/L has the benefit of eliminating the effect of dilution due to cover soil. However, some synthetic organic materials such as plastics and rubber have been shown to interfere with the traditional Klason lignin analytical method, resulting in measures of lignin concentration that are artificially high. Thus, a more robust analytical method for lignin is required. CuO oxidation involves breaking down complex lignin molecules by alkaline digestion and then analyzing its phenolic monomer building blocks by HPLC or GC/MS.

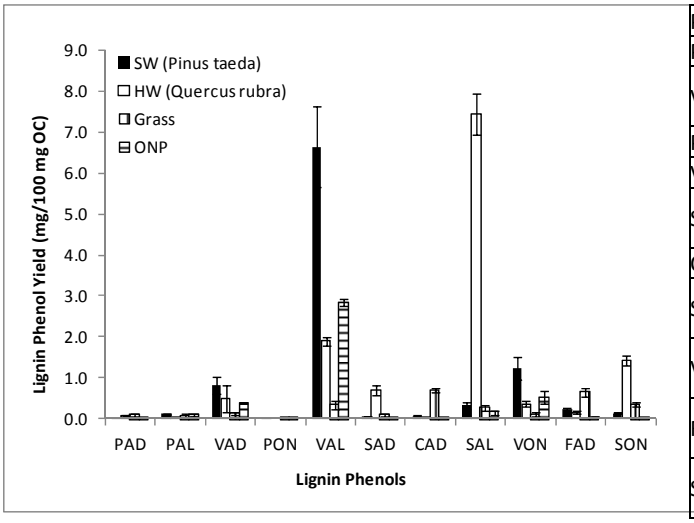
The overall objective of this research is to apply the CuO oxidation method to evaluate the behaviour of lignin under anaerobic conditions. In the CuO oxidation method, phenolic monomer building blocks that are unique to lignin are measured. Initial work was conducted to implement the CuO oxidation method for lignin analysis. Analysis based on 10 hardwood and 10 softwood samples showed that the traditional Klason lignin (KL) can be estimated using equation 1. This equation was derived using multiple linear regressions with coefficient of multiple determination (R) equals 0.91.

$$KL = 147(C) - 13(S)(C) \qquad (1)$$

Where C is the sum of cinnamyl phenols (CAD and FAD) and S is the sum of syringyl phenols (SAL, SON and SAD) in mg lignin phenols 100 mg organic C^{-1}. Abbreviations for individual lignin phenol are defined in Figure 1.

The second objective of this study was to determine the effect of temperature on lignin degradation under mesophilic (37 °C) and thermophilic (55 °C) conditions. Shifts in CuO

oxidation products will be used as indicator of lignin modification during anaerobic decomposition. Batch decomposition experiments are being conducted on different representative types of lignocellulosic materials in MSW (hardwood, softwood, grass, newspaper and office paper). The CuO oxide oxidation products of these materials are given in Figure 1. These batch reactors will be operated until no measureable methane production is observed at which time the solids will be analysed by CuO oxidation to evaluate whether there is evidence of lignin transformation under anaerobic conditions.



PAD	4-hydroxybenzoic acid
PAL	4-hydroxybenzaldehyde
VAD	4-hydroxy-3-methoxy-benzoic acid (vanillic acid)
PON	4-hydroxyacetophenone
VAL	4-hydroxy-3-methoxy-benzaldehyde (vanillin)
SAD	3,5-dimethoxy-4-hydroxy-benzoic acid (syringic acid)
CAD	4-hydroxy-cinnamic acid (p-coumaric acid)
SAL	3,5-dimethoxy-4-hydroxy-benzaldehyde (syringaldehyde)
VON	4-hydroxy-3-methoxy-acetophenone (acetovanillone)
FAD	4-hydroxy-3-methoxy cinnamic acid (ferulic acid)
SON	3,5-dimethoxy-4-hydroxy-acetophenone (acetosyringone)

Figure 1. CuO oxidation products of representative lignocellulosic materials in landfills.

References

Barlaz, M. A. (2006). Forest products decomposition in municipal solid waste landfills. *Waste Management, 26*(4), 321-333.

Colberg, P. (1988). Anaerobic microbial degradation of cellulose, lignin, oligolignols and monoaromatic lignin derivatives. In A. J. B. Zehnder (Ed.), *Biology of anaerobic microorganisms* (pp. 886-889). New York: Wiley-Interscience.

Wang, Y. S., Byrd, C. S., & Barlaz, M. (1994). Anaerobic biodegradability of cellulose and hemicellulose in excavated refuse samples using a biochemical methane potential assay. *Journal of Industrial Microbiology, 13*(3), 147-153.

Hydrogen Sulfide Production from Different Sulfur-Containing Materials under Anaerobic Conditions

Florentino B. De la Cruz* and Morton A. Barlaz.

Over the past several years, fly ash from coal-fired power plants has been buried in landfills that also receive municipal solid waste (MSW). In the case of power plants that remove SOx from the stack gas, the ash contains sulfate ($SO4^{-2}$), either as a calcium salt or as sodium salt or mineral. When this material is exposed to either MSW or MSW leachate under anaerobic conditions in a biologically active system, hydrogen sulfide (H_2S) is produced as described by equation 1:

$$\text{Organic matter (waste)} + SO_4^{2-} + H_2O \rightarrow H_2S + CO_2 \qquad (2)$$

Concurrent with the production of H_2S, the MSW will also decompose to methane and carbon dioxide as described by equation 2.

$$\text{Organic matter (waste)} + H_2O \rightarrow CH_4 + CO_2 \qquad (3)$$

In addition to fly ash, other sources of sulfate in landfills may include pure gypsum, construction and demolition (C&D) waste, and fines from MRFs that process C&D waste. The presence of elevated concentrations of H_2S in landfill gas is problematic for several reasons as it is toxic, results in a strong objectionable odor and is corrosive to landfill gas treatment equipment. Given that many landfills buried sulfate-rich wastes with MSW, estimates of the rate at which the sulfate will be depleted are needed to plan appropriate strategies for the management of landfill gas that contains high concentrations of H_2S.
The objectives of this research are: (1) to estimate the rate of production of hydrogen sulfide using laboratory landfill decomposition experiments, (2) to compare laboratory- and field-derived rates of hydrogen sulfide production, and (3) to determine the effect of gaseous hydrogen sulfide concentration on both methane and hydrogen sulfide producton.
Results showed that methane production during the anaerobic decomposition of office paper is significantly reduced in the presence of sulfate-containing materials (Table 1 and Figure 1). Decay rates calculated on the basis of the lab reactors are also presented in Table 1. These rates will be converted to a field-scale decay rate on the basis of equation 3. The terms on the left side of equation 3 can be measured in the laboratory and the numerator on the rate side of equation three has been estimated in previous work as 0.029 yr^{-1}.

$$\frac{\text{decay rate of OFF at lab-scale}}{H_2S \text{ production rate at lab-scale}} = \frac{\text{decay rate of OFF in the field}}{H_2S \text{ production rate at field-scale}} \qquad (4)$$

Concurrent with the laboratory-study, field data from a northeastren landfill that accepted fines from a MRF that accepted C&D fines were also analyzed. The field-data suggest a decay rate of 0.4 to 0.9 based on the assumed sulfate concentration of the waste.

Table 1. CH_4 and H_2S production during decomposition of office paper in the presence of different sulfur –containing materials in laboratory-scale reactors through day 368[a].

Treatment[b]	pH (day 367)	Reactor CH_4 yield (mL CH_4/g OFF)	Reactor H_2S yield (mL H_2S/g sulfate)	k_{lab}[c] (1/yr)
MSW+Office Paper Control (R1 and R3)	7.4	92.7(7.2)	NC[d]	NC[d]
Trona Ash (R1 and R2)	7.6	89(0.1)	44(19.6)	26.0 (15.5)
Lime Ash (R1 and R3)	7.0	12.1(3.5)	43.2(2.1)	19.2 (2.8)
Trona Ash + Lime Ash (R1 and R2)	7.4	20.8(3.2)	49.9(19.1)	25.3 (9.1)
Gypsum (R1 and R3)	6.9	1.6(2)	7.2(2.6)	NC[e]
C&D Fines (R1, R2 and R3)	7.5	12(5.7)	55.2(48.7)	21.34 (5.2)

a/ Standard deviations are given in parentheses. b/ Average of two reactors unless noted. Outlier values were excluded from the calculation of average parameter. c/ Calculated based on data through day 144 (to be updated). d/ NC-Not calculated because the sulfate content of the office paper is unknown. e/ NC-Not calculated because of very low gas production.

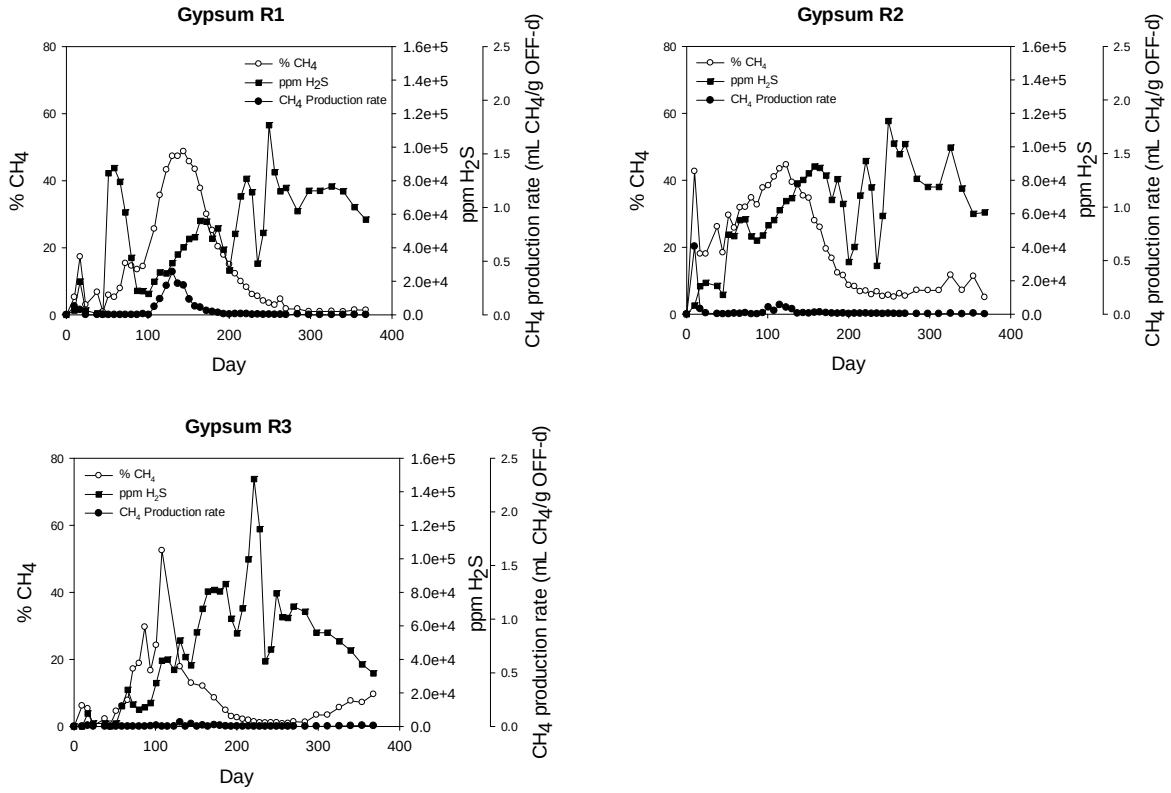


Figure 2. Decomposition data for gypsum in laboratory-scale landfill simulations (through day 368)

Dynamics in leachate quantity and quality at MSW landfills

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Introduction

Environmental concerns associated with Municipal Solid Waste (MSW) landfills, containing high content of biodegradable organic matter, are related to the generation of leachate and biogas. Both, quantity and quality of leachate and biogas formed depend upon the characteristics of the waste, the design and operation of the landfill and the climatic conditions (temperature, precipitation, and evapotranspiration).
In order to stabilize a landfill in a controlled and efficient way, so that environmental impacts are minimized on the short and long term, understanding of the processes in the landfill interior is crucial. In the last decades water and water flow were identified as the main factors determining the metabolism of landfills. Water is on the one hand essential for the biochemical decomposition of organic substances and on the other hand needed for leaching of soluble compounds. Different investigations showed that enhanced water flow through waste leads to an acceleration of biochemical processes, as water is the only carrier of substances within a landfill and only water flow facilitates the redistribution of chemicals, micro-organisms and nutrients. Water is also needed for hydrolysis which is the first step in the anaerobic degradation process.
Several researchers have pointed out, that in order to improve existing approaches for describing the landfill behavior, further research must focus on the analysis of leachate generation and its composition. Hence, the focus of the present paper is on the evaluation of dynamics in leachate quantity and quality. Thereto, leachate data are presented and analyzed for a closed MSW landfill. Finally, different phenomena of leachate generation characteristics are highlighted, and their potential impact on the long term evolution of leachate composition is discussed.

Evaluation of landfill leachate data

The landfill considered is located at a former gravel mining pit in Austria (47° 44′ 37″ N, 16° 08′ 28″ E) and has been constructed for research purposes. Between 1987 and 1988, around 95,000 tons of MSW were disposed of at this site and compacted (in layers of 1.0 m) to a wet waste density of approximately 1,200 kg/m³. The site is divided into three compartments with different capping systems (see Huber et al., 2004). In 2009 the permeable landfill cover initially installed (annual leachate generation rate of about 22% of precipitation) has been exchanged by a composite lining system (60 cm mineral liner and 2.5 mm HDPE liner). By this measure leachate generation has been reduced to about 2% of annual precipitation (630 mm/a) within a period of 2 years.
In the following figures different phenomena of leachate characteristics (regarding generation rate and composition) are illustrated. Figure 1 highlights the quick response in leachate generation due to single rainfall events, which can be explained only by preferential flow paths that shortcut a large bulk of the waste in landfills (see right hand side of Figure 1). Consequently, preferential flow of water in MSW landfills has been reported by different investigations in the recent years (e.g. Rosqvist, 1999).

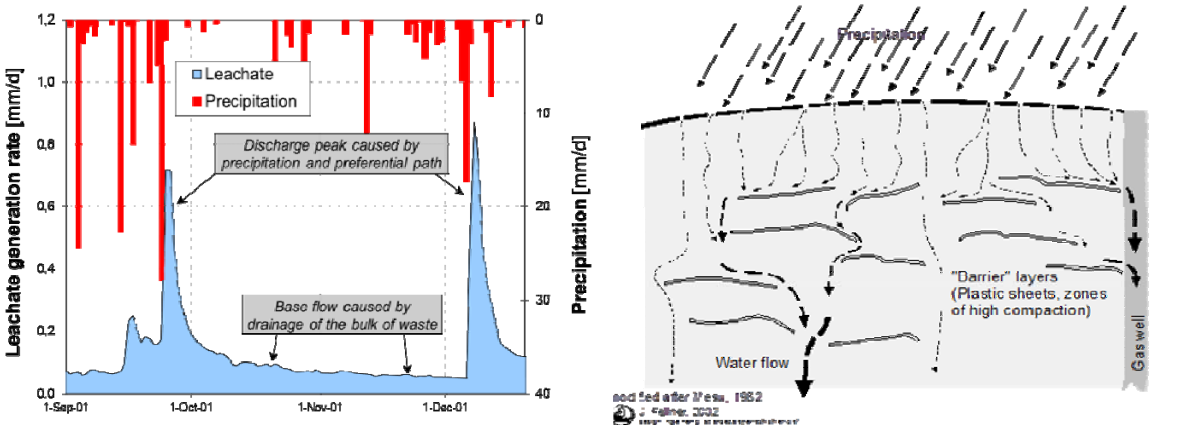

Figure 1: Leachate generation rate versus precipitation events (left hand side) and schematic illustration of water flow through landfills (right hand side)

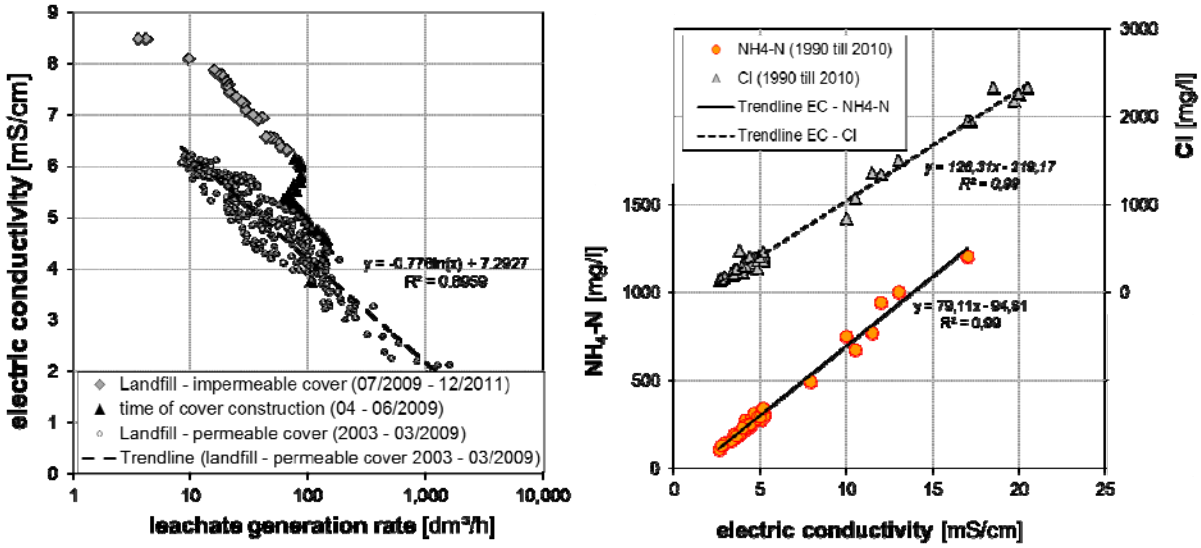

Figure 2: Electric conductivity vs. leachate generation (left hand side) and correlation between electric conductivity, NH_4-N and Cl concentration (right hand side) – (Laner et al., 2011)

The heterogeneity of water flow in landfills is also responsible for the distinct correlation between generation rate and quality of leachate (e.g., electric conductivity, Cl, NH_4). High generation rates inevitably result in significant dilution of leachate and hence lower pollutant concentration. However, a shift in the correlation between flow rate and substance concentrations can be observed if the landfill is disturbed, such as due to the installation of a new cover system at the case study landfill in spring 2009 (see Figure 2).
For the long term evolution of leachate quality this means that firstly information about the degree of preferential flow and secondly its potential changes over time are crucial.

Reference List

Huber, R., J. Fellner, et al. (2004). Water Flows of MSW Landfills and Implications for Long-Term Emissions. Journal of Environmental Science and Health, Part A – Toxic/Hazardous Substances & Environmental Engineering 39(4): 881-896.
Laner, D., J. Fellner, et al. (2011). Future landfill emissions and the effect of final cover installation – A case study. Waste Management 31(7): 1522-1531.
Rosqvist, H. (1999). Water Flow and Solute Transport in Landfills. Department of Civil and Environmental Engineering. Stockholm, Royal Institute of Technology: 40.

Requirements on gas permeability of gas distribution and methane oxidation layers in biological methane oxidation systems

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Introduction

Residual landfill methane emissions can be mitigated in optimised landfill covers (biocovers), biowindows or biofilters. In all cases, the effectiveness of the system is particularly dependent on the spatial evenness of the gas load to the methane oxidation layer (MOL), which has to be warranted by the correct choice of materials and an adequate design. Spatial distribution of the impinging gas flux is often realised by a gas distribution layer (GDL) of coarse material, underlying the MOL. Whether the preferred horizontal gas migration in the GDL occurs to the required extent depends on the difference between the permeability of the GDL and the MOL along the respective path lengths and is hence a function of material properties, compaction, and location of gas inlet points as well as factors that can change material permeability such as soil moisture. So far, guidance on these details is missing. With this paper, we will outline the boundary conditions under which spatially homogenous gas distribution can be achieved, considering soil properties, inclination and water balance. The model SWAP 3.2 (Kroes et al., 2008) will be used to simulate soil water balance and thus conditions of variable gas permeability under dry continental and moist atlantic climate.

Approach

Spatial harmonisation of the methane load can only be accomplished if the total gas flow resistance between inlet and emission into the atmosphere is nearly constant. This requires that the gas permeability (k_{Gas}) over the horizontal path length in the GDL significantly exceeds the permeability of the vertical path length in the MOL:

$$k_{Gas}\_GDL >> k_{Gas}\_MOL \qquad \text{Eq. 1}$$

From the point where the gas escapes from the underlying waste body it travels horizontally along the distance x1 in the GDL, vertically in the GDL along the distance x2 and finally through the MOL along path x3 (Figure 1).

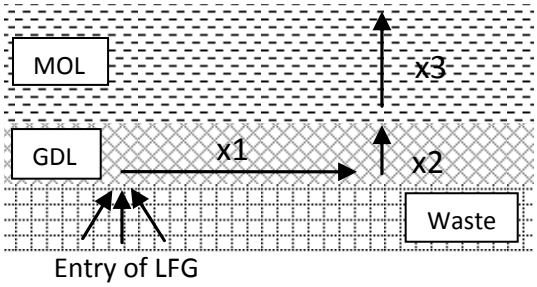


Figure 1: Schematic of gas transport in GDL and MOL. x1 = distance, over which gas travels in GDL; x2 =thickness of GDL; x3 = thickness of MOL.

The reciprocal value of the gas permeability coefficient ($1/k_{Gas}$) reflects the resistance (R) of a material towards gas flow (here: R_GDL and R_MOL). In order to enforce preferential horizontal distribution of gas, the total resistance $\Sigma(R)$ along the vertical path length (GDL

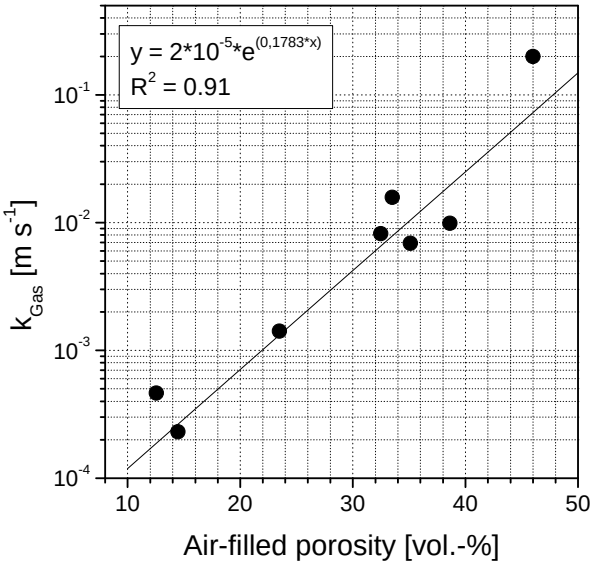
plus MOL) has to be significantly larger than the total resistance along the path length in the GDL:

$$\Sigma(R\_GDL*x1; R\_GDL*x2; R\_MOL*x3) >> \Sigma(R\_GDL*x1; R\_GDL*x2) \qquad \text{Eq. 2}$$

In a simplified notation, the left and the right term are referred to as Σ(RV) and Σ(RH). To warrant spatial evenness of the gas load, the combination of materials for the GDL and the MOL should aim at maximising the difference between $\Sigma(R_V)$ and $\Sigma(R_H)$ over as long path lengths as possible. The difference in permeability between the two layers is subject to variability over the season. For example, the moisture content in the MOL can be reduced due to enhanced evapotranspiration in summer (increasing k_{Gas}) or if seepage water accumulates in the GDL (decreasing k_{Gas}). In order to "buffer" these changes Gebert (2011) suggested to choose a combination of materials warranting a ratio of at least 10:1 for the resistance total of the vertical versus the horizontal gas transport:

$$\Sigma(R_V)/\Sigma(R_H) \geq 10. \qquad \text{Eq. 3}$$

Based on modelled variability of soil moisture for a range of soils and climates the authors aim to clarify this suggestion.



In practice, resistance ($1/k_{Gas}$) can be estimated from the share of air-filled porosity (AFP) of a given material. Figure 2 shows that k_{Gas} increases exponentially with AFP. According to equation 2 the required permeability and hence AFP of the GDL also depends on the permeability of the MOL and on the distance between the gas entry points into the GDL. Due to the likely inhomogeneity of the waste body, uneven spatial distribution gas pathways within the waste and hence uneven load to the gas distribution layer are to be expected.

Figure 2: Gas permeability in relation AFP. Data for mineral materials intended for application in gas distribution layers.

In addition to gas permeability, adverse capillary effects at the GDL/MOL-interface have to be considered. These occur in case of significant differences in the unsaturated hydraulic conductivity between GDL and MOL and can for example be avoided by a similarly textured sand fraction. Should these requirements not be compatible with the requirements on the difference in gas permeability, placement of intermediary layers may be necessary.

References

Gebert, J. (2011): Optimised CH_4 oxidation systems. Criteria for suitable materials. Contribution to the annual meeting o the IWWG task group CLEAR during the SARDINIA 2011 conference.

Kroes, J. G., Van Dam, J. C., Groenendijk, P., Hendriks, R. F. A., Jacobs, C. M. J. (2008): SWAP version 3.2. Theory description and user manual. Wageningen, Alterra, Alterra Report 1649, Swap32 Theory description and user manual.

Effect of diffusive gas transport on the fractionation of stable isotopes in landfill covers

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Introduction

Fractionation of carbon stable isotopes (SI) occurs during microbial CH_4 oxidation (Barker and Fritz, 1981). Provided that the fractionation factor α_ox is known, SI analysis enables the quantification of in situ CH_4 removal efficiencies, for example in landfill cover soils. As fractionation effects are also induced by gas transport processes such as diffusion (De Visscher et al., 2004), our study aimed at separating the biotic and abiotic fractionation effects, taking into account different soil physical properties.

Methods

Fractionation of carbon stable isotopes was investigated in a column study simulating a landfill cover soil using a sandy loam compacted to three different levels (75, 85, 95% of the Proctor density; setup described in Gebert et al., 2011). An artificial landfill gas mixture (60% CH_4, 40% CO_2) was circulated through the gas distribution layer at the base of the columns to maintain a constant concentration gradient between the column base and the surface. In this particular part of the experiment, the headspace was flushed with N_2 to preclude CH_4 oxidation. Soil gas composition and the change in the isotopic signature along the diffusion path were monitored by GC/GC-IRMS-analysis in different depths. The fraction factor for gas transport (α_trans) was approximated from the change in isotopic signature and the change in CH_4 concentration using the Raleigh equation for open systems.

Results

Diffusive gas transport through soils can result in significant depletion in ^{13}C along the path length (Figures 1), in this study up to 75 ‰. This is a result of the faster diffusive flux of the lighter isotopologue $^{12}CH_4$. The effect is increased with higher air-filled porosity (lower compaction) and hence increased high diffusivity (compare Figure 1 top and bottom). Correspondingly, calculated values for α_trans were 1.008 at high (95% D_{Pr}) and 1.016 at medium degree of compaction (85% D_{Pr}). Significant diffusive fractionation was also observed across the soil-atmosphere interface (Figures 1, change between 5 cm depth and headspace). Particularly in older landfills, diffusion is the main mode of gas transport through the cover soil. In the field, soil physical properties and hence diffusivity may vary significantly over small spatial scales due to heterogeneous bulk density, aggregation or soil moisture; hence α_trans will also vary strongly. These uncertainties may render the application of the SI method for the quantification of the oxidation process very difficult.

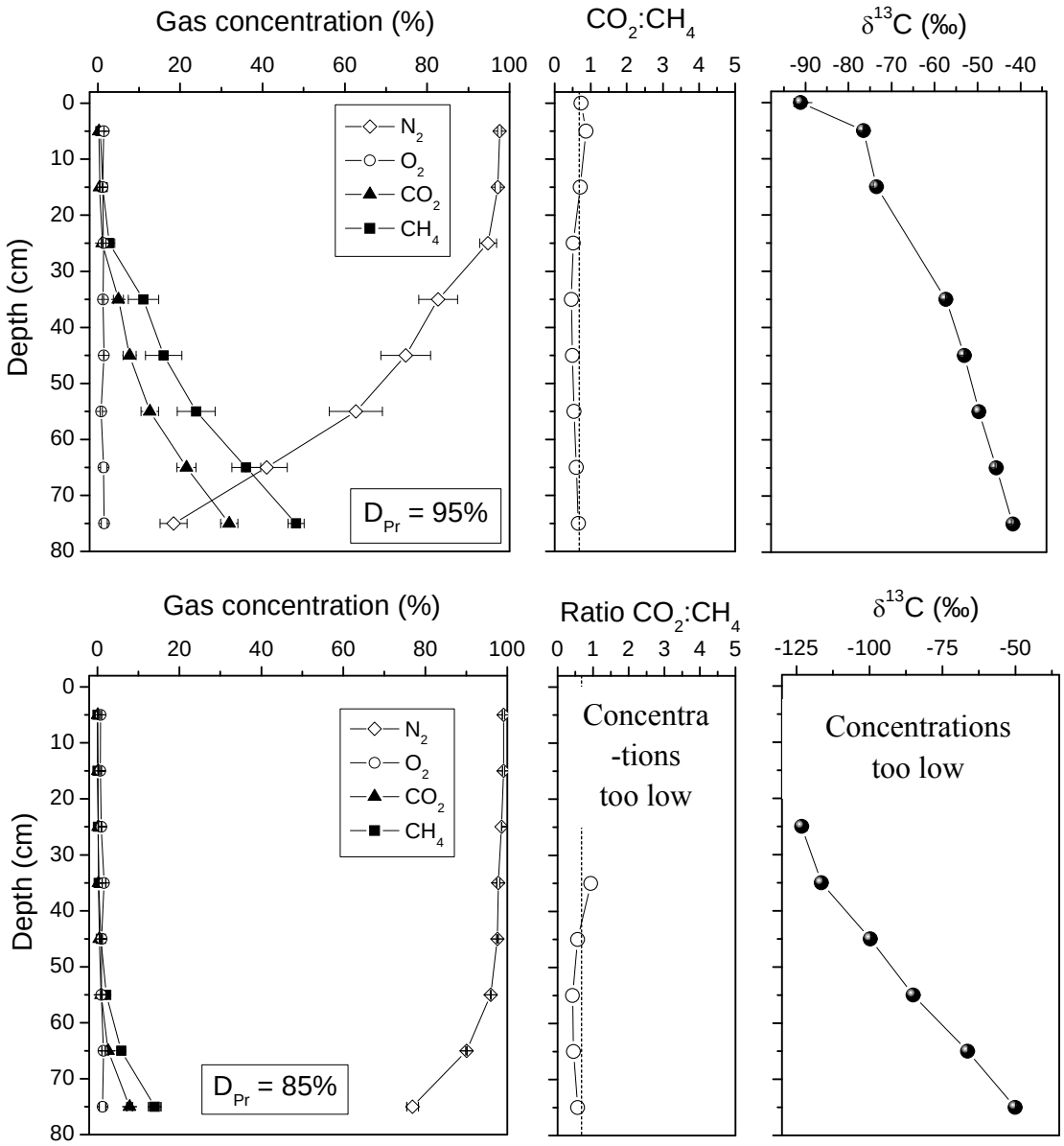

Figure 1. Gas composition and isotopic signature at a compaction of 85% Proctor. Data points: average of three replicates; error bars: min., max. $D_{eff} = 1.61 \times 10^{-6}$ m^2/s. Depth 0 = headspace. $D_{eff} = 5.82 \times 10^{-7}$ m^2/s

References

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

De Visscher, A., De Pourq, I., Chanton, J. (2004). Isotope fractionation effects by diffusion and methane oxidation in landfill cover soils. Journal of Geophysical Research 109, (D18111).

Gebert, J., Gröngröft, A., Pfeiffer, E.-M. (2011). Relevance of soil physical properties for the microbial oxidation of methane in landfill covers. Biology & Biochemistry 43, 1759-1767.

Heterogeneous gas distribution within a biocover designed for methane oxidation

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Introduction and Research Questions

Microbial oxidation is a cost efficient and sustainable technology to reduce landfill methane emissions after gas extraction systems have been shut down. Methane oxidation can take place in the cover layer, engineered as biocover, as well as in biowindows or biofilters. In any case, specific requirements exist with respect to physical and chemical properties of the gas distribution layer and the methane oxidation layer as well as to the dimensions and constructive setup of the system. The even distribution of the gas entering the oxidation layer is a prerequisite for avoiding preferential flow and achieving high methane oxidation efficiencies. On a landfill in the north-west of The Netherlands the efficiency of gas distribution and methane oxidation is being monitored in two large scale test fields with different constructive characteristics. Objective of the study presented here was to examine whether the desired spatial homogeneity of gas distribution at the basis of the oxidation layer was achieved.

Design of Test Fields

The two test fields have an area of 510 m^2 each and are situated on the northern slope of the landfill with an inclination of 1:5. They are sealed towards the waste body by a HDPE membrane. The controllable landfill gas flow into the fields is realized by six inlets distributed on the HDPE membrane. The fields differ concerning the nature of the gas distribution layer and the construction of the oxidation layer. The gas distribution of field C is realized by way of a gravel layer which is part of the capillary barrier to control leachate (20 cm capillary block of gravel (2-8 mm), 30 cm capillary layer of sand (1-2 mm)). The overlaying oxidation layer of 110 cm was placed with a long-stick excavator, achieving a mean bulk density of 1.34 g cm^{-3} and an air filled porosity of 21 vol.-%. In field G a 1 cm thick water drainage mat was used for gas distribution. The 110 cm oxidation layer in this field was installed with a bulldozer. A total mean bulk density of 1.50 g m^{-3} and an air filled porosity of 12 vol.-% were achieved here. Because of a remedial action, in the outer border the soil was built in with a lower compaction. Within the time frame of this study (starting in May 2011), the two fields were first loaded with 1 l CH_4 m^{-2} h^{-1}. In August 2011 the flux was increased to 2.5 l CH_4 m^{-2} h^{-1}. CH_4 concentration in the landfill gas was approx. 47 vol.-%, CO_2 concentration was about 33 vol.-%.

Methods

The soil gas phase was sampled by means of 108 permanently installed gas probes in 1 m depth directly under the oxidation layer but over the gas distribution layer. The probes consist

of aluminum tubing with an inner diameter of 7 mm sealed with a rubber septum. The probe volume is purged directly before gas samples are extracted by means of a syringe and were analyzed immediately with a biogas analyzer (BM 2000 "Biogas", Geotechnical Instruments (UK) Ltd.) for CH_4, CO_2 and O_2. Concentration of N_2 was calculated as difference to 100 %.

First Results

The obtained data were interpolated with Kriging and have to be considered qualitatively. Distribution and concentration of gases show clear differences between the two fields (Fig. 1). A considerably higher heterogeneity of soil gas composition can be seen in field G. As N_2 is inert, i.e. is neither consumed nor produced, it can be used as a measure of the aeration of the soil. At the high load of 2.5 l CH_4 m^{-2} h^{-1} the position of the six gas inlet points (circles) in field C become apparent through high methane concentrations. While the methane in field C stays inside the field under both flow rates, field G tends to show high concentration areas at the border of the field. The well-aerated area between the fields, not loaded with landfill gas, shows an elevated methane concentration at the border of field G and seems to be shifted towards field C. This points to a lateral migration of landfill gas from field G towards field C.

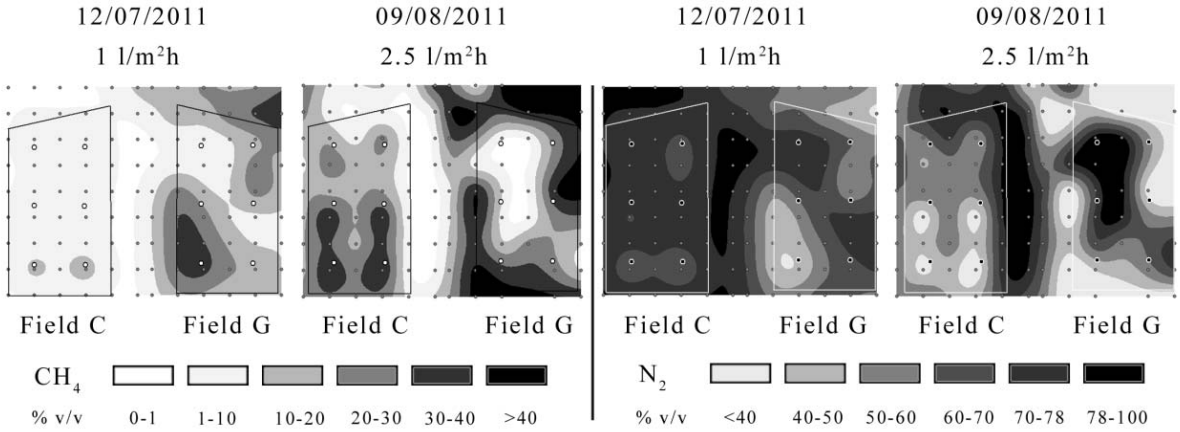

Fig. 1: Spatial distribution of soil gas concentration of CH_4 and N_2 in 1 m depth. Margins of test fields are displayed as lines. Small dots = gas probes, circles = gas injection points.

Conclusions and perspectives

The patterns of gas distribution show that none of the employed gas distribution layers allow for an even spatial distribution of the gas. In field C the gas seems to ascent directly from the inlets to the oxidation layer. This suggests that the difference in permeability between gas distribution layer and oxidation layer is not large enough to force the gas to a horizontal distribution. The results from field G suggest that the high compaction of the oxidation layer leads to preferential flow to the margins of the field. The vertical movement of the gas might be initialized by the HDPE barrier and then controlled by the different compactions of the surrounding and the different soil water conditions due to inclination.

Acetate oxidizing syntrophs functioning as methanogenic initiation center from acid crisis in anaerobic degradation

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BACKGROUND

Landfill is the major disposal route of municipal solid waste (MSW) in China. The Chinese MSW is characterized by a high moisture content (up to 65%) and degradability, its rapid decomposition easily lead to accumulation of VFAs (up to tens in g/L) and low pH (to a minimum ranging between 5.5 and 6.0), which results in a long lag phase before initiation of methanogenesis, thus extends the period for stabilization. To accelerate methane production and stabilization process in landfill, a deeper insight into methanogenesis initiation is important.

The release of methanogenesis from acid crisis depends on the rapid degradation of acetate by acetotrophic methanogens or syntrophs of acetate-oxidizing bacteria and hydrogenotrophic methanogens. The mixotrophic *Methanosarcina* species have been proposed as an initiation center in overcoming the low pH, high VFA conditions (Staley et al., 2011). Meanwhile, the acetate-oxidizing syntrophs were also indicated to have a great potential in initiating methanogenesis, especially from highly acidic status (Lü et al., 2010; Kotsyurbenko et al., 2007), which was still scarcely documented and quantified. It is not clear how the microbes contribute to initiation of methanogenesis via acetoclastic methanogenesis (AM) or syntrophic acetate oxidation coupled with hydrogenotrophic methanogenesis (SAO-HM) from acid crisis.

In the present work, the metabolic pathways during methanogenic conversion of 100 mmol/L acetate from different pH status (5.0−6.5) were investigated using isotopic signature. The syntrophic pathway was separated using selective inhibition method and the abundance of the syntrophic acetate-oxidizing bacteria was quantified by QPCR, in order to determine the potential of the acetate-oxidizing syntrophs as an initiation center which was finally confirmed.

RESULTS AND DISCUSSIONS

Lower initial pH extended the lag phase time attributed to the inhibition of undissociated acetic acid, with values of 1.6 d, 3.4 d and 16.9 d being observed at initial pH values of 6.5, 6.0 and 5.5 respectively in the un-inhibited controls. The addition of CH_3F depressed the AM pathway; however, methanogenesis started slowly 1−2 d later than the control via SAO-HM. At initial pH 5.0, methanogenesis was inhibited throughout the 39-day incubation (**Fig. 1a**).

In the absence of CH_3F at initial pH 6.5 and 6.0 sets, the stable carbon isotope was slightly fractionated (α_{app} being calculated as 1.008 and 1.014, respectively) (**Fig.1b**), with SAO-HM only accounting for 21−22% of the total CH_4 production. Conversely, at initial pH 5.5 set, the isotope fractionation was much stronger (α_{app} = 1.034), and the fraction of SAO-HM pathway increased to 51%. In the CH_3F-treated pH 5.5 set, the carbon isotope was strongly fractionated (α_{app} = 1.083), with demonstrating the initiation of methanogenesis via SAO-HM independently. The abundance of ACS-gene, reflecting the syntrophic acetate-oxidizing

bacteria, remarkably increased in pH 5.5 set (5.1×10^3 copies/ng-DNA) (**Fig.1c**), followed by the pH 6.0 set (1.1×10^3 copies/ng-DNA), while that in pH 6.5 and pH 5.0 sets decreased to lower levels (47 and 23 copies/ng-DNA), which supported the shift of dominant pathway from AM (pH 6.0 and 6.5) to SAO-HM (pH 5.5) during initiation of methanogenesis.

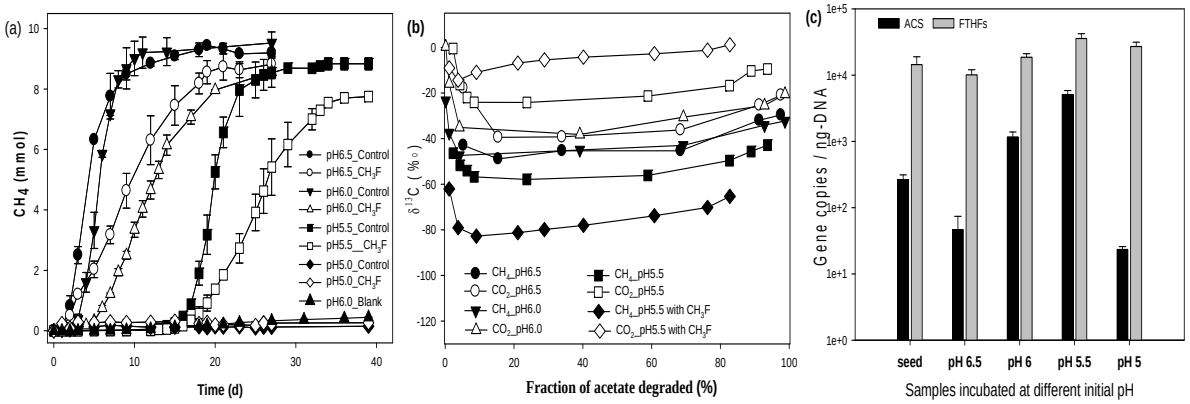

Fig.1 (a) CH_4 accumulation, (b) stable carbon isotopic signatures of CH_4 and CO_2 and (c) FTHFS-gene and ACS-gene copy numbers

CONCLUSIONS

The dominant pathway shifted to SAO-HM under highly acidic conditions (pH 5.5) from AM in neutral and slightly acidic (initial pH 6.0−6.5) environments. The acetate-oxidizing syntrophs could function as the initiation center of methanogenesis from low-pH acid crisis.

REFERENCES

Lü, F., Hao, L.P., Shao, L.M., Bouchez, T., He, P.J. Release from acid crisis by shifting methanogenesis pathway under thermophilic anaerobic digestion. Proceedings of Venice 2010, Third International Symposium on Energy from Biomass and Waste, Nov.7-11, 2010. Venice, Italy.

Kotsyurbenko, O.R., Friedrich, M.W., Simankova, M.V., Nozhevnikova, A.N., Golyshin, P.N., Timmis, K.N., Conrad, R., 2007. Shift from acetoclastic to H_2-dependent methanogenesis in a west siberian peat Bog at low pH values and isolation of an acidophilic methanobacterium strain. Appl. Environ. Microbiol. 73, 2344–2348.

Staley, B.F., de los Reyes III, F.L., Barlaz, M.A., 2011. Effect of spatial differences in microbial activity, pH and substrate levels on methanogenesis initiation in refuse. Appl. Environ. Microbiol. doi:10.1128/AEM.02349-10.

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Is it possible to quantify emission potential from high resolution monitoring of leachate dynamics?

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Introduction

Modern sanitary landfills are one the most important final storage solutions for safely storing waste in modern society. Many countries around the world have implemented regulations in order to protect the environment from adverse emissions from the landfills. Generally these regulations require the installation of protective barriers which inhibit contaminants within the waste from migrating to the surroundings. After the active phase of landfilling, regulations require after-care to be carried out the landfill in order to ensure that no migration of contaminants take place. The after-care can be considered to be completed when the landfill no longer poses a threat to human health and the environment. The question that has to be answered is, when does a landfill no longer pose a threat to human health and the environment?
Although it is not easy to answer to this last question, we would like to try. First we assume that any technical barrier implemented in order to prevent migration of contaminants to the surroundings of the landfill will eventually fail. If in such a case the landfill still contains a significant amount of emission potential, we can be certain that a threat to the surroundings will occur. Therefore, the only way we can assume that the landfill no longer poses a threat to human health and the environment is when we can ascertain that the emission potential within the waste has reduced to levels at which the migration of contaminants has become very low. It is important to realise that emission levels will never be really zero so “acceptable” emission levels need to be defined. These levels have to be based on criteria which protect the surrounding ecosystem and need to be implemented in regulations. Quantification of the remaining emission potential is a major challenge. This paper presents the approach we are investigating in order to address this challenge.

Leachate emission

Leachate develops because rain infiltrates into the landfill and migrates through the waste from which it drains as leachate. During the time water is flowing through the waste body it interacts with the waste and many processes occur. Minerals present in the waste dissolve or precipitate depending on the local chemical thermodynamic conditions. Therefore, flow of water leads to the flow of solutes within the waste body. Many dissolved species increasingly accumulate in the flowing water while if flows through the waste body. In addition, the flow of water has a significant impact of the activity of micro-organisms living off the organic compounds present in the waste. Enhancing the flow of water through landfills has been shown to significantly enhance the microbial production of methane. Therefore, the longer water remains within the waste body the more influenced by the waste it will be.

Rainfall dynamics lead to variable flow rates within the landfill. Because leachate levels in modern landfills are controlled at low levels (about 1 meter above the bottom liner), the waste in the landfill will be present under unsaturated conditions. Variation in flow rates will therefore also lead to significant variations in local water pressures and water contents. As a result, the relative permeability of the waste will be highly variable, both in time as well as in space. This local variability is enhanced by the intrinsic heterogeneity of the waste itself, impermeable materials such as plastics will block and diverge the flow path of water and maybe even lead to local ponding. We hypothesize that as the water moves deeper in to the waste, presence of impermeable barriers will increasingly lead to a funneling of water in to a increasingly smaller volume of the landfill. As a result the water will more and more flow thorough a limited number of preferential flow paths. The volume fraction of these preferential flow paths is dynamic and depends on the rainfall rates. High rainfall rates lead to significant preferential flow in which the rain is funneled through the landfill in a relatively short time. Leachate produced during dry periods, however, has drained slowly from the bulk of the waste. Water present in the waste is therefore moving at a wide range of velocities and therefore individual water drops have a wide range of residence times in the waste. As a result the leachate produced during high rainfall rates will have relatively low concentrations of waste related species, whereas, leachate produced during dry conditions has been influenced by the waste for a significant period and as a result the concentration will be high.

Assessment of Leachate Emission and Landfill Emission Potential

Coupling detailed high frequency measurements of leachate production and leachate quality to rainfall rates and rain quality will allow us to develop an understanding of the dynamics in leachate production and leachate quality. We aim to use these measurements, possibly combined with a set of simple experiments, in order to characterize the probability distributions of residence times of water drops in the waste. Our hypothesis is that residence time is a key parameter relating the the final concentration of waste related species in the leachate to the emission potential of these species in the waste. In this context we consider the emission potential to be similar to an emission source term which slowly loads the leachate with waste related species during the residence time of the leachate in the waste. The rate of loading depends on the bio-geochemical processes that take place in the waste and the challenge is now to quantify the loading rates from the dynamics in leachate production and leachate quality.
The reason why we want to quantify the landfill emission potential from the leachate data is related to the scale of the landfill and the heterogeneous nature of the waste. A landfill cell has a surface area of several hectares and often heights of several tens of meters. This leads to a waste volume of several hundred thousands of cubic meters. Taking samples from such a volume is not a trivial activity. Questions such as what size should the samples have and how many samples should be taken, are not easy to answer. In addition there are significant technical issues that need to be overcome in order to take samples of the deeper sections of the landfill. The consequence of these challenges will be that sampling will have a high cost. We are currently developing a probabilistic modeling framework in which we combine process based models with a data assimilation approach to quantify unknown parameters, of which the source term in the waste is one of the most important.

Experiences from biocover implementations in Austria regarding leachate minimization and enhancement of methane oxidation

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Introduction and Background

During the past few years it could be shown that enhancing microbial methane oxidation in suitable and engineered landfill covers (biocovers) can be a simple and low-cost measure to mitigate methane emissions, in particular on older landfill sites, on landfills with mechanically-biologically pretreated waste and also as a practical method for developing countries. However, since biocovers, biofiltres and bio-windows are usually one part of a capping system, they must not disturb any of the several different functions of the capping system, i.e., control leachate generation; minimize methane and odor emissions; provide a suitable basis for vegetation or for other intended after-use options; and protect against erosion. In order to answer to this complex question, we have to put all different pieces of this puzzle together to get an overall look on how such systems will act over the long-term, while considering the conflicting needs embedded in the design of methane oxidizing structures, for example, how to reconcile water infiltration control with methane oxidation. Indeed, sufficient infiltration is necessary during the early life of MSW-landfills to trigger the degradation processes. However, over the longterm, leachate minimization is desired. Moreover, when a high methane turnover takes place, water is produced, which may have influence on the water balance of the top cover and the landfill, as well as the vegetation and recultivation.
On the basis of findings from lysimeters and field trials with biocovers, experiences and data obtained from first full-scale applications of biocovers and biowindows on recently closed landfills (currently about five sites in Austria) and on old dumps (currently two sites in Austria) in Austria, recommendations for a "multifunctional cover" addressing all relevant aspects are summarized.

Scope of study

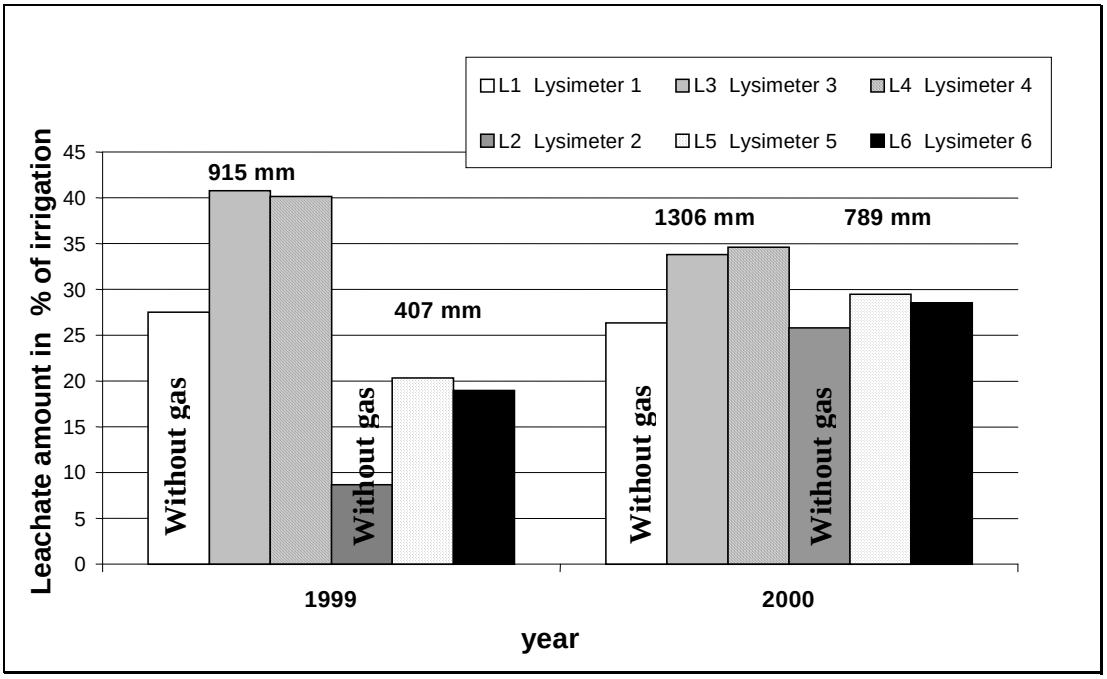
Over a period of two years, lysimeters (1 m^3) were used to study detailed the impact of methane oxidation in a 90-cm-thick layer of sewage sludge compost on water and gas budgets, leachate production and on the growth and contaminant transfer of grass vegetation (Miscanthus). The interaction between vegetation and the methane oxidation process and the impact of methane oxidation on leachate quality was also monitored. One set of lysimeters was ventilated with artificial landfill gas (60% methane, 40 % carbon dioxide), the second set was not. The amount of supplied methane was equivalent to that generated in a 15 to 20-meter-high, older municipal solid waste (MSW) landfill.

The interactions between methane oxidation, gas and water budget, plant growth and leachate production have been also monitored over some years on a closed MSW-landfill in Austria, where a biocover made of a compost/sand mixture was applied as the final top cover.
Moreover, experiences gathered from other full-scale applications of diverse biocovers in Austria were evaluated and will be presented.

Main findings and experiences
Some of the main findings of the lysiemter test are sumarized:

- The lysimeters that were supplied with landfill gas produced more leachate, mainly due to the water formed via methane oxidation.
- At low irrigation levels, when plants were limited by lack of water, plant growth was indeed higher in the gas-ventilated trials because the methane oxidation made additional water available to the plants.
- Methane oxidation has no negative impact on plant growth. In contrast, plant roots facilitate the oxygen transport necessary for methane oxidation, especially at greater depths.

In the figure below, the leachate amount of the various lysimeters (with and without gas supply) during the first and second vegetation period are shown.



Field experiences from the full-scale applications show, that biocover combined with suitable vegetation are also capable to act as a "evapotranspiration" to reduce leachate amount. In Austria's landfill directive a limit value for the leachate rate of a landfill of 5% of the annual precipitation (measured over 5 years) is required for evapotranspiration caps as final covers. Previous experiences show that this requirement can be achieved, particularly in the eastern part of Austria (mean annual precipitation between 500 and 800 mm) with sufficiently designed biocovers combined with proper vegetation possessing high transpiration rates.

Biocover Properties Affecting Spatial Variability of Methane Oxidation in Landfill Covers

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Two well-controlled test cells were constructed in the field to evaluate the short-term (0-18 mo.) performance of two alternative materials for landfill biocovers: fresh and aged green material, i.e., yard waste. The field experiments were also used to quantify biocover properties that result in temporal and spatial variability in total upward gas flux, methane (CH_4) oxidation, and CH_4 emissions.

The fresh and aged green materials were packed in biocover tests cells constructed near a landfill module at Yolo County Central Landfill, CA USA where landfill gas could be diverted to each biocover cell. Each biocover cell foot print was 2.4 m × 1.8 m × 1.2 m (length × width × height). The layout of each test cell is shown in Figure 1. During the operational period, profiles of gas composition were measured continuously in both cells, and the influent landfill gas (LFG) flow rate and composition were monitored. Spatially variable biocover moisture content was also measured periodically. At selected measurement periods, flux chambers were used to cover the entire cells to quantify total upward gas flux, CH_4 production, CH_4 oxidation, and CH_4 emissions. Smaller inverted "bucket" flux chambers were sometimes used to determine the spatial variability of LFG flux and CH_4 oxidation. In addition to these field tests, laboratory experiments were used to quantify CH_4 oxidation rates and gas transport properties for these materials, i.e., diffusion coefficients and gas permeabilities, using intact cores from the field. Water flow properties of fresh and aged green materials were determined using a combination of water retention cell measurements in the laboratory and disk infiltrometer data from field experiments.
The focus of this presentation is on the influence of biocover gas flow and water retention properties on CH_4 oxidation. Figure 2A shows the spatial variability of upward flow of a conservative tracer, sulfur hexafluoride (SF_6) introduced in the influent LFG for Biocover Cell 1 (aged green material). SF_6 flux varies by a factor of four over the domain: it is highest in the bottom left corner, but lower and relatively uniform over the rest of the domain. At the same time as these flux measurements, moisture contents were also determined and are shown in Figure 2B. Here, moisture content varies between 0.20 and 0.40 over the domain: the biocover is driest in the bottom left corner and is wettest in the center, with other regions of intermediate moisture content. Comparing these two figures suggests that regions of higher

gas flow are correlated with regions of low moisture content. The presence of moisture inhibits the upward flux of LFG, which is captured with tracer gas flux data.

The spatial variability of upward LFG into the biocover, which appears associated with variable moisture conditions, resulted in variable CH_4 emission and CH_4 oxidation. The percent CH_4 oxidized varied between 3 and 31%, while CH_4 emissions varied from 74 to 511 $g/m^2/day$ over the domain. The highest and lowest percent oxidation corresponded to the lowest and highest CH_4 emissions, which is consistent with other recently reported studies. In this presentation the impact of spatially variable gas and water flow properties of the green material on moisture content, upward LFG flow and CH_4 emissions will be discussed.

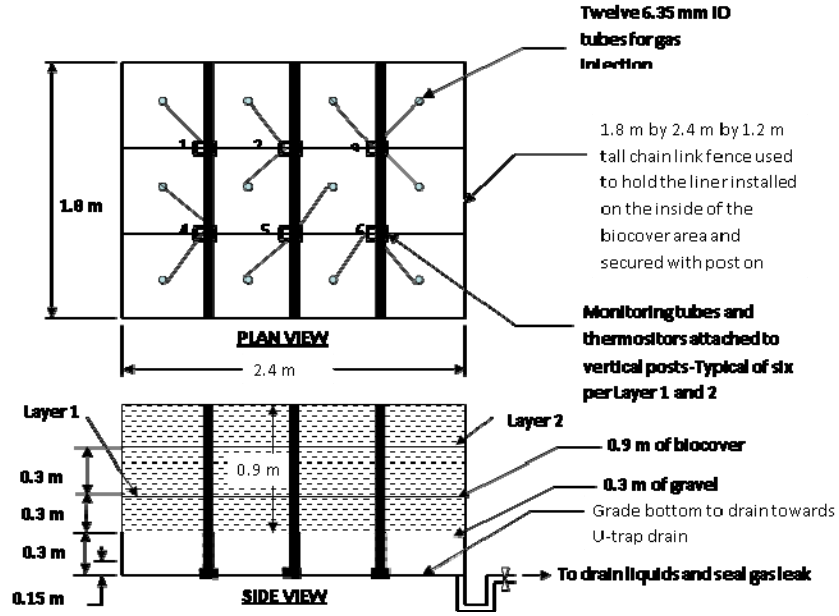

Figure 1. Layout design of each biocover test cell.

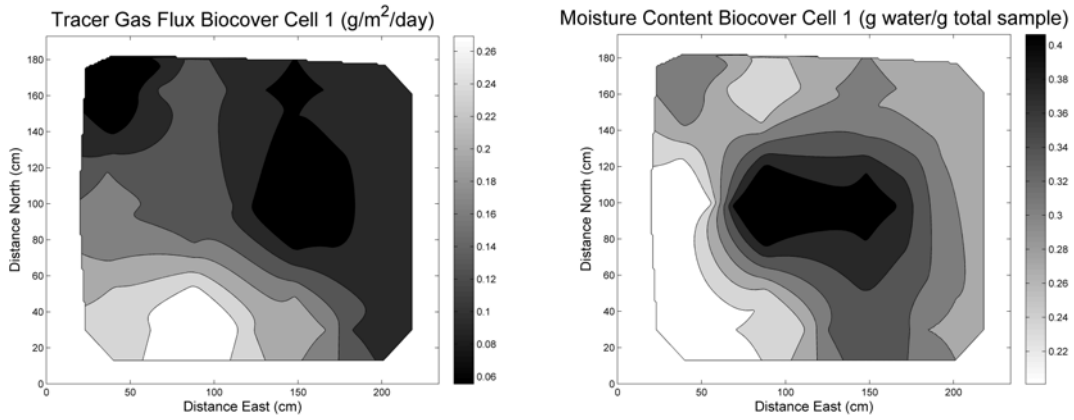

Figure 2. Surface contour of upward flux of SF_6 tracer gas (left) and corresponding vertically-averaged moisture content (right) for Biocover Cell 1 (aged green material). Drier regions correspond to regions of large upward gas flox.

Reducing methane emissions from Icelandic landfills by use of passive oxidizing biocovers

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Introduction

With the adoption of EU directives and local regulation no. 738/2003 on landfill waste, Icelandic landfills receiving biodegradable waste were required to collect landfill gas after July 16, 2009. A study carried out in 2010 (Júlíusson, 2011) revealed that most landfills in Iceland generate too little methane for it to be technically or economically feasible to collect biogas, as required by the above mentioned regulation. Landfill after-care with oxidizing biocovers is considered among key mitigating measures to reduce greenhouse gas emissions from landfills, according to the report of Working Group II of the Intergovernmental Panel on Climate Change (IPCC, Bogner et al., 2007). The methane oxidation process can furthermore be exploited in engineering systems developed for general methane emission mitigation (Scheutz et al., 2009), whether gas collection systems are in place or not. The method however needs to be verified with respect to specific soil and climate conditions in Iceland before it can be subjected for further implementation. In support of this study, a recent TAIEX mission report (Scharff, Hansen and Gústafsson, 2011) recommends a demonstration project to gather information on microbial oxidation in a boreal climate such as Iceland, since methane oxidizing covers might provide a feasible alternative for methane emission reduction on small Icelandic landfills. The study results will be of direct use to all Icelandic landfill operators and would contribute to ongoing research in the field.

Objectives and project plan

The project has three main objectives; 1) to develop and test methodology to assess and evaluate methane oxidation in Icelandic landfill covers, 2) to evaluate a suitable composition of soil materials in order to maximize the oxidation capacity of landfill covers with respect to Icelandic conditions, possibly with the aid of local geothermal energy and 3) to develop a general proposal or guideline for the closure of Icelandic landfills with the aim of minimizing methane emissions. A methodology will be developed based on international research (Gebert et al., 2011, Cabral et al., 2010, Jugnia et al., 2009, Jugnia et al., 2008), to assess methane oxidation through the use of a) gas profile methods (CO2/CH4 ratio) and b) mass balance methods (CH4), using gas flux measurements (dynamic chamber method) and methane generation measurements and estimations in Icelandic landfills (Júlíusson, 2011). International standards will be employed for all measurements. Improvements proposed recently will also be considered. Experimental plots will be set up at the Fíflholt landfill in Borgarfjörður, Western Iceland, following an initial grid-based field scan. A search for suitable local soil materials will be carried out in collaboration with local authorities and the Solid Waste Management of the West Iceland Regional Office (Sorpurðun Vesturlands). Instruments and equipment for field measurements are provided by the University of Iceland (Háskóli Íslands), EFLA Consulting Engineers and Iceland

GeoSurvey (ÍSOR). The study entails soil sample analyses, grid-based surface scans, design and setup of experimental plots and periodic measurements of surface fluxes and gas profiles from June to November 2012. The oxidation capacity and oxidation rate of the experimental biocovers will be assessed and the results and methodology evaluated. A general proposal will be developed on the closure or remediation of Icelandic landfills using oxidizing biocovers, with the aim of minimizing methane emissions to the atmosphere.

References

Gebert et al., 2011. Can soil gas profiles be used to assess microbial CH4 oxidation in landfill covers? *Waste Management* 31: 987-994.

Cabral et al., 2010. Biocover Performance of Landfill Methane Oxidation: Experimental Results. *Journal of Environmental Engineering*, 08.2010:786.

Jugnia et al., 2009. Diversity and Dynamics of Methanotrophs within an Experimental Landfill Cover Soil. *Soil Biology And Biochemistry*. SSSAJ: Volume 73: Number 5.

Jugnia et al., 2008. Biotic methane oxidation within an instrumented experimental landfill cover. *Ecological Engineering*, 33: 102-109.

Júlíusson, A.G. 2011. *Hauggasrannsóknir á urðunarstöðum á Íslandi* (e. Landfill gas generation at Icelandic landfills). MS-thesis, Faculty of Civil and Environmental Engineering, University of Iceland, 123 p.

IPCC, Bogner et al., 2007. *Mitigation of global greenhouse gas emissions from waste: conclusions and strategies from the Intergovernmental Panel on Climate Change (IPCC)* Fourth Assessment Report. Working Group III: Mitigation of Climate Change.

Scharff, Hansen & Gústafsson, 2011. *Mission report. TAIEX Assistance Iceland concerning the implementation of the EU Landfill Directive and landfill gas control*. TAIEX reference 45985.

Scheutz et al., 2009. Microbial methane oxidation processes and technologies for mitigation of landfill gas emissions. *Waste Management & Research* 27: 409-455.

Geophysical monitoring of in situ redox processes

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Introduction

Hazards associated with municipal solid waste landfills have increased the interest of population for a development in the treatment of landfills. The main problems of landfills are gas and leachate emissions. Bioreactor landfills that use treatment methods, such as recirculation of leachate and aeration for minimizing the production of leachate and the stabilization period of a landfill, are already in use [ITRC, 2006]. However, aftercare periods are still in the range of 30 years and there is no guarantee that the emissions will cease after this period. Therefore a method to quantify the leachate emissions and stabilization of a landfill is required [Heimovaara et al., 2012 submitted].
To be able to quantify these emissions it is necessary to understand the processes occurring inside a landfill. Our ambition is to be able to quantify the processes occurring by connecting the physical properties measured with geophysical measurements, with chemical properties. Geophysical measurements map the structure and provide information about the properties of the subsurface. Using geophysical measurements will allow us not only to map the leachate but to extract information about physical properties of the waste and leachate as well. An understanding of the physical and chemical properties, not only in stationary conditions, but also in time, allows us to improve our predictions of emission potential which will eventually improve emission potential reduction treatment of landfills.
This specific abstract focuses on the first step of our approach which includes the understanding of the processes occurring by conducting laboratory experiments. In detail, a biobattery is set up is adopted to understand the coupling between redox gradients present in waste and geophysical measurements.

The method

We focus on the electrical properties measurable by a variety of geophysical measurements and aim to include those processes occuring within the landfill that affect the measurable signal; electro-kinetic (fluid flow), electro-thermal and electrochemical (junction and redox potential) signals. As a starting point we will use Self Potential (SP) measurements in order to connect the physical properties measured with chemical properties. SP are passive electrical methods that can measure the electrical signal obtained from the processes occurring inside the landfill. We focus on the electrochemical and the electrokinetic processes and aim to calculate the chemical and redox potential.
Studies by Revil [2004] have already shown that there is a coupling between fluid flow, junction potential and electric potential. However we aim to add the term of redox potential and try to uncouple these processes in order to estimate redox and chemical potential. The innovation of our method is the addition of this term. Equation 1 shows the suggested model. $J_{junction}$ is the current density due to concentration differences, J is the electric current density, U the Darcy velocity and J_{redox} the current density due to redox processes. The coefficients in the second matrix are the Onsager's coefficients that show the relation between the different processes. σ is the electrical conductivity, e the number of electrons, σ_+ and σ_- the ionic conductivities , C_+ and C_- the ionic concentrations and k and η the permeability and viscosity respectively. In the last matrix the gradients are shown with μ the chemical potential, φ the electrical potential, p the pressure and E the redox potential.

(1)
$$\begin{bmatrix} 2J_{junction} \\ J \\ U \\ J_{redox} \end{bmatrix} = \begin{bmatrix} \frac{\sigma}{e^2} & \frac{1}{e}(\sigma_+ - \sigma_-) & \frac{k}{\eta}(C_+ + C_-) & ? \\ \frac{1}{e}(\sigma_+ - \sigma_-) & \sigma & \frac{ek}{\eta}(C_+ + C_-) & \sigma \\ \frac{k}{\eta}(C_+ + C_-) & \frac{ek}{\eta}(C_+ + C_-) & \frac{k}{\eta} & ? \\ ? & \sigma & ? & \sigma \end{bmatrix} \begin{bmatrix} \nabla\mu \\ \nabla\varphi \\ \nabla p \\ \nabla E_H \end{bmatrix}$$

The coefficients depicted with a question mark are the ones we aim to determine. These show the relation of the redox processes with fluid flow and with chemical, diffusion processes. In order to do that, we will conduct a number of experiments based on a biobattery set up. As shown in figure 1 the biobattery will consist of soil in a temperature controlled glass box, creating oxic (top) and anoxic (bottom) conditions. Redox sensitive (Ag/AgCl) electrodes will be connected to a voltmeter to measure the redox potential. By applying a fluid flow in our system the change of the redox potential due to fluid flow can be tested. In addition, adding concentration will change the chemical potential of the system and measuring again the redox can show us the connection between those two processes.

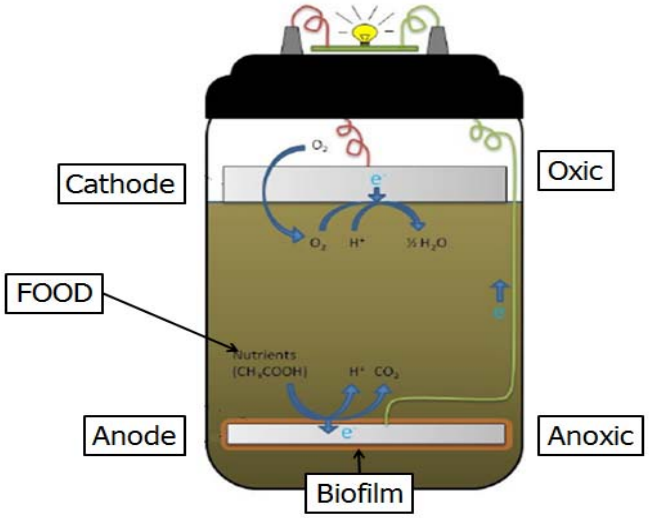

Figure 1: Biobattery set up. (Modified from [Keego, 2010])

Conclusions

The results will provide us with substantial information regarding the relation between the redox and the other processes occuring in the landfill, which we can use further on to develop our model and in the future for application on the field. Having the ability to calculate redox and chemical potential will allow us to make estimations regarding concentration gradients, Gibbs energy and pH, which will help us understand the system in detail and will allow for a development of a better treatment method.

References

ITRC (2006). Characterization, Design, Construction and Monitoring of Bioreactor Landfills.
Heimovaara T.J., Konstantaki, L.A., van Turnhout, A.G., Bun, A., Baviskar, S.M. (2012 submitted).Reduction of the emission potential through leachate from sanitary landfills.
Revil, A., Leroy, P. (2004). Constitutive equations for ionic transport in porous shales. Journal of Geophysical research, v.109

Arsenic leaching in landfilled soil

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Aim

The aim of the study was to assess the leaching of arsenic (As) from contaminated soil under simulated landfill conditions (low redox environment). In addition, soil samples were amended with sulfur-containing materials to evaluate changes in As leaching. Admixture of sulfur-containing compounds might lead to formation of stable arsenic sulfides that would render arsenic immobile at low redox conditions.

Materials and methods

Three soils from former wood impregnation plants containing 270-4590 mg/kg As were mixed either with coal fly ash (CFA), calcium sulfate (CaSO4) or calcium sulfide (CaS) (3 wt% each) and incubated for three months under methane environment. Soil solution was sampled and analyzed for dissolved As.

Preliminary results

Leaching of As increased significantly in reducing environment (Fig. 1). None of the soil amendments were effective in reducing As leaching in anaerobic environment to the levels that were measured under aerobic conditions. It is likely that the time of experiment was too short for the sufficiently low redox to establish in the samples. However, for one out of three soils (Innansjö), the leaching of As decreased in all amended samples as compared with the unamended soil in both environments. Further research is being carried out to understand the reasons of such a different behavior of arsenic in different soils.

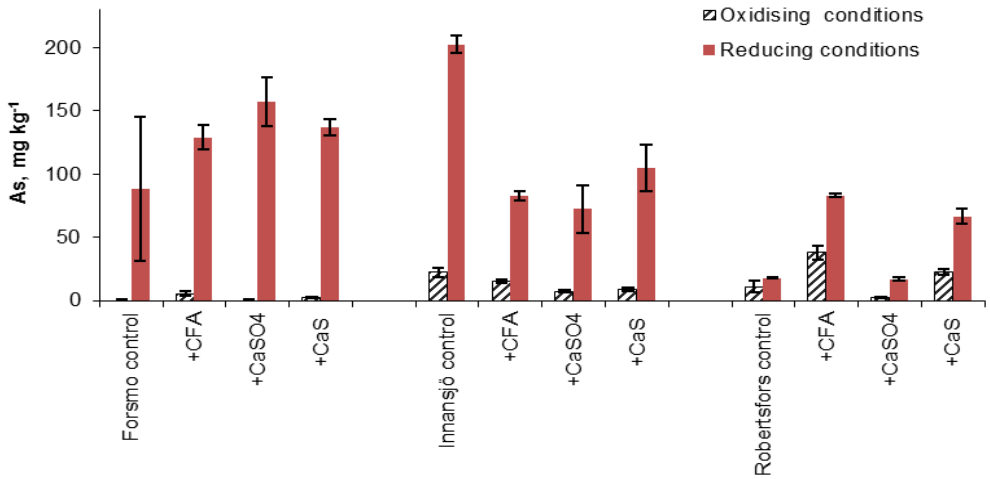

Figure 1. Leaching of arsenic in contaminated soils and soils amended with sulfur-containing materials under oxidising and reducing conditions.

Some simple and frequent errors is landfill research

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In landfill studies, methods are regularly employed that stem from other areas of research. For example standard metods from sanitary engineering may be used to analyse leachates. Some problems with total solids (TS), volatile solids (VS), chemical oxygen demand (COD), biological oxygen demand (BOD), and total organic carbon (TOC) has been discussed e g by Lagerkvist and Chen (1993), Kylefors et al. (2003) and Kumpiene et al. (2011). However, it seems relevant to bring it up now and again, because these basic techniques and their results does seems to be used uncritically quit often.
In this presentation some common errors will be discussed and some simple checks that can be used to assess data will be presented.
The analyses mentioned above will be focussed. The aim of the communication is to encourage a discussion on possible remedies, e g the creation of a methods collection for use in landfill research labs.

References

Lagerkvist, A., & Chen, H. (1993). Control of two step anaerobic degradation of municipal solid waste (MSW) by enzyme addition. *Water Science and Technology, 27*(2), 47-56.

Kylefors, K., Ecke, H., & Lagerkvist, A. (2003). Accuracy of cod test for landfill leachates. *Water, Air, and Soil Pollution, 146*(1-4), 153-169.

Kumpiene, J., Robinson, R., Brännvall, E., Nordmark, D., Bjurström, H., Andreas, L., Lagerkvist, A., Ecke, H. (2011). Carbon speciation in ash, residual waste and contaminated soil by thermal and chemical analyses. *Waste Management, 31*(1), 18-25.

How to consider long-term technical barrier performance in an aftercare completion framework?

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Introduction

After closure, landfills need to be managed and controlled until a landfill is not likely to present a threat to human health and the environment in the absence of care. Criteria to define the state of a landfill at the end of aftercare need to address different aspects significant to the environmental risk associated with a landfill. These include the behavior of the deposited waste and associated emissions, the (long-term) performance of the technical barrier system, and the migration of pollutants in the surrounding environment and potential adverse effects on environmental media (e.g. Laner 2011). Substantial uncertainties have to be dealt with in the risk analysis, due to limited availability of representative data, incomplete process understanding, and unknown future conditions. One of the major challenges in such an endeavor is the evaluation of long-term technical barrier performance (cf. Pivato 2011).
In this work, a scenario-based approach to consider different future performance levels of technical barriers is used to derive site-specific aftercare completion criteria. The whole procedure to evaluate landfill aftercare and define completion criteria is schematically illustrated in Figure 1. In this paper, the focus is on the development of different (long-term) performance scenarios for the technical barriers and the use of these scenarios in the developed aftercare completion framework. Finally, a critical discussion of long-term technical barrier performance evaluations in an aftercare completion framework is provided.

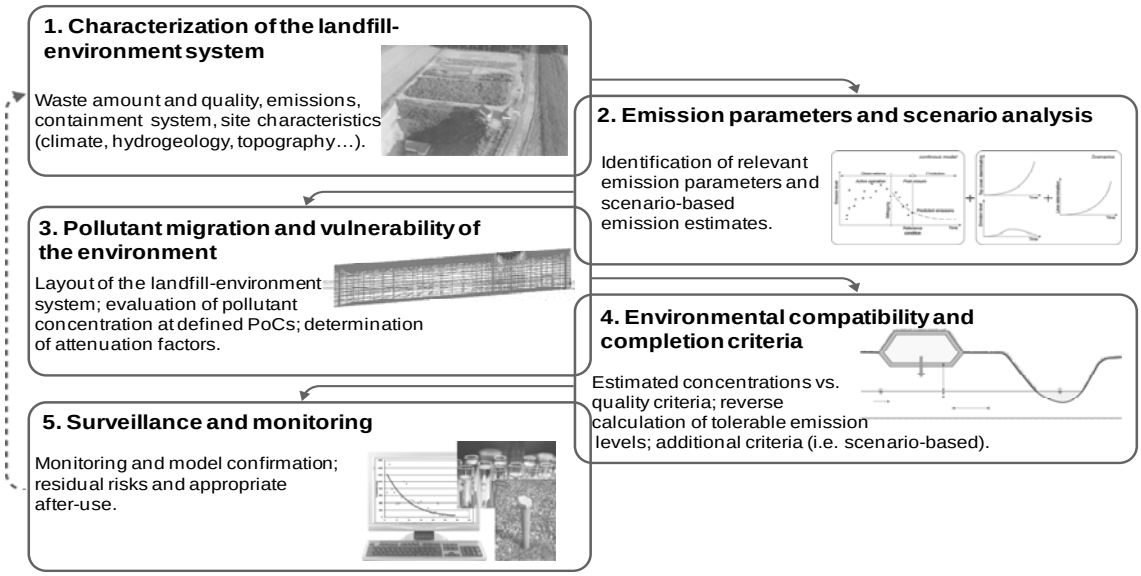

Figure 3: Schematic illustration of the procedure to derive aftercare completion criteria at a closed landfill site (Source: Laner 2011)

Scenario-based approach to evaluate technical barrier performance

At modern landfills technical barriers typically consist of several components such as low-permeability mineral layers, geosynthetic materials, drainage layers or recultivation layers. The performance of the technical barrier system is the product of the interaction of the different elements. In the short term, the most important factors with respect to barrier performance are proper design and construction of the system and geotechnical stability

issues (cf. Bonaparte et al. 2002), while the long-term performance is primarily dependent on the stresses induced on the system at the site compared to the resistance built into the system. The elements of a technical barrier system have different service lives (without continuous maintenance and repair) ranging from several decades (e.g. leachate drainage systems) to thousands of years (e.g. low-permeability mineral liners) (cf. Rowe 2005). However, due to the very limited period of field observations for technical barrier systems (several decades) and the complex interactions within the system and at the boundaries of the system, the evaluation of long-term barrier performance is associated with substantial uncertainties. Therefore, a scenario-based approach was used to evaluate future barrier performance in the assessment of environmental risks from a landfill. Three scenarios are developed to estimate future landfill emissions for a bandwidth of potential performance levels of technical barriers and deterioration patterns. In the best case scenario, the effect of constant barrier performance (i.e. the current performance level is perpetually maintained) on landfill emission levels is investigated. In the worst case scenario, a complete failure of the technical barrier system (e.g. percolation rates equal to the local groundwater recharge rate adapted for landfill relief and soil cover) is assumed. The third scenario assumes a gradual barrier deterioration pattern, based on an evaluation of different factors addressing the conditions at the landfill site and the resistance built into the system. An evaluation score is assigned to each factor (e.g. climate, expected settlements, relief, soil cover thickness, etc.) and a weighting algorithm (based on a survey involving experts in the field of landfill engineering) is used to derive performance levels (i.e. percolation rates) for specific time periods (cf. Laner 2011).

Discussion and outlook

The definition of performance scenarios for technical barriers allows for explicitly including assumptions on long-term barrier performance as additional aftercare completion criteria in the aftercare completion framework. For instance, if the aftercare completion criteria specified for a landfill were derived from a landfill emission scenario based on worst case barrier performance, no more maintenance of the technical barrier system will be necessary after landfill closure, because even the complete failure of the system would not cause intolerable impacts at the site. However, although technical barrier performance scenarios represent a transparent way to deal with assumptions on future interaction between the landfill and its surroundings, currently we do not have the data (especially at the field-scale) or the process understanding to adequately predict the performance of technical barrier systems. Therefore, as this is a crucial issue in the long-term environmental risk analysis from landfills, it is necessary to be vigilant in the assessments and to openly communicate inherent assumptions to decision makers involved with landfill aftercare and its completion.

Reference List

Bonaparte, R., D. E. Daniel and R. M. Koerner (2002): Assessment and Recommendations for Improving the Performance of Waste Containment Systems. Cincinnati, OH, US EPA: 1039.

Laner, D. (2011): Understanding and evaluating long-term environmental risks from landfills. Dissertation thesis, Institute for Water Quality, Resources and Waste Management, Vienna University of Technology, Vienna.

Pivato, A. (2011): Landfill Liner Failure: an Open Question for Landfill Risk Analysis. Journal of Environmental Protection, 2011 (2): 287-297.

Rowe, R. K. (2005): Long-term performance of contaminant barrier systems. Geotechnique 55(9): 631-678.

Fluorochemical Signatures in Municipal Waste and Landfill Leachate

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Concern about fluorochemicals as environmental contaminants has increased due to reports on their presence in humans and wildlife, most notably in species in remote regions of the environment distant from points of known release. Fluorocarbon chains are both hydrophobic and oleophobic, which makes them useful in a wide range of applications in which they repel water, oil and potential stains. Fluorochemicals enter landfills through the disposal of materials that have these compounds applied to their surface (e.g. carpet, textiles, paper, food packaging). In addition, these compounds are present in biosolids which are sometimes disposed in landfills. Unfortunately, the same properties that render fluorochemicals so useful, also render some forms, including perfluorooctane sulfonate and perfluorooctanoate resistant to degradation by acids, bases, and microbial processes. The spread of fluorochemicals is of concern because several are classified as persistent, bioaccumulative, and toxic. The presence of fluorochemicals in landfill leachate has been documented which leads to questions about the significance of landfill leachate as a source of fluorochemicals in surface water (Huset et al., 2011). While landfill leachate is typically treated, many fluorochemicals are not attenuated in traditional biological treatment processes.

The overall objectives of this research are to (1) quantify the concentration and composition of fluorochemicals released from refuse and refuse components by leaching and biodegradation of these refuse components, (2) characterize fluorochemical concentrations in leachates from a representative cross section of U.S. landfills as a function of refuse age, climate, and landfill operating strategy, and (3) estimate the mass of fluorochemicals released to wastewater treatment plants on a national scale using an inventory model developed in this research.

In the first phase of the experiment, the concentration of several fluorochemicals in samples of individual refuse components will be determined using a methanol extraction. These results will be used to identify waste components that are sources of fluorochemicals in the waste stream. Next, leachate concentrations will be measured in duplicate laboratory-scale reactors filled with samples of mixed residential refuse and in reactors containing refuse samples in which carpet, textiles, and paper have been removed as these are known sources of fluorochemicals. In each case, materials will be tested under both abiotic conditions and in the presence of a methanogenic inoculum to differentiate abiotic and biological processes that may contribute to fluorochemical release.

A second objective of this research is to characterize the concentrations of fluorochemicals in leachate from currently-operating and closed landfills that are operated in arid and non-arid climates, under different operating conditions and containing waste of varying ages. These data will be used to estimate the mass of individual fluorochemicals that enter and are released from wastewater treatment plants (WWTPs) attributable to landfill leachate.

An inventory model will be developed to evaluate the significance of landfills leachate as a source of fluorochemicals to WWTPs and to surface water. The inventory model requires estimates of (1) the total volume of leachate treated in WWTPs and (2) representative fluorochemical concentrations in leachate. The total volume of leachate will be estimated from (a) the total mass of refuse disposed in U.S. landfills, (b) information on the landfill area that is occupied per mass of refuse (tons/acre surface area) and (c) the volume of leachate that is generated per unit area (volume leachate/surface area). The mass buried will be divided into the mass in arid and wet regions. This is important because leachate generation rates vary with rainfall. In addition to climate and its impact on leachate generation, the model will be formulated so that the volume of leachate generated in landfills that are operated in bioreactors can be accounted for separately. This is important because a large percentage of the leachate generated in bioreactor landfills is recirculated and not shipped off-site for treatment. Estimates of leachate generation will be combined with measurements of fluorochemical concentrations in landfill leachate to estimate the total mass sent to WWTPs. As described above, leachate concentrations will be measured as a function of several variables (age, climate, operating strategy). The inventory model will be formulated to allow for as much differentiation attributable to landfill conditions as appropriate. Inventory model development requires numerous approximations to represent the population of landfills and leachate generation. The model will developed in Microsoft Excel and all parameters will be set up as inputs so that the sensitivity of various parameters can be explored.

Huset, C. A., Barlaz, M. A., Barofsky, D. F. and J. A. Field, 2011, Quantitative Determination of Fluorochemicals in Municipal Landfill Leachates, Chemosphere, 82, 10, p. 1380 – 6.

Landfill Leachate Treatment by Plant-Cover Soil Irrigation System: Field Monitoring on Ecophysiological Response of *Nerium indicum*. Mill

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Background

For the integrated management of municipal solid waste (MSW) landfills, vegetation planted on the final cover of landfill site is essential for the safety, stabilization and ecological remediation of closed landfill. Landfill leachate treatment can then be conducted by irrigation onto plant-soil systems, because of the high efficiency of volume reduction by solar evapotranspiration and uptake. Meanwhile, the COD and ammonium pollutants of biorefractory landfill leachate could be effectively reduced by plant-soil ecosystem on final cover when irrigated with leachate, based on soil interception, soil microorganism utilization and plant utilization. For this kind of process, the vegetation should be able to tolerate the stress from leachate irrigation. The impacts of leachate irrigation on physiological and biochemical characteristics of vegetation have been reported in the literature, but the coordinated stresses of natural circumstance and leachate irrigation on vegetation are less known. In the present paper, the influence of leachate irrigation on vegetable growth and physiological response were studied throughout one year, with dogbane oleander (*Nerium indicum*. Mill) as vegetable materials for final cover. It was aimed at evaluating the long-term effect of leachate irrigation by testing selected plants and leachate loading in fields.

Materials and Methods

The tested leachate irrigation section of 150 m^2 was located in the closed cells of the Laogang Solid Waste Landfilling Site, Shanghai. The tested section was separated from other sections with HDPE membrane. The cover soil was constructed from bottom to top with 1 mm HDPE membrane, 300 mm clay and 500 mm nutrient soil (stabilized waste). The leachate irrigation pipes were set at 20 mm below the surface, and the leachate collection pipe were set on the HDPE membrane. The control section without leachate irrigation has the similar soil construction. In the tested and control section, *Nerium indicum* Mill (dogbane oleander) was transplanted for landfill closure. The transplanted saplings were about 40 cm tall and have 3 or 4 branches and planted every 1 m. As long as the saplings survived from October, the tested section was irrigated with leachate at an hydraulic loading of 10 mm/d. Comparatively, the control section had the natural precipitation as the only water source for vegetation growth. The leachate for irrigation came from the effluent of leachate aerated lagoon. It's characteristics were like long age landfill leachate: pH 7.5~8.9, COD 439.3~1738.1 mg/L, BOD 45~160 mg/L, NH_3-N 216.6~1215.2 mg/L, NO_3-N 3.81~41.64 mg/L, TP 2.4~3.1 mg/L, Cl^- 723.4~1467.3 mg/L, and Na 1100~1298 mg/L. The daily atmospheric temperature, precipitation, light duration was recorded. Each 3 months, the stem length and branch numbers were measured. Each two months in one year, the soils and the fresh leaves of

dogbane oleander were collected. The ecophysiological parameters were measured including positive indicators (biomass production, chlorophyll (a+b), ascorbic acid (ASA), glutathione (GSH), peroxidase (POD), dismutase (SOD)) and negative indicators (chlorophyll a/b, H_2O_2, malondialdehyde (MDA), permeability, proline (pro)). The activity of soil microorganisms were assessed by the positive indicators (biomass production, iverlase, urease, phosphares, dehydrogenase, proteinase, nitrate reductase, nitrate reductase, respiration, catalase, $C_{microbes}/C_{soil\text{-}organics}$. The leachate treatment efficiency was demonstrated by volume reduction, COD removal and ammonia removal.

Results and Discussion

The one-year field results showed that the removal efficiencies of both COD and ammonia were higher than 90% during 95% of the tested period. Dogbane oleander could survive when irrigated with leachate at a hydraulic loading of 10 mm/d, grow faster than the control group. The growth trend of dogbane oleander evolved seasonally. The dynamic changes of physiological parameters such as malondialdehyde (MDA), proline (Pro) of leachate irrigation group and control group were likewise and regulated by the atmospheric temperature, while the parameters such as superoxide dismutase (SOD) , peroxidase (POD) , ascorbic acid (AsA) and glutathione (GSH) fluctuated almost seasonally. For example , the activity of SOD and POD was enhanced obviously during the period of June to August and period of October to April. At the same time, AsA and GSH accumulated greatly. Although the seasonal evolution of the physiochemical parameters of leachate irrigation group was more sensitive and fluctuant than that of control group , no obvious discrepancy could be observed between the temporal trends of two groups. It was noted that MDA positively relating to stress were higher than that of control group during the period of June to August , while the activities of SOD and POD positively relating to the stress-resistant ability were lower. Combined with the experimental results, the multivariate statistics using principal component analysis further showed that the ecophysiological responses of dogbane oleander were mainly regulated by seasonal change of climate, but not influenced by leachate irrigation. Therefore, the leachate irrigation would not impose observable stress on the growth of dogbane oleander. However , atrocious circumstance might aggregate the coordinated stresses of environmental factor and leachate irrigation. A similar field test was conducted again for another one year. The ecophysiological responses showed the same trends. It further proved the feasibility of long-term leachate treatment using plant-soil irrigation system.

Key words :landfill leachate; ecophysiology; *Nerium indicum*. Mill; irrigation

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STATUS OF LANDFILLS IN DEVELOPING COUNTRIES: A CASE STUDY ON PAKISTAN.

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INTRODUCTION

Landfills have evolved from the old tradition of land disposal of wastes, a phenomenon in place from pre-historic times. Currently, in many parts of the world the construction and operation of landfills is strictly monitored by the government bodies. Along with the regulations for careful management of landfills, targets are set (e.g. EU Landfill Directive) to reduce the amounts of biodegradable wastes in landfills. Most landfill sites in developed countries are designed to ensure efficient leachate management and landfill gas collection systems to reduce risk of potential hazards.

Despite the efforts in developed countries for reduction in harmful impacts of landfills, many developing countries still have no or ineffective landfill regulations. In Pakistan, for example, with a population of 173,593,380 million people, almost no sanitary landfill sites exist and people are unaware of the connection between poor waste management and health hazards. Waste is either dumped in open spaces or a designated landfill site, which is practically an open dump. These grim conditions require urgent attention and solutions.

METHODOLOGY

Data is currently being collected from waste management companies operating in Lahore, one of the metropolitan cities in Pakistan. Further data will be collected on the conditions of landfill sites and the potential health hazards attached with unregulated landfills, together with the potential for landfill gas generation and landfill mining. Site visits to major landfill sites in Lahore are in planning. A survey questionnaire has been developed to understand the level of awareness and involvement of the public in waste management practices. The data will be analysed focusing on the factors of government legislation and the effectiveness of its implementation, potential health hazards, utilization of landfill sites and public awareness.

SOLID WASTE MANAGEMENT (SWM) IN PAKISTAN

Improper waste disposal and lack of efficient management have been the major issues regarding waste management in Pakistan. According to a study conducted by JICA in 2005,

only 60% of the generated waste is collected in Pakistan. Waste streams vary according to the demographics. Also the waste management systems (collection, storage etc) vary greatly.

Most of the collected waste in Pakistan is indiscriminately landfilled, with no separation. Scavengers collect the recyclable materials by manually sorting the waste in landfills/open dumps. No leachate management or landfill gas collection systems are in place. Poor operation and maintenance conditions are detrimental both for environment and for the health of the general public.

CURRENT LANDFILL PRACTICES

Most of the landfill sites are not scientifically managed sites, but involve illegal open dumping at many unplanned locations scattered in and around the city. There is no official dumpsite for the safe disposal of solid waste. No regulations are observed in the construction of these landfill sites. Scavengers operating on these sites at times burn the wastes in open environment to retrieve metals etc, which is very dangerous.

The leachate that develops in landfills due to degradation of bio-degradable wastes in the absence of any collection system seeps into the groundwater and pollutes it. In poorly managed landfills additional issues include soil contamination, vermin, disease vectors (e.g. rats, flies etc), odour and methane emissions are very common. Such conditions pose a serious threat to the health conditions of people living in communities close to landfills.

CONCLUSIONS

The operating conditions of landfill sites in Pakistan are very poor and the main reason observed is the absence of landfill regulations. Data regarding the number of operating landfill sites needs to be updated to support policy making. Improvements will be suggested after data analysis to ensure that best practice from developed countries is transferred to Pakistan. Utilization of landfills for landfill gas-to-energy projects and material reclamation will prove environmentally and economically beneficial.

Inexperienced Challenge on Tsunami Disaster Waste in Japan

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The Earthquake and Tsunami hit the east coast of Tohoku area in March 11, 2011. The tsunami claimed lives over 20,000 people (5000 are still missing), and left a huge amount of disaster waste. In three prefectures damaged by the tsunami, the amounts of waste produced are equivalent to that for 11 years in Iwate, and that for 19 years in Miyagi. Although Japan is frequently fit by earthquake, tsunami waste in 2011 is very unusual in terms of waste management by the following reasons.

1) Huge amount of waste and vast size of damaged area. Total waste generated is estimated 13-28 million ton, which is equivalent to a half of MSW generation a year in Japan. The area of 450 square kilometer is damaged by the tsunami, and 120 thousand and 200 thousand houses were fully and half destroyed.
2) Wide variety of condition in suffered area. Northern area is characterized by a saw-tooth coastline, which has 1-5 km-wide flat area along coast, and Southern area, Fukushima, was polluted by nuclear power plant accident.
3) Various types of waste to be managed and disposed, including inexperienced type of waste in solid waste management. Waste Mixed waste with high salinity, sludge transported from the sea; contaminated with various hazardous product and substance, such as medical waste, PCB, asbestos; fishes from fish processing plant; WEEE and ELV products.
4) A lot of constraints and restriction. Space for temporary storage, solid waste management facility, human resource and experience, heavy machinery and dump truck etc.
5) Loss of administrative function. Waste management is planed and performed by local government, but government office was lost and peoples were died or missed.

Presentation will be made up of two parts:

A. General aspect
i) Estimation of waste amount, and
ii) Special waste found in the disaster: fish stored in refrigerators, hazardous product and material
iii) Recycling of WEEE and ELV product.

B. Our study on disaster waste
iv) Composition analysis of disaster waste
v) Estimation of density in temporary storage
vi) Estimation of volume in temporary storage
vii) Analysis of chorine content in wood for recycling

INFLUENCE OF VEGETATION TYPE ON MICROBIAL OXIDATION OF METHANE IN LANDFILL COVER SOILS

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Biocovers have been identified as one of the key mitigation technologies and practices to reduce greenhouse gas emissions from landfills. Therefore, the need to increase its efficiency is paramount.
Observations made during the last 4 years at an experimental site have shown that vegetation plays a key role in enhancing efficiency. This triggered a study to determine the specific role of vegetation in methane oxidation by biocovers, under the controlled conditions of the laboratory.

Four 0.3 m^2 columns (0.61 x 0.46 x 0.52 m) were filled with the modeled biocover, which is composed of a layer of sand topped by a layer of top soil enriched with compost. Columns I, II, and III were seeded, respectively, with millet (Class C4, grass, non-nitrogen-fixing, supplier in carbon), white clover (Class C3, leguminous, N-fixing) and a mixture of millet and white clover according to the ratio 2/3 / 1/3 w/w. Column IV was the control column and was therefore not seeded. Synthetic biogas (50% CO_2; 50% CH_4) was applied to the bottom of each column. The initial loading was 16 $g_{CH4}\,m^{-2}d^{-1}$ and the final one was 180 $g_{CH4}\,m^{-2}d^{-1}$. Gas samples were collected frequently at several depths and on the head-space (HS). Temperature and water content were monitored at the same depths.

The results of the first of a series of five sequential tests are shown in **Error! Reference source not found.** and Table 1. Each loading was applied at least 2 weeks before gas samples were taken and analyzed by gas chromatography. It can be observed in Table 1 that the biocover seeded with white clover provided the highest methane oxidation efficiency (MOE > 90%), while the least effective biocover was the one seeded with a mix of clover and millet, which was nonetheless able to abate nearly 70% of the CH_4 for the highest loading. The results for the column seeded with millet only - which is a non-nitrogen-fixing - show clearly that nitrogen availability was not a limiting factor.

It is hypothesized that the extent of white clover's roots, larger than those of the millet's, contributes to a more effective oxygenation of the soil, therefore to the greater MOE. In the biocover seeded with a mix of clover and millet (column 3; Figure 4), N_2 and O_2 concentrations were higher throughout the profiles, despite the lower efficiency of this column. It is hypothesized that microbial diversity changes with type of plant; there is a competition between bacteria developed close to the root system of the millet and of the white clover. Based on the fact that the control column had a greater efficiency than that seeded with millet, it is hypothesized that millet's root allows for the development of bacterial colonies that compete with methanotrophic bacteria, decrease their quantity, therefore decrease the MOE.

The results obtained, as well 5 years of quantitative field data and qualitative field observations seem to indicate that the rhizosphere layer plays a very important role in CH_4 oxidation. Further methane oxidation tests with the columns, metagenomic analyses and physico-chemical analysis are in progress to further investigate the specific role of vegetation and, in particular, two of the most frequent species found at the site we have been monitoring for five years.

Table 1 : Methane oxidation efficiency in the HS = ($CH_{4\ in}$ - $CH_{4\ out}$)/$CH_{4\ in}$

Loading (g CH_4 m^{-2} d^{-1})	Column 1	Column 2	Column 3	Column 4
16	100%	100%	100%	100%
32	100%	100%	100%	100%
64	100%	100%	100%	100%
100	81%	92%	79%	89%
128	100%	100%	76%	100%
180	79%	90%	69%	84%

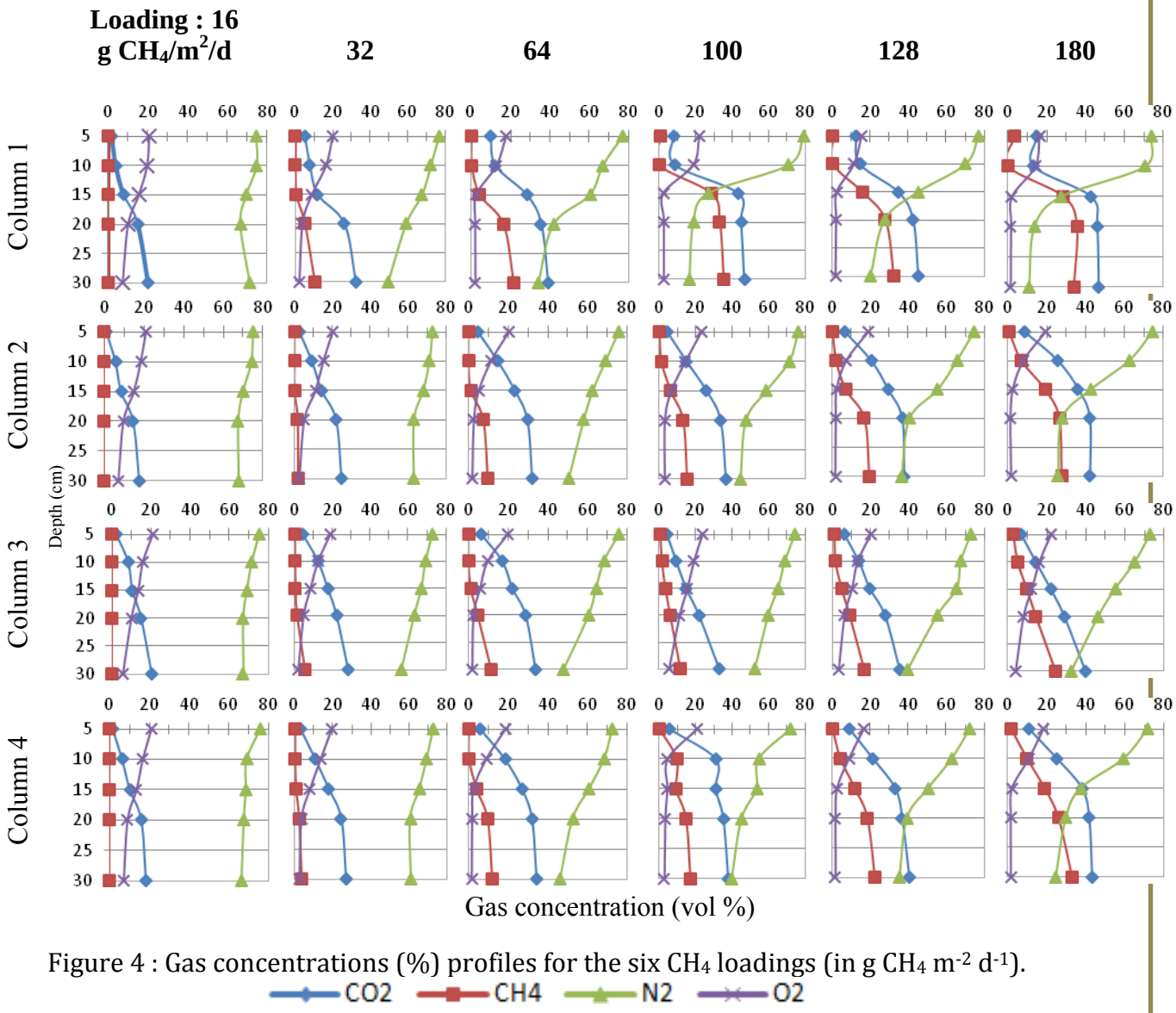

Figure 4 : Gas concentrations (%) profiles for the six CH_4 loadings (in g CH_4 m^{-2} d^{-1}).

Mobility of redox sensitive elements due to organic matter in contaminated soil; bottom ash and residual waste fraction

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Introduction

According to the EU criteria for waste acceptance at landfills (2003/33/EC), concentration of total organic carbon (TOC) should not exceed 3% for inert waste, 5% for non hazardous waste and 6% for hazardous waste. For waste from incineration (e.g. ashes) the limit is 18% TOC. The lowered limit values might affect the redox conditions and hence, the mobility of redox sensitive elements in landfilled waste. The aim of this study was to investigate the distribution of elements between different chemical fractions in wastes and to investigate OM effect on the mobility of redox sensitive elements.

Materials and methods

Three types of materials representing emerging waste streams to Swedish landfills were used in the experiment: chromium, copper and arsenic (CCA) contaminated soil; bottom ash from waste incineration and residual waste fraction, after material reclamation, containing lots of gypsum. Sequential extraction (Tessier*, et al.*, 1979) and a 3-step leaching test described below were performed on the materials with and without addition of OM (sewage sludge).
3-steps leaching test: soil, ash and residual fraction, respectively were filled in glass bottles in triplicates, mixed with distilled water and sealed with butyl rubber stoppers.

– Air in the bottles was replaced with methane. Bottles were stored in darkness at 30 °C for 2 weeks. After 24 h rotation, redox was measured and 5 ml leachate was sampled.
– The bottles were left open for 50 h until redox had stabilized. Then another 5 ml leachate was sampled.
– OM was added and the air in the bottles was replaced with methane again. The bottles were placed in darkness at 30 °C for 2 weeks. Redox was measured three times during the period. After 24 h rotation, redox was measured and 5 ml leachate was sampled.

Leachates <0.45 µm were analyzed for As, Cr, Cu, Pb, Zn, Fe and Mn with ICP-OES.

Results and discussion

The redox potential in the leachates varied during the 3-steps leaching test (Fig. 1). After addition of OM and 2 weeks incubation (step 3) the redox decreased to levels lower than after 2 weeks initial incubation without addition of OM (step 1).

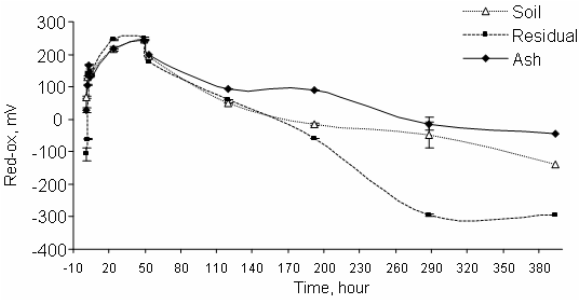

Figure 1 Variations in redox potential in wastes due to atmospheric conditions with and without addition of OM.

Leachability of risk elements varied according to changes in redox potential and OM. Leaching of Cr and Cu decreased in all materials in a reduced environment while the leaching of As and Pb increased (Table 1). The mobility of Zn decreased in soil and residual fraction and increased in ash.

Table 1 Partitioning of elements in wastes according to their leaching pattern under low redox conditions during 3-steps leaching test.

Waste	Decreased mobility (low redox)	Increased mobility (low redox)
Soil	Cr, Cu, Zn	As, Pb
Residual waste	Cr, Cu, Zn	As, Pb
Ash	Cr, Cu	As, Pb, Zn

Arsenic is often bound to redox sensitive Fe and Mn oxides in soil. Increased mobility of As due to addition of OM could be a result of reduced redox potential and a reductive dissolution of Fe and Mn oxides. According to the sequential extraction, the largest amount of Pb in the soil, the ash and a significant amount in the residual fraction was mobilized under reducing conditions. The ash also had a large amount of Zn in that fraction. Pb and Zn have amphoteric properties with minimum of leachability between pH 9 and 10 for Pb and between 9 and 11 for Zn. A successive decrease in pH was observed in the ash samples which also could explain the elevated mobility of Zn and Pb. Lower redox in the wastes with elevated levels of OM caused decreased mobility of Cr, Cu and Zn (except Zn in ash). Those metals occurred mostly in the non- reactive and oxidising phases in the wastes according to the sequential extraction which means that the addition of OM contributed to the immobilization of those elements. The difference between Cr mobility in the samples with and without addition of OM was particularly apparent in the ash. In ash Cr occurs mostly as Cr^{+6}, which is more mobile than the reduced Cr^{+3}.

Conclusions

OM contributed to a reduced environment in specific waste streams and increased the mobility of As and Pb and decreased the mobility of Cr and Cu. Less amount of OM in emerging waste may affect the redox ratio in landfills. Changes in leachate quality should be considered.

References

Tessier, A., Campbell, P.G.C. and Bisson, M. (1979) Sequential extraction procedure for the speciation of particulate trace metals. Analytical Chemistry, vol.:51, n:7, pp:844.

Enhanced biodegradation at the Landgraaf test-cell

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Hans Woelders

Introduction

At the landfill of Landgraaf, a test-cell was operated between 2001 and 2011, with the objective to demonstrate technologies to enhance biodegradation of freshly deposited waste. The test-cell contained 22.000 ton of mainly household waste, industrial waste and shredder waste. Enhancing of biodegradation was promoted by (i) choice of the mix of biodegradable, inert and biologically active wastes; (ii) homogenization of the waste and (iii) recirculation of leachate. At a later stage air was injected to complete biodegradation.
Prior to deposition waste was analysed, a.o. for its moisture content and ignition loss, and a fractionation was done for both household and industrial waste in e.g. organic material, paper, wood, plastic, inerts.
Biodegradation during anaerobic decay was monitored quantitatively by measuring the amount of biogas produced. The effects of aeration monitored in a more qualitative way on basis of gas composition in monitoring wells.
In 2011 the bioreactor was excavated, enabling an sampling and analysis of the residue. Separation tests were performed on the residue, along with respiration and fermentation tests.

Results

The demonstration proved the feasibility to enhance biodegradation to a large extent. The methane generation potential seems to be reduced by more than 95%. The residue meets the German and Austrian criteria on respiration (in 4 days, AT_4) and gas generation potential (in 21 days, GB_{21}).
The enhanced anaerobic phase proved to be very successful and resulted in an almost complete biostabilization of the waste. The subsequent aerobic phase also seemed successful, at least aerobic conditions were observed at all monitoring wells. However due to the successful anaerobic first stage, additional biodegradation under aerobic conditions seems to be limited. Instead anaerobic conditions seem to have resulted in inorganic reactions, e.g. iron-cementation which results in generation of large, stone-like agglomerates of waste particles
The excavation of the test-cell and the sampling of the residue also illustrates the difficulties in determining the remaining methane potential. The differences in particle size are enormous, ranging from sandy grains of about 1 mm, to parts of waste of about 10-50 cm diameter to large rock-hard agglomerates of 30-100 in diameter. Even from excavated waste it is difficult to grab a representative sample from the waste and also take samples from iron-agglomerates. The results of the excavations also give reason to reconsider the way we are modelling biodegradation and especially the biodegradation of lumps of paper and wood. Even under enhanced conditions for biodegradation, these seem to be non-reactive to a large part.

Long term leachate management based on anaerobic/aerobic landfill simulator studies

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Background and research questions

Leachate management and treatment costs form the most remarkable part of the landfill operation costs during the aftercare period. As the aftercare period can be very long before leachate emission limits to surface waters can be achieved, it is question of considerable total costs during the life cycle of the landfill. Therefore it is of importance to find out alternatives based on appropriate engineering solutions to achieve the leachate emission limits (Wang et al. 2011). Related to this the research questions in this paper are as follows:

- how does the L/S-ratio (liquid to solid) affect the aftercare period length at different temperatures to achieve the leachate emission limits ?
- what is the impact of initialization of aerobic conditions at different stabilization stages on the aftercare ?
- based on the above, what are the impacts on the costs and how the costs can be minimized during the whole aftercare period ?

Materials and methods

Waste samples were taken from a big landfill near Helsinki, Finland at different depths (2 – 30 m) and combined to obtain an average composition of the waste in the landfill. Estimated age of the samples was approximately 1 – 9 years, average around 3 years. Around 85 kg of waste was filled to each simulator, of which 2 were run at 20 °C, 3 at 33 °C and 2 at 46 °C. Deoxygenated tap water was added 0.2 – 1.5 l/week (depending on the simulator) preceding equal leachate volume removal. Leachate was recirculated 2.2 l/d during working days, to keep the waste wet enough for near optimal degradation conditions. Organic fraction (VS) of the dry solids (TS) was 45 %. Leachate and gas quality was analysed frequently and gas production was determined.
Air pumping was started in one thermophilic simulator at L/S=4.6, in two mesophilic simulators at L/S 1.8 and 4.6 and in one psychrophilic simulator at L/S 3.6, other 3 simulators were operated in anaerobic conditions the whole period, total of 4.5 years.

Results and discussion

In this study direct leachate discharge emission limits for COD 200 mg/l, Ntot 70 mg N/l and chlorides 100 mg/l were applied. The highest L/S ratios needed were for nitrogen and are shown in Table 1. Anaerobic reactors were run until L/S 5.3, and higher L/S values given are extrapolated from the estimated slopes of the decreasing curves. Liquid to solid ratio around 5.9 was needed in the psychrophilic and mesophilic ranges to achieve the limit value in anaerobic conditions, and in the thermophilic range it was considerable higher. Next highest L/S ratios were needed for chlorides and lowest ones for COD. After the initialization of aerobic conditions nitrogen and COD limit values were achieved within 0.4 - 0.6 L/S units, which showed a clearly faster stabilization and a shorter aftercare period, but chlorides may still remain as a problem.

Table 1. Liquid to solid ratios (L/S) needed to achieve direct discharge limit for nitrogen

Psychrophilic	
anaerobic/aerobic from L/S 3.6	4.0
anaerobic	5.9
Mesophilic	
anaerobic/aerobic from L/S 1.8	2.3
anaerobic/aerobic from L/S 4.6	5.0
anaerobic	5.9
Thermophilic	
anaerobic/aerobic from L/S 4.6	5.2
anaerobic	6.7

The simulator results were applied to calculate the length of aftercare period for a big landfill (50 ha, average waste height 25 m) and a medium size landfill (25 ha, height 10 m) in Nordic conditions. In the scenarios infiltrations of 200 mm/a (base case), 600 mm/a or 1200 mm/a were assumed, including fresh water and pretreated leachate (with nitrification). To find out most economic management alternatives, cost calculation was made for the scenarios and different level of local treatment was included: a) on-site biological treatment with activated carbon filtration, b) pumping to municipal wastewater treatment plant (WWTP) or c-d) two alternatives with lower/higher nitrogen removal efficiency before pumping to WWTP.
Results show that aftercare period length is 75 – 145 years for the big landfill even with elevated infiltration in anaerobic conditions, but by introducing aerobic conditions at L/S 1.8 the period length is 30 – 60 years. For the medium size landfill aftercare period length would decrease from 25 – 50 years (elevated infiltration, anaerobic) to 10 – 20 years (aerobic respectively). Lowest total costs were estimated for scenarios with highest infiltration in both landfills sizes with efficient nitrogen removal before pumping to WWTP, next lowest costs with similar infiltration and on-site treatment with direct discharge. Direct pumping to sewer (WWTP) showed highest costs in all scenarios. Initialization of aerobic conditions (from L/S 1.8) would decrease the total costs clearly, but a later start did not bring economic advantages.
In conclusion, aftercare period length was shortest in scenarios with highest water infiltration with initialization of aerobic conditions at L/S around 1.8. The total costs of leachate management during aftercare period were lowest in scenarios with shortest aftercare period together with high nitrogen removal before pumping leachate to WWTP. Hence both economic and environmental benefits can be combined.

References

Wang Y, Pelkonen M et al. (2011). Optimization of landfill leachate management in the aftercare period. Waste Management & Research (submitted)

Modeling Landfill Gas Generation in Landfills Receiving Biodegradable Waste

Use of Field-Scale Data in Modeling Landfill Gas Generation

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Control of landfill gas (LFG) emissions, generated from the decomposition of biodegradable components, is a challenge in sustainable landfilling and requires construction, operation and closure of landfills in such a manner to minimize fugitive emissions. LFG emissions consist of carbon dioxide (CO_2; 40-50% by volume), methane (CH_4; 50-60%), and other trace gases (<1%). Landfill CO_2 emissions are biogenic and do not contribute to climate change.

The generated LFG that is not collected or oxidized is emitted to the atmosphere. Currently many landfill covers are designed to promote methane oxidation potential and reduce the fugitive emissions. LFG oxidation is a function of several factors, including gas loading rate, cover type, cover thickness, and ambient temperature. Various methods have been developed and used to measure fugitive LFG emissions. The flux chamber and tracer gas methods are the most common in LFG emission studies, mainly due to simplicity and cost-effectiveness. Emissions vary according to ambient and in-cell conditions. The US EPA, however, recommends using the optical remote sensing vertical radial plume mapping (VRPM) method to measure fugitive LFG emissions. Uncertainty in landfill gas collection rates are related to cover type, cover conditions, gas well pressure, barometric pressure, precipitation, temperature, and wind conditions.

Methane generation is often estimated using empirical models. For example, LandGEM is a first-order kinetics model created and recommended by the US Environmental Protection Agency (EPA), and is widely used to estimate landfill methane generation (US EPA, 2008; US EPA AP-42, 1997). First-order models are more commonly used because of a higher reliability compared to zero-order models and greater simplicity compared to second-order models (Oonk et al., 1994). From these models, quantities of landfill gas that is collected, emitted or oxidized can be determined as well.

Validation of landfill gas models using field-scale data is important to ensure that the models are accurate and reflect real-world conditions. However these data are not generally available because of challenges related to data collection. These include:

- Spatial and temporal variability of landfill gas emissions, oxidation, and collection rates,
- Expense of field-scale landfill gas data collection,
- Inconsistency in quality and reporting of landfill gas data,
- Proprietary nature of landfill gas data,
- Incomplete landfill gas data sets,
- Uncertainty in waste composition and acceptance rates.

Recent research to determine model paramters using field-scale data from US case-study landfills will be presented. Fugitive emissions have been quantified for three US municipal solid waste case-study landfills, using the VRPM method, by ARCADIS n collaboration with the US EPA in 2009-2010 (USEPA 2012). Additional information has been gathered from case-study landfills and the information has been used to compare measured and modeled landfill gas collection, emission, and oxidation rates for each case-study landfill. Monte Carlo computational method were used to determine uncertainties in these values by generating numerous scenarios using randomly selected values from a specified range and probability distribution for LFG generation, emission, and oxidation.

Oonk, H., Weenk, A., Coops, O., Luning, L. (1994). Validation of landfill gas formation models. TNO Institute of Environmental and Energy Technology Report. Grontmij. US EPA AP-42 (1997). Compilation of air pollutant emission factors, AP-42. Volume I

US EPA (2008). Background information document for updating AP42 Section 2.4 for estimating emissions from municipal solid waste landfills. EPA/600/R-08-116.

US EPA (2012). Quantifying methane abatement efficiency at three municipal solid waste landfills using optical remote sensing technology, EPA/600/R-11/033.

How does landfill aeration impact on leachate composition?

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Introduction

Landfill in situ aeration aims beside both, a reduction in landfill gas (LFG) generation rates as well as LFG generation potential, at an efficient biodegradation of organic compounds and inorganic nitrogen compounds (mainly ammonia-nitrogen) in the leachate. The amount and kinds of organic and inorganic leachate compounds indicate whether or not the aeration enabled a sufficient biological and chemical conversion and/or biodegradation under the altered environmental conditions. Decreasing concentrations of total and dissolved organic carbon (TOC / DOC) and biological oxygen demand (BOD) are indications of organic substances degradation while the reduction of ammonium nitrogen (NH_4^+-N) and the simultaneous appearance of nitrite / nitrate are indications of nitrification processes. If nitrite / nitrate do not remain at elevated levels, subsequent de-nitrification processes can be assumed. However, ammonium nitrogen might also be incorporated into the increasing amount of microbial biomass during aeration or being stripped out as free ammonia (NH_3) via the gas phase. The intensity of these processes is mainly subject to climatic conditions (i.e. the amount of precipitation / infiltration and temperatures of the aerated waste material) as well as the intensity of aeration. When reviewing the results of several investigations both, at lab scale and full scale it becomes apparent that the above mentioned processes occur in a variety of intensities. So far it remains unclear to what extend ammonia volatilisation, nitrification / de-nitrification or nitrogen incorporation into biomass are accounting for the reduction in ammonia-nitrogen concentrations.

Experimental approach and observations

Investigations in simulated bioreactor landfills under intermittent aeration (column tests) showed that both, N_2O and NH_3 emissions are very small (less than 0.19%) in relation to the nitrogen contained in the waste materials. It is assumed that the observed significant reduction in nitrogen mass (up to 61.4%) might be explained by simultaneous nitrification / de-nitrification processes and the subsequent release of N_2 via the off-gases. Through other investigations in simulated MSW bioreactor landfills (column tests) under intermittent aeration it could be demonstrated that up to 90% of the ammonium leachate concentrations are removed under rapidly increasing pH values, reaching final levels of 8.5 to 8.8. Under these conditions air stripping was the primary pathway of ammonia removal (up to 258 mg N/l was detected in the vapour) whereas nitrification was inhibited by the high level of free ammonia. Finally, based on published data from landfill aeration projects as well as composting research results an evaluation of the long term behaviour of nitrogen in aerated MSW landfills has been made. According to these calculations the microbial nitrogen uptake (incorporation) is in the same order of magnitude as the nitrogen release via off-gas and leachate. Consequently the potential long term N-emissions have to be considered in future research as ammonification rates of the biomass-incorporated nitrogen (after completion of aeration) remain unclear so far.

Results

Applying aeration to a landfill has a significant impact on the accordant leachate quality. However, simulation experiments in laboratory scale do often not adequately consider the conditions to arise when aeration is performed at a full scale landfill. One of the major factors to determine the leachate composition during aeration is temperature. Raising temperatures stimulate / intensify both, nitrogen mineralisation rates (ammonification) as well as bioconversion of organic compounds. In consequence pH values are increasing and becoming the major driver of further processes. Due to a shift in the ammonium – ammonia equilibrium (towards free ammonia) nitrification processes are further inhibited and the same applies for microbial bioconversion processes of organic compounds. With the beginning of NH_3-N volatilisation both, organic and nitrogen compounds in the leachate are reduced and with the transition into the long term cooling phase (at reduced microbiological activity) the positive impact of aeration on the leachate composition (quality) becomes apparent. The question of a potential long term release of ammonium nitrogen (incorporated in the microbial biomass) once the aeration has been completed seems to be circumstantial. Investigations in laboratory scale indicate that the nitrogen mineralisation rates (ammonification) after a widely bio-stabilisation are in a range of 30% in comparison with the situation at the start of aeration. In any case leachate concentrations (ammonium nitrogen) are expected to adjust at low levels after completion of aeration.

Open questions:

Although for some of the aspects related to both, landfill aeration and its potential impact on the leachate quality, theories could be provided, there still remain a couple of questions demanding further investigations:

- What is the capacity of water addition and/or leachate recirculation towards temperature regulation during aeration?
- What are the possibilities and constrains to regulate temperatures and processes through application of intermittent aeration concepts?
- What is an accurate estimation of the nitrogen mineralisation rate both, at the beginning and after completion of aeration?
- What about the behaviour of further leachate compounds, like e.g. heavy metals?
- Are we able to set up a complete nitrogen balance for an aerated landfill and would this be of additional value?
- What is in reality (at full scale) happen to the leachate quality in the long term?

Within the session “Status quo and future perspectives of landfill aeration” the questions raised above are going to be subjects of the discussion. Additionally findings and implications from other aeration projects and lab scale tests shall be presented in order to get a broader data basis to draw conclusions upon.

Improvement of waste moisture content during filling phase: consequences on biogas and leachate production

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Introduction

During the last decade, the composition of French solid waste showed a strong evolution. The part represented by commercial and industrial waste (C&I waste) has increased while the municipal waste composed of 25.1 % of organic fraction in 2007 has decreased. Industrial waste are known to be dry (water content of 20 -25 %) compared to organic fraction of municipal waste (water content of 63 %). Therefore the global water content of waste has decreased from 35 % to a range of 22 to 30 % in the last ten years. In addition, many landfills are using HDPE membrane as final cover to limit fugitive emissions. These two elements lead to very low initial waste moisture content that is not favourable to biodegradation.
Several injection systems such as vertical wells, injection blankets or horizontal drains were evaluated through years in order to reinject leachates in landfills and therefore increase the water content. But these techniques step in after the filling of cells and they have shown limits to impact the entire waste due mainly to the heterogeneity.
Based on the difficulty to impact the waste once the filling of the cell is finished, Veolia Environment Research and Innovation (VERI) started to study a technique to humidify waste just after its landfilling in order to impact all the surface of the cell. A research project was started in 2009 to assess the possibility to use this technique on a real scale landfill and to measure the impact on waste characteristics and biodegradation. Several questions are raised and need to be answered before considering a possible deployment:

- Is there a real improvement of water content of the waste with on-site humidification?
- Is there consequences on the ability to extract biogas from landfill?
- Is there consequences on settlements?

This abstract present the strategy employed and the instrumentations implemented and the followed parameters.

Landfill equipment

A full scale landfill receiving 80000 tons/year of waste was equipped and divided in two areas to compare the behaviour of waste managed in a conventional way with waste humidified during filling phase. A hydraulic separation made of vertical HDPE membrane was set up at the bottom of the cell to separately collect leachate from conventional area and from humidified area. Leachate are collected by two drains and then pumped out of the cell to be counted. Rain water entries and evaporation are continuously measured on site by a meteorological station. Biogas is also collected separately. During the first months of filling gas was collected via the bottom drainage material with two drains. Later, two horizontal drains were added to complete the system. Biogas composition is continuously analysed. Pressure, gas flow and temperature of the gas are also measured. On the humidified area, 22 m^3 of leachate is spread every day on the surface of the cell.
At the end of the filling, a drilling campaign is planned in order to excavate waste in both areas and compare their water content and their degradation state.

Results

Water balance
The water content of excavated waste will be measured and compared between conventional area and daily humidified one. Excavated water content values are also compared with the initial water content of waste.
The measured water content will also be compared with an estimated water content based on the measurement of all water entries and exits during the landfill. This should allow us to calculate the volume of leachate that has been absorbed by waste and therefore the absorption capacity of waste. The variability between different depths and different drillings will also give us an indication on the ability of the humidification technique to homogeneously influence waste water content.
Gas balance
The biogas production of the two areas will be compared in order to detect the effect of humidification on the start of methanogenesis. The BMP of excavated waste of both areas will also be compared Moreover, the quantity of biogas collected will be related with the decrease of the biological methane potential (BMP) of the excavated waste compared to initial BMP in order to evaluate the efficiency of the gas collection system. Finally, comparative tests will be practiced to determine if the same depression enable to collect the same gas flow between the conventional area and the humidified area.
VERI is currently working on the management of the filling phase of the landfill. It is considered as a mean to increase the initial moisture content. After the filling, complementary injection systems should be considered to maintain the convenient moisture content and therefore optimize the biodegradation during operating period of the landfill. Only an acute reflection about the management of moisture during the period of landfill management will allow complete waste degradation.

Heterogeneous emission from a biocover designed for methane oxidation

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Introduction and Study Aims

Methane oxidation in landfill cover soils is a promising technique to reduce emissions from landfills with low gas product ion. Preferential landfill gas (LFG) flow, however, often leads to locally high emissions. Here, the bottom CH_4 flux exceeds the oxidation capacity of the soil and impairs the necessary ingress of atmospheric air from the top. On Wieringermeer landfill in The Netherlands two test fields (510 m² each) have been designed to validate CH_4 oxidation as a tool for the mitigation of residual CH_4 emissions. The construction was intended to achieve homogeneity in soil physical conditions and correspondingly homogeneous gas flux over the entire area of the test fields. Thereby it should allow for an even LFG flow to the cover, adequate atmospheric air ingress and a CH_4 load below oxidation capacity. However, distinct emissive areas were identified by repeated FID screenings. The magnitude of emissions within these "hot areas" varied spatially. In order to capture the emissions in a representative manner, a chamber covering a large surface of 17.6 m² was built. The aim of the study presented here was (1) to characterize the spatial patterns of emissions from the individual test fields and relate them to the constructive properties of the test fields and to (2) establish a CH_4 and a total C-balance using the known inlet flux and the emitted fluxes.

Study Site and Methods

Test field setup. The test fields differ in their degree of compaction, and their gas distribution system. Test field C (capillary barrier, soil placed with long stick excavator, soil bulk density 1.34 g cm^{-3}, air capacity 21 vol.-%) and test field G (soil built in with a bulldozer, lined by a drainage mat, soil bulk density 1.50 g cm^{-3}, air capacity 12 vol.-%) are built on a 1:5 slope, sealed from the landfill and surrounding cover with a HDPE liner. Landfill gas (47% CH_4, 33% CO_2 in 11/2011, load to the cover 2.5 L CH_4 m^{-2} h^{-1} since 08/2011) is introduced at each test field's base at six points. Test fields C and G are bounded by 40 cm and 15 cm vertical HDPE profiles beyond which a boundary zone extends with lower soil compaction, resulting from an excavation. In order to enable systematic and repeatable FID screenings and emission measurements, a 4.25 × 4.25 m grid was established on the fields and permanently marked with pegs. *Sampling design.* The data presented here was generated on two consecutive days in 11/2011. The following criteria were applied to select the grid fields to be measured in 4-weekly intervals: (1) all grid fields on which the FID-screening performed directly prior to the chamber measurements revealed CH_4 emission, (2) 4 designated grid fields repeatedly measured in every campaign (to allow for comparability) and (3) 4 grid fields that show zero CH_4 emission. In order establish a carbon balance, the CO_2 emission of the zero CH_4 emission grid fields is extrapolated to those grid fields without FID-detected CH_4 that were not

measured. *Chamber measurements*. The chamber (area 17.6 m², volume 8.8 m³) was constructed of aluminum beams, covered with an aluminum coated plastic foil. A tarp extends from each side, and was weighted down with roof battens to seal the chamber volume against the outside atmosphere. The chamber atmosphere was sampled continuously through a tube with 18 evenly distributed sampling tubes of equal length using a portable FID (TVA 1000, Thermo), to measure CH_4 concentration increase (Measurement duration 6 min., 15 s intervals). CO_2 concentrations were measured with a non-dispersive IR sensor (IAQ-CALCTM Model 7525, TSI Inc.) placed under the chamber. To ensure homogenization of the chamber volume during measurement, two fans were fixed to the cross struts.

First Results

The FID screening on which the emission sampling was based suggested that CH_4 emitted in a spatially heterogeneous manner. The large part of CH_4 emitted from the upslope fringe of test field C and from the boundary zones of test field G (Figure 1, bottom is upslope). The grid fields selected for their "zero CH_4 emission" emitted considerable amounts of CO_2, suggesting CH_4 oxidation occurring here. The CH_4 balance suggested an oxidation efficiency of 86 % (C) and 91 % (G), asssuming that the entire area emitting CH_4 had been covered. Total C-emission (measured CH_4 and CO_2 plus extrapolated CO_2) amounted to only 33 % (C) and 12 % (G) of the carbon introduced. Thus, the C-balance gap of C is 67 % and G 88 %.

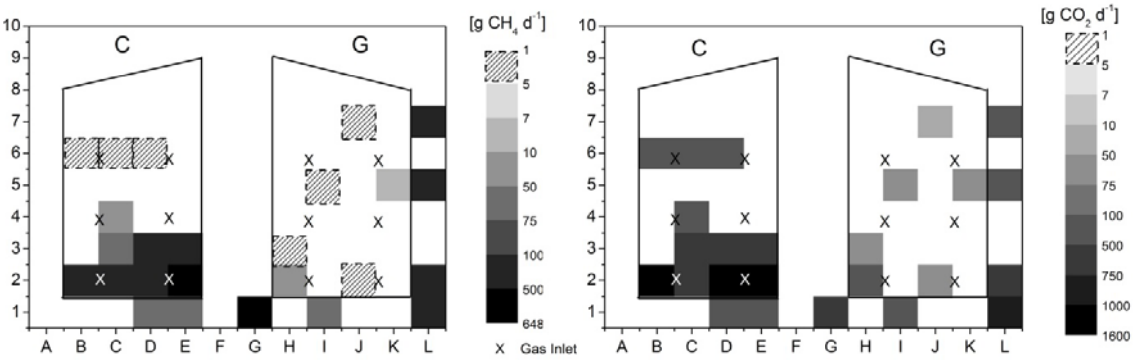

Figure 1: Methane (right) and carbon dioxide (left) emission from two test fields (area of 510 m² each, marked by trapezoid lines). Each chamber measurement covered one grid field (A1-L10, 4.2 x 4.2 m). Sampling design was based on prior FID screening. White areas: no elevated surface concentration, thus no emission measurement conducted. CH_4 load: 2.5 L CH_4 m^{-2} h^{-1}. Row 1 is upslope, row 10 downslope. Scheme is north oriented.

Conclusions

The gap in the carbon balance suggests that the extrapolation underestimates CO_2 emission. Sampling a larger number of "zero CH_4 emission" grid fields will help to close the carbon balance and further clarify the spatial emission patterns. The test field C emission pattern can be understood as an effect of LFG migrating up the slope underneath a water-saturated seam within the capillary barrier, first breaking upslope, allowing gas to migrate through the water-free pores at the capillary block/layer-interface. The compaction of test field G resulted in a low water-free pore volume for gas to migrate in. As a result, LFG escapes through the less compacted boundary zones. Due to the heterogeneous distribution of the gas load, which was observed in spite of the endeavor to construct a homogeneous biocover, the load at these specific locations exceeded the existing CH_4 oxidation capacity in the soil and lead to emissions.

Mistakes in construction of a controlled field trial - and how they were (not yet) fixed

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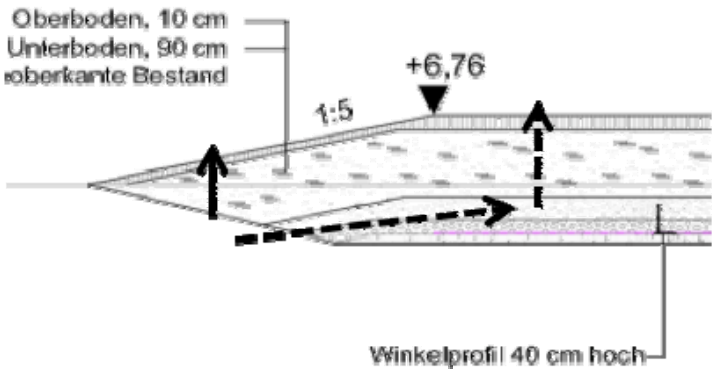
In the summer of 2009 Afvalzorg constructed two testfields for research of methane oxidation in cover soil. The research is carried out with support of Agentschap NL (programme Reduction of Other Greenhouse Gases), the Institute of Soil Science of the University of Hamburg and the Technical University of Hamburg-Harburg in the framework of the MiMethox project (www.mimethox.de). The melchior + wittpohl Ingenieurgesellschaft (Hamburg) is involved for water household issues.

The testfields were constructed on a membrane. This enables catchment and measurement of water flow through the cover soil. The membrane blocks the 'natural gas supply' from the landfill. The design included a boundary zone of 5 m. The membrane continued more than 3 m outside the testfields. Gas from the landfill was anticipated to migrate 0.7 m vertically rather than migrate horizontally for 5 m. In the first months with low methane loading vegetation damage in the boundary zone indicated migration from the landfill. The boundary zone had to be dug up and the membrane was extended up to the top of the recultivation layer.



In order to 'evenly feed' the soil microorganisms it is important to evenly distribute the landfill gas over the entire surface. A gas distribution layer is required. Two types were installed. One was the 20 cm gravel layer of a capillary barrier. The other was a rainwater drainage mat. Although the utmost was done to construct the different layers in both testfields as homogeneous as possible, there was no even gas distribution. A capillary effect between soil and geotextile of the drainagemat caused a water film. The occurrence of this water film first became apparent due to leaching of iron indicating anaerobic conditions in the soil. A test excavation down to the drainagemat verified the occurrence of a water-saturated layer of several cm. The water-saturated layer obstructs even gas distribution. Investigation of gas distribution was intensified. Ideas to 'save' this testfield are being developed.

Concluding remarks

Engineers, contractors and supervisors are human beings. No human being is free of mistakes. Checking work of human beings is a must. Double checking is fine. Pay extra attention to aspects that can no longer be reached after construction. Triple checking will make a lot of work impossible or unaffordable. Allow for mistakes to be made in terms of extra time and money to fix the unavoidable mistakes.

Why trace elements are often immobilized in ashes and slags. On the role of solid solution in iron (hydr)oxides.

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Objective, scope and main conclusions

The objective of the work was to determine the role of solid solution in iron (hydr)oxides for the immobilization of trace elements in ashes and slags.

The scope of the work included the following:

- To study the availability of elements to the water phase for different ageing times
- To search and analyse literature
- To make thermodynamic calculations which include the features of solid solubility and aqueous solution

Iron (hydr)oxides are of paramount importance for efficient long-term immobilization of many transition elements and other elements in ashes and slags.

Results and conclusions

The major elements in ashes and slags form phases in which they are major elements. The minor elements do not usually form such phases. Instead, they become incorporated - atom by atom - in the crystal lattices of the phases formed by the major elements. This makes the minor elements as inaccessible as the major elements into which they have been incorporated.

Iron-rich phases are well known scavengers for many transition elements and other elements. Iron- and manganese (hydr)oxides are largely responsible for continually cleaning up the sea. The reason for this is that the trace elements go into solid solution. (This has been known for decades among inorganic chemists, geochemists and geologists).

The low content of transition and heavy elements in the sea cannot be explained by thermodynamic calculations in which it is assumed that the trace elements form phases in which they are major elements. Nonetheless, such assumptions are often used in the areas of waste and environmental impact assessment. In fact, commercial programmes are not available that simultaneously include the features of solid solution and aqueous solution.

Such calculations have been performed in the present work and they clearly show a strong effect for Zn. Scavenging effects have also been observed experimentally. The above conclusions are supported also by the literature search and analysis.

Solid solution in iron-rich phases presupposes a surplus of iron relative to the trace element in question. This condition is usually met in ashes and slags. Cr-III is stabilized even at high pH conditions.

Leach tests

Leach tests were carried out on four ashes moistened at two different levels, and aged at three different times without access to air. The leach rates of most elements decreased very substantially, and e. g. for zinc, the leaching decreased with up to three orders of magnitude compared to fresh fly ash. Leaching of zinc is frequently below what may be expected from its oxide. The observations thus support a conclusion that the trace elements are immobilized through solid solution.

Literature survey

In general, the conditions for substitution in phases rich in iron and manganese are as follows:

1 The charge of the replacing atom should not differ by more than one unit.

2 The effective radius of the replacing atom should not differ by more than ± 15 %.

Effective ionic radii for various elements are tabulated in the literature. Such values in combination with the requirements 1 and 2 above clearly indicate that many elements have a strong tendency to form solid solutions with iron.

These effects take place at high as well as at low temperature. They are observed in soil, and it was found that ash and slag behave in the same way as soil. It has been reported in the literature that iron-rich phases in ash are enriched with other elements of similar ionic radii.

Thermodynamic calculations

Thermodynamic calculations were conducted for the system $(Zn_xFe_{1-x})Fe_2O_4$, and for various values for x. The end points x=0 and x=1 correspond to magnetite (Fe_3O_4) and franklinite ($ZnFe_2O_4$), respectively. Magnetite and franklinite both occur in nature. Magnetite is a common mineral while franklinite is rare.

The results show that the availability for zinc in the pore water is drastically influenced by the formation of a solid solution. Moreover, the effect is larger, the lower the total concentration of zinc. This concentration behaviour is quite different from that of when a trace element is assumed to be a major element in a phase of low abundance.

References

1 Sjöblom, R. and Noläng, B., *On the significance of solid solution in iron (hydr)oxides for immobilisation of potentially polluting elements in ashes*. Proceedings of Ash Utilisation 2012, Ashes in a Sustainable Society, Värmeforsk (the Swedish Thermal Engineering Research Association), January 25-27, 2012, Stockholm, Sweden.

2 Sjöblom, R. and Noläng, B., *The significance of solid solution in iron (hydr)oxides for stabilization of elements potentially harmful to health and environment*. (Swedish title: *Betydelsen av fast löslighet i järn(hydr)oxider för fastläggning av potentiellt miljöstörande ämnen i askor*). Värmeforsk (the Swedish Thermal Engineering Research Association), Report 1198, November, 2011.

3 Sjöblom, R., *Pertinent methodology for basic characterization of ash for acceptance for landfilling.* (Swedish title: *Lämplig metodik för grundläggande karakterisering av aska för acceptans på deponi*). Avfall Sverige (Swedish Waste Management), Report U2011:22, October, 2011.

Landfill aeration worldwide: Concepts, indications and findings

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Introduction

Landfill aeration as a methodology for the fast, controlled and sustainable conversion of landfills into a biological stabilised state attracts attention worldwide. In many cases landfill aeration can be considered the only technically and economically feasible method to significantly reduce the aftercare both, by time and complexity. In other cases aeration contributes towards the long term reduction of greenhouse gas and leachate emissions (semi-aerobic concept) or is applied for the fast recovery of landfill volume (aerated bioreactor concept). Numerous studies in laboratory scale have been conducted over the past two decades and full scale examples are increasing by number in particular since 15 years ago. However, although comprehensive data has been published until today, there is a great demand for systematisation of the results in order to gain generally admitted indications.

Despite the fact that landfill aeration is a more recent concept it has already been successfully applied to several landfills in Europe, North America and Asia. Some of the most comprehensive investigations have been made in Germany and, due to the high interest in this technology; more investigations are underway in different European countries (e.g. The Netherlands) nowadays. This is certainly essential as in-situ aeration stands for more than just injecting air into a landfill. Aspects as e.g. well design and spacing, selection of appropriate air volume and pressure, control of air distribution, temperature and moisture control as well as potential pollutants mobilisation into both, gas and liquid phase have to be considered. Finally, the question of a particular point in time to terminate the aeration process has to be answered. The latter includes indications about the biological landfill stability achieved during aeration and leads back to the initial consideration on sustainability. Thus, not only technical and ecological aspects have to be considered but also economic issues.

Landfill aeration concepts

The term “landfill aeration” comprises a variety of different concepts and methodologies. Over the past decades many of these concepts have proven to be applicable at full scale, even though in the majority of cases a clear proof of success has not been provided. The reasons for this are manifold and include, beside others, a lack of general parameters and target values for the successful completion of aeration as well as the application of insufficient monitoring programs during aeration. Furthermore, the aeration of landfills has to follow a number of very specific objectives in accordance with the local situation, framework requirements and the specific problems. Table 1 provides an overview of different landfill aeration concepts and their specific areas of application.

Table 1: Overview of different landfill aeration concepts, application areas and related off-gas treatment methods.

Aeration concept	Sub-concept	Main area of application	Off-gas treatment method
high pressure	-	in preparation for landfill mining	biofilter
low pressure	active aeration & off-gas extraction	accelerated bio-stabilisation and GHG emissions minimisation	thermal oxidation (subsequently biofilter)
	active aeration w/o off-gas extraction	accelerated bio-stabilisation	biocover (landfill surface)
	passive aeration (air venting)	increasing gas extraction rates / subsequent aerobic biostabilisation	biofilter
	energy self-sufficient long term aeration	avoidance of long term gas generation	none / landfill surface
semi-aerobic	connection between leachate collection pipes and gas vents included in the design layout	long term reduction of GHG emissions & improvement of leachate quality	none
	subsequent installation of passive gas vents (w/o connection to leachate pipes)	landfill remediation	none

Findings and conclusions

Landfill aeration as a methodology for the fast, controlled and sustainable conversion of landfills into a biological stabilised state attracts attention worldwide. In many cases landfill aeration can be considered the only technically and economically feasible method to significantly reduce the aftercare both, by time and complexity. Numerous studies in laboratory scale have been conducted over the past two decades and full scale examples are increasing by number since 10 years ago. However, although comprehensive data has been published until today there is a great demand for a systematisation of the results in order to gain overall indications. Against this background the International Waste Working Group (IWWG) has set up a task group on the topic of landfill aeration to bring together international perspectives and expertise in the area in order to better define the technology and to disseminate experiences to the public. Currently the group is focussing on the compilation of an overview on the status of landfill aeration worldwide, definitions of landfill aeration methods, stabilization and quality criteria for landfill aeration as well as the set-up of a database. The latter is of special interest in connection with the intended systematisation approach.

Furthermore it becomes apparent that there is still a lack of understanding regarding specific processes to occur during aeration (e.g. the nitrogen and temperature dynamic under full scale conditions) and a deficit in transparency regarding costs. Obviously the great variety of landfill aeration concepts and the increasing number of realised aeration projects should be used as a chance and basis for a broad discussion and exchange among experts in this field. Based on the existing experiences potential combinations of concepts might be further developed, such as e.g. the inclusion of wind driven aspirators in the semi-aerobic landfill concept, the application of deep-filtered gas wells for active aeration, intermittent operation in low pressure concepts as well as the adjustment of temperatures by a controlled addition of water or leachate.

Stable isotope fractionation during microbial methane oxidation – new insights from gas push-pull tests

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Introduction

Quantitative assessment of microbial methane oxidation in landfill cover soils is an important task for verification and optimisation of biocover performance.
The method of stable isotope analysis is based on the fact that methane containing the lighter isotope ^{12}C is consumed at a higher rate than methane containing the heavier ^{13}C. Theoretically, the fraction of methane oxidised can be assessed from the ratio of the stable carbon isotopes ^{13}C and ^{12}C following Equation 1:

$$f_{OX} = \frac{\delta_{out} - \delta_{in}}{(\alpha_{ox} - \alpha_{trans}) \cdot 1000} \quad \textbf{(1)}$$

where f_{OX} is the fraction of methane oxidised, δ_{in} and δ_{out} are the carbon isotope ratios of the methane going into and out of the system, respectively, given in the Vienna Peedee Belemnite expression, α_{ox} is the isotopic fractionation factor associated with microbial oxidation, and α_{trans} is the isotopic fractionation associated with (diffusive) gas transport. The latter effects are determined by the concentration gradient and the effective diffusion coefficient of the soil and proved to be significant even in systems where mass transport is controlled by advection (Gebert et al, 2011). The oxidative fractionation factor is influenced by various parameters, including temperature, methane concentration, turnover rate and composition of the microbial community. Therefore, the transfer of fractionation factors determined in the laboratory to biocovers in the field is – at least – problematic.
The method of Gas Push-Pull Tests (GPPTs) can be used to measure potential methane oxidation rates in the field by injection of a mixture of methane, air and argon as a non-reactive tracer into the soil and subsequent extraction and analysis of the soil gas mixture (Streese-Kleeberg et al., 2011). Besides the methane oxidation rate, the fraction of methane oxidised can be calculated from the GPPT results. Thus, the term (α_{ox} - α_{trans}) can be determined directly in the field when stable isotope measurements are conducted during a GPPT.

Methods

During the evaluation of a GPPT, a mixing factor is calculated from the ratio of the current to the initially injected tracer concentrations, each of which is corrected for the background concentration:

$$f = (C_{tr} - C_{tr}^{b}) / (C_{tr}^{0} - C_{tr}^{b}) \quad \textbf{(2)}$$

with: f: mixing factor; C_{tr} : current tracer concentration; C_{tr}^{b} : background tracer concentration; C_{tr}^{0} : injected tracer concentration.

Without methane oxidation, the methane concentration would be the product of the mixing factor and the initial methane concentration. From the ratio of the actual methane concentra-

tion to the methane concentration that would be expected from mixing and dispersion only, the fraction of oxidised methane can be calculated:

$$f_{OX} = 1 - \frac{C_m - C_m^b}{f \cdot \left(C_m^0 - C_m^b\right)} \quad \textbf{(3)}$$

with: f_{OX}: oxidised fraction of methane; C_m: current methane concentration; C_m^b: background methane concentration; C_m^0: injected methane concentration; f: mixing factor, see Eq. (2).

First Results

During a measurement campaign at five old landfills in Germany, samples for stable isotope measurements were taken during 27 GPPTs. Oxidation efficiencies ranged between 0.03% and 94.92%, corresponding to 2.3 and 91.3 g CH_4 $m^{-3}_{soil\ air}$ h^{-1}. Given the different molecular masses of the methane isotopologues, α_{trans} was assumed to be 1.0195. By resolving Eq. (1) for α_{ox}, the apparent (in situ) oxidative fractionation factors were calculated for all data points (Figure 1).

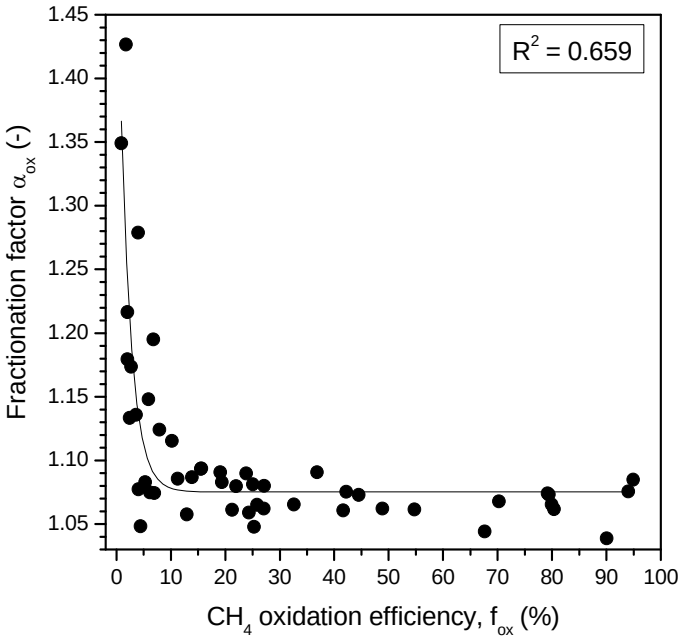

Figure 1: Relationship between CH_4 oxidation efficiency (f_{ox}) and apparent oxidative fractionation factor α_{ox}. Line = exponential fit.

Large variation of α_{ox} was found for oxidation efficiencies below 15%, with α_{ox} assuming values between 1.05 and 1.43. As also transport fractionation is neither negligible nor constant, depending on physical soil properties, the quantification of methane oxidation using the stable isotope approach is subject to significant uncertainty. The resulting limitations with respect to method applicability have yet to be defined.

References

Streese-Kleeberg J, Rachor I, Gebert J and Stegmann R (2011): Use of gas push-pull tests for the measurement of methane oxidation in different landfill cover soils. Waste Management 31, 995-1001.

Gebert, J.; Streblow, C.; Knoblauch, C.; Pfeiffer, E.-M. (2011): Effect of gas transport on fractionation of carbon stable isotopes related to the microbial oxidation of methane in landfill cover soils. Proceedings Sardinia 2011, 13th International Waste Management and Landfill Symposium

Role of mobile sorbent nanoparticles in transport of toxic metals in clay mineral barriers

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Introduction

The most widely used strategy for containing toxic substances found in leachate is through the use of low hydraulic conductivity clay liners. Initial models of such systems were largely predicated on solute transport models based on batch sorption data, in which species were defined as partitioned between liquid and solid phases. It is apparent, however, that colloidal materials (ranging from 1nm-1μm) also play an important role in transport of dissolved organic matter and metals.

The structural diversity and high concentration of functional groups present in colloids (or mobile sorbent nanoparticles (MSNs) mean that they can interact with components of both solid and liquid phases to facilitate pollutant transport. The major physicochemical interactive processes between metals in solution and immobile solid phases (e.g. sorption, partitioning, speciation, ion-exchange) can be directly influenced by interaction with dissolved organic carbon (DOC) and MSNs, due to processes such as complexation, solubilization, carrier association and the solvophobic effect. It is now recognized that colloids are important in facilitating transport of strongly sorbed trace metal and organic contaminants further and faster than predicted from the batch-sorption based, solute transport models that are commonly employed to describe these environments. Conversely, MSNs may also lead to "cumulative sorption", whereby colloids contribute to an increase of the organic carbon content of the solid phase, thus increasing potential metal binding sites and sorption. Colloids may also retard metal transport by plugging pore constrictions.

Organic matter (OM) comprises the major component of MSNs isolated from landfill leachate, of which humic substances (HS) constitute the greatest proportion (40-90%). The nature of OM in leachates varies with landfill age, becoming more dilute and less biodegradable with time, with corresponding increases in the aromaticity of the HS. The EU Landfill Directive 99/31/EC will lead to further changes in the physicochemical properties of leachate OM as landfill management practices adapt due to the requirement of a phased reduction in the proportion of biodegradable municipal solid waste (MSW) going to landfill. The impact of dilution of MSW leachate and the influences of the new Landfill Directive compliant practices may profoundly affect the amount and nature of DOM and MSNs found

in the leachates, leading to the mobilization of contaminants previously stored within the barrier system, for example through the lowering of the ionic strength of the leachate below the critical salt concentration, which has been shown to lead to the release of contaminant laden particles.

Experimental work and modelling

In evaluating the impact of MSNs on toxic metal transport in landfills and contaminated land, there is a need for more quantitative data that can be used to develop generic models to describe and predict the transport of potentially toxic species in these environments. This will require study of MSNs derived from landfill leachates of different ages and management practices, together with experimental study of commercially manufactured MSNs of defined size, charge and concentration under simulated landfill conditions. Size, charge and concentration are the key attributes influencing straining and filtering of MSNs by clay liner materials and are parameters that can be manipulated through the use of manufactured microspheres. Quantification of interactions between manufactured nanoparticles and clay liners thus permits elucidation of the effects of these parameters on the potential for MSNs to facilitate or prevent cross-barrier transport of contaminants, therefore providing a means through which containment failure can be more accurately predicted. The data derived from these studies are an absolute pre-requisite for developing the models required to make accurate and precise predictions of toxic metal behaviour in contaminated sites, and thus develop robust and cost-effective strategies for ensuring compliance of landfill practice with legislative requirements, in particular those set out in EU Groundwater Directive 80/68/EEC and the new Groundwater Daughter Directive (2006/118/EC).

Environmental Characterization of "Salt Cake": A Secondary Aluminum Processing Waste

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Introduction and Research Objectives

Aluminum is one of the most utilized metals in a wide range of products (Green, 2007). Aluminum is being recycled into new products through a series of processes that re-melt aluminum to produce aluminum ingots at secondary aluminum processing (SAP) facilities. Solid residues generated by the aluminum melting processes are often referred as aluminum dross which is formed on the surface of molten aluminum by contact with oxygen during melting in a rotary salt furnace. The nonmetallic byproduct residue is frequently termed "salt cake", and is traditionally being disposed of in landfills (Schlesinger, 2007). Almost one million tons of salt cake is landfilled annually in the United States (USDOE, 1999). The many forms of aluminum contained in salt cake (i.e., metallic Al, aluminum nitride (AlN), aluminum carbide (Al_4C_3), aluminum sulfide (Al_2S_3), and aluminum oxynitride (Al_5O_6N)) can undergo a wide range of potential reactions after interacting with water. These reactions are generally exothermic (produce heat) and result in the generation of gases, some of which are flammable (e.g., hydrogen and methane) (European commission, 1991). Thus, there are potential dangers with the disposal of salt cake in an environment where water may be present. Research is needed to better understand salt cake waste material, its potential risks, and strategies for safe management in municipal solid waste (MSW) landfills. Therefore, the current study aimed at 1) characterizing the metal constituents as well as the mineral phases in the salt cake, 2) investigating the metal leachability from salt cake after its reaction with water, and 3) examining the temperature changes, the quantity and composition of the gas generated as a result of the reaction of salt cake with water.

Materials and Methods

To meet the study objectives, 38 salt cake samples were collected and shipped from 10 SAP facilities across the U.S. The facilities were identified by the U.S. Aluminum Association to cover a wide range of processes. Upon receipt, the samples were crushed and sieved to obtain < 2 mm and 0.05 mm particle sizes to be utilized for the physical and chemical analysis. The mineral phases of the salt cake (< 0.05 mm) were determined using X-ray diffraction (XRD) analysis. The total metals concentrations in the salt cake as well as in the salt cake leachate were determined using the U.S. EPA acid digestion Methods 3051A/3015 followed by Atomic absorption spectroscopy analysis.
To obtain the temperature profile after reaction with water, 10 gm of the < 2 mm samples were placed in 60 ml vials (connected to 1 L Tedlar bag for pressure relief) and the vials were purged with argon for 10 min and then preheated to either 37^0C or 50^0C. A 10 mL of preheated DI water was added to the vials and the temperature change profile was monitored over time using thermocouple wires placed in the samples. For the determination of gas quantity and composition, similar setup with slight modifications was used. The quantity of gas generated was measured using a gas tight syringe and the gas composition was measured

using an Agilent Gas Chromatograph.

Results

The XRD analysis showed that the dominant minerals in the salt cake were metallic aluminum (Al), aluminum oxide (Al_2O_3), aluminum nitride (AlN), potassium chloride (KCl), sodium chloride (NaCl), and magnesium aluminum oxide (Al_2MgO_4). The average content of total Al in the salt cake was approximately 14.2%. Other metals (Na, K, Mg, Ca, and Fe) and trace metals (As, Cd, Cr, Cu, Pb, Mn, Se, and Zn) were also detected in the samples (data not shown). Significant positive correlations were found between the concentrations of trace metals and the aluminum content in the salt cake. Leachable concentrations of aluminum were detected in all 38 leachate samples (average percentage of the leachable Al was 0.6% of the total Al). Other trace metals were also detected in the leachate (< 0.1% to 2% of their respective total concentration).
The results indicated that environmental temperature, mass of salt cake, and particle size can significantly influence the value of these two key descriptors. The change of temperature profile was sample specific. A temperature increase was observed in all 38 salt cake samples analyzed. On average, the maximum temperature increase (ΔT_{max}) for both reaction temperatures (37 °C and 50 °C) were 22.1 and 28.6 °C, respectively. The average time to reach the highest temperature for reaction temperatures of 37 °C and 50 °C were 1.6 and 0.9 hrs, respectively. The chemical composition of salt cake played a key role for the temperature change. Furthermore, a significant positive correlation between the ΔT_{max} and the content of metallic Al in salt cake was also observed. Hydrogen, methane and ammonia were detected in the gas phase with H_2 being the dominant one. It was also confirmed that there was a positive correlation between AlN content in the salt cake and the production of total ammonia.

Conclusions

The major conclusions to date are: 1) metallic aluminum in salt cake does react with water, 2) AlN and potentially Al4C3 trigger the reaction of salt cake with water, and 3) there is a potential for an incomplete hydrolysis of metallic Al and AlN in salt cake, which may react at a later time given preferable environmental conditions.

References

Green, J. A. S., *Aluminum Recycling and Processing for Energy Conservation and Sustainability*. ASM International: Materials Park, OH, 2007; p 288.

Schlesinger, M. E., *Aluminum Recycling*. CRC Press: Boca Raton, FL, 2007.

USDOE *Recycling of aluminum dross/salt cake*; Office of industrial technologies,Energy Efficiency and Renewable Energy, U.S. Department of Energy: 1999; p 2.

European-Commission, Council Directive of 12. December 1991 on hazardous waste (91/689/EEC), ANNEX III: Properties of wastes which render them hazardous. In *L0689-EN-22.07.1994-001.001-10*, European-Commission, Ed. Official Journal of the European Communities, 1991; p 11.

Quantification of (bio)geochemical heterogeneous activity in full-scale landfills

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Introduction

The main focus of this paper is the assessment of landfill leachate/gas emissions and long-term prediction of these emissions. Due to variation in environmental conditions, like rainfall events, leachate concentrations are very dynamic on a short-time scale. The measured concentrations in the leachate oscillate between maximum and minimum concentrations. Our assumption is that the maximum concentrations occur under slow flow conditions when the leachate is in equilibrium with the relatively immobile water fraction present in the bulk of the waste. The minimum concentrations are the result of strong dilution during heavy rainfall events, where the bulk of the waste is by-passed due to preferential flow.. For accurate data-analysis and further modeling it is important to obtain a good estimation of these two types of concentrations. With these two bounding concentration values the effectiveness of natural and induced landfill stabilization (by recirculation or aeration) can be monitored and assessed. In addition we aim to develop a method to predict long-term emissions in which these bounding concentration values play an important role.

Research question

Until now prediction of long-term emissions are made by extrapolation of empirical relations with a high degree of uncertainty. These predictions cannot be used for assessment of landfill after care. More accurate predictions can be made when models are based on the landfill processes responsible for these emissions. But the problem for a process-based modeling approach is that landfills are considered as black boxes. It is known that important processes, influencing emissions, are biological degradation, speciation/precipitation, adsorption/leaching, diffusive/convective transport (influenced by preferential flow). But information on these processes can only be obtained through emission and waste sample measurements. The key research question is to model the processes (opening the black box) in such a way that calculated emission data is comparable to measured emission data. Once these processes are modeled sufficiently accurate, the model can be extrapolated for estimation of future emissions. Our hypothesis is that the processes in a landfill, due to the heterogeneity, have to be modeled by a stochastic approach. This way every process has its own probability of occurrence. The combination of process probabilities will result in an emission pattern specific for each landfill. This stochastic modeling approach is illustrated in more detail and is presented in Figure 1.
In order to model its heterogeneity in processes the landfill is divided in many cells or sub-models. Each cell represents a particular state of the landfill. For example, one cell can be dominated by organic acid fermentation in combination with diffusive transport while another cell is dominated by aerobic (bio)degradation in combination with convective transport. Parameters to model such particular cell/states (like pH, Eh, DOC, flow etc.) are obtained from databases like LeachXS or experiments. All these cells are combined in a stochastic

model with probabilities assigned to each cell. This lumped model will yield a emission pattern specific for its cell probability distribution. To obtain this cell probability distribution for a landfill, model parameters are fitted such that the calculated emissions correlate well with measured baseline emissions. If a good correlation between the measured and calculated emissions exists, this will indicate that the modeled processes and their probabilities are a representation of the processes occurring in the landfill. Then future emissions can be predicted with this stochastic model by process-based extrapolation.

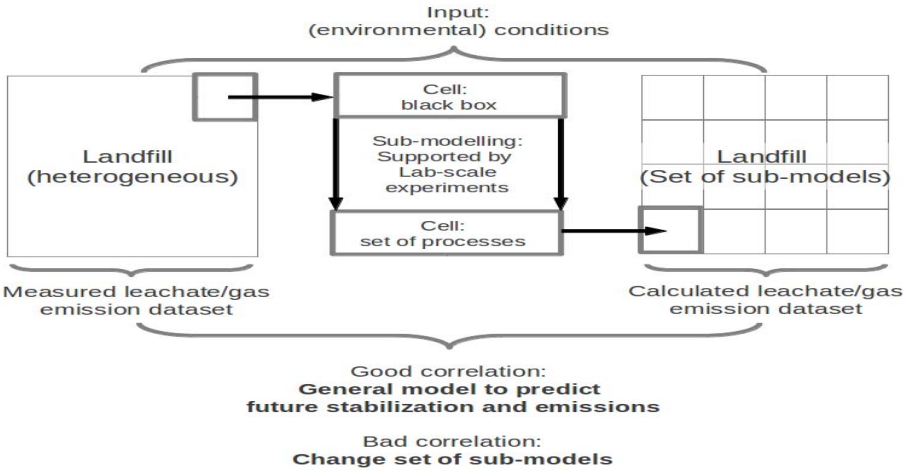

Figure1: Representation of a heterogeneous landfill by a set of cells (sub-models). Good correlation of the calculated and measured leachate/gas emissions proves a good description of the dominant processes in the landfill.

Within each cell (sub-model) Java-based modeling programs, each specialized to model certain processes, are coupled. For instance, geochemical equilibria in cells can be calculated very accurate with ORCHESTRA, while transport or biological degradation is better simulated with COMSOL. MATLAB serves as a general platform to transform and exchange the simulated data between the different programs for each time step. A schematic representation of this coupling is represented in Figure 2.

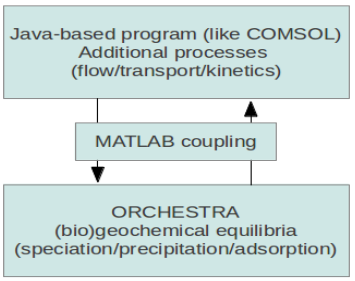

Figure 2: Coupling scheme of Java-based modelling programs for a sub-model.

Preliminary results of sub-models, databases and experiments will be presented during the congress. For example a simple sub-model (cell) was constructed representing glucose (batch) fermentation buffered with calcium carbonate. In this model biological degradation was simulated with MATLAB coupled to geochemical equilibrium calculations in ORCHESTRA.

Modernizing Methane Generation Models and Data from U.S. Landfills

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Introduction

The amount of municipal refuse generated in the U.S. is estimated at 413 million tons, of which 64.5% is disposed by landfill, while recycling, composting and combustion account for the balance (Arsova et al., 2008). Based on present trends and the economics of waste management in most of the U.S., reliance on landfills is likely for the foreseeable future. The U.S. EPA's Landfill Gas Emissions Model (LandGEM) is a simple yet powerful tool for predicting methane production from landfills and it is used heavily at many levels (e.g., site-specific, statewide). In LandGEM, gas generation is modeled using a first-order decay equation as presented in Equation 1 (U.S. EPA, 2005):

$$Q_n = k \cdot L_0 \cdot \sum_{i=0}^{n} \sum_{j=0.0}^{0.9} \frac{M_i}{10} \cdot e^{-k \cdot t_{i,j}} \quad , \qquad (1)$$

where Q_n is the CH_4 generation rate (m^3 yr^{-1}) in year n; k is the first-order waste decay rate (yr^{-1}); L_o is the CH_4 generation potential (m^3 Mg^{-1} wet waste [Mg = metric ton]); M_i is waste mass placement in year i (Mg); and t is time (yr).

Methods for Data Collection and Analysis

The overall objective of this research is to provide an improved data set and model to predict methane production from U.S. landfills. The ability to predict landfill methane generation is important for several reasons including its use to evaluate the feasibility of energy projects, its role in national methane emissions inventories, and its use to guide capital investments in energy recovery equipment. Specifically, predictive models are needed to determine the number and size of engines for which there will be sufficient gas for energy recovery. The current default parameters that are used in LandGEM rely on data that were collected over ten years ago and reflect landfill management practices (e.g., waste composition, daily cover, gas collection) that are representative of the 1990s (U.S. EPA, 1998). In addition to an improved understanding of gas collection, there have been changes in waste composition over the past decade as increasing quantities of fiber are recycled.

Field-scale data from about 13 landfills will be included. As of November, 2011, we have obtained data for 11 landfills including:

- gas collection and composition data for a period of 5 to 10 years. Sites were only selected for inclusion if data were available for at least 5 years;
- data on the mass of waste disposed and the fraction of this mass that is biodegradable (e.g., MSW, commercial waste) as opposed to other wastes that will not generate methane (e.g., auto shredder fluff, contaminated soil);
- data on the history of gas collection system installation and operation; and
- data on the history of intermediate and final cover installation.

Regression analysis will be conducted to identify the best fit decay rate with L_o treated as an intrinsic property of the waste. L_o will be calculated from published waste composition data and measured methane yields for MSW components (Staley and Barlaz, 2009). In addition, uncertainty in decay rate estimates will be characterized to develop guidance on the probability that a predicted level of gas flow will be realized (i.e., there is an 80% probability of at least 5000 cfm and a 70% probability of 6000 cfm). Next, we will explore alternative formulations to LandGEM that could provide an improved representation of landfill gas data and again characterize uncertainty.

Concurrently, we are processing data and will continue data analysis through the second quarter of 2012 such that significant results will be available before the GWMS. An example of typical gas collection data from one landfill is presented in Figure 1, in which the importance of gas well installation is illustrated. A significant increase in gas collection was measured after the installation of additional gas wells at a northeastern landfill. Without this information, one is left to guess why a non-linear change in methane production occurs in 2003. Issues like this are typical and must be considered during data analysis.

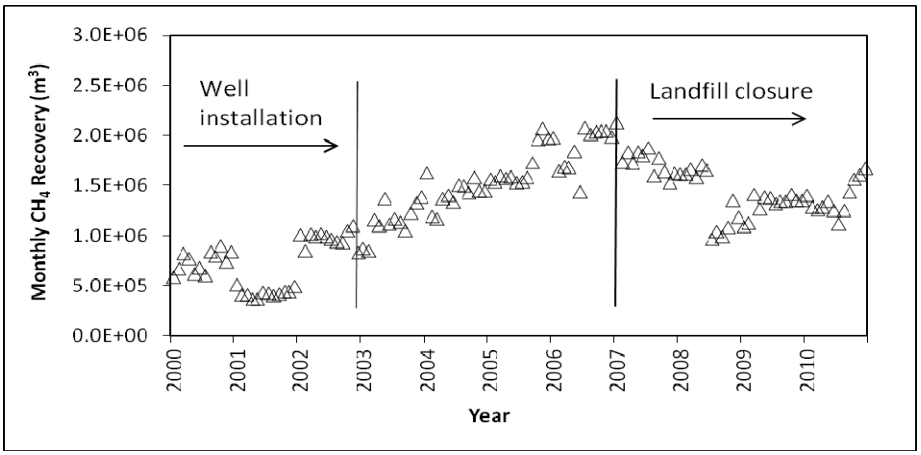

Figure 1. Effect of Landfill Gas Management Practice on Methane Collection

Expected Results

An example of regression results for first-order decay rate, k is presented for a mid-Atlantic landfill in Table 1. Values were calculated at L_o=100 m^3 Mg^{-1}, the AP-42 default, and at L_o=59 m^3 Mg^{-1} which was calculated using national average waste composition data (Staley and Barlaz, 2009). The results illustrate the variation in the optimal decay rate with varying assumptions of L_o and collection efficiency.

Table 1. First-order Decay Rates Estimates from Gas Data at a Mid-Atlantic Landfill[a].

Landfill	CH_4 generation potential, L_o (m^3 Mg^{-1} wet waste)	First-order Decay Rate, k (yr^{-1})[a]	First-order Decay Rate, k (yr^{-1})[b]
P	100	0.035	0.029
	59	0.114	0.086

a. A final cover was in place on this landfill and a collection efficiency of 85% was assumed to convert gas collection to gas production.
b. Assuming a collection efficiency of 95% to convert gas collection to gas production.

Decomposition and Carbon Storage in Forest Products in Laboratory- and Field-Scale Landfills

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Introduction

It is estimated that approximately 25.9 million tons of paper and paperboard and 13.6 million tons of wood were discarded in municipal solid waste (MSW) in 2009, which accounts for 24.6% of MSW (US EPA, 2010). The decomposition of these forest products under anaerobic conditions in landfills results in the generation of carbon dioxide and methane. Some of the generated methane is released into the environment as it is produced prior to gas collection system installation, and collection efficiencies are less than 100%. On the other hand, a significant portion of paper and wood do not decompose in landfills and therefore represent a source of biogenic carbon storage. Information on the biodegradability of these forest products in landfills can be used to estimate the feasibility of gas recovery projects and provide data for methane emission and carbon storage inventories in landfills. The results are also important for life-cycle assessments of greenhouse gas impacts of wood and paper products.

Experimental Methods

The objective of the research was to measure the decomposition and carbon storage of several types of wood and paper products under both laboratory and field conditions. Tests were conducted under field conditions at two bioreactor landfills as well as under laboratory conditions where ultimate biodegradability can be determined. Several wood and paper types were selected as the dominant types of forest products in the U.S. (Table 1). At lab-scale, anaerobic decomposition of each material was measured separately in 8-L reactors that were operated to maximize the rate and extent of anaerobic biodegradation. The study includes (1) measurement of the loss of cellulose and hemicellulose for each product, and (2) measurement of the methane yield and carbon storage factor for each material over the test period. Samples of wood and paper were also buried in full-scale bioreactor landfills and excavated after approximately 1 and 2 years for chemical analysis.

Results and Discussion

As shown in Table 1, the extent of decomposition across different wood types ranged from zero to 19.9% (still ongoing) conversion of C to CH_4 and CO_2. Hardwood degradation was more extensive than softwood except for eucalyptus. The relatively low fraction of carbon

conversion (DOC_f) and high range of carbon storage for wood products documents their role in carbon storage in landfills. The extent of decomposition measured at laboratory-scale represents an upper limit on what would be expected at field-scale. An example of what we have excavated from a landfill is presented in Figure 1. On the left, a sample of old corrugated containers (OCC) is illustrated. A piece of excavated particle board is illustrated on the right. In the case of particle board, solids composition was measured separately on sections of the board that did and did not show evidence of degradation. A decomposition index (DI) was used to eliminate the influence of dirt and detect degradation. The DI is computed with respect to the ratio of (C+H) /L in initial and final solid states, and increases with increasing degradation.

$$\text{Decomposition Index (\%)} = [(C+H)/L_{init} - (C+H)/L_{final}\ ] / (C+H)/L_{init} \qquad (1)$$

The ratio of lignin to volatile solids was also used to evaluate whether the excavated samples exhibited degradation as this ratio would be expected to increase as cellulose and hemicellulose are degraded. For example, this ratio increased from 0.16 to 0.65 in an excavated OCC sample. The field experiment observations, in general, correlated well with the laboratory results. Most of the wood samples excavated after 1 to 2 years of burial in actively decomposing landfills did not exhibit decomposition compared with the fresh materials.

Table 1. CH_4 Yield, Carbon Conversion and Carbon Storage Factors of Test Materials

Material Type	**CH_4 Yield (mL/dry g)**	**DOC_f [a] (%)**	**CSF (g biogenic C stored dry g sample^{-1})[b]**
HW-Red Oak	32.5	7.8	0.41
HW-Eucalyptus	-0.3	0.0	0.45
SW-Spruce	7.5	1.8	0.41
SW-Radiata Pine	0.5	0.1	0.46
OSB-HW	84.5	19.9	TBD[c]
OSB-SW	-0.1	0.0	0.47
PW	6.3	1.4	0.46
PB	5.6	1.3	0.38
MDF	4.6	1.1	0.37

a. DOC_f, in IPCC terminology, represents carbon recovered as CH_4 and CO_2 divided by the mass of the organic carbon added to a reactor initially.
b. Units are mass of biogenic carbon stored per dry mass of sample.
c. TBD = To be determined as the reactors continue to produce methane.

OCC Particle board

Figure 1. Excavated Samples of Old Corrugated Containers and Particle Board

Fluorescent *in situ* hybridization technique in anaerobic process studies

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Introduction

FISH (Fluorescent *in situ* hybridization) is a technique that is used to identify and enumerate specific microbial groups. This technique can be used to determine whether specific genetic elements exist in a sample. This is useful for determining microbes that have a particular gene (rRNA) present (Amann, 2008). Fluorescent oligonucleotide probes (15-25 bases) are designed to attach to specific genetic regions of microbes that will differentiate them from other groups, by using different fluorescent dyes. Epifluorescent microscopy is then used to detect the presence or absence of individual microbial groups.

FISH can be applied to analyze microbial characteristics on general level or on more specific level, like different groups in landfill environment such as methane oxidizing bacteria in landfill cover and ammonia and nitrite oxidizing bacteria in harsh environment in landfill leachate treatment plant (Pelkonen, 2002). In this study microbiological communities in different anaerobic sludges were studied with FISH technique, aiming at to characterize biological phenomena in more detail together with other analysis methods.

Materials and methods

The sludges used in this study were from a slaughterhouse pig farm (referred to as sludge 1) and from a sewage treatment plant also containing municipal solid food waste as co-substrate (referred to as sludge 2). Both sludges were digested anaerobically in thermophilic digesters, with hydralic retention time over 15 days.

A general probe for *Archaea*, ARC915, was used to identify all methanogens, MSMX860 to *Methanosarcinales* and MB311 to *Methanobacteriales* (Crocetti et al. 2006). The probes were labeled at 5'-end with indocyanine dye (Cy3). Microscope used was Diaphot-300 epifluorescence microscope (Nikon, Japan) with 100 W mercury lamp, equipped with filter sets (Chroma Tech, USA), and used with 1000x magnification. Also metabolic tests were used by adding acetate or by producing conditions with hydrogen atmosphere in 110 ml serum bottles with sludge and incubated for weeks.

Results and discussion

The number of methanogens was in the same range. Microbial characterization showed only one methanogen type present in sludge 1 (Fig 1) and metabolic tests with acetate showed clearly lower methane production rate than in sludge 2. As hydrogen consumption was considerable in metabolic tests with sludge 1, results indicate that hydrogenotrophic activity was responsible on the methane production. Sludge 1 contained slaughterhouse waste feed

with high nitrogen content and ammonium-nitrogen in the liquid was as high as 2.4 g/l. When pH was high (around 8.4), these conditions combined with high temperature indicate strong ammonia concentration and inhibition to methane production. Absence of acetotrophic methanogens is indication on this, as they are more sensitive to ammonia inhibition. Methanogens in sludge 2 consisted of two dominating groups (Fig 2) with more diverse use of substrate. Ammonia inhibition was less probable (pH 7.4, NH_4-N < 1 g/l). This also gives more favorable conditions in possible transitions.

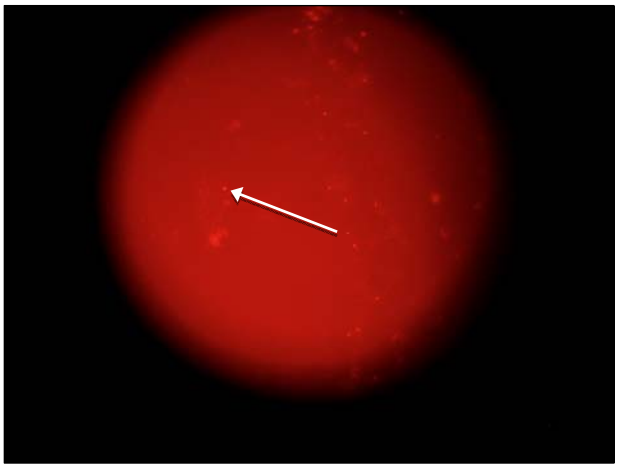

Figure 1. Sludge 1: hydrogenotrophic methanogens (probe ARC915).

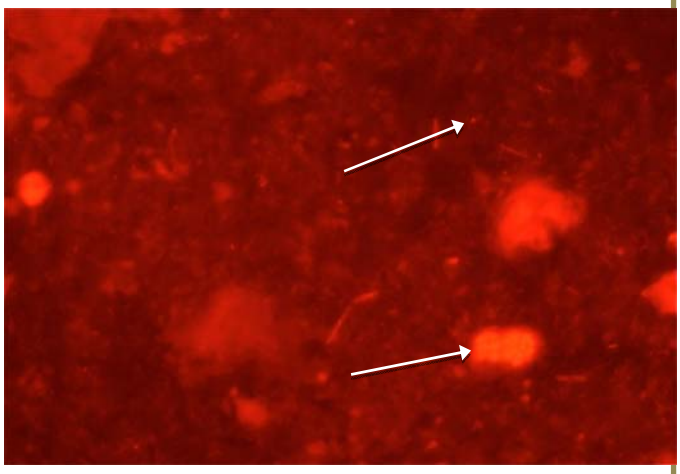

Figure 2. Sludge 2: diverse species of methanogens (*Methanosarcinales*, *Methanobacteriales*; probe ARC915).

Conclusions

Compared to the study of enrichment cultures, FISH gives faster results with less effort. In the cases studied it was fairly easy to identify a stressed anaerobic culture, in the one case, through the reduced variation of micro-organisms. This could be linked to high pH and high concentration of nitrogen resulting from a protein rich feedstock. The other reactor received a more varied substrate and displayed a good variation of the microbial community and hence a good biogas production.

References

Amann, R. (2008). Single-cell identification in microbial communities by improved fluorescence in situ hybridization techniques. *Nature Reviews Microbiology, 6*(5), 339-348.

Crocetti, G. (2006). An update and optimisation of oligonucleotide probes targeting methanogenic archaea for use in fluorescence in situ hybridisation (FISH). *Journal of Microbiological Methods, 65*(1), 194-201.

Pelkonen, M. (2002). Evaluation of microbial populations in biological treatment of landfill leachate at low temperature and in a methane oxidation structure. 2nd Intercontinent. Landfill Research Symposium, Asheville NC USA, 13 – 16.10.2002. Proceedings: p. 196-7.

Sustainable emission reduction of Dutch landfills in the aftercare period.

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Introduction

In 2010 the Dutch Ministry of Infrastructure and Environment started the project "Introduction of sustainable landfill management in the aftercare period". The aim is to prove the feasibility to reduce the emissions to the environment to an acceptable flux within a period of ten years. Next year the operation of three pilots will start with the establishment of the existing landfill conditions and emissions and also the design and planning of a ten year operation period. The emission reduction will be established by an enhancement of the anaerobic degradation by recirculation and flushing of leachate and followed by forced low pressure aeration. The design and control system will be developed by the use of international experience (e.g. Germany, Austria, and France) and the Dutch experience from the Sustainable Landfill research group (www.sustainablelandfilling.com).

At the landfill of Landgraaf (NL), the test-cell was operated between 2001 and 2011, with the objective to demonstrate technologies to enhance biodegradation of freshly deposited waste. The experimental data of this pilot are compared with the aims of the starting full scale experiments. The focus of this abstract is the leachate emission from landfills and the operation measures to reduce these emissions to an acceptable level.

In spring 2012 the Dutch Ministry of Infrastructure and Environment will publish the acceptable (leachate-) flux of pollutants (test values) to the underground and especially to the freatic groundwater (point of compliance, POC 1.) and the first groundwater layer (POC 2.).

The methodology will be comparable with the method to establish the waste input to inert landfills (EU Landfill Directive, annex II), but now used for the leachate emission from mixed waste. The test values will be site specific considering natural attenuation (soil conditions, natural background composition of the groundwater and the flow). The test values comprise chloride, sulphate, NKj, As, Cd, Cr, Cu, Ni, Hg, Pb and Zn.

Lessons learned from the bioreactor landfill Landgraaf (NL)

The pilot had been in operation over eight years and has been excavated in summer 2011.
Waste samples on 19 spots have been taken and analysed.

Figure 1. Excavation of the Landgraaf bioreactor pilot.

The demonstration proved the feasibility to enhance biodegradation to a large extent. The methane generation potential seems to be reduced by more than 95%. The residue meets the German and Austrian criteria on respiration (in 4 days, AT_4) and gas generation potential (in 21 days, GB_{21}). During the running period fresh water have been added additional to net precipitation and recirculated; the flushing performed with intervals. The gross infiltration resulted in an L/S of approx. 0.6 and the net L/S by flushing 0.2 (litre/kg ds). Leaching tests were performed by ECN (NL) according the procedures of CEN EN 12457: pH dependent leaching tests, batch leaching tests at native pH and percolation leaching test.

In the figures 2 and 3 the results for DOC and chromium (from the total of 30 considered elements) are presented: the upper lines are the initial mixed waste and the underlines the excavated waste. A preliminary conclusion can be drawn from the DOC results if these results are compared with annex II of the EU LD inert landfill: DOC < 160 mg/l. Although the waste has been biodegraded in a large extent DOC concentrations still are too high with an L/S of 0.2; an additional flushing up to an L/S of 1 seems to be necessary. The chromium concentrations meet the Annex II: Cr < 0.1 mg

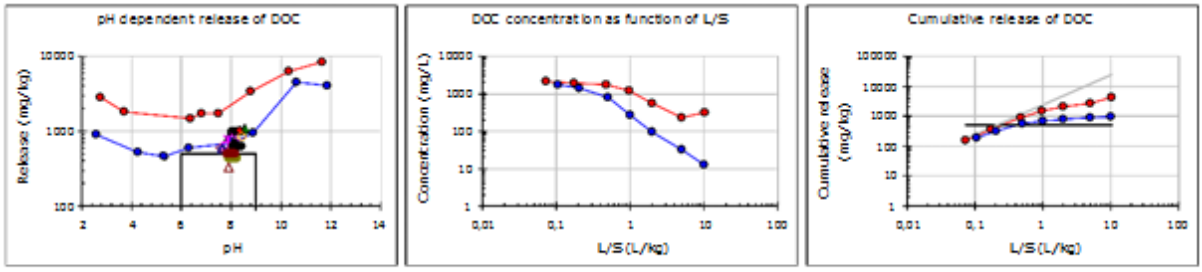

Figure 2. DOC

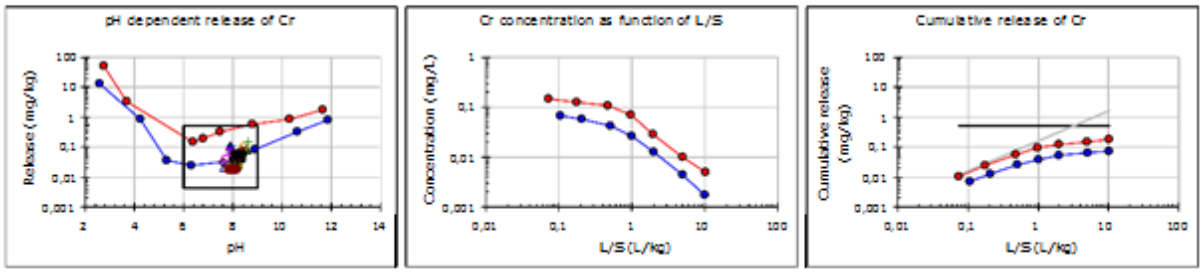

Figure 3. Chromium

Sustainable emission reduction of Dutch landfills; test values for leachate

The presentation and poster for the ICLRS conference will comprise:

- the methodology of assessment of test values for the emission flux to the subsoil of landfills;
- the site specific flux test values for the three demonstration sites;
- the assessment of leachate quality;
- comparison of the assessed flux values with existing values (Germany, Austria, Annex II, experimental data pilot Landgraaf);
- implications for design and operation of the demonstration site Bergen op Zoom (NL);
- preliminary conclusions, recommendations and outlook of the project "Sustainable emission reduction of Dutch landfills in the aftercare period".

Reference.

W. Kattenberg cs.: Policy process of allowing research pilots for sustainable emission reduction at landfills in The Netherlands, Sardinia 2011.

Decomposition of composite wood products and paper products under controlled anaerobic conditions

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Introduction

The Australian Department of Climate Change and Energy Efficiency (DCCEE) has recently decided to adopt waste-specific decomposition factors (DOCf) for organic waste materials in landfills (DCCEE 20100. Previously in their Guidelines a default DOCf of 0.5 was applied to all organic materials in landfills (DCC 2008). Whilst this may have been adequate for an initial estimate of greenhouse emissions from the landfill as a whole, it would be wrong to apply it to individual components of the waste stream. In addition to national greenhouse estimates, a greater understanding of the dnamics of the decay of a range of forest products in landfills is important for both carbon accounting and Life Cycle Assessment purposes.

The key objective of the experimental work described here was to determine long-term carbon storage in paper and composite wood products in anaerobic reactors in the laboratory under optimal decay conditions.

Methods

Reactors (8-L) containing a range of paper and composite wood products were set up and kept in a room at 39 ºC, with frequent recirculation of leachate (sourced from active cells in a bioreactor landfill) to stimulate decay, neutralisation of pH and addition of nutrients as required.
The following samples were placed in the bioreactors:

Composite wood products (Particleboard , MDF and high-pressure laminate)
Copy paper (3 types: Eucalyptus fibre, Acacia fibre and Recycled fibre)
Cardboard (ex builders and ex landfill Brisbane (buried for 18 years in a landfill in Brisbane, Queensland).

Results and Discussion

Methane quickly became the predominant gas in all reactors for the initial two months. The particleboard and MDF reactors produced gas for two months and then stopped. As the production of gas slowed down, additional phosphorus was inserted into all reactors (target of 10 mg P/L). This led to a spike in gas production across all reactors filed with paper products, with a change in gas composition to an approximately equal ratio of carbon dioxide and methane.

A follow-up experiment where the leachate in the composite wood product reactors was twice replaced with fresh leachate did not lead to additional gas production. The results for the composite wood products in bioreactors suggest that virtually all of the carbon in

composite wood products commonly disposed off in landfills in Australia can be considered to be retained in storage indefinitely.

It is premature to make a final assessment on the implications of this project for paper products as all reactors filled with paper are still producing gas (an update will be provided for the presentation). However, the maximum carbon loss observed so far has been 32%%, for one of the reactors filled with Eucalyptus copy paper, and the lowest carbon loss so far has been reported for one of the cardboard reactors (5%). Given that most of gas production in the bioreactors is likely to have occurred alreafy, this result suggests that regardless of the paper type a significant proportion of the carbon will be retained in long-term storage, even when exposed to optimum anaerobic decay conditions.

Table 1. Carbon storage and carbon loss (range) from paper and composite wood products in bioreactors

Product type	Carbon storage (%)	Carbon loss (%)
Particleboard	98.25	1.75
MDF	100	0
High-pressure laminate	100	0
Copy paper (Eucalyptus)	TBA	TBA
Copy paper (Acacia)	TBA	TBA
Copy paper (Recycled)	TBA	TBA
Cardboard (fresh)	TBA	TBA
Cardboard (landfill, 18 years)	TBA	TBA

References

DCCEE 2010. Review of the NGER (Measurement) Determination Discussion Paper. Department of Climate Change and Energy Efficiency August 2010.
DCC 2008. National Greenhouse Accounts Factors. Australian Government. Department of Climate Change. November 2008.
Ximenes, F.A., Gardner, W.D and Cowie, A. 2008. The decomposition of wood products in landfills in Sydney, Australia. Waste Management. Vol. 28 (11): 2344-54.

Characteristics of temperatures and gas components in debris storage sites in East Japan Earthquake disaster areas

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The devastating earthquake and tsunami of March 11, 2011, killed as many as 20,000 people and left more than 20 millons of tons of debris in the stricken areas. The debris have been temporarily stored in the areas for more than a year after the disaster. Only less than 16% of them were recycled, incinerated or disposed until May 2012. A long storage of the debris has risks for the people who live in the surrounding areas. Espacially wildfires burned more than 10 debris storage sites so far.

Wildfires at debris storage sites may result from temperature rising by biological or chemical reactions. Landfill fires are mainly caused by spontaneous combustion which are initiated by solid materials with lower ignition points (Moqbel et al., 2010). Temperature rinsing of wood chips and RDF material are caused by moisture adsorption(Kakuta et al., 2009). Several factors may effect on wildfires at debris storage sites. As first stage, temperatures of debris storage sites may rise by biological or chemical heat generation like moisture adsorption, aerobic biodegradation, etc and result in spontaneous combustion by further chemical heat generation like chemical self-heating.

In this study temperatures and gas components were measured at the debris storage sites of a devastated city in Iwate prefecture. Figure 1 shows the surveyed area. The mixed debris includes much of organic matters like wood, textile, plastic, and sand or soil after metal, rubber, timber and electric appliances are basically separated and picked up at the clearance process of debris in the disaster areas. Such organic materials may result in active biodegradation at the storage sites. At the sites vapours were emitted from the surface of the storage sites when heavy machineries excavated them because the temperatures were high.

Figure 2 shows the surface temperature of mixed debris measured by thermography instrument. The maximum temperature was 43 degrees Celsius. The gas at the surface consisted of 0.2% methane, 6.1% carbon dioxide, and 17 ppm carbon monoxide. Aerobic biodegradation is active and maximum temperature inside of the debris may be more higher than 43 degrees Celsius. The temperature rising may result in spontaneous wide combustion and wildfire. So temperature and gas monitoring are essential for managing the safety of debris storage sites.

The Ministry of the Environment, JAPAN, published the guideline for prevention of fires at debris storage sites. Finally this paper attempts to propose additional methods to manage debris sites safely.

Figure 1 Surveyed area.

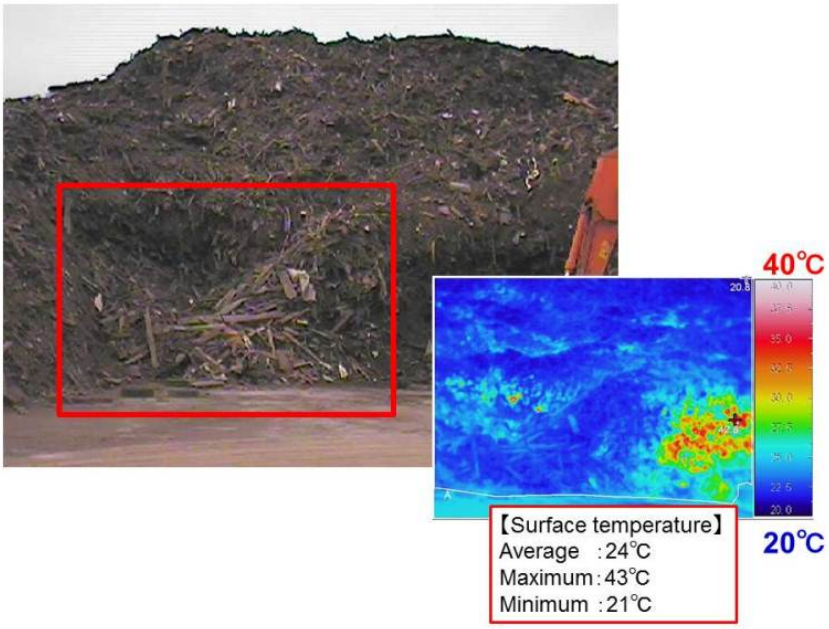

Figure 2 Surface temperature (Thermography)

REFERENCES

1) Moqbel S., Reinhart D., Chen R.H., Factors influencing spontaneous combustion of solid waste, Waste Management 30, 1600-1607, 2010,
2) Kakuta Y., Machida T., Matsuto T., Tojo Y., Matsuo T., Study on spontaneous combustion of wastes caused moisture adsorption during storage, Journal of the Japan Society of Material Cycles and Waste Management, Vol.20, No.1, pp.39-51, 2009 (in Japanese)

Enhanced Abiotic Humification of Organics and its Implication on Carbon Sequestration in Landfills

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Introduction

Natural metal oxide minerals in soil have been found to be highly reactive in facilitating browning reactions of organic compounds such as amino acids, polyphenols, and sugars. These browning reactions, which are called humification process, mostly involve carbonyl-amine reactions (including polymerization and/or polycondensation of phenolic compounds, consuming sugars and amino acids) and result in the generation of highly colored, high and low-molecular weight polymers (humic substances, HS). The organic matter gets more and more stabilized during the humification process and organic carbon is simultaneously captured into these advanced polymers. The humification process also plays a significant role in stabilization of municipal solid waste (MSW) treatment. There is significant potential to sequestrate carbon via humification within MSW landfills. Steel slag is industrial waste that often contains high quantities of oxide minerals. We examined its ability to facilitate humification reaction and sequestration of organic carbon. The production and characteristics of humic-like substances (HLS) were identified. Experiments performed with zeolite and birnessite were also conducted for comparison.

Materials and methods

A 300 ml aliquot of autoclaved phosphate buffer (pH=6, 0.2 M) containing 0.02% (w/v) thimerosal was placed in a 500-ml Erlenmeyer flask, and then 4 grams of mineral (steel slag, zeolite, or birnessite) were added. Then a certain amount of catechol, glycine and glucose were added to the suspensions to make each concentration 0.01 M. The reaction mixture was shaken in the incubator in the dark for 400h (100 rpm, 30 ℃). A mixture of catechol, glycine and glucose in the absence of mineral was conducted as control 1, while a reaction system with steel slag but without organic compounds was conducted as control 2. Sterile conditions were maintained throughout the experiment. All treatments were in triplicates.
During the incubation, a 5ml aliquot of the reaction mixture was withdrawn and analyzed. The extinction coefficient at a wavelength of 600nm (E600) was used to describe the reaction, and it was calculated as follows:

$$E_{600} = \frac{A_{600}}{TOC(mg/l) \times L(cm)} \times 1000$$

Where L is the length of the light path. E600 indices always increase with increasing extent of humification.

Results

E600 was monitored during the experiments and is shown in Figure 1. Steel slag exhibits excellent accelerating effect on the darken process. In the absence of minerals, a significant proportion of organic carbon was transformed to stable humic-like substances (24.9-38.0%) within 400 hrs. Steel slag can transform more organic carbon into humic-like substances (38%) and leave less organic carbon (47.7%) in reaction system than birnessite and zeolite do.

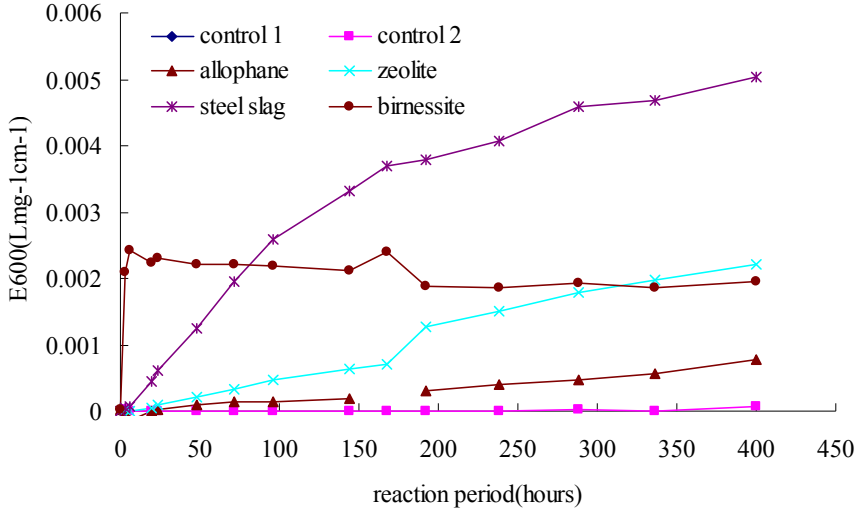

Figure 1 Variation of E600 during humification experiment

Table 1 Carbon balance of humification systems (%)

	HLS	CO_2	Remained organic carbon
Zeolite	24.9	12.9	62.2
Birnessite	29.3	19.6	51.1
Steel slag	38.0	14.3	47.7

Acknowledgement

This work was supported by the Key Laboratory for Solid Waste Management and Environment Safety, Ministry of Education of China (No. SWMES2010-12).

MEASUREMENT OF PERSISTENT ORGANIC POLLUTANTS IN LANDFILL LEACHATES

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Introduction

Leachates from two French landfills which receive biologically stabilized municipal solid waste –MSW- (and the process waters from one of the MBT plants) have been previously studied (Zdanevitch *et al*, 2009a). Classical parameters were measured on all samples and values were compared between leachates of the same site, and between sites. Leachates showed different behaviours depending on the measured compounds: some values were typical of young waste, while others were more representative of old, stabilized waste.
During ICLRS 2010, a discussion on emerging contaminants covered two aspects: nanomaterials and trace organic compounds. The conclusions on organic compounds was that "we were somewhere in the middle", that there was already some knowledge on the fate of organics in landfills, but that more research was needed. This paper deals with the measurement of persistent organic pollutants (POPs) in landfill leachates.

Experimental setup

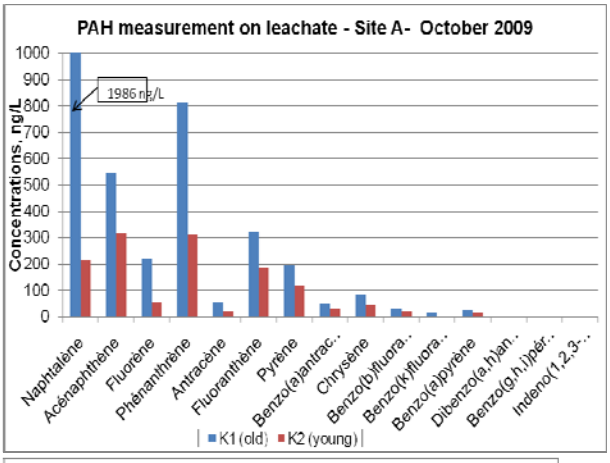
Samples were taken on landfill previously studied, both for biogas emissions (Zdanevitch et al, 2009b) and classical parameters in leachates (op. cited). The sites receive the outputs of aerobic treatments. Leachate samples were taken on two landfills:
- site A has two landfill zones: the old one –K1- received untreated MSW until 2006, the new one –K2, organized in cells- receives only stabilized waste, since 2006.
- site B has a high grade separation of the biodegradable fraction of MSW; this fraction is treated by a composting process. The compost (which has a high quality) is used by local farmers. The landfill receives only the refuses of sorting, which contain very little biodegradable material.

The analytical Laboratory in INERIS is used to measuring POPs at trace levels in surface waters and ambient air. Yet, measuring these compounds at low concentrations in biogas or leachates (heavy polluted matrixes) is not very easy. Samples were carefully taken and quickly returned to the Laboratory. In addition, "classical" parameters (suspended matter, metals, salts...) were measured outside INERIS, in the Laboratory which already measured the leachates for the previous study.

Discussion

Comparison of the measured values, for each of the classical parameters, taking into account the previous study, is still under progress (there are 6 complete sets on 3 sites, measured over 2008-2009). The treatment process must be taken into account for interpreting the results.
Meanwhile, the leachate from site B (receiving little organic material), though having a rather high concentration of suspended matter, has much lower values than leachates of

site A for organic compounds, which is shown by lower values of COD, BOD5, adsorbable halogens...
On site B, pH is rather acidic (value of 6.5), although leachates of site A (stabilized mixed waste) have rather basic pH (7.6 to 8.6).
Leachate of site B has a higher content in sulphates than both samples of site A, and lower values for nearly all the other salts. Therefore, the total ion content of leachate B is lower than on site A.
Content of leachates in heavy metals is usually rather low, except for aluminium and iron (which are not really toxic). Here, leachate of site B has, for most metals, intermediate concentrations between leachates of K1 and K2.
Organic compounds and persistent organic pollutants which were measured are: aromatics (benzene-toluene-ethylbenzene-xylenes) and chlorinated compounds, volatile fatty acids, aldehydes, ketones, 15 polycyclic aromatic compounds, 7 PCB, 8 PBDE, phtalates. Results are plotted for Site A for PAH and PCB: see figures 1 & 2. Some of the results can be compared to European Limit values for waters for human consumption ou WHO guidelines: for site A, the sums of 2 chlorinated (tetrachlorethene, trichlorethene), 4 PAH (BbF, BkF, BghiP, I123cdP), and 7 PCB fall within these limit values (10 µg/L, 100 ng/L, 500 ng/L respectively), even for the more polluted leachate, though these liquids are all treated before they return to the water grid. Leachate of site B has a much lower content in all the organics and POPs, therefore it also complies to the limit values stated, though this leachate, like the others, is treated. Due to the low content in certain hazardous pollutants, the treatment of these leachates may be rather simple. Nevertheless, the content in “classical” pollutants (especially salts) is still high, which will require a further processing.



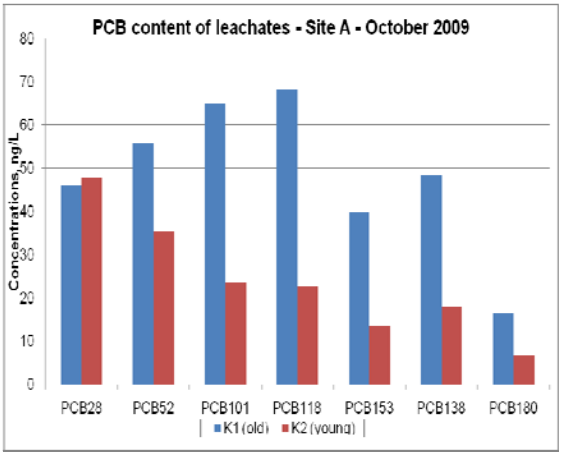


Figure 1 : PAH for leachates of site A

Figure 2 : PCB for leachates of site A

Conclusions

Leachates emitted by landfills receiving biologically treated municipal solid waste show different patterns for the metals, salt and organic content compared to classical landfills, and between sites, taking into account the treatment process. The POPs and volatile organic content of these leachates could allow a rather light purification treatment, but the rather high content in salts (which do not easily undergo biodegradation) will require more attention.

References

Zdanevitch I., Bour O., Lejal S., Llinas L. (2009) (a): "Comparison of polluting potentials of liquid emissions from MBT plants". Proceedings Sardinia 2009, Twelfth International Waste Management and Landfill Symposium, S. Margherita di Pula, Cagliari, Italy; 5 - 9 October 2009

Zdanevitch I., Bour O., Mallard P., Briand M. (2009) (b): "Gaseous emissions reduction from aerobic MBT of municipal solid waste". Waste-to-Resources 2009 – III° International Symposium MBT & MRF, Hannover, May 2009

Three-dimensional Modeling of Tracer Gas Emissions over a Landfill

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Short-term methane emissions from the Sandtown Landfill (Delaware, USA) were measured in March 2010. During the field campaign, acetylene, a tracer gas, was released at selected locations on the landfill surface. Based on the tracer dilution technique, mass fluxes of methane were calculated using measured downwind concentrations of acetylene and methane. Surface and airborne measurements of both gases at time T_o were found to be positively correlated with the measured wind speed on the landfill at time $T_o - T_t$. The time difference T_t is called travel time, and is loosely defined as the time for a gas parcel to travel from the release point to the measurement location. While T_t was initially "estimated" using field data, an accurate estimation of T_t requires complete representation of the complex flow over the Sandtown Landfill. Therefore, we conducted a 3D numerical study to investigate flow over the landfill. Model results were used to evaluate the spatial and temporal variability of the tracer plume and the associated travel time.

To resolve the local flow around the landfill, we adopted a multi-scale approach. Simulations were first performed on a 200 x 200 km^2 area that covered the entire Delmarva Peninsula. Realistic initial and lateral boundary conditions were obtained from a numerical weather prediction model, the North American Mesoscale Forecast System. The computational domain was then refined using grid nesting to a 5 x 5 km^2 area on a fine 30 m-horizontally and 20 m-vertically spaced grid, where the land surface was represented with high resolution terrain (10 m) and land cover (30 m) data from the U.S. Geological Survey. **Error! Reference source not found.** presents an example of the grid nesting methodology used (see also Zhong and Chow 2012). The landfill is surrounded by 30 m pine trees that break the wind. To represent the induced aerodynamic drag, a vegetation canopy model was implemented on the 30 m grid.

The Advanced Regional Prediction System (ARPS) was used for the simulations. ARPS is an open-source atmospheric model developed by the Center for Analysis and Prediction of Storms at the University of Oklahoma. The modeled wind on the landfill achieved very good agreement with observations (without any tuning or fitting). The temporal record of downwind measured gas concentrations displayed high, sharp peaks aloft, and low, broad peaks at the surface, consistent with field data. Simulations suggest that this behavior was due to the turbulence caused by the trees surrounding the landfill, since simulations without explicit representation of canopy did not reproduce this effect. The numerical technique of Deleersnijder et al. (2001) was applied to

determine the travel time of the tracer gas, and a contour map of mean travel time is shown in Figure 2.

In this presentation, results from the wind modeling study are used to help interpret tracer dilution measurements of methane emission at this site, helping to explain plume meandering and differences in tracer concentrations measured at the surface versus aloft with a weather balloon. Given the success of wind modeling at the Sandtown Landfill, similar modeling studies might prove useful for understanding methane emission at other landfills.

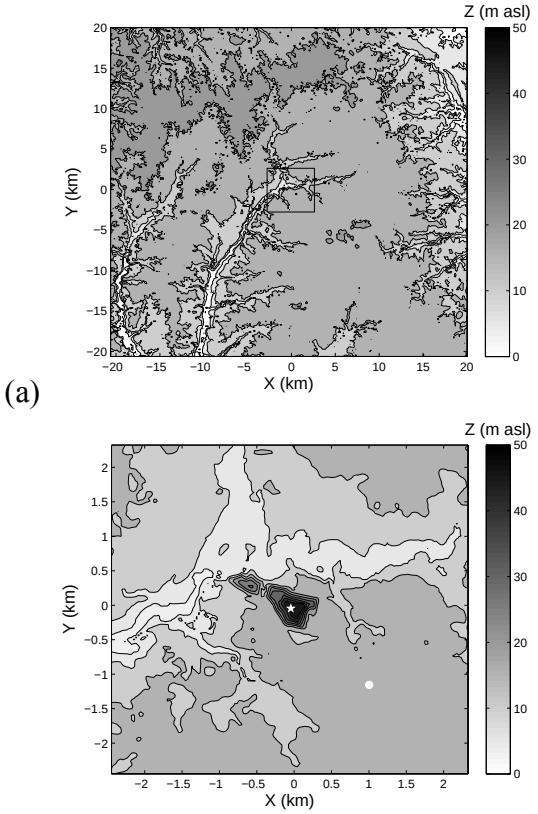

Figure 5. Elevation contours for the (a) 150 m grid with the 30 m grid shown within, and (b) the 30 m grid. Contour interval is 5 m. The Sandtown Landfill is the elevated hill in the center of (b): the tracer release point is marked with a star, and the stationary downwind measurement location of the tracer and methane is marked with a solid white circle.

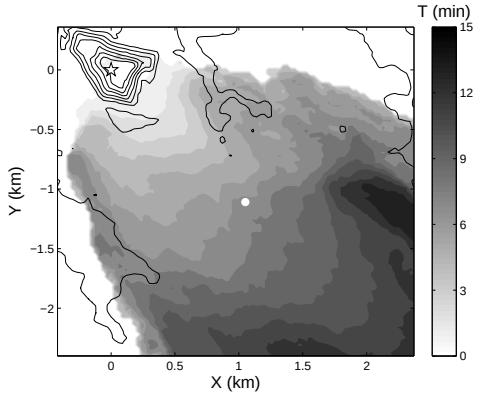

Figure 6. Contour of time-averaged (from 1400 to 1600 LST) travel time of tracer at 80 m asl. The downwind measurement location of the tracer and methane is marked with a solid white circle. Contour interval is 1 min. Terrain contours (black lines) shown with 5 m contour interval.

References

Deleersnijder, E., Campin, J., & Delhez, E. (2001). The concept of age in marine modelling I. Theory and preliminary model results. *Journal of Marine Systems*, 229-267.

Zhong, S., & Chow, F. (2012). Meso- and fine-scale numerical modeling over complex terrrain: parameterizations and applications. In F. Chow, S. De Wekker, & B. Synder, *Mountain Weather Research and Forecasting: Recent Progress and Current Challenges. in press.* Berlin: Springer.