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Dissertation

Understanding and evaluating long-term environmental risks from landfills

Verständnis und Beurteilung langfristiger
Umweltrisiken von Deponien

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Left to themselves, things tend to go from bad to worse.

Murphy's Law

"… we are aware of the fact that landfilling with contemporary wastes is a large scale long-term experiment for which we have poor knowledge. We also know that in the field of production (supply of consumer goods) we would not run chemical and biological processes under such circumstances."

Peter Baccini and Thomas Lichtensteiger (The Landfill, 1989)

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Abstract

After closure, landfills need to be managed (i.e. monitoring and maintenance) and controlled to avoid adverse effects on humans and the environment. The aftercare period ends when a landfill is no longer likely to represent a threat to human health or the environment. As quantitative criteria to determine the state of a landfill at the end of aftercare have been missing, it was the aim of this work to develop a methodology for evaluating the environmental compatibility of a closed landfill in view of aftercare completion. The methodology takes into account the behavior of the deposited waste and associated emissions, the performance of technical barriers, the potential for the migration of pollutants in the environment, and the impact of landfill-borne pollutants on environmental media.

Landfill behavior and associated emissions were analyzed for different types of landfills. Based on literature data and landfill data collected at sites in Austria and Switzerland, emission characteristics of landfills and their evolution over time, remaining emission potentials, and approaches to predict future landfill emissions were evaluated. Though organic leachate emissions and constituents such as ammonium or chloride are of potential significance at many closed landfills, the type of pollutants and emission levels vary from one site to another. The impact of substances mobilized from the deposited waste on the surrounding environment is potentially mitigated by technical and natural barriers at the landfill. The long-term efficiency of technical barriers (i.e. landfill containment) to limit and control the interaction of the deposited waste with the surrounding environment was evaluated based on literature data and expert surveys using a Delphi approach. Due to the large uncertainties inherent in long-term evaluations, scenarios are used to illustrate the effect of different technical barrier performance levels on emissions. The potential of the natural environment to attenuate adverse effects of released landfill-borne pollutants was evaluated by pollutant migration models. In general, simple and robust models were used to describe pollutant migration in the environment, as data to establish and calibrate more complex models are typically not available at closed landfills. Finally, scenario- and pollutant-specific aftercare completion criteria were derived based on the combination of the different models and the definition of an acceptable level of impact on the environment (e.g. groundwater) at the site.

The developed methodology was applied at three closed landfill sites to derive aftercare completion criteria and estimate the duration of aftercare. For a large municipal solid waste (MSW) landfill, the aftercare period is estimated to last from one century to several centuries, depending on the long-term management concept at the site. At a smaller MSW landfill with high water infiltration rates (annual precipitation of 2000 mm/yr), aftercare might be terminated within a few decades, provided that intensified leaching of the waste is continued for another decade before final cover installation. For a closed landfill containing building and construction (C&D) waste, aftercare completion is found to be environmentally tolerable already after a few years, provided that the assumptions and models in the evaluation are confirmed by further monitoring. The main reasons for the low environmental risk associated with the C&D landfill are the small emission potential of the deposited waste, the arid climate (and low amount of leachate), and the large dilution potential of the local groundwater stream. The case study findings highlight the importance of site-specific factors for the evaluation of landfill aftercare and underline the site-specific nature of the evaluation of landfill environmental compatibility.

The evaluation methodology allows for a comparison of different long-term management strategies in view of the desired state of the landfill at the end of aftercare. Thus, it links the duration and intensity of landfill management to the environmental risks associated with a closed landfill, which is a prerequisite to developing cost-effective long-term landfill management concepts.

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List of Acronyms

AF	Attenuation factor
AOX	Absorbable organic halides
BOD_5	Biological oxygen demand in 5 days
BTEX	Benzene, toluene, ethylbenzene, and xylenes
C&D	Construction and demolition
COD	Chemical oxygen demand
DM	Dry matter
DOC	Dissolved organic carbon
EC	Electric conductivity
GCL	Geosynthetic clay liner
HDPE	High density polyethylene
HHE	Human health and the environment
L/S	Liquid-to-solid ratio
LDPE	Low density polyethylene
LSR	Landfill simulation reactor
MBT	Mechanical biological treatment
MSW	Municipal solid waste
MSWI	Municipal solid waste incineration
PAH	Polycyclic aromatic hydrocarbons
PCC	Post-closure care
PVC	Polyvinylchloride
QA&C	Quality assurance and control
SP	Service period
TKN	Total Kjeldahl nitrogen
TOC	Total organic carbon
VOC	Volatile organic compounds

Glossary

Aftercare completion: The point in time when the authorities accept the end of aftercare, because the landfill is no longer likely to present a threat to human health and the environment in the absence of aftercare.

C&D waste landfill: A landfill which contains primarily construction and demolition (C&D) waste.

Contaminant: A substance, the presence of which is unwanted in a material.

De-minimus care: Low-intensity landfill management comprising levels and types of activities also required at non-landfill sites (e.g. maintenance of a vegetation cover).

Environmental risk: The integration of the probability of occurrence of a release of pollutants from the landfill and the damage caused by the pollutant in the environment.

Field capacity: The volumetric water content of a porous medium 2-3 days after saturation and after gravity drainage has ceased.

Final storage quality: Expresses that a material can be deposited without containment and without treatment of emissions into air and water.

Hydraulic conductivity: A property which describes the ease with which water can move through a porous medium.

Landfill aftercare: The management of a closed landfill (typically monitoring, maintenance, and treatment of emissions) until no more aftercare measures are necessary (landfill aftercare = landfill post-closure care).

Landfill after-use: The use of a landfill site which has been released from aftercare.

Landfill closure: The point where the landfill has reached the layout according to the aftercare requirements (i.e. installation of top cover) and is transferred to the aftercare period.

Landfill operation period: The time period during which waste is deposited at the landfill.

Landfill owner: The individual who has operated the landfill and is responsible for landfill management (landfill owner = landfill operator).

Leachate: A liquid generated in a landfill that contains dissolved and/or suspended contaminants from the waste.

Mobilizable substance potential: The amount of substances which can be released from the waste via a specific emission pathway. The mobilizable substance potential depends on the considered release mechanisms and thus, is a function of the investigation timeframe.

MSW landfill: A landfill which contains primarily municipal solid waste (MSW).

MSWI bottom ash landfill: A landfill which contains primarily bottom ash from municipal solid waste incineration (MSWI).

Natural attenuation: The reduction of mass, toxicity, mobility, volume, or concentration of pollutants in soil or groundwater due to naturally occurring physical, chemical, or biological processes.

Point of compliance: The point where certain criteria defining the acceptable level of contamination for the affected natural environment have to be fulfilled.

Pollutant: A substance that pollutes air, water, or soil, and is the cause of pollution.

Preferential flow: The movement of mobile water and its constituents in preferred pathways through porous media.

Short-, mid-, and long-term: In this work, short-term periods comprise years to decades, mid-term periods comprise decades to centuries, and long-term periods comprise centuries to millennia.

Site surveillance: The monitoring of the landfill site to assure appropriate after-use.

Waste stabilization: A process to reduce the emission potential of the deposited waste to produce an environmentally tolerable residue.

1 Introduction

1.1 Background and motivation

Of all man-made structures landfills are among those with the longest life times in the environment. Landfills were built without containment systems until the 1970s, because the attenuation and buffering capacity of the natural barriers (i.e. geologic conditions) was thought to be sufficient to protect humans and the environment (e.g. Gray et al. 1974; Bagchi 1987). However, as significant groundwater contamination was observed at such landfill sites (e.g. Andersen and Dornbusch 1967; Zanoni 1972; Kerndorff et al. 1980), landfill regulations generally required low-permeability liners at the base of new landfills starting from the 80s-90s (e.g. Austrian landfill directive (MoE 1996b)). In addition to a lining system at the landfill base, a low-permeability cover is typically installed at modern landfills after final closure. As a consequence of increasingly stringent regulations with respect to the quality of the deposited waste, the level of waste containment, and landfill operations, the complexity and cost of landfilling has increased significantly during the last decades (e.g. Environment Agency 2010b). An implication of this trend is the decreasing number of landfills, particularly with respect to MSW landfills in operation, in many industrialized countries (e.g. in the U.S. the number of MSW landfills decreased from 6300 in 1990 to 1750 in 2006 (U.S. EPA 2006); in Germany the number of MSW landfills decreased from 560 in 1993 to 330 in 2000 (BMU 2006). As many of these landfills represent a potential source of significant emissions for long time periods (cf. Belevi and Baccini 1989; Krümpelbeck 2000; Kjeldsen et al. 2002), landfill management needs to be prolonged beyond closure to protect humans and the environment. Consequently, as the diversion of waste from landfills to recycling, composting and combustion with energy recovery is increasing and the number of operating MSW landfills is on the decline (also due to the increasing size of modern facilities (cf. Krümpelbeck 2000)), the development of long-term management schemes for closed landfills is necessary to prevent adverse effects on humans and the environment.

The management of closed landfills typically includes monitoring (e.g. emissions, groundwater), treatment of emissions (e.g. leachate treatment), and maintenance (e.g. cover, leachate and gas collection) of the landfill facilities and is referred to as the aftercare (or post-closure care) period. Aftercare can be ended at a site, when the landfill is not likely to pose a threat to human health and the environment (e.g. EC 1999) anymore. As there is a lack of technical criteria to evaluate landfill aftercare and associated environmental risks (Barlaz et al. 2002; Stegmann et al. 2003; Stegmann et al. 2006), it is difficult to develop strategies for the cost-effective protection of human health and the environment at closed landfill sites (Morris and Barlaz 2011). The development of appropriate long-term landfill management strategies is in the interest of both regulatory agencies and landfill owners for several reasons. First, aftercare funding accrual mechanisms currently in place do not typically consider the possibility of aftercare periods exceeding 30 years (minimum period of aftercare for MSW landfills in Europe (cf. EC 1999)). If necessary, reform of the current time-based systems would be most effective if changes were made while landfills are still in active operation and accruing funds. Second, appropriate fiscal management of existing aftercare funds is critical to provide for the proper protection of human health and the environment, the financial well-being of landfill owners, and to prevent the development of landfills without aftercare funding.

1.2 Objectives and scope of the work

The objective of this work is to investigate long-term landfill management with respect to environmental risks associated with closed landfills. The goal is to develop a methodology to define the state of an environmentally compatible landfill at the end of aftercare. The methodological framework is developed in consideration of typical emission profiles and emission models for different types of landfills, the (long-tem) performance of landfill containment systems, and considerations about the migration of pollutants in the surrounding environment. The application of the method shall provide a basis for the comparison of different aftercare strategies with respect to the duration and intensity of aftercare and the assessment of the environmental risks associated with a specific management strategy.

The following questions are addressed in this work in order to attain the objectives outlined above:

- What is an appropriate method to assess the environmental compatibility of a closed landfill? What would be suitable evaluation criteria?
- How can one investigate and analyse the emission behavior of a landfill? What are relevant pollutants at different landfill types?
- How can landfill stability (in an environmental sense) be defined and evaluated with respect to estimating future emission levels?
- How can technical barriers installed at the landfill and their respective service lives be taken into account?
- How can the potential for natural attenuation processes in the subsurface in terms of environmentally compatible landfill emissions be considered?
- How can substance release from the waste body, barrier performance and reactive transport in the surrounding environment be integrated to determine criteria for terminating landfill aftercare?

The analysis in this work focuses on environmental aspects of closed landfills. Consequently, despite their potential significance, the economic and social dimensions of long-term landfill management are not addressed in this work.

1.3 Structure of the thesis

In chapter 1 the issue of “long-term landfill management” is briefly highlighted, the objective of the work is described, and the structure of the thesis is outlined (see Figure 1-1). A critical review of approaches for evaluating landfill aftercare including a discussion of their limitations and strengths is provided in the first section of chapter 2. Based on this discussion, a framework for a new evaluation methodology is presented and the procedure to establish the evaluation framework is described in the last section of chapter 2. In Chapter 3 emission characteristics of municipal solid waste (MSW) landfills, municipal solid waste incineration (MSWI) bottom ash landfills, and construction and demolition (C&D) waste landfills are analyzed to evaluate future emission behavior at closed landfills. Studies of the (long-term) performance of technical barriers (i.e. containment systems) are discussed in chapter 4 in the context of estimating future landfill emissions. Chapter 4 also addresses the potentially attenuating effect of natural barriers (i.e. (hydro)geologic conditions at the site) on the spread of

contaminants in the environment. Based on the framework introduced in chapter 2, the analysis of landfill emissions, and the effect of barriers on pollutant migration in chapters 3 and 4, a methodology to derive completion criteria for landfill aftercare and to evaluate the environmental risks associated with long-term landfill management is presented in chapter 5. The application of the methodology is illustrated in chapter 6 for two closed MSW landfills and for a closed C&D waste landfill. Finally, the main findings of the work are summarized and conclusions are provided in chapter 7.

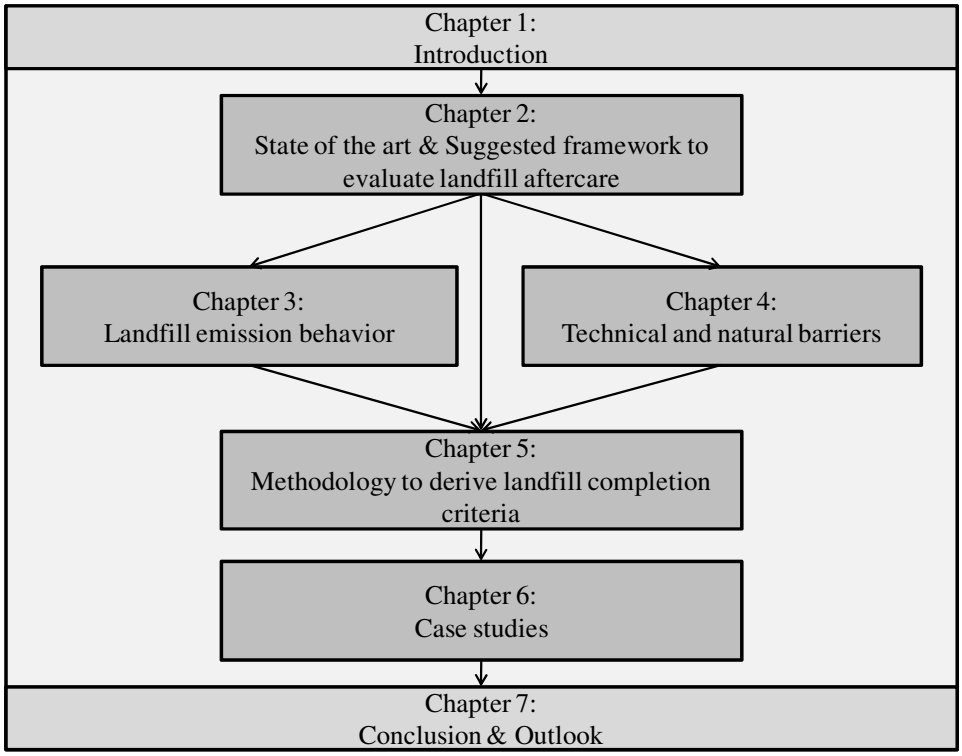


Figure 1-1: Schematic illustration of thesis structure

The work presented in this thesis is largely based on a research project, which was conducted primarily by the author at the Institute for Water Quality, Resources and Waste Management at the Vienna University of Technology. Reports and articles that have been published as part of this research project or independently by the author may contain some of the text or work included in this thesis. Therefore, the work by Laner (2009), Laner et al. (2009), Laner et al. (2010), Laner et al. (2011a), Laner et al. (2011b), and Laner et al. (2011c) may serve as a source of additional information. Further data on landfill emissions (chapter 3) and for the case studies (chapter 6) can be also obtained directly from the author.

2 Aftercare completion criteria and landfill stability

2.1 Concepts for aftercare and completion

2.1.1 General overview

Apart from a lack of technical criteria and procedures for the evaluation of aftercare and associated environmental risks, different terms and concepts referring to the condition of closed landfills add to the confusion on long-term landfill management. After the operational period of a landfill, different management phases might occur before a final cover is installed (e.g. enhanced emission reduction at the site). In Figure 2-1 it is illustrated that while the post-operational care period starts directly after the end of waste disposal, the aftercare period starts after the landfill has reached its final layout (i.e. cover has been installed). Although the focus of this work is on the aftercare period (=post-closure care period), the management of the landfill during the post operational period is addressed when this is relevant to the approach proposed for aftercare. This would be the case, if measures for enhanced emission reduction after the end of waste deposition are essential for the management concept during landfill aftercare. It would be also the case, if the state of a landfill at the end of aftercare is different from one approach to another (e.g. top cover maintenance as an afteruse activity vs. no more maintenance of the cover). Consequently, aftercare completion may not always mean the same due to different long-term management concepts. Therefore, aftercare completion is defined as the point when the competent authorities accept the end of regulated aftercare, because the closed landfill is not likely to present a threat to human health and the environment (HHE) in the absence of aftercare. In consideration of the underlying management approach this can be a definitive endpoint (i.e. requiring no further care) or the transformation of a closed landfill from post-closure care into a "de minimus care" program (e.g. site surveillance and protection of top cover) (cf. Barlaz et al. 2002). Whereas the former would also imply the end of post-operational care, the latter would leave the landfill in the post-operational care period (cf. Figure 2-1).

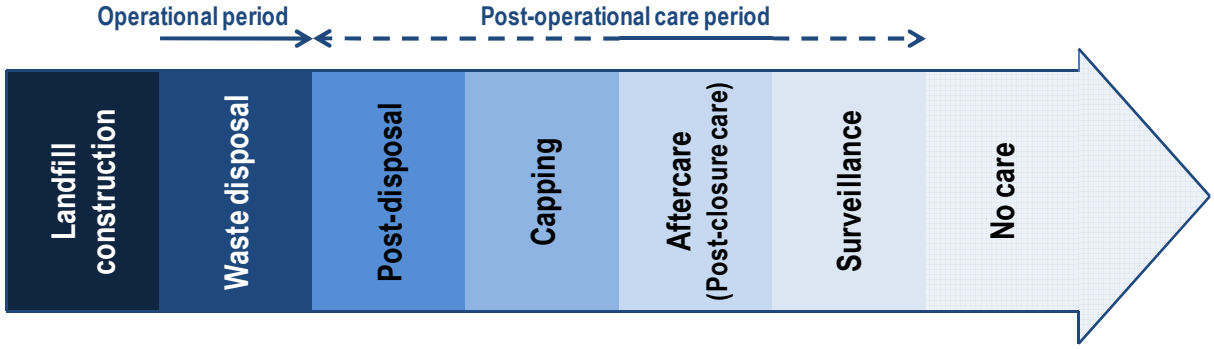


Figure 2-1: Different management phases throughout the existence of a MSW landfill

Several alternatives have been discussed for the long-term management of landfills. These alternatives include (1) termination of aftercare after a definite time period, (2) perpetual care, (3) complete waste stabilization, (4) specific chemical criteria and (5) a performance based approach (cf. Morris and Barlaz 2011). The first alternative describes a situation in which landfill management by the landfill owner is terminated after specific time period. This period could be 30 years or any other time specified by the regulator. The advantage of this alternative is that it is predictable and the owner

knows what is required for what period of time. However, this alternative leaves society responsible for problems that could arise after the owner is no longer responsible for aftercare at the site. In the termination alternative, the biological, chemical or physical status of a landfill and its potential threat to HHE are not addressed. If termination is at one extreme, perpetual care is considered at the other extreme. In this alternative the owner is required to monitor and maintain the landfill forever. Although this removes uncertainty about the duration of aftercare for owners and authorities and it offers maximum protection of HHE, it is not necessarily cost efficient. If funds are spent to protect against insignificant risks, this is not an optimal use of societal resources (cf. Scharff et al. 2011). In the complete waste stabilization alternative, the landfill is managed until the waste is completely stable with respect to chemical, biological and physical characteristics. At the point of complete stabilization, the waste itself (i.e. without containment) does not pose a threat to HHE. While geotechnical stability may be addressed in terms of post-closure settlement and material properties, chemical stability will be harder to evaluate (e.g. Brunner 1992). Some issues making the evaluation of biochemical stability a difficult task are the accumulation of ammonia in anaerobic leachate of MSW landfills (cf. Kjeldsen et al. 2002), the presence of trace organics and persistent chemicals in the leachate (e.g. Öman and Junestedt 2008; Weber et al. 2011), the heterogeneity of the deposited waste, the heterogeneity of water flow and its effect on waste degradability (cf. Maloszewski et al. 1995; Fellner 2004; Rosqvist et al. 2005), and the challenge of representatively characterizing a landfill (e.g. Chiampo et al. 1996). The fourth alternative is to operate a landfill until specific criteria for the leachate, solids and gas are reached. In the case of leachate from MSW landfills, a BOD_5/COD ratio of less than 0.1 has been suggested as in indicator of stable leachate (Reinhart and Townsend 1997), but it has been recognized to be a necessary but insufficient condition for chemical stability (cf. Barlaz et al. 2002). Criteria for landfill gas might relate to a decrease of gas production rate or a minimum fraction of the expected total gas production has taken place at the site (e.g. Wisconsin Department of Natural Resources 2007). However, typically the lack of consistent (over time) data on landfill gas production and varying collection efficiencies of the gas extraction system will make such evaluation difficult. For waste quality, criteria such as the cellulose plus hemicellulose to lignin (CH/L) ratio have been suggested as indicators for waste stability (e.g. Kelly et al. 2006), again facing challenges of representative sampling and consistent test methods (cf. Knox et al. 2005). A limitation of many of these criteria is that they do not consider site-specific conditions such as local climate, as the requirements for the long-term management of landfills will probably be different in a region with arid climate and in a humid region. The fifth alternative is to adapt post-closure management at the site based on the actual landfill performance. The focus of this alternative is to provide specific guidance on post-closure landfill management to achieve a sequential reduction of the necessary post-closure care intensity. A desired state of the landfill at the end of aftercare is not defined, but aftercare intensity is reduced as warranted by site-specific data ultimately leading to de-minimus care requirements (e.g. activities necessary at any property such as vegetation maintenance) at the site.

Based on the general overview of management alternatives for closed landfills given above, the next section analyzes specific approaches which have been suggested for the evaluation and potential completion of aftercare at landfills with a focus on MSW landfills. Finally, the major findings from the analysis of long-term management approaches are highlighted, an evaluation framework is developed, and the conceptual foundation of the approach is described in section 2.2.

2.1.2 Review of approaches for landfill aftercare and its completion

Given the range of alternatives for the long-term management of closed landfills, three specific types of approaches to evaluate and manage MSW landfills after closure have been suggested: i) target value approaches define criteria which have to be met to complete aftercare; ii) risk assessment approaches evaluate aftercare in view of the local conditions at the landfill site to quantify risks associated with aftercare activities and aftercare completion; iii) performance based systems for aftercare draw on site-specific data, which are used to evaluate the actual level of aftercare and to provide guidance on the progressive reduction of aftercare intensity. A review of specific approaches suggested by different authors (largely) adhering to one of these approaches is presented in this section.

2.1.2.1 Target values to evaluate PCC

In the target value approach, the state of the MSW landfill at which aftercare can be terminated is defined via generic standards. This approach could address the quality of the deposited waste, the tolerable levels of emissions (landfill gas and leachate), and criteria such as the maximum rate of settlement at the landfill surface. As the specified criteria are generic, they can be applied without further evaluations.

The target values for aftercare completion suggested by Stegmann et al. (2006) are based on approaches to significantly reduce the emission potential of a MSW landfill before the final cover is installed at the landfill. According to investigations by the authors, a MSW landfill will require aftercare for many decades to several centuries (cf. Heyer 2003; Heyer et al. 2005a), which does not correspond with the goal of aftercare completion within one generation. The reduction of the remaining emission potential within the waste body is aspired to minimize potential environmental impacts associated with a release of pollutants if containment systems fail at a later time. Controlled infiltration of water into the waste body and the in-situ aeration of the landfill are the main strategies to achieve a low emission potential (cf. Stegmann et al. 2003). In-situ aeration is presented as a promising technology to reduce leachate concentrations, accelerate biochemical degradation processes within the landfill, and complete major settlements within several years (cf. Heyer et al. 2005b).

Stegmann et al. (2006) state that at the end of aftercare the biochemical processes and other conversion processes have to be largely completed and unlikely to be reactivated in the future due to changing conditions at the landfill. The quantitative criteria to terminate aftercare at a site are based on the technical (i.e. enhanced emission reduction technologies), the legal (primarily with respect to German regulations, cf. BMU (2009)), and the economic (i.e. the costs of different aftercare strategies) conditions concerning landfill aftercare. The suggested target values address tolerable emission levels for landfill gas and leachate (primarily pollutant loads), the quality of the deposited waste (leaching tests and biological degradability), and settlement rates (see Table 2-1). However, Stegmann et al. (2006) emphasize that additional criteria need to be defined on a site-specific basis which relate to engineered landfill elements (e.g. condition of the containment system), the potential of natural attenuation processes taking place in the subsurface below the landfill, and potentially affected vulnerable uses in the vicinity of the landfill. These site-specific criteria are addressed on a conceptual level and tangible criteria or evaluation procedures are not included within the presented approach.

Similar to the above approach, Cossu et al. (2007) proposed a methodology to evaluate the "final storage quality" of a landfill based on a combination of generic criteria and site-specific risk analysis. A landfill is considered to have reached final storage quality when it contains materials of such quality

that no further treatment of emissions into air and water is necessary (Baccini 1989). Starting from a review of different performance parameters, emissions and waste quality, and a comparison of attainable values for defined landfill management technologies, a set of parameters (landfill gas, leachate, and deposited waste) was chosen to evaluate aftercare. Several quantitative criteria are presented (see Table 2-1) for a screening evaluation of aftercare at a landfill. If the landfill complies with these criteria, then a site-specific risk analysis (cf. Pivato 2003) is suggested to evaluate environmental compatibility of the emissions at the site.

The evaluation of final storage quality was also proposed by Knox et al. (2005) for 14 year old MSW landfill test cells. The authors state that the organic degradation of MSW is the key process with respect to achieving final storage quality. The criteria to assess final storage quality are likely to require the removal of 95 to 99.5% of degradable organics, but corresponding biodegradability tests are still rather imprecise and sometimes inconsistent with each other. The target values suggested by Knox et al. (2005) include landfill gas emissions, leachate emissions, and tests to assess the biodegradability of the deposited waste (Table 2-1). The criteria are defined generically based on literature data and regulatory standards. The authors conclude, however, that criteria to assess final storage quality will necessarily be site-specific and that current test methods are not suitable to appropriately characterize final storage quality for a landfill. Thus, operative criteria are still to be developed and should be based on documented cases, where MSW landfills have achieved final storage quality. A case study is presented by Valencia et al. (2009), who evaluate waste from a pilot scale bioreactor test cell by applying the waste acceptance criteria for inert waste landfills (EC 2003). They reported that the quality of the waste did not comply with waste acceptance criteria values after two years of operation (note that this would not be expected anyhow). However, as the waste acceptance criteria were not designed as target values for aftercare completion at MSW landfills, as for example ammonium is ignored as a leachate quality parameter, these criteria are not included in the discussion on target value approaches.

Krümpelbeck (2000) proposed another approach to evaluate the state of the landfill at which aftercare might be completed using generic values. Based on an evaluation of emission data from German MSW landfills, Krümpelbeck (2000) identifies leachate emissions as the primary long-term threat at MSW landfills. Consequently, the suggested target values refer to the load of chemical oxygen demand and ammonia-nitrogen in the leachate (see Table 2-1). Although the loads are suggested as the primary indicators, critical concentration levels are also presented for these parameters. These criteria are based on the assumption that 9% of the original emission potential of the deposited MSW, calculated via trend extrapolations of leachate data, will be released from the landfill over 500 years. In addition to these criteria, other values are discussed as leachate-based completion criteria, such as water quality standards for the inflow of drinking water purification plants, and it is emphasized that the application of the presented criteria is only suitable if the landfill base is located at least several meters above the groundwater and emissions are compatible with the planned future use of the site. Krümpelbeck (2000) concludes that the target values enable an initial assessment that must be complemented by site-specific criteria to determine when completion of aftercare is possible. Hence, the approach provides screening level indicators for leachate, but does not represent an operative procedure to evaluate aftercare.

In general, all of the authors of target value approaches emphasize that additional site-specific evaluations are needed to complement the suggested values before a decision on aftercare completion can be made. Although no specific procedure is suggested, the authors agree that a site-specific assessment

needs to address the risk of environmental pollution at the site, especially with respect to groundwater. Most of the approaches include criteria for the quality of the deposited waste with a focus on its biodegradability. However, the issue of a representative characterization of the landfill body (cf. Chiampo et al. 1996; Sormunen et al. 2008) and the consistency of different test methods (e.g. Laner et al. 2011a) has not yet been handled on an operative level. In the approaches described by Stegmann et al. (2006) and Cossu et al. (2007), the target values are derived in consideration of achievable landfill properties (based on enhanced emission reduction measures) and they may not necessarily be associated with environmentally compatible emissions. The latter is discussed by Ehrig (2002) who points out the danger of specifying target values based on achievable rather than on environmentally protective levels.

Apart from the approach suggested by Stegmann et al. (2006), the approaches presented in Table 2-1are conceptual and not specific with respect to the underlying definition of the state at which a landfill might be released from aftercare. Although, Knox et al. (2005), Cossu et al. (2007), and Valencia et al. (2009) use “final storage quality” to describe this desired state, it has been previously concluded that this is not yet an operative definition and that it probably cannot be defined in a generic way (e.g. Baccini 1989; Brunner 1992; Hjelmar and Nedenskov 2007). In conclusion, the presented target value approaches range from specific guidance on aftercare (cf. Stegmann et al. 2006) to screening level assessments (e.g. Knox et al. 2005). All authors acknowledge that aftercare completion at MSW landfills is a site-specific exercise and thus supplementary evaluations to generic target values are necessary to evaluate completion.

Table 2-1: Completion criteria suggested within different target value approaches

	Stegmann et al. (2006)	**Cossu et al. (2007)**	**Knox et al. (2005)**	**Krümpelbeck (2000)**
Leachate	COD: 5 – 20 g/m²*yr, NH_4-N: 2.5 – 10 g/m²*yr, Cl: 10 – 20 g/m²*yr, AOX: 0.01 – 0.05 g/m²*yr (Emissions to subsurface)	COD: < 200 mg/l BOD_5/COD ratio & NH_4-N concentrations: mentioned as indicators, but no value provided.	NH_4-N: ≤ 10 mg/l	Concentrations & loads: COD: ≤ 16 – 70 mg/l, NH_4-N: ≤ 9 – 20 mg/l COD: ≤ 3 - 14 g/m²*yr, NH_4-N: ≤ 1.8 – 4 g/m²*yr
Landfill gas	Methane production rate: < 25 m³ CH_4/h and <0.5 liters CH_4/m²*h Hydrocarbon emissions (flame ionization detector measurements): < 25 ppm	Landfill gas generation rate: < 25 m³/h Area-specific methane generation rate: < 1 liter CH_4/m²*h (CO_2+CH_4)/N_2 ratio: no value given	Landfill gas emission rate: ≤ 8.4 liters/m²*h Trace gas concentrations: ≤ 0.5 H_2S ppbv	-
Waste quality	Eluate (L/S=10 l/kg): TOC: ≤ 150 mg/l, NH_4-N ≤ 50 mg/l, cyanide ≤ 0.1 mg/l, AOX ≤ 0.5 mg/l (additionally: heavy metals, organic compounds, pH, EC) Biodegradability: Respiratory index (RI_4): ≤ 2.5 mg O_2/g dry matter (DM) Methane generation potential in 21 days: 10 liter/kg DM	Biodegradability: Respiratory index (RI_4): ≤ 2.5 mg O_2/g DM Methane generation potential in 21 days: 10 liter/kg DM	Biodegradability: Loss on Ignition (corrected for plastics): ≤ 25 % Acid detergent fiber (corrected for plastics): ≤ 2.5 % Cellulose/lignin ratio (corrected for plastics): < 0.2 Biochemical methane potential (BMP): ≤ 0.2 m³ CH_4/Mg DM	-
Landfill settlement	Site-specific assessment (90 % of overall settlement completed)	-	-	-
Comments	Site-specific criteria addressing geotechnical stability, landfill elements (e.g. base lining system), and hydrogeology to be set.	All target values are intended for screening purposes to decide whether a subsequent risk analysis should be implemented.	Biodegradability tests are not precise and site-specific assessment is required in addition.	Concentration levels are applicable only if leachate generation rate is below 200 mm/yr.

2.1.2.2 Environmental risk assessment to evaluate PCC

As long as a landfill represents a hazard and there is a good which can be damaged (e.g. human health or environmental compartments), some level of risk is associated with the landfill. Risk assessment has been applied to landfills, often with a focus on leachate emission levels and groundwater pollution (e.g. Butt and Oduyemi 2003). With respect to landfill aftercare, an acceptable level of risk has to be defined for the state of the landfill at the end of aftercare (cf. Pivato 2003). Hence, to evaluate the risk of a closed landfill, potential negative effects on HHE and the probability of their occurrence need to be assessed. Specific approaches which adhere to the concept of landfill risk assessment for evaluating aftercare are described below and summarized in Table 2-2.

Scharff et al. (2007) state that a robust risk assessment is needed to end aftercare at landfills. Within a pilot study on the landfilling of primarily inorganic waste, Scharff et al. (2011) developed an approach for determining aftercare completion criteria as a part of an integrated procedure to evaluate aftercare. Though recognizing the importance of additional evaluation criteria such as landfill gas parameters (for landfills containing biodegradable waste) or parameters associated with functional stability[1], the work presented by Scharff et al. (2011) focuses on leachate risk assessment. The long-term impact of leachate emissions is evaluated based on geochemical modeling and the subsequent comparison of maximum concentration levels to water quality criteria. The procedure is comparable to the source-pathway-receptor approach, which was applied to determine the waste acceptance criteria within the EU landfill directive (cf. Hjelmar et al. 2001; Hjelmar et al. 2005). The major difference is that the suggested approach employs a more sophisticated reactive transport model to describe contaminant migration through the soil towards the point of compliance. The modeling scenario presented in Scharff et al. (2011) is based on a largely inorganic waste landfill with an average waste thickness of 20 meters, a net water infiltration rate of 300 mm/yr, and an assumption that the liner system at the landfill base failed completely. The application of the approach revealed chloride and sulfate as the contaminants potentially not complying with the quality standards at the point of compliance (PoC - a hypothetical drinking water well in the groundwater downstream of the landfill). It should be emphasized that these results relate to leachate from largely inorganic landfills. For landfills containing biodegradable waste, they suggest that after the organic matter has been degraded, the behavior of the landfill body may converge towards the behavior of a largely inorganic landfill (cf. Mathlener et al. 2006). Parameters typically important in the leachate from MSW landfills (e.g. NH_4-N, COD) have not been addressed. Scharff et al. (2011) conclude that a robust risk assessment approach is needed to determine aftercare completion criteria in view of the conditions at the site and that the evaluation of aftercare completion cannot be based solely on absolute numbers.

Another risk assessment approach primarily based on modeling of landfill leachate release and forward modeling of contaminant transport processes in the subsurface has been presented by Hall et al. (2006; 2007a). The modeling is carried out with LandSim2.5 (Environment Agency 2004),

[1] defined in Morris and Barlaz (2011) as a landfill that is no longer a threat to HHE based on its performance and data trends

which is used to predict the time period until "equilibrium status" with the environment can be reached at the landfill site, based on specific assumptions with respect to long-term liner and cap performance. Equilibrium is defined as the state when emissions from a landfill occur at a rate that allows sufficient natural attenuation in the surrounding environment to prevent environmental harm, so management is no longer required (Hall et al. 2007a). Consequently, if equilibrium conditions are reached at a site, landfill aftercare can be terminated (Hall et al. 2007b). Within the presented study, the risk assessment methodology has been primarily applied to demonstrate the effect of waste diversion and pretreatment options (e.g. MBT residues, incineration) in the UK on the time period until environmental equilibrium can be achieved. The authors state that aftercare completion can only be achieved if a landfill has stabilized physically, chemically, and biologically to a degree that the undisturbed contents are unlikely to pose a pollution risk at the site, which implies that the presented modeling approach needs to be complemented with other criteria to form a framework to address aftercare completion. However, this framework has not yet been proposed.

In contrast to the approaches described above, the methodology suggested by Boerboom et al. (2003) does not assess the pollution risk emanating from a closed landfill but rather the costs of environmental risks associated with aftercare. The methodology is designed to estimate appropriate funding for aftercare activities, which should be based on the potential financial risks present at a site. The risk assessment builds on expert estimates of failure probabilities for specific events (e.g. unsatisfactory inspection of the landfill cover) that are linked to the occurrence of unwanted events in a fault tree analysis. The investigated unwanted events do not comprise exceptional risks (e.g. earthquake) and relate to a large and uncontrolled contamination of groundwater, damage to the final cover, or an early need for replacement of the final cover. The probability of occurrence for an unwanted event is calculated via Monte Carlo analysis and the sum of costs to undo the unwanted events for a defined period is calculated as the financial risk during aftercare. Although the methodology is not intended to specify a procedure for landfill completion (risks are calculated for specific aftercare durations), it may represent a valuable tool to provide appropriate funding for aftercare activities, especially if it is widely applied and the model is parameterized from a growing database.

Out of the risk assessment approaches summarized in Table 2-2, only the approach suggested by Boerboom et al. (2003) directly addresses MSW landfills, however, only in terms of financial risks during aftercare. The other approaches were developed to assess (primarily) the risk of groundwater pollution for largely inorganic landfills (Scharff et al. 2011) or for virtual landfills to analyze different waste pretreatment options before landfilling (Hall et al. 2007), respectively. Although these approaches might prove valuable for an assessment of environmental risks associated with closed MSW landfills, they have not yet been integrated to form an evaluation framework for aftercare. There is a high level of uncertainty involved in a risk assessment and the transparency of modeling assumptions is significant for an understanding of the outcome. Hence, there is a need for consistent procedures that allow for an objective evaluation of the environmental risk at a closed landfill. This is highlighted by the work of Hall et al. (2006; 2007a), who point out the complexity in their modeling due to assumptions on containment system performance and subsequent effects on the interrelationships between leachate quality, release rates and concentrations at the PoC in the groundwater. Consequently, sensitivity analysis and the handling of inherent uncertainties are

essential for the comprehensiveness of the modeling results. In conclusion, while the presented risk assessment approaches could be a substantial part of an aftercare evaluation procedure, they are expensive and uncertain. As such, the utility of risk assessments in the development of aftercare completion remains to be determined.

Table 2-2: Methodological basis and application of risk assessment suggested to evaluate PCC

Authors	Modeling approach	Application	Comments
Scharff et al. (2011)	Long-term modeling of contaminant migration to a point of compliance and comparison to specific standards	Case study on a largely inorganic landfill (Cl and SO_4 identified as critical leachate parameters)	Suggested as a part of the toolbox to assess aftercare completion
Hall et al. (2007)	Long-term modeling of contaminant migration to a point of compliance and comparison to specific standard	Hypothetical landfills with different waste qualities sited in a specific model environment (leachate parameters at equilibrium level are presented)	A procedure for aftercare completion incorporating the modeling results has not been suggested.
Boerboom et al. (2003)	Probabilistic risk assessment to estimate potential costs associated with environmental risks during aftercare	Used as a cost calculation method for aftercare in several provinces of The Netherlands	No guidance on aftercare or completion, but could be a basis to accrue appropriate aftercare funding

2.1.2.3 Performance-based system for PCC

As a consequence of potentially long aftercare periods for MSW landfills, performance-based evaluations have been suggested for long-term management of closed MSW landfills (Barlaz et al. 2002). In general, such approaches are based on landfill monitoring and performance data, which are used to evaluate the actual state of the landfill and to determine appropriate aftercare activities. Hence, the evaluation procedures are site-specific and provide guidance on the sequential reduction of aftercare intensity that may, if warranted by performance data, ultimately lead to aftercare completion. Different approaches adhering to this concept are described in this section and summarized in Table 2-3.

A performance-based system for evaluating aftercare requirements at MSW landfills referred to as the evaluation of post-closure care (EPCC) methodology has been presented by Morris and Barlaz (2011). The EPCC methodology addresses the four primary aspects of landfill aftercare monitoring and maintenance (i.e. leachate quality and quantity, gas management, groundwater monitoring and protection, and cover maintenance) in a user-friendly modular system. It provides long-term stewardship of landfills by assessing their current and future impacts to the environment based on the evaluation of "functional stability". Functional stability is used to define a closed landfill that does not present an unacceptable threat to HHE in the absence of aftercare, but with some remaining level of control to protect the cover (Solid Waste Association of North America Bioreactor Committee (2004) cited in GeoSyntec (2006)). The methodology establishes a site-specific basis for decisions on maintaining, extending, reducing, or modifying aftercare activities while being protective of the environment. Logic diagrams are used for each aspect of aftercare (i.e. for each

module) and the modules can be sequentially evaluated (prerequisites module, leachate module, gas module, groundwater module, and cover module). The aspired end-use and final condition of the landfill are defined from the outset of the evaluation. Conservative assumptions are used within the evaluation procedure and proposals to modify an aftercare activity are driven by data. The application of the methodology generally involves analyzing statistical trends in leachate, landfill gas generation, and groundwater quality, as well as additional biological, chemical, and physical data, to demonstrate that gas production is stable or decreasing and leachate quality is constant or improving. Once a change in aftercare is implemented, the owner is expected to verify no adverse effect by "confirmation" monitoring followed by surveillance monitoring at a decreasing frequency. Monitoring procedures also identify high and low level "trigger" conditions requiring immediate responsive action to be taken to resolve the condition. The system allows for multiple outcomes in which the requirements for each aspect of aftercare may vary significantly. Three possible levels of analysis are built into the methodology: a) in a source evaluation the compliance with target values may be demonstrated at the source (e.g. leachate quality < drinking water standards); b) in a point of compliance evaluation, it is demonstrated that the landfill does not pose an unacceptable threat at the PoC; and c) in a point of exposure (PoE) evaluation it is demonstrated that the landfill does not pose an unacceptable threat at the PoE (Morris and Barlaz 2011). The potential for a harmful release to the environment is analyzed before any aspect of aftercare may be reduced or discontinued. Thus, the EPCC methodology determines a level of aftercare which is consistent with the actual state of the landfill and the sensitivity of the surrounding environment. This is the basis for progressively reducing aftercare and finally turning the landfill into a custodial care program (i.e. aftercare completion). The custodial care program represents a level and type of activities which would be required at other sites and thus would not be specific for MSW landfills (cf. Pivato and Morris 2005; ITRC 2006).

Application of some components of the EPCC methodology to evaluate appropriate aftercare at a specific MSW landfill in Florida has been described by Sizirici (2009). A set of relevant parameters (e.g. leachate: ammonia-nitrogen, chloride, iron, VOCs or landfill gas: remaining gas generation potential) was identified and estimates of future emission levels were derived for the case study landfill (cf. Sizirici and Tansel 2010). Based on a risk assessment at the site, Sizirici (2009) concluded that landfill gas management is no longer necessary, whereas monitoring of leachate and groundwater has to be continued. Surveillance of cap integrity also remains an integral part of aftercare activities at the landfill. Hence, the aftercare period at the site can be reduced for some landfill elements (e.g. gas collection) but is required for others (e.g. leachate management) to be protective of HHE.

Recently, Sizirici et al. (2011) presented a screening procedure for a preliminary assessment of aftercare needs at closed landfills. The procedure is based on expert evaluations (scale from 1 to 10) of site-specific parameters (e.g. climate, operational factors, leachate management, gas management, etc.), which are combined using a ranking algorithm (i.e. by assigning weights to different factors). The overall score from the algorithm is used to categorize the landfill conditions as critical, acceptable, or good. "Critical condition" at a site indicates that the landfill may represent a threat to HHE and thus, aftercare activities need to be adjusted. "Acceptable condition" as a result of the evaluation confirms current aftercare intensity and "good condition" as an outcome of the

evaluation indicates the potential for a reduction of aftercare activities. However, the suggested aftercare evaluation procedure does not provide specific guidance on the intensity of aftercare or the completion of aftercare activities.

A performance-based approach for aftercare based on the potential of naturally occurring or enhanced natural attenuation (NA) processes (e.g. via waste aeration) at a landfill has been described by van Vossen (2010). This approach aims at lasting emission reduction at closed MSW landfills and is based on the full scale demonstration of the methodology at three case study landfills in The Netherlands (cf. van Vossen et al. 2007; van Vossen et al. 2009a; van Vossen et al. 2009b). The model to demonstrate the capability of natural attenuation processes to phase out unacceptable levels of contaminants is referred to as the SANA (Sustainable Aftercare of landfills based upon Natural Attenuation) model (van Vossen 2010). As one of the demonstration landfills has been partly built without a base lining system, the SANA model has the capability to analyze the potential for natural attenuation in the landfill body as well as in the pollution plume below the landfill and in the groundwater. The conditions in the landfill are characterized and the model evaluates the state of biochemical, geochemical, and hydrological processes in consideration of performance indicators (e.g. in-situ temperature, settlements, BOD_5/COD ratio, ammonia-nitrogen, gas production rate, and CH_4/CO_2 ratio) that are used to describe the extent of organic matter stabilization. The goal of enhanced emission reduction activities within the SANA model is to replace the need for an impermeable final cover by a low emission potential of the waste, which is expected to result in landfill emissions meeting acceptable levels and consequently allowing for aftercare completion. If acceptable leachate emission levels cannot be achieved at a site, an environmental risk assessment is suggested to determine the necessity of mitigation measures for preventing inacceptable groundwater pollution. In essence, the suggested approach promotes a specific aftercare strategy (i.e. enhanced emission reduction based on NA processes) and describes its application to demonstration landfills, but does not yet constitute a consistent methodology to evaluate aftercare and aftercare completion. Van Vossen (2010), states that such a methodology will be based on the final results of the demonstration projects and is currently under development.

The performance-based evaluations of aftercare shown Table 2-3 are essentially site-specific, allowing for long-term landfill management to be adapted to local conditions. Among the presented approaches, the EPCC methodology represents the most tangible evaluation procedure, providing operative assessment protocols to decide on an appropriate level of aftercare. Whereas the EPCC methodology establishes a consistent aftercare evaluation framework, the approach suggested by van Vossen (2010) follows a bottom-up strategy of demonstrating the performance of natural attenuation processes to reduce emissions at full scale landfills and developing a consistent evaluation methodology based on the outcome of current demonstration projects. However, within the evaluation methodology an environmental risk assessment and potential site remediation may be necessary if admissible emission levels could not be met with the emission reduction approach. In addition, for those landfills not equipped with a bottom liner no operative protocol to monitor NA at MSW landfills has been suggested yet and regulators may not accept the local groundwater as a long-term treatment filter for waste management processes (cf. Christensen et al. 2000).

Table 2-3: Methodological basis and application of performance-based systems to evaluate PCC

Authors	Methodology	Application	Comments
EPCC methodology (Morris and Barlaz 2011)	Modular approach for the evaluation of environmental impacts associated with aftercare addressing data collection, leachate management, gas management, groundwater monitoring, and cover performance.	Evaluation is used as a basis to progressively reduce aftercare, ultimately leading to completion while being protective of the environment. Application was demonstrated at a hypothetical case study site.	Operative methodology providing specific protocols for long-term landfill management.
Sizirici et al. (2011)	Ranking algorithm based on expert evaluations of site-specific factors resulting in a preliminary assessment of appropriate aftercare at a site.	Factor-specific ordinal scores based on site performance assigned by experts and then weighted to derive an overall landfill score. Application at two sites was presented.	Tool is dependent on expert judgment and qualitative analysis.
SANA model (van Vossen (2010))	Demonstration of natural attenuation as a feasible approach for aftercare to minimize long-term environmental risks at MSW landfills based on performance data.	A natural attenuation (or bioreactor) approach is promoted to achieve a lasting reduction of the emission potential at three full-scale demonstration landfills.	Bottom-up approach to provide guidance on aftercare. Consistent methodology is to be developed based on the case study results.

Another fundamental difference between the concept suggested by van Vossen (2010) and the EPCC methodology is that the former embraces a strategy to enhance emission reduction to decrease emissions to low levels without a long-term low-permeability cover, whereas the latter does not require a strategy to decrease the emission potential of the waste body, as the focus lies on actual emission levels and emission trends at the landfill. Where no steps are taken to decrease the emission potential of the waste body, maintenance of a low-permeability top cover is expected to be part of the custodial care program within the EPCC methodology. Indeed, regardless of the long-term management approach at a landfill, some level of cover maintenance and associated site care activities will always be required using the EPCC methodology.

The approach presented by Sizirici et al. (2011) does not rely on measured landfill performance-data like the other approaches listed in Table 2-3, but on expert evaluations of different factors addressing aftercare at the landfill. The procedure also does not give guidance on specific aftercare activities or completion criteria. In general, the application at case study sites is of paramount importance for all performance-based approaches, as this will be necessary to assess their practicability with respect to aftercare duration and completion.

2.1.3 Discussion of approaches to evaluate aftercare

The aim of any evaluation of landfill aftercare and completion is to address the risks associated with a closed landfill in a consistent and transparent manner and thereby allow decision makers to comprehend underlying assumptions as well as associated residual risks. Table 2-4 summarizes the major characteristics of the target value, risk, and performance-based approaches. As described in

this section, each approach has strengths and limitations. Although target value approaches are not site-specific per se, all of the approaches ultimately depend on site-specific data. This implies that generic target values without site-specific assessments are not a feasible strategy for evaluating aftercare at existing MSW landfills. In addition to the need for site specific evaluations, target value approaches are confronted with unresolved issues of representative waste sampling and the inconsistency of tests to estimate potential (residual) release rates. A single test is also not able to consider the variation of substance release mechanisms dominating in a landfill over time, highlighting the need for complementary approaches. Risk assessment approaches have been largely presented with respect to the evaluation of specific risks (i.e. groundwater pollution), but not as an integrated framework to allow for the confident evaluation of aftercare completion. Although risk assessment approaches address the very core of the issue with respect to acceptable risk and tolerable emissions (cf. Scharff et al. 2011), the evaluation requires a high level of expertise and comes with large uncertainties (e.g. Hall et al. 2006). The performance-based systems for aftercare evaluation represent a more pragmatic approach, as they sequentially adjust aftercare activities to the actual risk allowing for more confident evaluations of aftercare. However, as a desired state of the landfill does not consider the condition of the waste body, such as. current emission levels and observed emission trends as a basis of the evaluation, residual environmental risks might be neglected within such an assessment. In general, to demonstrate the feasibility of aftercare completion based on any of these approaches case studies that lead to the completion of aftercare need to be conducted.

In conclusion, a consistent framework to evaluate landfill aftercare and define the condition of a closed landfill at the end of aftercare needs to integrate different aspects of the approaches discussed above. As the evaluation of landfill environmental compatibility addresses long time periods with potentially changing conditions, the notion of "stability" is often used in the context of long-term landfill management. Therefore, the concept of environmental stability from a dynamic system perspective is outlined in the next section as a basis for the evaluation of aftercare and aftercare completion.

Table 2-4: Overview on main characteristics for different types of approaches to evaluate PCC

Characteristic	Target value approaches*	Risk assessment approaches	Performance-based approaches
Site specific	No	Yes	Yes
Data collection in addition to standard emission monitoring	▪ Waste sampling and analysis (i.e. leaching and biodegradability tests)	▪ Waste sampling and analysis (potentially) ▪ More extensive monitoring (potentially)	▪ More extensive monitoring
Outcome verification	▪ Not necessary	▪ Not included	▪ Included
Status of approach	▪ Conceptual to partially operative	▪ Partially operative	▪ Partially operative to operative
Level of expertise	▪ Low	▪ Very high	▪ Very high
Usability of approach	▪ Mostly prescriptive	▪ Flexible	▪ Flexible

*The target value approaches acknowledge the need for additional site-specific evaluations, but do not provide detailed guidance. Thus, the characteristics refer to generic target values only.

2.2 Landfill stability – definition and metric

From the review of approaches for the long-term management of landfills in the previous section it is apparent that the environmental compatibility of a closed landfill is often associated with the concept of a "stable landfill" (cf. Baccini 1989). A "stable landfill" is considered not to pose a threat to humans or the environment, and hence, it represents the state when aftercare at the landfill can be ended. However, the specific conditions under which a landfill is called "stable" are often not clear and rigorous definitions as well as evaluation frameworks are missing (cf. Barlaz et al. 2002). Clarity about the concept of landfill stability underlying the evaluation of post-closure care is needed to establish an appropriate modeling framework addressing the environmental risks associated with long-term landfill management. Major aspects and the conceptual basis of the consequent framework to evaluate landfill long-term management developed in this work are outlined in this section.

2.2.1 The notion of "landfill stability"

The review of existing approaches to evaluate aftercare in the previous section highlighted that an evaluation of the environmental risk at a landfill is an integration of different aspects addressing the condition of the landfill including biological, chemical, and physical properties, as well as the conditions at the site. The notion of "stability" has been used with respect to all of these landfill properties specifically (e.g. biological stability: Barrena et al. (2009), chemical and physical stability: Boda (2002), geotechnical stability: Sabatini et al. (2002)), but not yet in an integrated way to evaluate landfill environmental compatibility.

The complete stabilization of waste produces a material which does not represent a hazard in any environmental setting (i.e. inert waste in Figure 2-2). For instance, this has been described as "solid residues with properties very similar to the earth crust" (Baccini 1989). In such a case, the environmental compatibility of the landfill is guaranteed solely by the waste quality without additional barriers (e.g. containment system). Consequently, a substantial amount of work has been done to define waste stabilization and derive indicators to evaluate the stability of deposited waste. On the one hand, the definition of environmentally tolerable waste qualities for disposal has been suggested as a basis to achieve the goal of landfills not requiring any aftercare after closure (cf. criteria to assess final storage quality of deposited wastes in Switzerland in the 1980ies (EKA 1986; Baccini 1989; Brunner 1989; Brunner 1992). On the other hand, waste stabilization can refer to the transformation of reactive waste in existing landfills into stabilized residues, which are deemed environmentally compatible. Such a transformation has been intensively discussed with respect to landfills containing organic waste. Lichtensteiger et al. (1989) discuss the transformation of sewage sludge to ultimately achieve a final storage stage when material fluxes are no longer harmful for the environment even on a long-term basis. In order to evaluate long-term landfill behavior they compare the transformation of sewage sludge to natural sediments with analogous characteristics (e.g. sapropel, peat and organic soils). They found that about one third of the organic material in the sewage sludge was mineralized within the first 10 to 20 years, with the remaining fraction of organic matter to be mineralized or transformed into insoluble, refractory organic compounds over an extended time period of thousands to hundred thousands of years. Persistent organic pollutants are expected to be present in the landfill at least for decades to centuries. The observed mobility of

heavy metals in the sewage sludge landfills is low. In similarity to the investigation of sewage sludge landfills, it has been suggested to investigate the behavior of MSW landfills based on processes taking place in natural analogues such as peat bogs (cf. Bozkurt et al. 2001; Döberl 2004). Bozkurt et al. (2000) concluded on such an analysis of long-term processes in MSW landfills that, based on the alkalinity of the waste and its pH-buffering capacity, an increased mobilization of heavy metals might not be expected in organic waste landfills for thousands of years. However, projections over such time periods are highly speculative and difficult to include in an evaluation of landfill environmental compatibility.

Several criteria were suggested to evaluate waste stabilization in MSW landfills. For instance, the evaluation of the stability of organic waste fractions (i.e. the extent of degradable organic matter which has been already decomposed) was suggested based on biogas production tests (e.g. Wisconsin Department of Natural Resources 2007; Barrena et al. 2009). In addition, the portion of carbohydrates still present in organic matter and the content of lignin, which is supposed to increase with increasing biodegradation of the waste, were suggested as indicators (Bäumler et al. 2001). The ratio of cellulose (and hemicelluloses) to lignin (C(+H)/L) was also used to characterize waste stability, whereby a C/L-ratio of 0.2 was proposed as an indicator for stable waste (cf. Boda 2002; Kelly et al. 2006), as C/L-ratios between 0.02 and 0.2 were observed in old landfills with well degraded waste (Bookter and Ham 1982; Wang et al. 1994). Another method to indicate the extent of waste stabilization is the black index (cf. Cossu and Raga 2008), which is based on the reaction of lead acetate on a test paper with hydrogen sulfide (resulting in black lead sulfide on the paper). Further tests and indicators to determine the stabilization of organic waste include for example the FT-IR (Fourier Transform Infrared) spectroscopy (cf. Smidt et al. 2002), the paper cardboard content or the SUVA index (UV absorbance to DOC ratio (see Francois et al. 2006)). Although there are numerous tests and indicators available to evaluate waste stability, they are often not consistent with each other (Knox et al. 2005; Wagland et al. 2009), face challenges of representatively sampling and analyzing deposited waste (e.g. Chiampo et al. 1996; Sormunen et al. 2008), and the parameters cannot be directly linked to actual emission levels from the waste body (e.g. water flow heterogeneity and its implication for the emission behavior (see Fellner et al. 2009a)).

Although complete waste stabilization (i.e. the production of inert, solid residues) is desirable due to the absence of environmental risk at such landfills (no hazard), it is hard to achieve for existing landfills (especially with respect to landfills containing organic waste) and not yet measurable (e.g. Brunner 1992; Knox et al. 2005). Hence, for existing landfills the conditions at the landfill site will be of crucial importance to assess the environmental risk at a landfill. If the deposited material represents some level of hazard to the environment, the environmental setting and its vulnerability are essential to evaluate the environmental risk. The main elements of an evaluation of environmental compatibility of a landfill are the quality of the deposited waste, the level of interaction between waste and environment, and the vulnerability of the local environment (cf. Nienhaus 2001; Scharff et al. 2011). Thus, to assess landfill environmental compatibility in an integrated way, a dynamic, system-based concept of landfill stability is introduced as a basis for the evaluation of the environmental risk associated with a closed landfill. The suggested evaluation framework addresses the landfill hazard as well as the potential damage at the site.

A definition of dynamic stability has been suggested by Godoy (2000) and refers to the ability of a system to resist a disturbance and return to an equilibrium state. The following discussion and the underlying concept to evaluate long-term environmental compatibility of landfills is based on landfill stability as a system property to restore the original condition of a system after it has been disturbed from equilibrium or steady motion. Different aspects of this definition are listed below (adapted from Godoy (2000)):

- It is assumed that the environmental system is at equilibrium state before the landfill is built.
- The original system is disturbed by landfill emissions (interventions).
- The environmental system responds to the disturbance (environmental media are affected).
- There is an assessment of the system behavior following the deposition of waste.

The list above indicates the need for an evaluation of landfill stability in view of the specific environmental situation at the landfill site. Depending on the original condition of the system and the interactions of the waste with the surrounding environment, the landfill will cause a disturbance of the system, controlled by the deposited material, the local climate, the applied technology (i.e. landfill design and technical barriers), and the management of the landfill (e.g. collection and treatment of landfill leachate). As long as the landfilled material has different properties from its environment, there is some interaction between the landfill and its surrounding (e.g. landfill gas emissions, deterioration of barriers) which causes a response of the surrounding environment. The response could be a change of pollutant concentrations in different environmental media which might be tolerable or not based on the criteria applied to evaluate the system response. Hence, the assessment of the system response is related to a point of compliance, where certain criteria have to be fulfilled. A system is said to be stable, when the interventions (i.e. emissions) do not cause a response of the system which violates defined criteria at the specified point of compliance. Thus, the point of compliance represents the boundary of the relevant environmental system. The closer the system boundaries are placed to the landfill, the lesser buffering capacities of the surrounding environment are taken into account and the more conservative a respective assessment would be (provided the same criterion is applied). Apart from the spatial system boundary also a temporal system boundary has to be specified, during which the stability of the local environmental system is evaluated. This is the time period until the landfill does not represent a disturbance of the system any more. In conclusion, from an environmental perspective a stable landfill disturbs the environmental system only to such a degree that the response of the system at any time complies with defined criteria at the point of compliance.

From the discussion above it is evident that the evaluation of landfill stability includes several models which have to be established and linked to describe the system behavior over time:

- a pollutant release model of the waste to characterize the source term
- a model to describe the performance of the waste containment,
- a model to predict the substance concentrations in the affected environment

The methodological framework to evaluate landfill long-term management and assess environmental risks at a closed landfill based on the system-based concept of landfill stability is outlined below.

2.2.2 Framework to evaluate landfill PCC

The potential disturbance of the environmental system, as the ability to cause a response of the system which does not meet given criteria at the point of compliance, is illustrated schematically in Figure 2-2. It depends on the emission potential of the landfilled waste and the performance of technical barriers, which were built to control the interaction between waste and surrounding environment. The evaluation, if a potential disturbance is acceptable or not, is carried out with respect to the specific situation at the site (sensitivity of the system) and with respect to societal norms, forming the basis for delineating the relevant system (i.e. defining the point of compliance) and for specifying compliance criteria. A low disturbance will probably be acceptable in most environments even if high levels of protection are aspired for environmental assets, with the extreme of the waste itself not causing any potential disturbance (indicated in Figure 2-2 as inert waste). Medium disturbances might be tolerable under specific circumstances, e.g. in case of an insensitive environmental system and a lack of high valued environmental goods. However, a very high emission potential even combined with highly reliable containment systems might hardly be acceptable, if great societal value is assigned to environmental assets and a long-view on potential contamination is taken. This can be observed for instance with respect to nuclear waste repositories, as even the siting of low-level nuclear waste repositories is an intractable problem in most societies (cf. Sjöberg 2009).

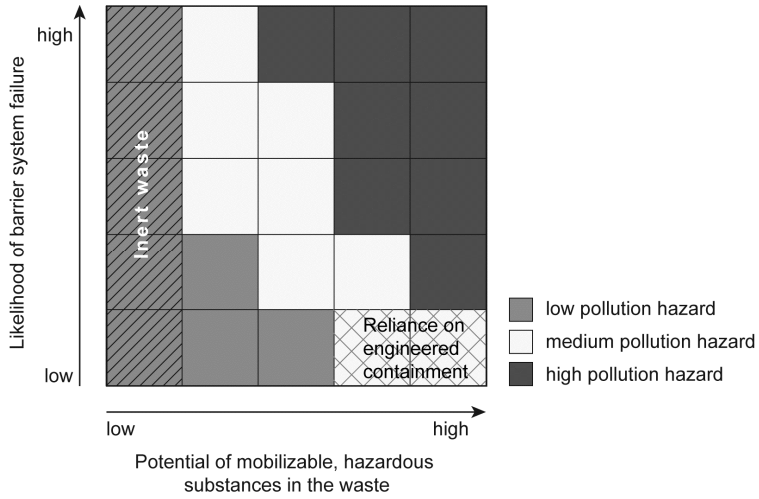


Figure 2-2: Evaluation matrix of the potential disturbance imposed on an environmental system by a landfill

The qualitative diagram in Figure 2-2 has to be transferred into a quantitative evaluation methodology to assess the response of the environmental system to a specific landfill. The response is evaluated by applying compliance criteria which depend on affected subjects of protection, the regulatory framework for protecting environmental goods (societal value), and the intended afteruse of the site. Some critical aspects within such a framework are discussed briefly below.

2.2.2.1 Landfill emissions

Information on the emission behavior of a landfill can be drawn from observed emission levels at full-scale landfills (e.g. Ehrig and Krümpelbeck 2001), experiments in the lab via landfill simulation reactors or waste leaching tests (e.g. Kylefors et al. 2003), physically and chemically based mathematical models (e.g. White et al. 2004), and the analysis of the behavior of natural analogues (e.g. Bozkurt et al. 2001). Based on this kind of data, a model for predicting the substance release from deposited waste can be established and an emission profile for the landfill can be developed. Whereas monitoring data of real landfills are rather limited with respect to the duration of the observation period, lab experiments can be used to simulate longer time periods via equivalent liquid-to-solid ratios (amount of water that passed through the waste relative to the waste mass). However, such approaches are facing limitations with respect to the reproducibility of the extent and rate of biochemical transformation processes, the preferential flow regime in landfills, and the heterogeneity of the waste itself. Fellner et al. (2009a) for instance, conclude that due to the differences in water flow and distribution between real-scale landfills and lab-scale simulations, leachate emissions from full-scale landfills tend to decrease faster than expected from laboratory experiments. Consequently, the stock of materials still remaining in the landfill body after a certain period is likely to be underestimated by lab-scale landfill simulations. Hence, landfill characterization limited to observed leachate and gas emissions is likely to underestimate the remaining emission potential in the landfill. Gas generation is dependent on prevailing conditions at a rather small scale, e.g. in dry zones of the landfill there might be easily degradable organic matter present, but there will be no significant gas generation due to the lack of moisture (cf. Klink and Ham 1982). Also leachate quality provides information only about the part of the landfill participating in water flow. Although observed emission data are a valuable basis for simple estimates on future emission levels as long as conditions within the landfill remain constant (e.g. by the extrapolation of observed trends), they do not provide an appropriate basis for estimating release rates with the prospect of potentially changing conditions such as a change of water flow regime in the waste or the intrusion of air into the waste body.

The combination of emission models and models on the future barrier performance will allow for determining future emission levels from a specific landfill (see Figure 2-3). The emission models need to take into account the conditions at the site (e.g. precipitation, waste deposition height), the pollutants of concern at the site (e.g. based on screening analysis and routine monitoring data), the actual performance of the system (e.g. current and historic monitoring data), and the quality of the deposited waste (e.g. mobilizable amount of substances). The models on future barrier performance are based on the resistance built into the system and the stresses induced on the system (Inyang 2004). This includes an evaluation of the specific system installed at the landfill, the available performance data, the system characteristics at the time of evaluation, and the potential impacts on the system expected at the site. A schematic illustration of the extrapolation of emission trends at the site, on the one hand, and the combination of different models within scenarios to illustrate the effect of changing conditions on landfill emission levels, on the other hand, is shown in Figure 2-3.

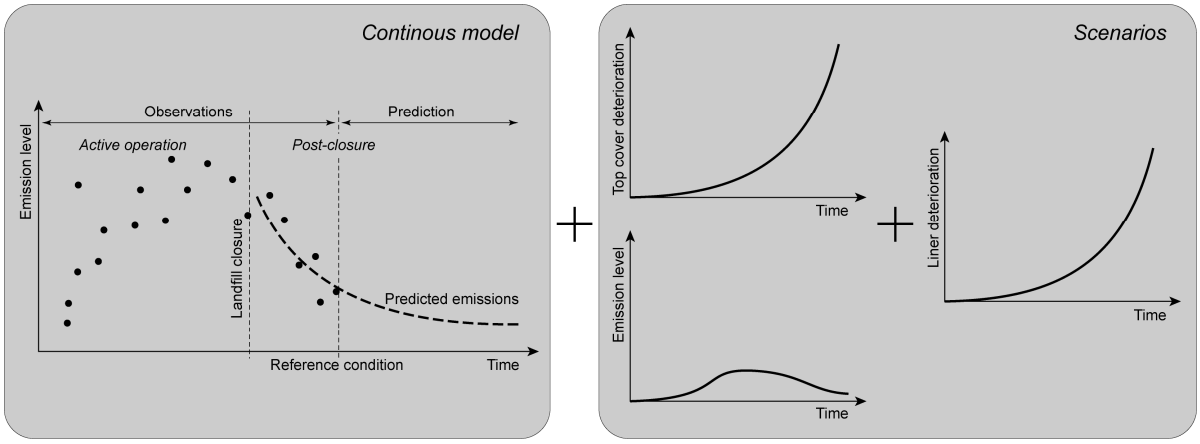


Figure 2-3: Combination of models to establish a disturbance term for the environmental system (the scheme is related to leachate emissions)

2.2.2.2 System response and vulnerability

If leachate seeps from a landfill (i.e. no containment is 100% effective) and enters the zone below the landfill, it potentially contaminates the groundwater underneath. Naturally occurring processes in these environmental media can attenuate contaminants present in the leachate. Although, natural attenuation (cf. U.S. Environmental Protection Agency 1997) has been shown to be substantial for leachate in the subsurface at different sites and for several pollutants (e.g. Christensen et al. 1994), there is no easy-to-use protocol available on how to assess attenuation potentials for a specific environment (Christensen et al. 2000). Nevertheless, to evaluate the environmental risk a landfill poses to the groundwater such processes should be included in the assessment, as they are occurring naturally. However, due to the potentially large uncertainty associated with the evaluation of natural attenuation processes, conservative assumptions should be used in respective models. The part of the environmental system which is accepted as a buffering zone in the evaluation is defined by the critical point of compliance (PoC). For instance, a PoC defined directly outside the containment system would exclude attenuation processes potentially occurring in the natural environment from the evaluation, whereas a PoC defined at some distance in the groundwater downstream of the landfill, would allow for the consideration of natural attenuation in the unsaturated and saturated zone in the evaluation. If considered, the potential for natural attenuation needs to be demonstrated at the site and confirmed by appropriate monitoring protocols. In addition to the release of leachate to the subsurface, the impact of leachate released to surface waters needs to be considered. However, as collected leachate will be managed according to regulatory discharge standards, the controlled discharge of leachate is considered to be compatible with the environment.

Landfill gas emissions are an issue at sites where waste containing degradable organic matter has been deposited (i.e. MSW landfills). Landfill gas is environmentally relevant on a global scale due to the contained methane as a potent greenhouse gas (global warming potential of CH_4 is 25 times higher as CO_2 (IPCC 2007)) and on a local scale due to potential vegetation damage, odors, and landfill gas migration off site (i.e. explosion hazards). Oxidation of methane has been reported for landfill cover systems in numerous studies (e.g. Oonk and Boom 1995; Huber-Humer 2004;

Chanton et al. 2010) with observed reductions of methane emissions during cover passage between 0 and 99 % (cf. Oonk and Boom 1995; Huber-Humer et al. 2008; Kühle-Weidemeier and Bogon 2008; Rachor et al. 2011). At many landfills with low methane generation rates (below 48 liters of CH_4 per day), an oxidation of at least 15% of the generated methane can be expected even without specifically designed top covers (Kühle-Weidemeier and Bogon 2008).

The vulnerability of the environmental system to the potential disturbance represented by the landfill is dependent on actual and planned uses at the site and the aspired level of protection for potentially affected subjects. The vulnerability of the system is reflected by the definition of the points of compliance, where certain quality standards have to be met, and by the stringency of the quality standards applied at these points. Hence, in order to evaluate the environmental compatibility of a landfill, a decision about the acceptable risk at the site has to be made (cf. Scharff et al. 2011). In essence, this decision is based on a societal consensus and expressed by the authorities, who decide on the extent and type of long-term management at the landfill site.

2.2.2.3 Approach to evaluate landfill PCC and environmental risk

Within the concept of stability, the landfill represents a (potential) disturbance of the environmental system. Corresponding stability criteria are applied in consideration of the response of the environmental system and its vulnerability. Consequently, to evaluate the environmental compatibility of a closed landfill, models to describe the behavior of the dynamic landfill-environment system have to be established and integrated to an evaluation method. The hypotheses underlying a respective evaluation is that it is possible to assess the environmental risk emanating from a landfill based on information about the deposited waste, the emission behavior of the landfill, and the current conditions at the site. A respective methodology represents a risk-based approach which is not limited to one functional state of the landfill, but which integrates the evolution of the landfill over time. Clearly, there are large uncertainties associated with such an approach, as system understanding is limited (e.g. natural variability, quality and quantity of measurements, process interactions, etc.) and future conditions are unknown. Among other factors, these uncertainties and the complexity of the evaluation might have hampered the development of an integrated methodology to evaluate long-term landfill management so far (e.g. Inyang and Lee 2005).

3 Emission behavior of landfills

The emission behavior of landfills is discussed for three different landfill types based on literature data and the analysis of data collected from landfills in Austria and Switzerland. Numerous studies have investigated emissions from MSW landfills and the results underlined their long-term environmental significance (e.g. Belevi and Baccini 1989; Kruse 1994; Krümpelbeck 2000; Kjeldsen et al. 2002; Heyer 2003). As many of these landfills have been closed during the last decade due to increasingly stringent regulations in the landfill sector and the ban on landfilling untreated MSW (EC 1999), MSW landfills are of particular interest with respect to post-closure care and long-term management concepts. In addition to emissions from MSW landfills, the emission behavior of municipal solid waste incineration (MSWI) bottom ash landfills is discussed in this chapter. Deposits of MSWI bottom ash are of growing importance due to increasing thermal waste treatment capacities in many parts of the world and because of the substances contained in and potentially emitted from these landfills (e.g. heavy metal contents of MSWI bottom ash). The third type of landfills addressed in this chapter is construction and demolition (C&D) waste landfills. C&D waste landfills are chosen for analysis, because approximately 25% of the waste to landfill is deposited at this type of landfill in Austria (Lebensministerium 2006) and leachate from C&D landfills needs to be treated during operation and potentially after landfill closure to comply with regulatory standards for direct discharge into surface waters.

3.1 Emissions from MSW landfills

3.1.1 General description of emission behavior

The metabolism of MSW landfills during operation and in the decades after landfill closure is determined by the biochemical degradation of organic matter, which results in the production of biogas (i.e. CH_4 and CO_2) and organically polluted leachate. In addition, the elution of soluble salts and nitrogen compounds (i.e. ammonium) represent major substance release processes (e.g. Farquhar and Rovers 1973; El-Fadel et al. 1997; Kjeldsen et al. 2002). One of the first schematic descriptions of MSW landfill emission characteristics was presented by Farquhar and Rovers (1973), see Figure 3-1. Based on the composition of landfill gas they distinguished four emission periods: An initial phase with aerobic degradation (Aerobic phase, I) is followed by an anaerobic phase without methane production (Acidic phase, II) after the oxygen within the deposited waste has been depleted. In the third phase methane generation is initialized, because accumulated organic acids are metabolized by methanogenic bacteria to CH_4 and CO_2. In the stable methanogenic phase (IV) methane concentrations are typically around 50 to 60% in the landfill gas and the concentration of easily degradable organic compounds in the leachate is low. The schematic sequence of emission characteristics has been confirmed by various studies and extended with respect to the long-term behavior of MSW landfills (e.g. Stegmann 1981; Ehrig 1986; Pohland and Harper 1986; Christensen and Kjeldsen 1989; Kjeldsen et al. 2002). However, the descriptions of emission characteristics after the stable methanogenic phase are not based on observed data at the lab- or field-scale, but on theoretical considerations. Therefore, the subsequent descriptions of

phases V-VIII are hypothetical and to some degree based on speculation. Due to the continuous decrease of degradable organic matter in the landfill, it is expected that air starts to infiltrate into the waste body during the methane oxidation phase (phase V in Figure 3-1). Whereas air intrusion is restricted to the landfill boundaries during the methane oxidation phase, the inner areas of the landfill get in contact with air during the air intrusion phase (VI) too. At some point the air flow into the landfill is sufficient to oxidize all the methane which is produced from residual anaerobic degradation. Subsequently, the remaining degradable organic matter is oxidized during aerobic degradation in the carbon dioxide phase (VII). In this phase the pH value might decrease causing increased metals mobility (cf. Kjeldsen et al. 2002). Finally, the composition of the landfill gas is equal to the composition of soil air in the last phase (soil air phase, VIII). It is speculated that it will take many hundred to several hundred thousands of years until this phase can be observed at real landfills, depending on the conditions at the site (e.g. waste composition, cover, degree of water saturation, landfill relief, etc.), see Bozkurt et al. (1999).

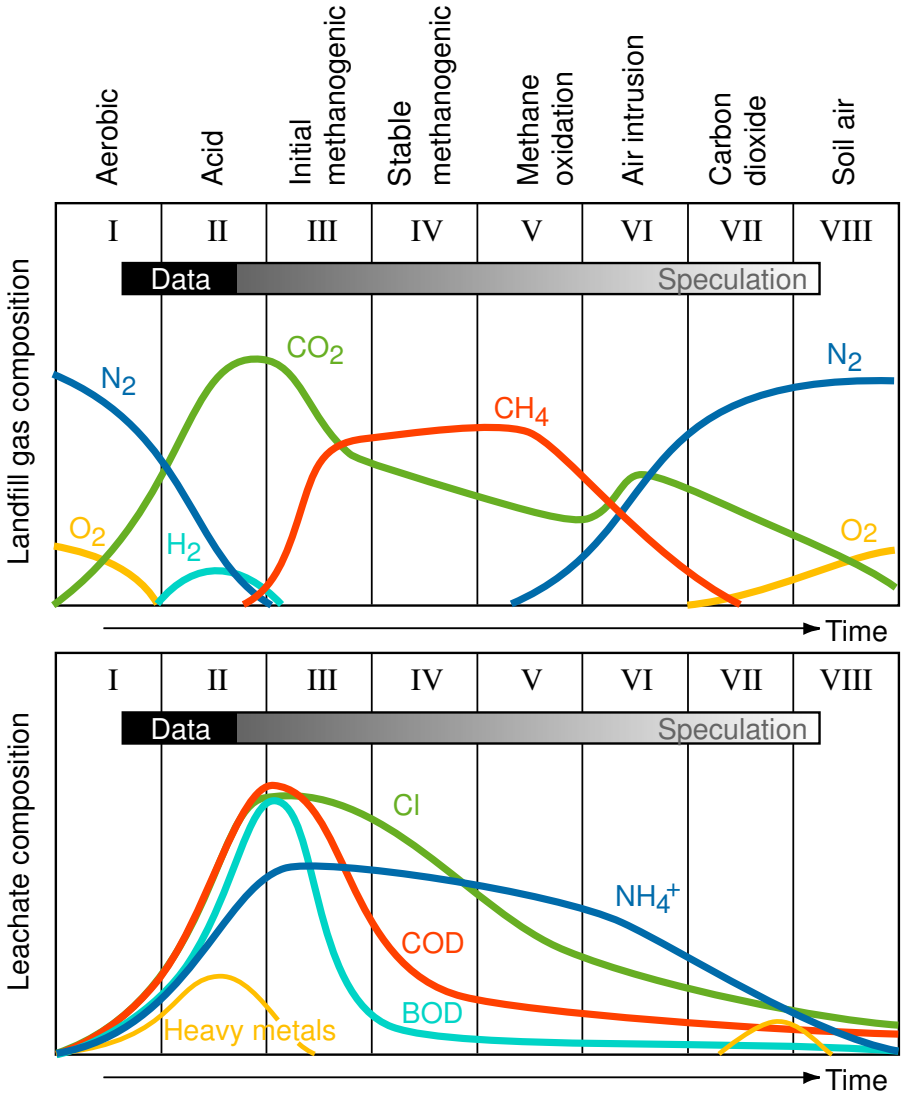


Figure 3-1: *Characteristics of landfill gas emissions (top) and leachate emissions (bottom) for an idealized MSW landfill (from Kjeldsen et al. 2002 based on Farquhar and Rovers 1973)*

It should be emphasized that the emission characteristics shown in Figure 3-1 and described above simplify the conditions at real-scale landfills and, hence, are to be understood schematically. Apart from different conditions at landfill sites (e.g. climate) and different characteristics of the landfilled waste (e.g. composition), several emission phases will be encountered at the same time at a landfill (cf. Ehrig 1991). This is due to the operation of landfills over an extended period of time (e.g. sections with old, well degraded refuse and younger sections with relatively fresh MSW) and the heterogeneity of the waste body. As the presence of mobile water is of major importance for the degradation of organic matter in landfills (e.g. Pohland 1975; Leckie et al. 1979), the heterogeneity of the water flow within landfills has been found to be of major importance for its emission behavior (cf. Öman and Rosqvist 1999; Huber et al. 2004; Rosqvist et al. 2005; Fellner et al. 2009a).

3.1.1.1 Landfill gas

Table 3-1: Selected mathematical models to estimate landfill gas generation at MSW landfills

Model formulation	Author suggestions
Tabasaran and Rettenberger 1987: $L_0 = 1.868*TOC_{deg}*(0.014*T+0.28)$ $G(t)_{cum} = L_0*(1-10^{-(t*k)})$	$k= 0.04\ yr^{-1}$ TOC_{deg} = 170-200 kg C/Mg MSW T= 30°C
Bundesamt für Umweltschutz (1983) (cited in Ehrig 1986): $G(t) = G_{max}*(1-e^{-(t/k_1)})*e^{-(t/k_2)}$	G_{max}= 13.61 m³/(Mg*yr) k_1= 1.1 yr (increasing trajectory) k_2= 15.72 yr (decreasing trajectory)
Ehrig (1986): $G(t) = G_{max}*e^{-k_1*(t_1-t)}$ $G(t) = G_{max}*e^{-k*(t-t_1)}$	L_0= 100-180 m³/Mg MSW $k= 0.139\ yr^{-1}$ $G_{max}= (L_0*k)$= 13.9-25.0 m³/(Mg*yr) $k_1= 3\ yr^{-1}$ (increasing trajectory) t_1= 3 yr (time of max. gas production)
LandGEM (U.S. EPA 2005): $G(t) = \sum_{i=1}^{n}\sum_{j=0.1}^{1} kL_0\left(\frac{M_i}{10}\right)e^{-kt_{ij}}$	Range of k values depending on landfill conditions (e.g. wet vs. arid)

Legend:
L_0: Landfill gas generation potential [m³/Mg], **TOC_{deg}**: degradable organic carbon [kg C/Mg MSW], **T**: Temperature [C°], **$G(t)_{cum}$**: Cumulative amount of generated landfill gas after t years [m³/Mg], **t**: time after deposition [yrs], **k**: first order rate [yr-1], **G(t)**: Landfill gas generation rate [m³/(Mg*yr)], **G_{max}**: Maximum landfill gas generation rate [m³/(Mg*yr)], **M_i**: mass of waste accepted in the i^{th} year [Mg]

The amount of landfill gas, which can be generated from municipal solid waste, depends on the composition of the waste. Based on theoretical estimates, gas generation potentials for MSW range between 120 and 450 liters per kg of dry MSW, whereas experimental studies indicate a range from close to zero up to 390 liters per kg of dry MSW (cf. El-Fadel et al. 1997). Numerous models have been presented to estimate the generation of landfill gas typically involving first order kinetics (cf. Barlaz et al. 1990). In Table 3-1 some landfill gas generation models are shown together with author suggestions on typical values for degradation rates and other model parameters. Typically, landfill gas generation models over-estimate gas generation rates and potentials at real-scale landfills (Fellner et al. 2004). Though such factors as the moisture content and distribution, temperature

fluctuations, the presence of inhibitors, and pH levels are of major importance for landfill gas generation (cf. El-Fadel et al. 1997), they are typically not addressed mechanistically by landfill gas generation models.

The major components of landfill gas are methane (40-70% of volume) and carbon dioxide (30-60% of volume). In addition, the volumetric contents of carbon monoxide, nitrogen, oxygen, hydrogen, hydrogen sulfide and trace gases can be in the range of several percent (cf. El-Fadel et al. 1997). For instance, Krümpelbeck (2000) reported levels of hydrocarbons between 1.8 and 166 mg/m³, chlorofluorocarbons between 0.06 and 119 mg/m³, or xylene from 0.04 to 257 mg/m³ in the landfill gas of closed German MSW landfills. It should be noted that the deposition of special wastes (e.g. industrial wastes) at the site may have a strong impact on trace gases present in the landfill gas (cf. Ehrig and Brinkmann 1999).

3.1.1.2 Landfill leachate

Leachate is generated from the preferential and transient flow of water through the deposited waste. The water balance at a landfill is dependent on the water content of the deposited waste, the precipitation at the site, the potential leachate recycling, the surface runoff, the evapotranspiration at the site, the biochemical degradation of waste, the water exported via leachate and landfill gas, and the water storage in the landfill (cf. Baccini et al. 1987). Typically, the water losses in the landfill gas as well as the water produced/consumed during biodegradation processes can be neglected at closed landfills under moderate conditions (Fellner 2004). The amount of leachate generated at closed landfills is dominated by the type and efficiency of the top cover system and the climatic conditions at the site. For instance, at old German landfills with earthen top covers of low thickness leachate generation amounted for 25 – 60% of the annual precipitation, at sites with thicker earthen covers leachate generation amounted for 15 – 40% of annual precipitation, and at sites with low-permeability clay liners leachate generation was in the range of 10 – 40% (Krümpelbeck 2000). Different top cover systems and their (long-term) effectiveness to control and minimize water infiltration are discussed in chapter 4.1.

The quality of leachate is dependent on the chemical composition of the deposited waste, the pH value and redox conditions, the water flow heterogeneity, as well as physical, chemical and/or biological transformation of the waste material. In Table 3-2 typical concentration ranges for leachate constituents are shown. It is apparent that concentrations vary over one to two orders of magnitude even during the same phase of landfill evolution. The differences between individual measurements and individual landfills may be due to the different waste compositions, different water infiltration rates (e.g. climatic conditions), different water flow distribution in the waste, or different landfill design (e.g. waste thickness) and management (e.g. leachate recirculation, compaction technology). For instance, the fraction of waste participating in water flow has been observed to vary from 60% to only 5% at different landfills (cf. Rosqvist and Bendz 1999; Döberl et al. 2005), which is highly significant for leachate emission levels as well as long-term emission potentials of these landfills (e.g. Rosqvist et al. 2005; Fellner et al. 2009a).

Table 3-2: Typical concentrations of selected leachate constituents during the acidic and methanogenic phase at a MSW landfill (from: Ehrig 1980 cited in Ehrig and Brinkmann 1999)

	Acidic Phase		Methanogenic phase		Independent from phase		
Parameter	**Mean**	**Range**	**Mean**	**Range**	**Parameter**	**Mean**	**Range**
pH [-]	6.1	4.5 – 7.5	8.0	7.5 – 9.0	AOX [mg/l]	2	0.32 – 3.35
BOD_5 [mg/l]	13000	4000 – 40000	180	20 – 550	Cl [mg/l]	2100	100 – 5000
COD [mg/l]	22000	6000 – 60000	3000	500 – 4500	NH_4-N [mg/l]	750	30 – 3000
SO_4 [mg/l]	500	70 – 1750	80	10 – 420	NO_3-N [mg/l]	3	0.1 – 50
Ca [mg/l]	1200	10 – 2500	60	20 – 60	NO_2-N [mg/l]	0.5	0 – 25
Mg [mg/l]	470	50 – 1150	180	40 – 350	P_{tot} [mg/l]	6	0.1 – 30
Fe [mg/l]	780	20 – 2100	15	3 – 280	Cd [mg/l]	0.006	0.0005 – 0.14
Mn [mg/l]	25	0.3 – 65	0.7	0.03 – 45	Cr [mg/l]	0.3	0.03 – 1.6
Zn [mg/l]	5	0.1 – 120	0.6	0.03 – 4	Cu [mg/l]	0.08	0.004 – 1.4
Sr [mg/l]	7	0.5 – 15	1	0.3 – 7	Ni [mg/l]	0.2	0.02 – 2.05

Although there are large variations at individual MSW landfills, a decreasing trend of concentrations can typically be observed for many leachate constituents with increasing landfill age, especially after landfill closure (e.g. Krümpelbeck 2000; Kjeldsen et al. 2002; Gibbons et al. 2007). The organic pollution of the leachate is often expressed as total organic carbon (TOC), dissolved organic carbon (DOC), chemical oxygen demand (COD), and biological oxygen demand in 5 days (BOD_5). The BOD_5/COD ratio has been suggested as an indicator for the extent of waste degradation, with a BOD_5/COD values below 0.1 indicating stabilized waste (cf. Ehrig 1989; Reinhart and Townsend 1997). However, this is a necessary but insufficient criterion, because easily degradable substances present in the leachate from less stable waste could be consumed while percolating through well degraded refuse at the bottom of the landfill (cf. Bookter and Ham 1982; Barlaz et al. 2002; Youcai et al. 2002). Thus, the leachate quality at the landfill bottom does not necessarily reflect to state of waste degradation, not even for the part of the landfill in contact with mobile water.

Apart from parameters for total organic leachate pollution, also the levels of specific organic compounds or groups of organic compounds have been measured for MSW landfill leachate. Frequently detected xenobiotic compounds include benzene, toluene, ethylbenzene, and halogenated hydrocarbons such as trichloroethane or tetrachloroethylene (Arneth et al. 1989; Kjeldsen et al. 2002). A model of the long-term behavior of xenobiotic organic leachate constituents has been suggested by Kjeldsen and Christensen (2001), which indicates that these substances may persist in landfill leachate for many decades. In general, a lot of different xenobiotic compounds have been identified in landfills (e.g. Öman and Junestedt 2008) and as persistent organic pollutants have been accumulated in consumer wastes, they may also be present in landfill leachate (Weber et al. 2011).

For most inorganic macrocomponents (e.g. chloride, sulfate) decreasing concentration trends can be observed in the leachate of MSW landfills after closure (Kjeldsen et al. 2002; Gibbons et al. 2007). As the concentrations of ammonium decrease only slowly or remain in the same range for decades, ammonium has been identified as a critical leachate constituent in terms of post-closure care and environmental compatibility (e.g. Kruse 1994; Krümpelbeck 2000; Kjeldsen et al. 2002).

Heavy metals are typically present at very low levels in MSW landfill leachate with concentrations in the range of or below drinking water quality standards (Kjeldsen et al. 2002). Although significant quantities of heavy metals are contained in MSW, release via leachate is very low due to low

solubility of present species, sorption, and other processes decreasing metals mobility (cf. Bozkurt et al. 1999; Döberl et al. 2002). As a decrease of pH and a consequent increase of metals mobility is speculated not to occur for thousands of years at MSW landfills (Belevi and Baccini 1989; Bozkurt et al. 2000), metals are primarily mobilized due to the complexation with organic and inorganic compounds, and due to metal association with colloids (Kjeldsen et al. 2002). Hence, heavy metal concentrations in leachate from MSW landfills are typically low, but may vary from one landfill to another, as well as from one landfill phase (i.e. environmental conditions such as redox in the waste body) to another.

3.1.2 Emission characteristics of MSW landfills

The emission profiles of the selected MSW landfills are investigated to describe the range of emission levels at Austrian MSW landfills, to analyze emission trends at closed or old MSW landfills, and to evaluate the emission potentials remaining in these landfills. The analysis is based on reported data on existing landfills (e.g. Lebensministerium 2006; Fellner et al. 2009b) and data available from landfill owners and authorities. Out of approximately 100 MSW landfills in Austria (Fellner et al. 2009b), 50 sites were chosen for further investigations, because of the preference for older landfills (age above 10 years) and for sites with a functional waste containment system (i.e. potential availability of monitoring data). Landfill owners have been contacted and further data on the specific landfills and emission monitoring data were gathered. Finally, 32 sites[2] were chosen to analyze emission profiles of older MSW landfills (see appendix 1). Together with landfill owners and authorities data on deposition history, climatic conditions, landfilling technology (intermediate covers, base liner, gas collection, etc.), and emissions (leachate and gas) were collected. All of the investigated sites are equipped with a lining system at the landfill base to collect the leachate percolating through the waste body. The average annual precipitation rate at the sites is 832 mm/yr and the average age of the landfills is 24 years (in 2008), with most of the landfills (30) being closed (see appendix 1). By 2008 the average duration of post-closure care at these landfills has been 16 years. In Table 3-3 the mean, minimum and maximum concentrations of selected leachate parameters are shown in comparison to current Austrian standards for leachate discharge into surface water bodies. Table 3-3 includes parameters, for which more than 5% of the measurements exceeded the stipulated discharge limits together with information about the measured pH values, the electric conductivities, and chloride concentrations. From Table 3-3 it is evident that the mean values of organic leachate pollution (chemical oxygen demand (COD), biological oxygen demand in 5 days (BOD_5), and total organic carbon (TOC)) and of ammonium-nitrogen concentrations are two orders of magnitude above the respective Austrian quality standards for direct discharge to surface water bodies and consequently, leachate needs to be treated (either on-site or in a waste water treatment plant off-site) before discharge. The limit values are exceeded also by the mean concentrations of iron, absorbable organic halides (AOX), and chromium. In addition, the relatively high chloride concentrations of 1390 mg/l should be noted. Although there is no concentration-based discharge limit, these concentrations are substantially higher than target values suggested for the termination of leachate management during post-closure care, for example a target value of 100

[2] 29 MSW landfills are located in Austria, 3 in Switzerland.

mg Cl per liter suggested by Stegmann et al. (2006). In general, the concentration levels shown in Table 3-3 are in accordance with the literature data discussed previously. Organic leachate pollution and ammonium concentrations are high and most critical with respect to the corresponding discharge limits. Leachate concentrations of chloride, AOX or iron might as well be problematic at many sites, whereas most heavy metal concentrations are typically below the discharge standards. However, the identification of potentially critical parameters is limited by the range of parameters measured at the sites. On average 50 parameters were determined for leachate samples at the sites mainly based on the regulatory requirements in Austria (see appendix 1). An analysis of trace organics and persistent chemicals or rare metals (e.g. Rubidium) is typically not included in routine monitoring and not covered by the analysis presented in this section. In addition, it should be noted that the emission characteristics of an individual landfill might diverge substantially from the data shown in Table 1, due to varying waste compositions, different conditions at the site (e.g. climate or technology) or the co-disposal of wastes other than MSW (cf. Gibbons et al. 1999; Slack et al. 2005).

Table 3-3: Statistical analysis of leachate characteristics of 32 MSW landfills and Austrian discharge limits into surface water bodies

Parameter	Number of landfills	Number of measurements	Mean*	Min. – Max.	Discharge limit (surface water)**
pH [-]	32	1262	7.9	5.5 – 9.6	6.5 – 8.5
EC [µS/cm]	32	1223	14055	352 – 504000	-
COD [mg O_2/l]	31	1152	4203	18 – 57000	50
BOD_5 [mg O_2/l]	30	964	1581	3 – 32000	10
TOC [mg C_{org}/l]	24	703	1460	0.5 – 28150	20
NH_4-N [mg N/l]	32	1284	1045	1.1 – 6200	10
Cl [mg/l]	31	1060	1388	11.7 – 9700	Toxicity
AOX [mg Cl/l]	27	626	1.71	0.01 – 38.4	0.5
Fe [mg/l]	27	792	29.2	ND – 3700	2.0
Zn [mg/l]	30	925	0.48	ND – 14.9	0.5
As [mg/l]	24	644	0.04	ND – 0.64	0.1
Cr_{tot} [mg/l]	28	803	0.58	ND – 9.2	0.5
Ni [mg/l]	24	561	0.18	ND – 8.8	0.5
Sulfide [mg S/l]	4	64	0.41	ND – 4	0.5

* A non-detect (ND) is considered 0 for calculating the mean value.

** Limits according to the ordinances on waste water discharge from landfills (Leachate ordinance (MoE 2003) and waste water discharge ordinance (MoE 1996a)).

The leachate generation rate at the investigated MSW landfills is primarily dependent on the landfill management period. During landfill operation on average 40% of the annual precipitation was collected as leachate, at sites with a temporary cover (typically a layer of soil) the mean leachate generation rate accounted for 23% of annual precipitation, and after the installation of a final cover the mean leachate generation rate decreased to 8% of annual precipitation. In general, leachate generation rates were on the decrease at sites with a final cover, due to the continuing drainage of wa-

ter stored within the waste body. However, due to different systems and conditions at the site, general statements on leachate generation rates are of limited value for the evaluation of individual landfills.

Out of the 32 MSW landfills in the data set, monitoring data on landfill gas emissions is available for 13 (see appendix 1). The data is of limited informative value, as time series are often incomplete, especially during different landfill management periods, cf. Figure 2-1, and the quality of data seems to be poor for many sites, e.g. unknown collection efficiency of the gas extraction system, suction of air into the gas extraction system. However, the mean concentrations of methane, carbon dioxide, and oxygen in the landfill gas collected at 13 MSW landfills were 50, 40, and 2 Vol%, respectively[3], which are typical values for MSW landfills during the stable methanogenic phase (cf. Figure 3-1). The cumulative amounts of landfill gas collected at the landfill sites (by the year 2008) range from 16 m³ of landfill gas per Mg of MSW to 162 m³ of landfill gas per Mg of MSW. Both extremes of this range relate to old landfills (36 and 28 years, respectively), but whereas active gas extraction started 22 years after initial waste deposition at the landfill with low cumulative gas yield (MSW4-MSW11), gas has been collected already three years after initial waste placement at the landfill with the highest cumulative yield (MSW1). This underlines the importance of a timely installation of gas collection systems to capture a large fraction of the generated gas and highlights the large potential variation of collection efficiencies between individual sites. In any case, landfill gas data should be handled cautiously and evaluated with respect to representativeness and trends over time.

3.1.2.1 Evolution of emission characteristics after landfill closure

The average concentrations and concentration ranges for selected leachate parameters for specific time periods after landfill closure are shown in Table 3-4. For most of the parameters the concentrations decrease with increasing post-closure care duration. For instance, the mean values of organic leachate pollution parameters (BOD_5, COD, TOC, AOX) are almost by an order of magnitude higher in the initial five-year period after closure than 20 years and more after the landfill has been closed (cf. Table 3-4). Similar trends can be observed for ammonium, iron or chloride. However, after more than 20 years post closure the mean values of ammonium or COD in the data set are still an order of magnitude above the corresponding quality standards for direct discharge into surface waters (see Table 3-3). Only the BOD_5/COD ratio does not exhibit any significant trend during the post closure care period in Table 3-4. Primarily, this is due to very high variations at a few individual landfills, which do not allow for a meaningful calculation of average BOD_5/COD rations for the data set.

[3] Volumetric concentrations measured at each site have been corrected for the suction of air based on the content of Nitrogen (N_2) in the landfill gas. High volumetric contents of Nitrogen infer significant suction of air into the gas extraction system.

Table 3-4: Statistical analysis of selected MSW landfill leachate parameters for consecutive time periods after landfill closure

Parameter	Mean	Min-Max	Mean	Min-Max
	0-5 years after closure		**5-10 years after closure**	
pH value [-]	8.0	6.9-8.8	7.9	7.0-8.8
COD [mg O_2/l]	3008	182-11618	1510	213-4680
BOD_5 [mg O_2/l]	565	19-4157	251	3-1671
BOD_5/COD [-]	0.2	0.01-0.58	0.2	0.004-0.86
TOC [mg C_{org}/l]	906	6-2700	670	65-3100
NH_4-N [mg N/l]	926	13-3287	571	5.8-2133
Chloride [mg Cl/l]	1812	140-5843	1485	122-6040
AOX [mg Cl/l]	3.0	0-35	1.08	0.1-3.9
Fe [mg/l]	6.1	0.01-70.9	3.6	0.68-8.7
Cr_{tot} [mg/l]	0.83	0.03-5.2	0.29	0.01-1.77
	10-20 years after closure		**>20 years after closure**	
pH value [-]	7.7	6.8-8.6	7.7	7.0-8.6
COD [mg O_2/l]	1028	18-6189	642	100-2530
BOD_5 [mg O_2/l]	140	3-890	93	24-240
BOD_5/COD [-]	0.2	0.006-0.58	0.2	0.03-0.70
TOC [mg C_{org}/l]	498	18-3000	-	-
NH_4-N [mg N/l]	369	2.8-2800	181	33-856
Chloride [mg Cl/l]	981	62-4500	537	116-2980
AOX [mg Cl/l]	0.9	0.05-6.3	0.39	0.08-0.79
Fe [mg/l]	4.3	0-45.0	1.03	0.33-2.31
Cr_{tot} [mg/l]	0.24	0.04-1.14	0.13	0.02-0.80

Figure 3-3 illustrates the concentrations of ammonium-nitrogen (left) and chloride (right) in the leachate of closed MSW landfills with increasing post-closure period. The average concentrations of ammonium-nitrogen gradually decrease from around 1000 mg/l subsequent to landfill closure to less than 100 mg/l 30 years post-closure. For chloride too, concentrations decrease from 1000-2500 mg/l directly after closure to 200-300 mg/l after 30 years of post-closure monitoring. Similar trends can be observed for BOD_5, COD, Fe or AOX concentrations in the leachate, but are not shown here (however, this is indicated by the aggregate analysis in Table 3-4). It should be noted, that the concentration data at landfills closed for more than 20 years refer to a few landfills only and might not be representative for each landfill included in the initial data set. Hence, although the analysis reflects some typical emission characteristics and trends at MSW landfills, it is not directly transferable to the behavior of individual landfills. For such an analysis, site-specific aspects like the water infiltration rate (i.e. climate, level of interaction), the waste deposition thickness and technology, and the heterogeneity of water flow need to be considered. Thus, the liquid-to-solid

ratio[4] may be better suited than landfill age as an independent variable to analyze emission behavior of MSW landfills (see Allgaier and Stegmann 2003; Heyer 2005). The concentrations of NH_4-N and Cl in the leachate after landfill closure are plotted against the liquid-to-solid ratio of the deposited waste in Figure 3-3. Decreasing concentrations can be observed with increasing liquid-to-solid-ratio (L/S) for both parameters. The concentrations vary across two orders of magnitude at individual landfills with similar L/S of the deposited waste. The large bandwidth indicates that factors such as the quality of waste, water management during landfill operation, and the heterogeneity of water flow may have a strong influence on actual leachate quality. Therefore, to evaluate the emission behavior of a landfill and to make meaningful predictions on leachate emissions, specific conditions at the site are to be considered. This is valid also for the transfer of results from laboratory experiments to field-scale conditions based on the L/S of the deposited waste (cf. Kylefors 2003; Fellner et al. 2009a).

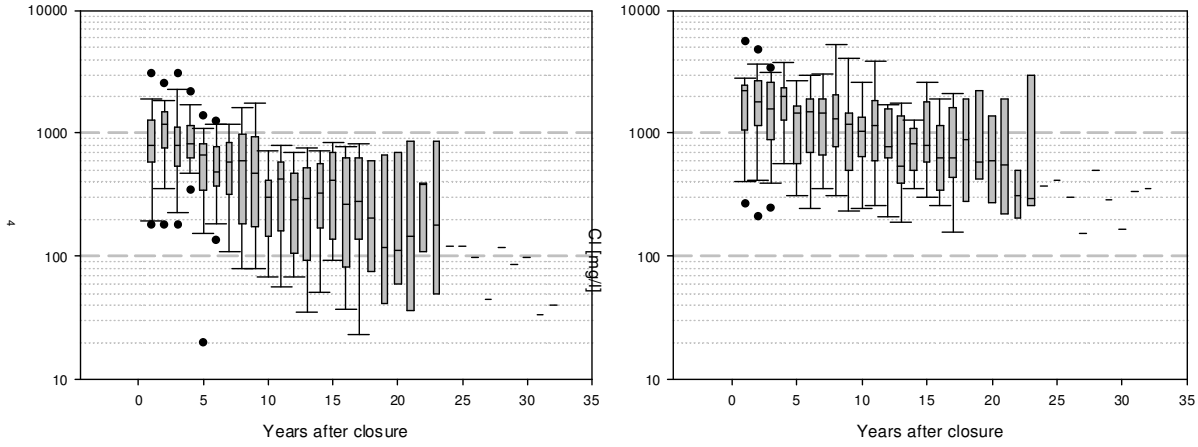


Figure 3-2: *Box plots[5] of ammonium-nitrogen concentrations and chloride concentrations in the leachate as a function of time after landfill closure*

Landfill gas data is available for 40% of the MSW landfills (see appendix 1), whereby MSW4-MSW11 have one common system for landfill gas extraction (only aggregate data available). In Figure 3-4 the cumulative amount of landfill gas collected per dry waste mass is shown for six gas extraction systems. The systems have been installed at different times after landfill operation had been started at the site and the amount of gas collected indicates strongly varying collection efficiencies. Whereas more than 100 m³ of landfill gas per ton of dry waste could be collected at the sites MSW1, MSW2, and MSW3, less than 30 m³ of landfill gas per ton of dry waste have been

[4] The liquid-to-solid ratio (L/S) after t years is calculated as: $\frac{L}{S}(t) = \frac{I_t}{m_{DM}}$, where I_t is the amount of water that passed through the landfill after t years [l] and m_{DM} is the total dry mass of waste deposited at the site [kg] (for a closed site).

[5] Box plots: The box is limited by the 25th and 75th percentile of the measurements and the median is shown as a horizontal line within the box. The ends of the whiskers indicate the 10th and 90th percentile. The black dots represent the 5th and 95th percentile. At least 3 values are required to calculate the box, 9 values to display the whiskers, and 18 values to calculate 5th and 95th percentiles.

collected at MSW4-MSW11, MSW16, and MSW18. As the composition of waste at these sites is assumed to be similar, the differences are mainly attributed to the landfill gas management at the sites. In any case, based on a typical landfill gas generation potential at MSW landfills of 280 m³/Mg DM (cf. Krümpelbeck 2000), even at sites with high specific landfill gas yields less than 50% of the theoretical landfill gas generation potential has been collected. Although there is still gas collected at the sites, the cumulative amount of collected landfill gas is not expected to increase significantly in the future as gas collection rates are on the decrease and will further decrease after final cover installation (see marked periods of the cumulative curves in Figure 3-4).

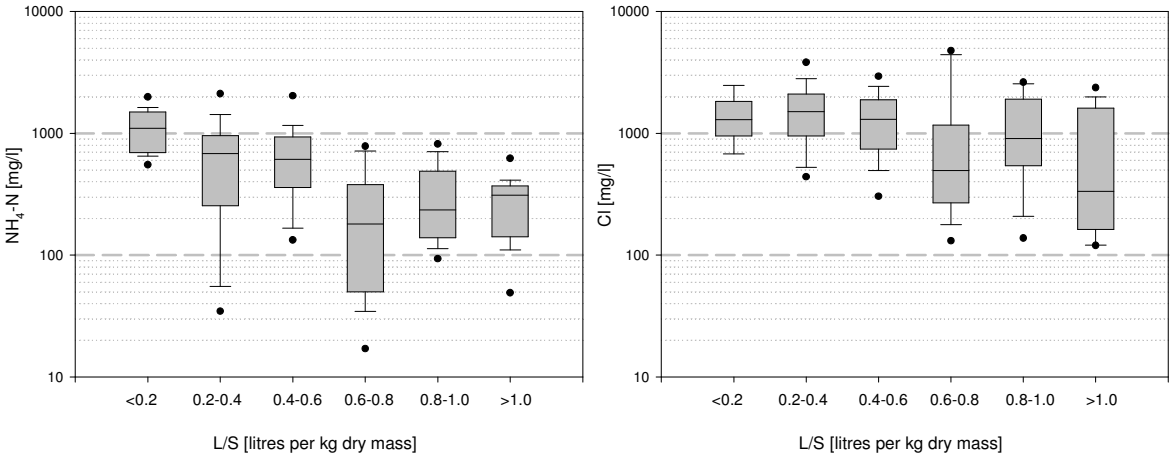


Figure 3-3: *Box plots of ammonium-nitrogen concentrations and chloride concentrations in the leachate as a function of the deposited waste's liquid-solid-ratio (L-S) at closed landfills*

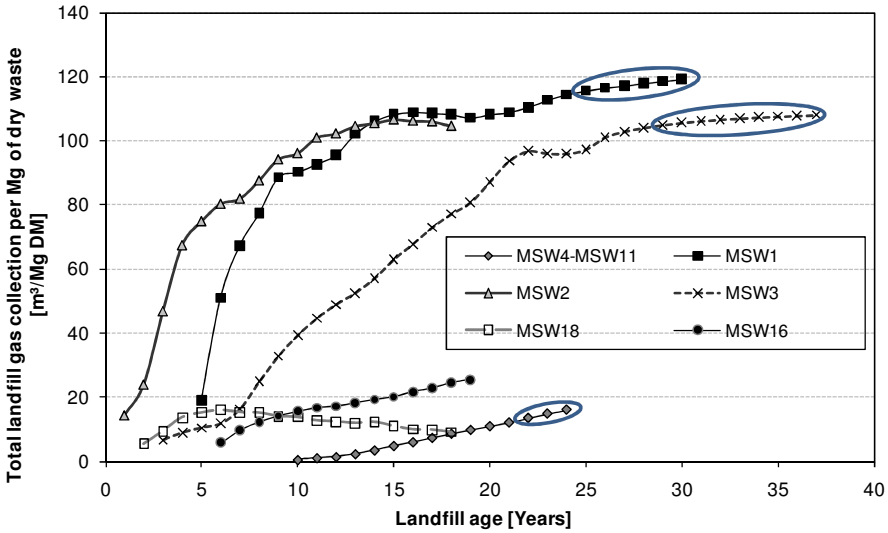


Figure 3-4: *Total amount of collected landfill gas per Mg of dry waste at selected MSW landfills (collection periods after landfill closure are marked)*

3.1.2.2 Emission loads and remaining substance potentials

Apart from a decrease in pollutant concentrations in landfill leachate after landfill closure, the amount of leachate typically decreases too after the installation of a top cover. Thus, the decrease is

even more pronounced for substance loads in the leachate than for substance concentrations. The annual substance load in the leachate is calculated as the product of the amount of leachate [m³/yr] and the average concentration of the substance in the leachate [g/m³]. The specific (per ton of dry waste) annual loads of nitrogen (primarily as ammonium-nitrogen) and chloride are plotted against landfill age for two well documented landfills in Figure 3-5. The indicated uncertainty ranges of the loads are based on an evaluation of the quality of data on the amount and composition of leachate. Whereby criteria, such as the completeness of time series data, the level of variation in leachate quality data, the number of measurements per year (for calculation of average values), or the method to measure leachate generation have been translated to quantitative uncertainty ranges for the available data. These estimates form the basis for the uncertainty ranges associated with the annual loads shown in Figure 3-5. The annual loads of N and Cl decrease from around 100 g/(Mg.year) during initial landfill operation to 10-50 g/(Mg.year) after the end of waste deposition. As both sites have been equipped with a temporary cover (soil layer of >2 meters thickness) after closure, the leachate generation rates during the post-closure period still amounted for around 30% of annual precipitation at MSW1 and around 15% of annual precipitation at MSW3. The installation of more or less permeable covers aimed at a continuous reduction of the emission potential remaining in the landfill before the final (low-permeability) cover is installed. After the installation of the final cover the substance loads in the leachate are expected to decrease even faster than previously observed after landfill closure. That is confirmed by significantly lower substance loads of N and Cl around 1 g/Mg DM at landfills with a final cover already in place for several years (e.g. MSW4 and MSW5).

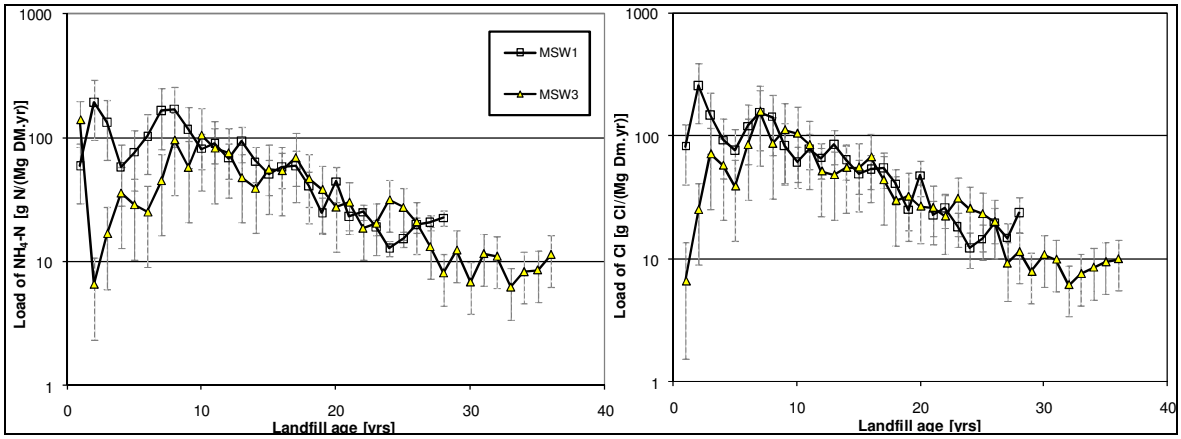


Figure 3-5: Specific annual loads of nitrogen (left) and chloride (right) for two well documented MSW landfills (MSW1 and MSW3) plotted against landfill age

Decreasing loads, especially after landfill closure, can be observed for many substances in the leachate (e.g. NH_4-N, Cl, C_{org}, Fe, Cr, etc.). However, the substance potential remaining in the landfill body after closure might be substantial even for the sites MSW1 and MSW3, where a temporary cover has been installed to continuously decrease the emission potential of the deposited waste after closure. To determine the remaining amount of selected substances in the landfills MSW1 and MSW3, the methodology of material flow analysis (Brunner and Rechberger 2004; ÖNORM 2005) is used. Substance balances are established for C_{org}, N, Cl, Fe, Zn, and Cd from the

beginning of waste deposition until 2007 (last year with a complete monitoring data set). The initial composition of the deposited waste is determined based on data for typical MSW composition in Austria and Switzerland during the last decades (see Baccini et al. 1987; Fehringer et al. 1997; Rolland and Scheibengraf 2003). Leachate monitoring data are available for the whole period of landfill existence at both sites. Landfill gas collection systems have been installed three years (MSW1) and two years (MSW3) after initial waste deposition. Apart from the amount of landfill gas produced during the first years, also the collection efficiency of the system is estimated. The gas generation rates before active collection are derived from a landfill gas generation model. The efficiency of gas collection is estimated based on measurements of fugitive emissions, on the one hand, and on collection efficiencies at other sites for those periods without fugitive emission monitoring, on the other hand. At MSW1 between 50 and 89% of the annually generated landfill gas have been collected with an average gas collection efficiency of 60% throughout the period of gas extraction. At MSW3 around 50 to 80% of the generated gases have been collected annually with an average gas collection efficiency of 52% throughout the period of gas extraction. Although uncertainties have been considered in the substance balance calculations (cf. Figure 3-5), they are not shown in Figure 3-6 and Figure 3-7. Nevertheless, it should be emphasized that the substance balances for MSW1 and MSW3 are based on uncertain and incomplete data (e.g. waste composition, average concentrations, data gaps in time series information, measurement errors). The results of the analysis provide an indication of remaining substance potentials in the waste body at these sites, but should not be regarded as exact numbers.

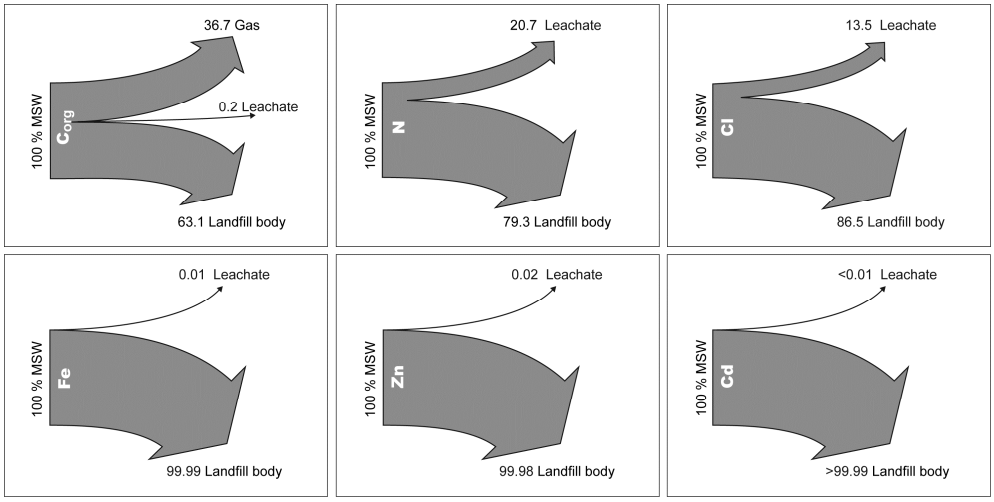


Figure 3-6: Substance balances for MSW1 with an average waste age of 19 years and a L/S ratio of 1.38 l/kg DM

Seven years after closure of the landfill MSW1 around 63% of the originally deposited organic carbon is still present in the waste (see Figure 3-6). Approximately 37% of the organic carbon has been exported via landfill gas and only a small fraction (0.2%) of C_{org} left the landfill in the leachate. For the other substances, leachate was investigated as the only emission pathway. In Figure 3-6 it is shown that 21% of the total nitrogen and 14% of the total chloride have been extracted during the balancing period. Thus, most of these substances are still contained in the deposited waste. Almost the whole deposited amounts of Fe, Cd, and Zn (99.98 to >99.99%)

remain in the landfill at an average waste age of 19 years. It can be stated, that in spite of the relatively high liquid-to-solid ratio of the deposited waste (1.38 l/kg DM) most of the substances (more than 60% of C_{org} to >99.99% of Cd) originally deposited remain in the waste of landfill MSW1 at a landfill age of 28 years.

At the landfill MSW3 a fraction of 35% of the organic carbon has been exported via landfill gas and 0.2% of organic carbon has left the landfill in the leachate (see Figure 3-7). Consequently, around 65% of the originally deposited organic carbon is still present in the landfill 36 years after the first waste deposition at the site. In Figure 3-7 it is shown that leaching removed approximately 14% of the deposited amount of nitrogen and 9% of the initial amount of chloride contained in the waste. The leached substance fractions of N and Cl at MSW3 are lower than at MSW1, which is in accordance with the lower liquid-to-solid ratio of the waste at MSW3 (0.45 l/kg DM) relative to MSW1 (1.38 l/kg DM). However, with respect to iron, zinc, and cadmium, the substance balances of MSW1 and MSW3 are similar, as at least 99.98% of the deposited substances remain in each of the landfills.

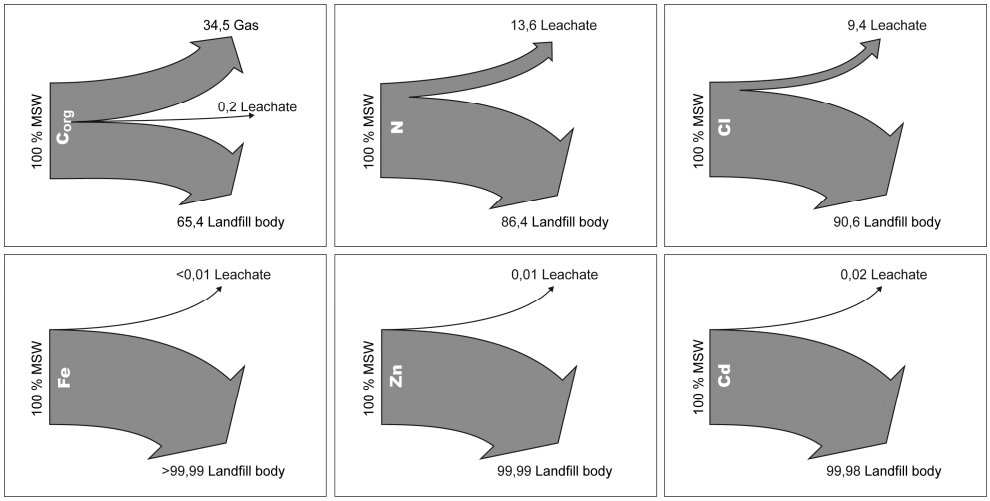


Figure 3-7: Substance balances for MSW3 with an average waste age of 23 years and L/S ratio of 0.45 l/kg DM

The substance balances in Figure 3-6 and Figure 3-7 illustrate that large fractions of the originally deposited materials are still present in the landfills after several decades (average waste age is 19 years at MSW1 and 23 years at MSW3), even though temporary covers were installed to allow for a further reduction of emission potentials after the end of waste deposition at these sites. After the installation of a final, impermeable cover, in 2008 at both sites, the substance loads are expected to decrease and the remaining substance potentials will persist for long times within the landfills. Thus, the material stock present in a closed landfill needs to be investigated with respect to its significance for mid- and long-term emission characteristics. A major aspect of such an assessment is the potential of mobilizable substances in the waste, whereby the mechanism of mobilization as well as the environmental conditions relevant to a mobilization may be different for different time-frames (e.g. Bozkurt et al. 2000; Kjeldsen et al. 2002). To estimate future emission levels, the mobilizable substance potentials need to be related to substance release rates. Different approaches to estimate future emissions from MSW landfills are discussed in the next section.

3.1.3 Evaluation of mid- to long-term emissions

Different approaches can be used to evaluate the mobilizable substance potentials of deposited MSW and consequent emissions from MSW landfills. The mobilizable substance potential is a function of the approach and timeframe of investigation and the considered release mechanisms. Therefore, mobilizable potentials may range from the total concentration of a substance in the waste (e.g. long-term erosion is considered as a mechanism) to the substance fraction readily mobile at the time of investigation (e.g. the directly leachable fraction of chloride). Approaches to determine mobilizable substance fractions of deposited waste have been categorized by Döberl (2004) into

- observations at existing MSW landfills (full-scale or pilot-scale),
- laboratory experiments (landfill simulation reactors, elution tests, etc.),
- mathematical models based on physical-chemical descriptions of landfill processes (different levels of complexity),
- observations of processes at analogous natural or anthropogenic systems (e.g. peat).

Monitoring data at MSW landfills (with deposited materials comparable to modern MSW) are available for several decades. Direct observations of landfill behavior are therefore restricted to such a timeframe (cf. Zweifel et al. 1999). Nevertheless, estimates on mid- to long-term emission behavior have been based on extrapolations of emission trends observed at full-scale MSW landfills (e.g. Krümpelbeck 2000; Gibbons et al. 2007). In many cases laboratory experiments have been used as a basis to evaluate the emission behavior of MSW landfills. The transformation of laboratory results to real-scale landfills based on the L/S ratio of the deposited waste is a possible approach to make predictions about future landfill behavior (e.g. Stegmann 1981), whereby such issues as the scale-dependency of water flow heterogeneity or the potential of accelerating biodegradation processes in the laboratory reactor need to be considered in such an evaluation (cf. Kylefors et al. 2003; Fellner et al. 2009a). Results from laboratory experiments are also used to establish mathematical models on the emission behavior of MSW landfills. Models range from an exponential decrease of substance concentrations in the leachate based on leachable substance fractions determined in an elution test (e.g. Belevi and Baccini 1989) to more complex models considering various processes in the landfill in a spatially distributed way (e.g. White et al. 2004). In general, emission models have hardly been calibrated with field data yet and the validation of model results is missing (also due to the prediction periods), see for example the discussion on gas generation models shown in Table 3-1. A possibility to handle the problem of limited observation periods or calibration and validation periods, is the analysis of old systems with a long-term behavior similar to MSW landfills. Peat bogs have been suggested to analyze the degradation of organic matter, because of their composition and oxygen availability (under water saturated conditions) similar to MSW landfills (cf. Lichtensteiger et al. 1989; Bozkurt et al. 2001; Döberl 2004). However, a transfer of process characteristics from peat systems to MSW landfills is confronted with serious limitations due to factors such as the more complex chemical composition of MSW (chemicals, heavy metals, etc.), the heterogeneity of waste materials and water flow in the landfill, or the influence of different initial and boundary conditions at the landfill (e.g. waste deposition method and rate or technical barriers to separate the landfill from the environment).

Accordingly, natural analogues have been used for qualitative statements on the degradation of organic matter in anaerobic environments, but not to derive quantitative estimates on emission levels from MSW landfills (cf. Lichtensteiger et al. 1989; Bozkurt et al. 2001).

The mobilizable potentials shown in Table 3-5 are based on the extrapolation of observed emission trends at closed German MSW landfills. Krümpelbeck (2000) observed an exponential decrease of leachate concentrations at old MSW landfills and calculated emission loads in the leachate for 10000 years based on this trend. The resulting emission potentials for leachate of MSW landfills are shown in Table 3-5. Although an exponential decrease of leachate concentrations has been observed for many leachate parameters after landfill closure (e.g. Zweifel et al. 1999; Gibbons et al. 2007), it is questionable if the leachate emission potentials can be estimated based on an extrapolation of these trends over several thousands of years, as conditions at the site might change throughout such a long period.

Table 3-5: Mobilizable substance potentials via leachate based on an extrapolation of observed leachate emission trends at German MSW landfills (from Krümpelbeck 2000)

Leachate parameter	Mobilizable potential [mg/kg DM]	Leachate parameter	Mobilizable potential [mg/kg DM]
COD	2500 – 11200	**Cl**	900 – 3800
NH_4-N	1400 - 3400	**Cd**	0.0016 – 0.0062
AOX	2 – 22	**Zn**	0.2 – 16

To evaluate the long-term emission behavior of MSW landfills, Belevi and Baccini (1989) developed a model on leachate characteristics following first-order kinetics (equation 3-1) with the input parameter m_0, mobilizable substance potentials, derived from leaching tests (see Table 3-6). The model is built on the assumptions that release mechanisms are constant throughout the modeling period, water flow is homogenous in the landfill, and biodegradation processes after the intensive reactor phase are insignificant. In the model formulation shown in equation 3-1 c(t) is the average concentration of a substance in the leachate t years after the intensive reactor phase (i.e. gas generation has dropped significantly and leachate concentrations exhibit a downward trend) in mg/l, c_0 is the substance concentration in the leachate directly after the intensive reactor phase in mg/l, m_0 is the mobilizable waste fraction in mg/kg of dry matter, V is the annual volume of leachate in liters/yr, M is the waste mass in kg of dry matter, and t is the time after the intensive reactor phase in years.

(3-1) $$c(t) = c_0 \bullet e^{-(\frac{c_0 \bullet V}{m_0 \bullet M}) \bullet t}$$

Table 3-6: Mobilizable substance potentials of landfilled MSW via leachate determined for waste samples from landfills 11 and 14 years after waste deposition via leaching tests (see Belevi and Baccini 1989)

Substance	C_{org}	N	P	Cl	Fe	Cu	Zn	Pb	Cd
m_0 [mg/kg DM]	2100-7100	200-310	5-33	1000-1500	20-39	1.0-6.7	14-98	0.1-2.5	0.06-0.22

The leachable amounts of substances shown in Table 3-6 were determined based on consecutive leaching experiments with ground waste samples excavated from MSW landfills 11 years and 14 years after waste deposition, respectively. As these experiments cannot account for biodegradation

processes potentially mobilizing substances contained in the waste (cf. Kylefors et al. 2003), it is not surprising that the determined mobilizable nitrogen potential is low compared to estimates by other authors (cf. Table 3-5, Table 3-7, Table 3-8, and Table 3-9). Contrarily, organic carbon mobilization via leachate is probably overestimated due to the neglect of the landfill gas emission pathway. In addition to the assumption on negligible biochemical reaction rates after the intensive reactor phase, the model assumption of homogeneous water flow in the waste hampers a realistic evaluation of future emission levels (cf. Rosqvist et al. 2005; Fellner et al. 2009a).

In a study on residual emission potentials of old MSW landfills in Germany the mobilizable substance potentials of deposited MSW for the leachate and landfill gas pathway were determined (Ehrig and Brinkmann 1999). The corresponding data in Table 3-7 on old MSW landfills refers to sites with landfill ages between 5 and 35 years. The emission potentials in MSW landfills in western Germany were found to be significantly higher than those in Eastern Germany, due to the different composition of the deposited waste. In the East, less organic material and a large amount of ash from households (domestic fuel) were deposited at the sites, because of the socioeconomic differences between East and West before the German reunification. In addition to landfill simulation reactor experiments with samples from old MSW landfills, they investigated also the emission potentials of fresh MSW and residues from mechanical biological treatment (MBT) via landfill simulation reactor experiments. The results in Table 3-7 illustrate that MBT may significantly reduce the potential of mobilizable substances in the deposited waste, as landfill gas generation potentials of MSW decreased up to 90% and the mobilizable amounts of organic carbon and nitrogen in the leachate were up to 95% lower than without treatment.

Table 3-7: Mobilizable substance potentials via leachate and landfill gas for old MSW landfills, fresh MSW, pretreated MSW (MBT) (source: Ehrig and Brinkmann 1999)

Emission potentials Parameter	Old MSW landfills (West)	Old MSW landfills (East)	MSW (without pretreatment)	MBT residue
Gas generation potential [liters/kg DM]	10 - 40	n. n.	110 - 240	15 - 115
TOC (leachate) [mg/kg DM]	1800 - 8400	60 - 325	8000 - 15000	600 - 1300
N_{tot} (leachate) [mg/kg DM]	1200 - 4100	50 - 950	4000 - 6000	500 - 800

Kruse (1994) also used landfill simulation reactors to investigate mobilizable substance potentials and long-term release of substances from MSW. Based on his observations, Kruse (1994) distinguished three phases of landfill gas production: a lag phase without gas production, a growth phase with increasing gas production, and a decay phase with decreasing gas production rates. The total amount of methane produced in the reactors filled with fresh MSW was 103 liters CH_4/kg DM after 300 days, i.e. further gas generation could be expected after the end of the experiments. The concentrations of COD, Cl, TKN, and AOX in the leachate of the simulation reactors were described by exponential decrease functions. Based on the observed concentrations and exponential trend extrapolations, Kruse (1994) estimated total mobilizable potentials for these leachate parameters (see Table 3-8).

Table 3-8: Mobilizable potentials of selected parameters for MSW via landfill gas and leachate (source: Kruse 1994)

Parameter	Methane potential	COD	Cl	TKN	AOX
Mobilizable potential	103 liters CH_4/kg DM	3 g/kg DM	2.1-2.5 g/kg DM	2.2 g/kg DM	0.6-1.2 mg Cl/kg DM

An investigation of mobilizable substance potentials of 18 years old MSW excavated from a landfill was carried out by Döberl et al. (2005). Waste samples were taken from an Austrian MSW landfill and placed in landfill simulation reactors to determine the remaining mobilizable substance potential via leachate (see Table 3-9). The experiments were terminated after 2.25 bed volumes of water had been exchanged and the total potentials of mobilizable substances in Table 3-9 were estimated based on trend extrapolations.

Table 3-9: Residual amounts of substances mobilizable via the leachate pathway from 18 years old waste excavated from an Austrian MSW landfill (source: Döberl et al. 2005)

Leachate parameter	Mobilizable potential [mg/kg DM]	Leachate parameter	Mobilizable potential [mg/kg DM]
COD	1540-1890	**Cl**	1560-1820
BOD_5	90-150	**SO_4**	3.1-9.4
TOC	520-620	**Na**	1130-1770
NH_4-N	790-940	**Fe**	5.1-8.8

From the review and discussion of mobilizable substance potentials and release rates above, it is apparent that large amounts of substances are potentially mobilizable even at old (several decades) MSW landfills. The emission potentials shown in Table 3-5, Table 3-6, Table 3-7, Table 3-8, and Table 3-9 for fresh and aged MSW samples from landfills diverge substantially for specific substances, due to different waste qualities, experimental settings or investigated systems, and model assumptions. Thus, mobilizable substance potential should be determined on a case-by-case basis for individual landfills, taking into account such factors as waste composition and heterogeneity, water flow heterogeneity, climatic conditions, landfill operation and design. If site-specific data on the remaining mobilizable substance potentials are missing, corresponding data from the literature may be used for screening evaluations.

In Figure 3-8 bandwidths of mobilizable substance potentials of N and Cl via leachate are compared to the emitted substance loads at the landfills MSW1 and MSW3 after 28 and 36 years of landfill age, respectively (see section 3.1.2). The most probable ranges of mobilizable substance potentials for nitrogen and chloride are assumed to be 2.0-3.4 g N/kg DM and 1.7-3.0 g Cl/kg DM (grey areas in Figure 3-8). In addition, maximum and minimum values for mobilizable potentials are marked by dotted lines. Figure 3-8 illustrates that only the cumulative chloride load of MSW1 lies in the range of probable values for the potential of mobilizable chloride. The cumulative loads of nitrogen at both landfills and of chloride at MSW3 did not reach probable ranges after an average waste age of two decades, even though both sites were equipped with a temporary cover to allow further infiltration of water after the end of waste deposition. Despite the efforts to reduce the remaining emission potential in the waste body at MSW1 and MSW3 after closure, a significant amount of mobilizable substances may be present in these landfills. After the installation of a final cover the increase of cumulative loads over time will slow down even further (cf. Figure 3-8) and mobilizable substances will be potentially present in the landfill for long periods of time.

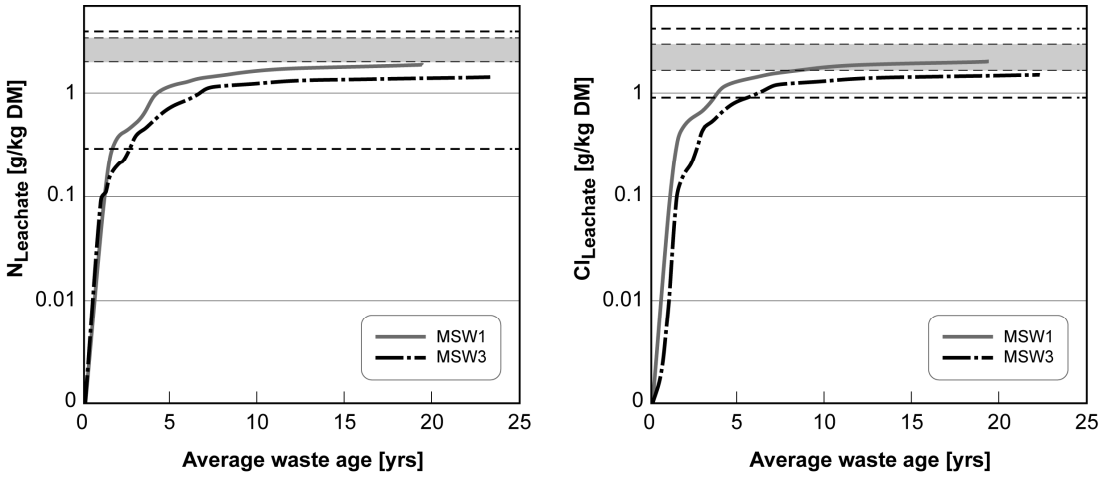


Figure 3-8: *Total cumulative loads of N (left) and Cl (right) in the leachate of MSW1 and MSW3 compared to estimates of corresponding mobilizable substance potentials (grey area: most probable range of mobilizable substance fractions; dotted lines at the bottom and at the top: maximum and minimum mobilizable substance potentials)*

In conclusion, although emissions of MSW landfills typically decrease after landfill closure, a significant amount of mobilizable substances remains in the landfill for an extended period of time (i.e. decades to centuries). Models to evaluate long-term emission characteristics at closed MSW landfills need to consider the level of interaction between the deposited waste and the environment, the mobilizable substance fraction in the waste, the water flow regime in the landfill, and a potential change of the landfill conditions (e.g. changing pH, aeration, changing water flow paths, etc.). On the short term (decades) emission trends and models based on observed emission characteristics can be used to predict emissions. On the long term (centuries to millennia) fundamental relationships could be used to establish emission models, however, associated with large uncertainties due to the long modeling periods. A procedure to establish emission models for closed MSW landfills and derive emission estimates on a site-specific basis is presented as a part of the evaluation methodology to derive aftercare completion criteria in chapter 5.2.

3.2 Emissions from landfills containing MSWI bottom ash

3.2.1 General description of emission behavior

From the incineration of one ton of MSW around 255 kg of bottom ash are produced. Municipal solid waste incineration (MSWI) bottom ash typically consists of mainly ash (~ 45%) and melted materials (~ 40%), glass (5%), metals (5%), organic components (2%), and fines (3%, mainly stones and earth) (Baccini and Brunner 1985). Matrix components in the bottom ash are Si, Ca, Al, and Fe, which account for 80% of dry mass and are present in oxidized form. Apart from these, bottom ash exhibits significant concentrations (grams per kg of dry matter) of Mg, CO_3, Na, K, S, Cl, organic compounds, Cu, Pb, Zn, Mn, and Cr. Other heavy metals such as Cd, Mo, Sb, V, and W are typically present at concentration levels of several mg per kg of dry bottom ash (cf. Johnson et al. 1999). Organic residues, such as paper, are found in MSWI bottom ash due to incomplete incineration processes, particularly at older waste incineration plants (e.g. Ferrari 1997). Depending

on the quality of the incineration process the content of organic materials may still be substantial (several mass %).

Numerous physical and chemical processes take place in MSWI bottom ash landfills, as the incineration residue is not in thermodynamic equilibrium at the conditions present in the landfill (temperatures during the incineration process are around 800 – 1000 °C) (cf. Johnson et al. 1995). Due to the contact with water, metal oxides are transformed to hydroxides in hydration reactions. Calcium-rich leachate forms minerals together with hyaline phases. The main products of this recrystallization process are clay minerals (rich in Si and Al), calcium aluminum silicate hydrate, and a crystalline phase rich in calcium and silicate (Johnson et al. 1995). Simultaneously with these reactions, carbonates are precipitated, as the leachate rich in calcium hydroxides gets in contact with carbon dioxide (air constituent or from organic degradation processes). In addition, elemental substances are involved in several reactions in the waste body (see Förstner and Grathwohl 2007). The reaction of aluminum and water produces aluminum hydroxide and hydrogen ($Al + 3H_2O \rightarrow Al(OH)_3 + 1.5\ H_2$). Biochemical degradation of organic matter may be relevant in the landfill if organic matter is present and the pH value decreases to a level allowing for microbial activity (initially, after waste deposition, the pH is in the range of 11 to 12).

Due to the large extent of exothermic processes occurring in the deposited waste (hydration of slag phases, corrosion of metals, and formation of calcium silicate hydrate), the temperature in the landfill body increases. In the first years after deposition, the temperatures in the waste body may increase up to 90°C and have been observed to remain in such ranges potentially for up two decades (cf. AGW 1992; Turk 1996). For instance, at a 20 years old MSWI bottom ash landfill temperatures of 60°C were measured at the landfill base (Turk 1996). However, the effect of exothermic reactions taking place in the landfill could be drastically reduced by effective separation of the metals contained in the bottom ash and intermediate storage of the bottom ash before deposition (Speiser 2001).

3.2.1.1 Gaseous emissions

The main components of gas from MSWI bottom ash landfills are water vapor and hydrogen gas. Water is transferred to the gaseous phase due to the high temperatures in the waste body and hydrogen is a product of elemental aluminum oxidation. As 1.25 liters of hydrogen gas can be produced from 1 g of elemental aluminum, approximately 25 liters of hydrogen may be generated from 1 kg of MSWI bottom ash with 20 grams of elemental aluminum per kg of dry matter (see Hutterer et al. 2000).

Carbon dioxide produced by anaerobic degradation of organic matter, may dissolve in the leachate (carbonate precipitation), but may also be released in the landfill gas, depending on reaction rates and contact with water. Methane may also be present in the landfill gas under specific environmental conditions and if degradable organics are present in the waste body. For instance, Speiser (2001) observed increasing methane concentrations with increasing depth at a MSWI bottom ash landfill. At the top of the landfill methane concentrations were found to be below 100 mg CH_4/m^3 of gas, whereas at the landfill bottom concentrations of 1600 mg CH_4/m^3 were measured. However, as many factors affect the occurrence of anaerobic degradation processes in

bottom ash landfills, the presence of methane may vary on a case-by-case basis, but CH_4 concentrations will typically be low at any site.

3.2.1.2 Leachate emissions

In the first years after waste deposition, the quality of leachate from MSWI bottom ash landfills is dominated by high concentrations of soluble salts (e.g. chloride concentrations of 10000 mg/l, see Hjelmar (1996), alkaline metals, and alkaline earth metals. Heavy metal concentrations in the leachate are very low during this period and typically comply with drinking water quality standards. However, the acid neutralizing capacity of the deposited material is expected to decrease over time and increased mobilization of heavy metals may occur after long time periods (see Figure 3-9). Predictions of heavy metal mobilization rates can be based on geochemical modeling, which may account for processes of dissolution and precipitation, sorption and complexation, cation exchange, and incorporation of substances into the solid phase (cf. Hellweg 2000; Dijkstra et al. 2006). The expected evolution of leachate quality based on geochemical considerations is schematically illustrated in Figure 3-9. After the leaching of soluble substances an extended phase of carbonate buffering keeps the pH value of the leachate above 7. During this period heavy metal concentrations in the leachate are very low. Due to the consumption of carbonate buffer (e.g. import of weak acids in rainwater, internal generation of acids from organic degradation, or the export of bases), the pH may drop below 7 (this was speculated to occur over thousands of years based on geochemical modelling). Initially increased mobilization of heavy metals takes place primarily for zinc and nickel, but after a further drop of the pH below 5 also the concentrations of copper and lead increase. Chromium mobilization is not expected at high rates even if the pH reaches a level of 4 (see Förstner and Grathwohl 2007). However, large uncertainties exist with respect to such estimates due to the very long timeframes involved

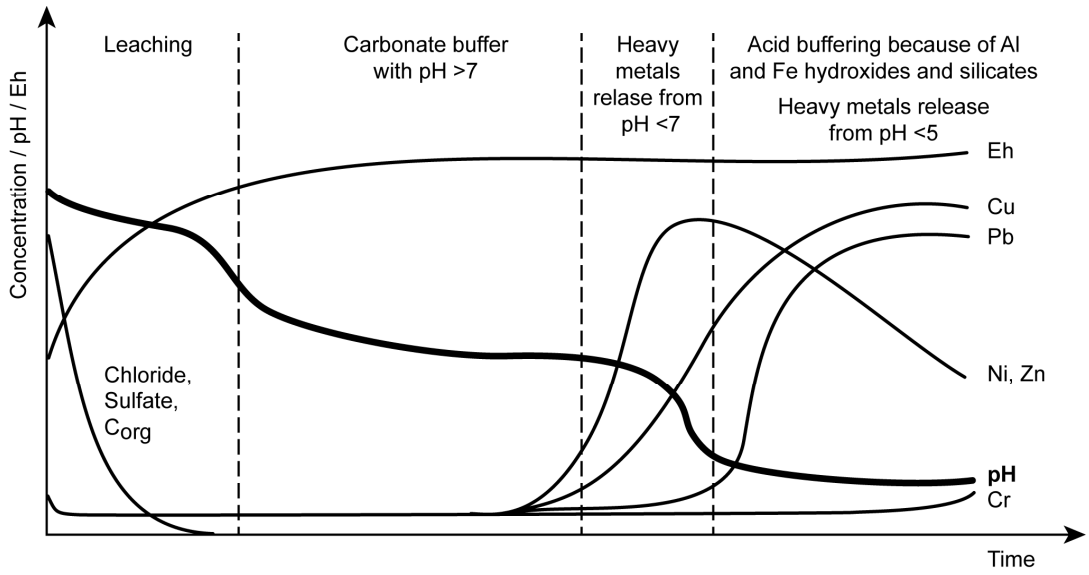


Figure 3-9: Schematic illustration of leachate quality from MSWI bottom ash landfills over time (source: Förstner and Grathwohl 2007)

As only a fraction of the deposited MSWI bottom ash gets in contact with mobile water, it is important to consider preferential water flow within geochemical models (Hellweg 2000). For

instance, Johnson et al. (1999) observed that 10-80% of the infiltrating water percolated through the deposited MSWI bottom ash in preferential flow. Assuming that around 70% of the infiltrating water is available for equilibrium reactions with the waste mass, Hellweg (2000) estimated that the release of heavy metals will occur over tens of thousands of years with copper concentrations in the leachate of 1.8 mg/l 100000 years after waste deposition.

3.2.2 Emission characteristics of MSWI bottom ash landfills

Although bottom ash from municipal solid waste incineration constitutes approximately half of the material deposited at Austrian landfills for non-hazardous largely inorganic waste ("Reststoffdeponien" (cf. Lebensministerium 2006)), there are no mono-landfills for MSWI bottom ash in Austria. Therefore, the analysis of typical landfill emission characteristics is based on data from Swiss mono-landfills for MSWI bottom ash. Together with authorities and landfill owners a screening of suitable sites to be included in the data set on emissions from MSWI bottom ash landfills was carried out. As data from closed mono landfills containing MSWI bottom ash had not been available (i.e. there are no engineered sites after closure in Switzerland), data from older landfills were primarily included in the analysis (see appendix 1). The four investigated MSWI bottom ash landfills have a lining system at the landfill base to collect the leachate percolating through the waste body and monitoring data (deposition history, leachate quality and amount, precipitation data, etc.) are available for the whole period of landfill existence. The average annual precipitation rate at the sites is 1070 mm/yr and the average age of the landfills is 15.5 years (in 2008). None of the investigated landfills is closed, but waste deposition has been taking place at very low rates at the sites BA3 (no deposition from 2001 to 2007) and BA4 (no deposition from 2003).

The mean, minimum and maximum concentrations of selected leachate parameters are shown in Table 3-10 together with the corresponding Austrian standards for leachate discharge into surface water bodies. Table 3-10 includes parameters with at least one concentration measurement at one of the landfills above the discharge quality standard. In addition, data are shown for electric conductivity, chloride, and sulfate, as these parameters are present at high levels in the leachate. The average concentrations in the leachate of the MSWI bottom ash landfills are 6870 mg/l for chloride and 2750 mg/l for sulfate. Apart from soluble salts, the mean concentrations of organic compounds (COD, DOC) and ammonium-nitrogen are factors 4-8 above the corresponding discharge standards. Heavy metal concentrations in the leachates are typically very low, but the maximum values of Cr_{tot} and Cu exceed the limit values.

Although a large number of parameters is measured at the landfills (in average 34 parameters, see appendix 1), substances not in the focus of environmental regulations might be present in the leachate without detection. For instance, concentrations of rubidium and strontium have been detected at levels of several mg/l during screening analysis of MSWI bottom ash leachates (Stünzi 2009), but are typically not included as monitoring parameters. Thus, the data in Table 3-10 give an overview about general emission characteristics typically monitored at MSWI bottom ash landfills throughout the first decades of landfill existence. However, critical leachate parameters may be different from one site to another and vary over time.

At the four MSWI bottom ash landfills on average 45-50% of the annual precipitation has been collected as leachate during the period of landfill operation (the annual fractions range from 40-70%).

Table 3-10: Statistical analysis of leachate characteristics of 4 landfills containing waste incineration bottom ash

Parameter	Number of landfills	Number of measurements	Mean*	Min. – Max.	Discharge limit (surface water)**
pH [-]	4	274	8.9	7.3 – 12.1	6.5 - 8.5
EC [µS/cm]	4	275	2667.9	231 – 8410	-
COD [mg O_2/l]	1	45	397.4	67 – 1830	50
DOC [mg C_{org}/l]	4	134	96.9	6.4 – 746	20
NH_4-N [mg N/l]	4	139	44.85	0.02 – 180	10
Cl [mg/l]	4	132	6872	2260 – 27700	Toxicity
Sulfate [mg SO_4/l]	4	138	2750	ND – 7920	site specific
Sulfide [mg S/l]	1	17	0.36	ND – 4.1	0.1
AOX [mg Cl/l]	2	64	0.49	0.01 – 4.7	0.5
As [mg As/l]	4	76	0.03	ND – 0.15	0.1
Cr_{tot} [mg Cr/l]	4	91	0.13	0.002 – 0.67	0.5
Cu [mg Cu/l]	4	104	0.19	0.01 – 2.0	0.5

* A non-detect (ND) is considered 0 for calculating the mean value.

** Limits according to the ordinance on waste water discharge from landfills (Leachate ordinance (MoE 2003) and waste water discharge ordinance (MoE 1996a)).

3.2.2.1 Evolution of emission characteristics at MSWI bottom ash landfills

The leachate data of the MSWI bottom ash landfills is longitudinally analyzed with respect to the average waste age. The liquid-to-solid ratio is not used as an independent variable to analyze leachate quality, because of the ongoing waste deposition at the landfills which blurs the relationship between overall L/S (for the whole landfill) and the actual leachate quality (potentially strong influence of substance releases from freshly deposited waste).

The box plots of DOC concentrations in the leachate of the MSWI bottom ash landfills in Figure 3-10 (left) illustrate a decreasing trend of organic leachate pollution with increasing duration of waste deposition. The reasons may be the continuous stabilization of organic matter contained in the waste and the decreasing content of organic matter in the deposited material due to the continuous optimization of the incineration processes. The concentrations of ammonium-nitrogen on the other hand, do not exhibit a significant trend with increasing waste age, but remain constant in the range between 10 and 100 mg/l (cf. Figure 3-10, right). However, depending on the oxygen availability in the landfill body and in the leachate collection system, a substantial fraction of nitrogen may be present as nitrate in the leachate.

The concentrations of chloride decrease with increasing waste age from initially around 6000-10000 mg/l to 4000-6000 mg/l after more than 10 years of mean deposition time of the bottom ash (Figure 3-11, left). Thus, although decreasing, the concentrations of Cl in the landfill leachate

remain high for more than a decade. In case of sulfate, concentrations are increasing with increasing waste age from an average of 1900 mg/l for waste ages of a few years (0-2) to an average of 4300 mg/l for waste ages of 12-15 years (Figure 3-11, left). The reasons for increasing sulfate concentrations in the leachate are not clear yet, as more significant analysis of the post-closure emission behavior of MSWI bottom ash landfills will be possible after closure and only with respect to individual landfill conditions.

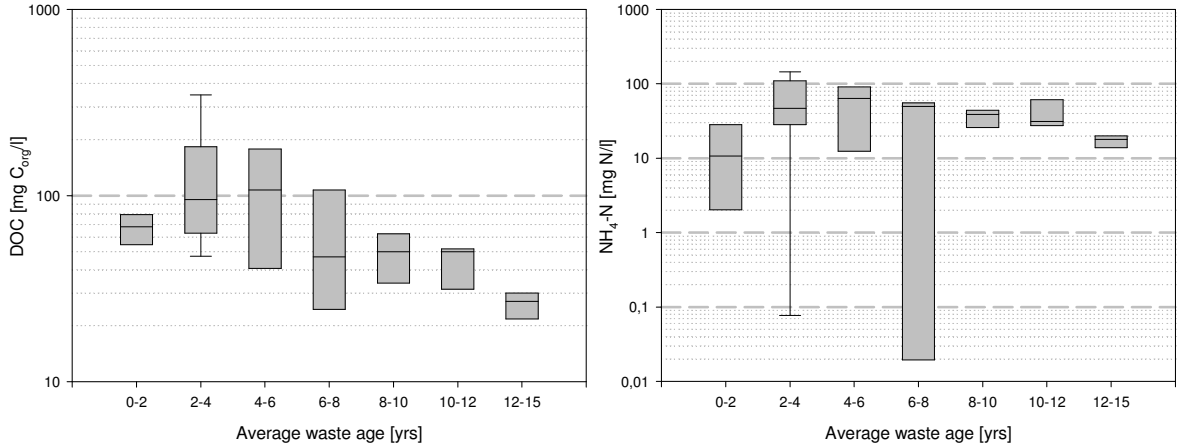


Figure 3-10: Box plots of the concentrations of dissolved organic carbon (left) and ammonium-nitrogen (right) in the landfill leachate plotted against average waste age

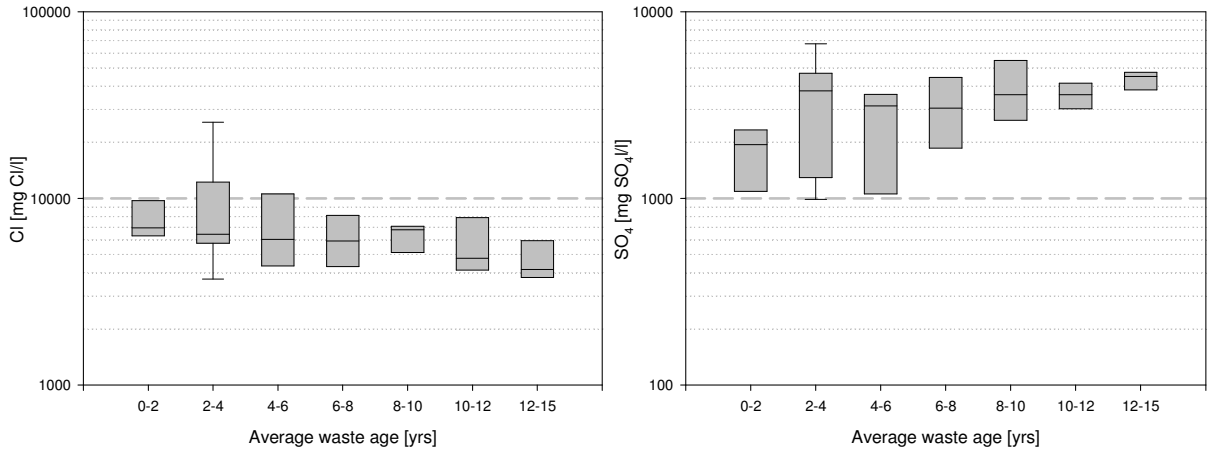


Figure 3-11: Box plots of the concentrations of chloride (left) and sulfate (right) in the landfill leachate plotted against average waste age

The concentrations of Cr and Cu in the leachate decrease with increasing average waste age from 0.1-0.5 mg/l at the beginning to around 0.01 mg/l at an average waste age of 12-15 years (Figure 3-12). Though the concentrations of heavy metals in leachate from aged MSWI bottom ash appear to be below drinking water quality standards, this may not hold for long time frames (millennia), because of changing landfill conditions (i.e. decrease of pH due to the consumption of acid buffers, cf. Figure 3-9).

In general, the longitudinal analysis of leachate characteristics will allow for more significant extrapolations when data from closed MSWI bottom ash landfills become available. Extrapolations may be useful for prediction periods of decades to centuries, whereas geochemical models are

appropriate for long-term estimates on emission characteristics, especially with respect to heavy metal mobilization.

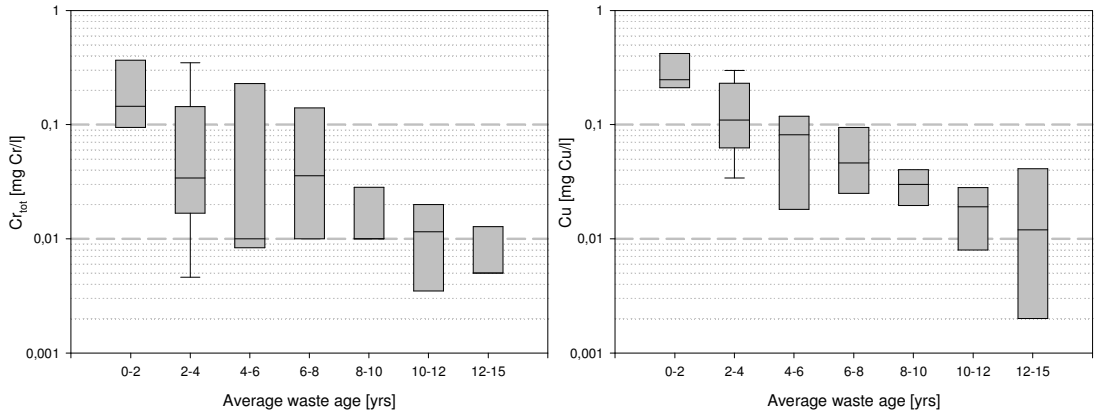


Figure 3-12: Box plots of the concentrations of chromium (left) and copper (right) in the landfill leachate plotted against average waste age

3.2.2.2 Emission loads and remaining substance potentials

The monitoring data on leachate quality and quantity is used to estimate loads of substances which were exported from the landfill body from the beginning of waste deposition till 2008. The uncertainties of the estimated annual loads are quantified based on criteria addressing the completeness of the time series data, the number of measurements per year, the method to determine leachate generation rate (e.g. climatic water balances vs. continuous discharge measurements), and others. In Figure 3-13 the annual loads of ammonium-nitrogen and chloride are plotted against the average waste age for the two MSWI bottom ash landfills BA1 and BA2, because a complete data set was available for these sites only, but not for the others. The uncertainties illustrated for the annual loads of NH_4-N and Cl at BA1 and BA2 in Figure 3-13 are mainly because of the limited amount of concentration measurements per year (1-2 values/yr) and the lack of measured leachate generation rates specifically for the bottom ash compartments (i.e. leachate generation data were only available in aggregate for several compartments). During the first years of waste deposition (average waste age below 2 years) the annual loads of ammonium-nitrogen decrease from around 10 g N/Mg DM and year to around 1 g N/Mg DM and year and remain on this level afterwards, with a slow decrease to be observed at BA2. Similarly, annual chloride loads decrease rapidly throughout the first years of landfilling to levels of a few hundred g Cl/Mg DM and year at BA1 and of around 100 g Cl/kg DM and year at BA2. The annual chloride loads in the leachate of BA2 are then constant for the remaining observation period.

The specific annual substance loads of ammonium-nitrogen and chloride in the leachate of BA1 and BA2 shown in Figure 3-13 decrease with increasing average waste age. Similar trends can be observed also for the other sites (BA3 and BA4) and for other substances (e.g. C_{org}, Cr, and Cu, for which data are not shown). However, as MSWI bottom ash deposition is taking place at all the investigated sites, the presented analysis provides an indication of the evolution of emission levels after closure, but should be backed up with monitoring data from closed MSWI bottom ash landfills in the future. Apart from that, emissions from individual landfills might vary significantly

due to different qualities of the deposited bottom ash (e.g. organic residues, metal separation technology), deposition techniques (e.g. intermediate storage before landfilling), or the intensity and extent of waste getting into contact with water (e.g. fraction of preferential flow).

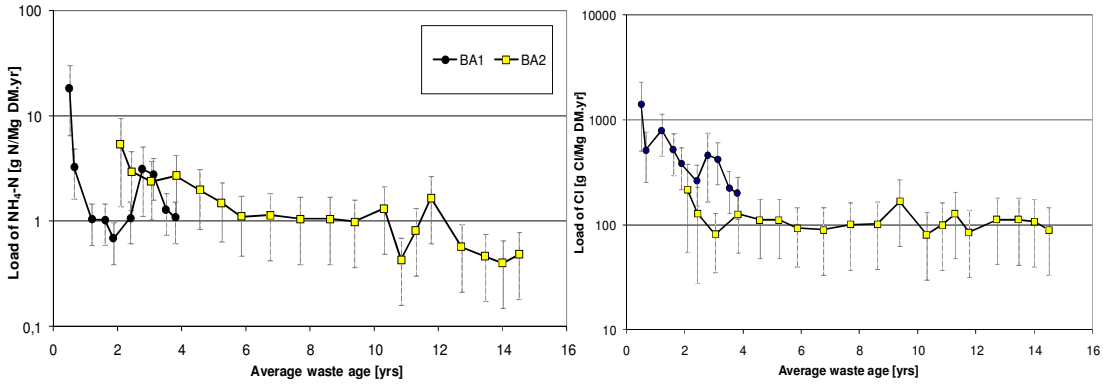


Figure 3-13: Annual loads of ammonium-nitrogen (left) and chloride (right) in the leachate of BA1 and BA2 plotted against the average waste age (=mean residence of the waste in the landfill) of the MSWI bottom ash

Based on the annual substance loads in the leachate of BA1 and BA2 (cf. Figure 3-13) and the reported data on MSWI bottom ash composition, substance balances are established for organic carbon, nitrogen, chlorine, chromium, and copper using material flow analysis (Brunner and Rechberger 2004). Substance flows are investigated from initial waste deposition at the sites until 2008. The initial contents of substances in the deposited MSWI bottom ash are 10 g/kg DM of C_{org}, 0.4 g/kg DM of N, 2.8 g/kg DM of Cl, 1 g/kg DM of Cr, and 2g/kg DM of Cu (cf. Belevi et al. 1992; Faulstich 1993; Huber et al. 1997). Although fresh waste is deposited at the sites, the results represent an interim balance to evaluate the emission trends shown above (see Figure 3-10 to Figure 3-13) with respect to the actual amount of substances remaining in the landfill after some time of waste deposition.

The fraction of substances exported from the waste body via leachate after 10 years of waste deposition or at an average waste age of 3.8 years, respectively, at BA1 are shown in Figure 3-14 for C_{org}, N, Cl, Cr, and Cu. Whereas a substantial part of the Cl originally contained in the waste has been leached (45%), only minor fractions of the other substances have been removed from the landfill via leachate. More than 99.99% of the heavy metals, around 99.8% of the organic carbon, and 97% of the deposited nitrogen are still present in the waste. However, as the initial substance concentrations are estimated from literature data and only leachate has been included as an emission pathway, the estimates are indicative and may deviate from the actual amounts of substances present in the waste.

The transfer coefficients into the leachate of landfill BA2 in Figure 3-15 exhibit similar characteristics to BA1. Because of the higher liquid-to-solid ratio at BA2, the exported fractions of Cl, N, and C_{org} are significantly larger than at BA1 (61% of Cl, 6% of N, and 0.4% of C_{org}, respectively). At an average waste age of 14.5 years, more than 99.99% of the heavy metals remain in the landfill. Literature data are used to determine the original waste composition and leachate is the sole emission pathway considered for the substance balances of BA2.

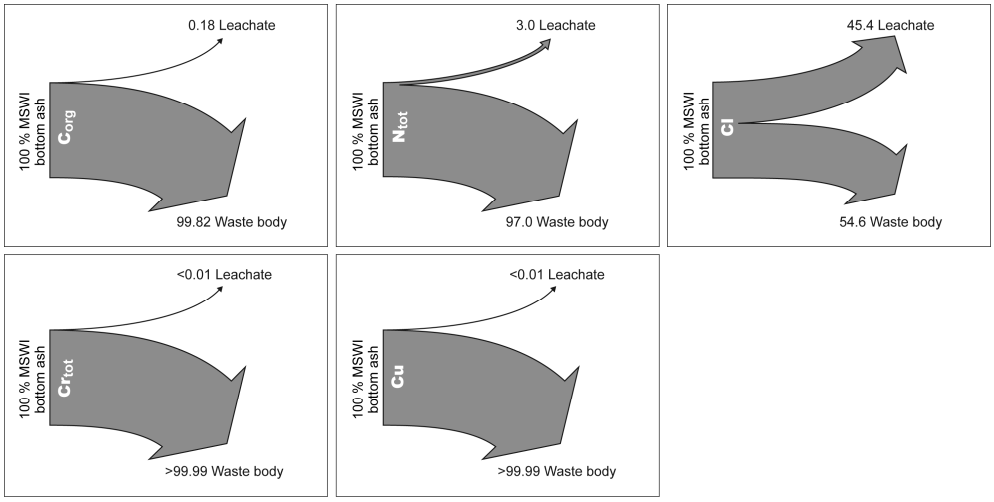


Figure 3-14: Substance balances for the MSWI bottom ash landfill BA1 with an average waste age of 3.8 years and L/S ratio of 0.24 l/kg DM

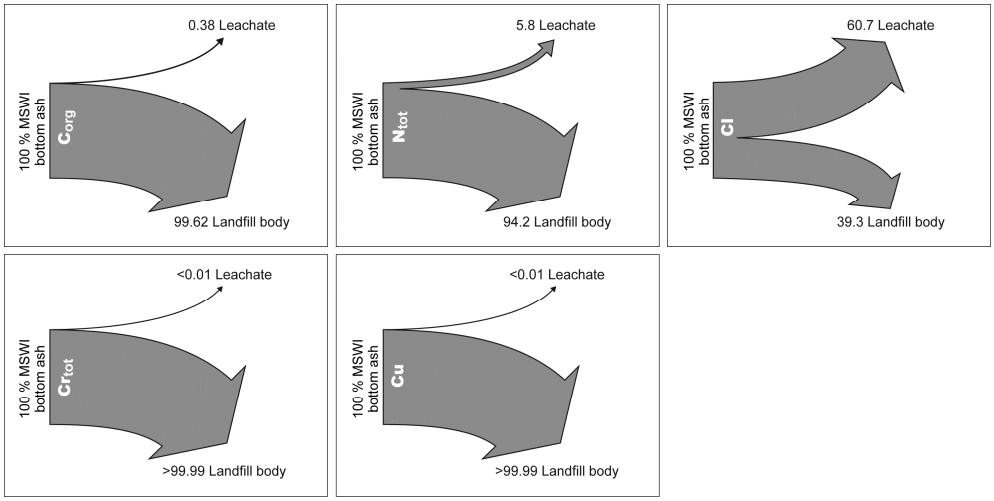


Figure 3-15: Substance balances for the MSWI bottom ash landfill BA2 with an average waste age of 14.5 years and L/S ratio of 0.37 l/kg DM

The analysis of leachate emissions and of remaining substance potentials illustrates that most of the substances deposited at MSWI bottom ash landfills are still present in the landfill decades after the deposition started. Although the observed concentrations in the leachate decrease with increasing average waste age (with the exception of sulfate), it may take several decades for the leachate quality to comply with quality standards for direct discharge into surface waters (e.g. NH_4-N or DOC). Very high concentrations of chloride in the leachate might be expected for an extended time period, too. With respect to the short- and mid-term (decades to centuries) emission characteristics of MSWI bottom ash landfills, data from closed sites will form a valuable basis to evaluate the landfill emission behavior at the full scale. For estimates on the long-term evolution of emission characteristics, geochemical modeling can be used. The buffering capacity of the deposited materials, the amount of mobilizable substances in the waste (at different pH levels), and the fraction of material in contact with water are major aspects to be addressed in such models (cf. Förstner and Hirschmann 1997; Hellweg 2000).

3.2.3 Evaluation of mid- to long-term emissions

As many heavy metals are present at high concentrations in MSWI bottom ash (e.g. 0.4-7.0 g Cu/kg DM or 0.15-9.6 g Cr/kg DM, from Faulstich (1993), knowledge about the mobilizable fraction of these substances as well as about the rate of release under different conditions (e.g. evolution of pH) are essential to evaluate landfill emission behavior. The assumption of constant conditions at a site allows for an extrapolation of trends observed in the monitoring data from closed or aged MSWI bottom ash landfills. As a change of conditions (i.e. continuous consumption and export of acid buffering capacity associated with a decrease of pH, cf. Figure 3-9) is expected over time (many centuries to many thousands of years), laboratory experiments (e.g. elution tests, column tests, lysimeter studies) can be used to evaluate long-term emission behavior, typically combined with geochemical modeling.

Table 3-11: Mobilizable fraction of selected substances contained in MSWI bottom ash (shown as % of total mass of the substance)

Mobilizable substance fraction	Extraction with water (from (1))	pH = 4 for 24 hours (from (2))	pH = 3-4 (from (3))	Maximum (from (3))
Ca [% of total Ca]	5	-	-	90
Cu [% of total Cu]	2	8	25	44.5
Cr [% of total Cr]	-	0.1		7
Zn [% of total Zn]	0.4	37	70	84
Pb [% of total Pb]	0.7	3	50	23
Cd [% of total Cd]	-	41	45	87.5
Hg [% of total Hg]	-	-		60
Cl [% of total Cl]	70	-		100
S [% of total S]	20	-		100
C_{org} [% of total C_{org}]	4	-		100

Sources: (1)Belevi et al. 1992; (2) Förstner and Grathwohl 2007; (3) Hutterer et al. 2000;

The fractions of mobilizable substances shown in Table 3-11 have been determined with different experimental settings and, thus, deviate substantially from one source to another. Belevi et al. (1992) determined the fraction of mobilizable substances based on consecutive leaching tests with water at neutral pH, whereas the data presented by Förstner and Grathwohl (2007) and Hutterer et al. (2000) relate to leaching experiments at significantly lower pHs. The variation of the mobilizable fractions determined from different leaching experiments underlines that a low pH is associated with intensified mobilization of heavy metals (e.g. Cu, Zn, Pb, and Cd). Geochemical models can be used to estimate the evolution of the pH in the waste based on its initial composition (mineral phases) and assumptions on the interaction of the waste with the surrounding environment (e.g. water infiltration, acidity of the inputs).

Investigations of long-term emissions from bottom ash landfills have focused on the effect of the continuous reduction of the acid buffering capacity on the release of heavy metals contained in the waste (e.g. Johnson et al. 1995; Hellweg 2000; Dijkstra et al. 2006). A simplified approach for an evaluation of long-term metal concentrations in the leachate of MSWI bottom ash landfills was suggested by Hellweg (2000). The approach is based on the mineral phase composition of the MSWI bottom ash and the modeling of equilibrium reactions in the landfill. In the model the complete dissolution of a mineral phase is directly linked to a change of pH, which allows for determining the concentrations of Mg, Cd, Zn, Ca, Pb, and Cu based on the modeling results in

Table 3-12 and the determination of mineral phases present in the landfill. The remaining mineral phases in the deposited material are calculated from the initial amount of minerals deposited (waste composition data) and the amount of water getting in contact with the waste for chemical equilibrium reactions. The time until a mineral phase is depleted in a MSWI bottom ash landfill is estimated from the equation shown in Table 3-12, whereby Hellweg (2000) assumed that 30% of the water infiltrating the landfill does not participate in equilibrium reactions with the waste matrix, but flows through the waste in preferential pathways. The data and the equation shown in Table 3-12 can be the basis for a rough estimate of long-term metal concentrations in the leachate of a MSWI bottom ash landfill, provided that the composition of the waste (mineral phases) and the water flow regime in the waste are known. Hellweg (2000) applied this procedure to a MSWI bottom ash landfill with a deposition height of 5 meters and 275 liters per m² and year of water flowing through the waste matrix (preferential flow has been excluded). The model results showed that metals will be released over a period of tens of thousands of years at the site and that the mineral content of $Cu(OH)_2$ will be finally depleted after 125,000 years at the landfill.

Table 3-12: Dissolved concentrations of selected substances as a function of pH and arranged in the sequence of depletion (after Hellweg 2000)

pH	Mg [mol/l]	Cd [mol/l]	Zn [mol/l]	Ca [mol/l]	Pb [mol/l]	Cu [mol/l]
9.11	0.4000	$4.28*10^{-8}$	$1.73*10^{-4}$	$8.34*10^{-5}$	$2.23*10^{-6}$	$9.28*10^{-6}$
9.50	0.0302	$1.17*10^{-8}$	$6.69*10^{-4}$	$8.87*10^{-6}$	$7.47*10^{-6}$	$1.29*10^{-5}$
8.28		$1.22*10^{-7}$	$1.50*10^{-5}$	$4.94*10^{-4}$	$5.88*10^{-7}$	$9.70*10^{-6}$
6.76					$5.50*10^{-6}$	$2.46*10^{-5}$
6.72						$2.80*10^{-5}$

$$t_n = \frac{Amount_{initial} - Water\ flow_{matrix} \bullet \sum_{i=1}^{n-1} t_i * c_i}{c_n \bullet Water\ flow_{matrix}}$$

where: t_n is time until the complete dissolution of a mineral [yrs], n is the number of pH changes until the dissolution of a mineral (see above) [-],t_i is the duration of period i [yrs], c_i is the dissolved concentration of the substance during period i [mol/l], c_n is the dissolved concentration in the last period of complete dissolution of the mineral [mol/l], $Amount_{initial}$ is the initial mass of a mineral [mol/m²], and water $flow_{matrix}$ is the amount of water flowing through the waste matrix per year [l/m² and yr]

In summary, to evaluate the long-term emission behavior of MSWI bottom ash landfills geochemical models including numerous simplifications and assumptions can be used. The respective modeling timeframes are many thousands of years, reducing the direct significance of such model estimates for aftercare considerations, as aftercare strategies are not able to address such time periods. Nevertheless, such evaluations are indicative of the long-term environmental hazards associated with the deposition of MSWI bottom ash. While the release of heavy metals represents a long-term threat, the immediate attention in terms of leachate quality from MSWI bottom ash landfills is more focused on soluble salts, organic pollution, and ammonium-nitrogen. To estimate the evolution of these parameters in the leachate over decades to centuries, trend extrapolations based on real-scale monitoring data or results from laboratory experiments simulating current landfill conditions may be appropriate. Estimates need to be based on the specific conditions at the site (and potential

changes of these) and will become more assessable when aftercare monitoring data become available at MSWI bottom ash landfills.

3.3 Emissions from landfills containing construction and demolition wastes

3.3.1 General description of emission behavior

Construction and demolition (C&D) wastes are generated during construction or demolition activities and do not comprise MSW-type wastes. C&D waste going to landfill consists primarily of concrete debris, crushed bricks, soil, metals, asphalt debris, drywall, and smaller fractions of materials such as (treated and untreated) wood, cardboard, paper, plastics, tarboard, and bitumen (Denner et al. 2005). The quality of material which can be accepted at Austrian C&D waste landfills is defined in the Austrian landfill directive (MoE 2008), which typically requires the selective demolition of buildings (e.g. removal of problematic or valuable materials before demolition), sorting (e.g. recovery of pure material fractions at the source) and pretreatment (e.g. crushing and metals separation) for C&D wastes to meet acceptance criteria. Nevertheless, as regulations have been changing over time, the quality of the C&D waste already present in the landfill may be different from current material qualities going to landfill. In addition, even with a high degree of sorting and treatment a (small) fraction of potentially problematic materials (e.g. treated wood or paint coatings containing lead, etc.) remains in the C&D waste, provided that these substances were already present in the generated waste (cf. Roussat et al. 2008).

With respect to emissions from C&D waste landfills, investigations have focused on leachate due to its potential for groundwater contamination (e.g. Townsend et al. 1999; Stegmann et al. 2006). Gaseous emissions from C&D waste landfills may contain methane from anaerobic degradation of organic materials present in the waste and hydrogen sulfide due to sulfate (present at high concentrations in the leachate primarily due to leaching of drywall materials) reducing processes in the landfill (i.e. anaerobic degradation of organic matter) (cf. Weber et al. 2002; Denner et al. 2005). Nevertheless, significant gas generation is not expected at landfills containing pre-sorted C&D wastes (according to the standards of the Austrian landfill directive (MoE 2008)). Therefore, landfill gas emissions are considered negligible with respect to the environmental impact of pure C&D waste landfills (cf. Denner et al. 2005).

Leachate characteristics of C&D waste landfills may vary substantially between different waste management systems (countries), because of the different qualities of the deposited materials. At German sites the leachate of C&D waste landfills typically contains significant concentrations of sulfate, calcium, chloride, nitrogen, and organic compounds (cf. Stegmann et al. 2006). In general, heavy metal concentrations are low (i.e. below drinking water quality standards), but high concentrations of zinc or iron have also been observed. Elevated levels of arsenic, chromium, and copper in the leachate may be associated with C&D wastes containing a substantial amount of treated wood, as this material is typically the major source of these contaminants in C&D waste (Weber et al. 2002). Hence, based on the composition of C&D wastes different pollutants and pollutant pathways may be significant at a landfill site.

3.3.2 Emission characteristics of C&D waste landfills

C&D waste has been sent to landfill already for several decades in Austria. Historically, mono landfills of this kind have not been equipped with liner systems at the landfill base and thus, time-series data from leachate monitoring over longer periods (decades) are scarce for C&D waste landfills, due to the lack of leachate collection and treatment. Together with public authorities and landfill owners 12 C&D waste mono landfills with data on leachate quality were identified in Austria and Switzerland (one Swiss site was included in the study) (see appendix 1). As the amount and quality of data varied drastically from one site to another, a large fraction of the data set is only used for statistical analysis (cf. Table 3-13), but cannot be used for more process oriented investigations (e.g. evolution of leachate characteristics with increasing average waste age). For 8 sites only punctual information about leachate characteristics (single measurements at an unknown point in time) are available and data on the deposition history or the amount of leachate are missing. For one more site (C&D4) monitoring data on leachate quality are available over time, but the amount of landfilled waste, the period of landfilling, or the leachate generation rate at the site are unknown. Consequently, detailed analysis of emission data is restricted to three sites (C&D1-C&D3), where data on annual waste deposition and leachate quantity are available.

Table 3-13: Statistical analysis of leachate characteristics measured at 12 construction and demolition (C&D) waste landfills

Parameter	Number of landfills	Number of measurements	Mean*	Min. – Max.	Discharge limit (surface water)**
pH [-]	12	87	7.7	5.8 – 8.9	6.5 - 8.5
EC [µS-cm]	12	87	3416.2	530 – 10010	-
COD [mg O_2-l]	4	63	132.7	16 – 644	50
DOC [mg C_{org}-l]	2	48	56.1	10 – 196	20
NH_4-N [mg N-l]	3	53	18.0	ND - 134	10
Cl [mg-l]	11	59	336.5	24 - 1760	Toxicity
Sulfate [mg SO_4-l]	10	58	1680	77 - 11970	Site-specific
Sulfide [mg S-l]	3	21	0.09	ND - 0.2	0.1
AOX [mg Cl-l]	11	41	0.23	ND - 2.3	0.5
Fe	1	14	0.41	0.05 - 3.0	2.0
Zn	11	50	0.14	ND – 5.8	0.5

*A non-detect (ND) is considered 0 for calculating the mean value.

** Limits according to the ordinance on waste water discharge from landfills (Leachate ordinance (MoE 2003) and waste water discharge ordinance (MoE 1996a)).

A statistical analysis of the data set about leachate parameters measured at 12 different C&D waste landfills is shown in Table 3-13. 28 leachate parameters have been measured on average at the landfills (see appendix 1), but Table 3-13 includes only parameters with at least one measurement in the data set above the respective quality standard for direct discharge into surface waters. Though electric conductivity and the concentrations of chloride and sulfate do not exceed discharge limits, these parameters are listed in Table 3-13, because of their high concentrations in the leachate of C&D waste landfills. Apart from high concentrations of sulfate in the leachate, the

mean values of organic leachate pollution parameters such as COD and DOC, as well as of the inorganic macro component ammonium-nitrogen are above the corresponding discharge standards (cf. Table 3-13). Sulfide, chloride, iron or zinc concentrations may be problematic at individual landfills or for individual measurements. The levels of other heavy metals typically comply with drinking water quality standards. However, problematic parameters may be present but not detected in the leachate (limited range of measured parameter), especially if the deposited C&D wastes contain significant fractions of hazardous materials.

3.3.2.1 Evolution of emission characteristics at C&D waste landfills

The longitudinal development of leachate emissions is investigated based on the data collected at three C&D waste landfills, because the data from the other nine sites do not allow for such an analysis (i.e. no information concerning deposition history and lack of monitoring data). At the landfill C&D1 waste has been deposited for 14 years (see appendix 1) with 120,000 m³ of occupied air space in 2008. The landfills C&D2 and C&D3 have been in operation for a shorter period (7 and 4 years, respectively, see appendix 1) and the cumulative waste volumes in 2008 were 100,000 m³ (C&D2) and 300,000 m³ (C&D3) at these sites. All of the sites are still in operation and annual waste deposition has been taking place during the last five years at an average rate of 9,000 tons per year at C&D1, 20,000 tons per year at C&D2, and 90,000 tons per year at C&D3.

The concentrations of selected leachate constituents at the sites C&D1, C&D2, and C&D3 are plotted against the average waste age in Figure 3-16. It is apparent that for many parameters, in particular for the chemical oxygen demand, the sum of ammonium-, nitrate- and nitrite-nitrogen (N_{tot})[6], chloride, and AOX, the pollution level of the leachate at C&D1 is higher throughout the whole monitoring period. For instance, COD concentrations at C&D1 range between 100 and 350 mg O_2/l, whereas COD levels at C&D2 and C&D3 are between 30 and 100 mg O_2/l. Similarly, nitrogen concentrations in the leachate of C&D1 range from 40 to 120 mg/l compared to concentration ranges of 1 to 15 mg/l at C&D2 and C&D3. The observations could be explained by the different composition of the C&D waste deposited at the C&D1 in the past (low intensity of pre-sorting and higher contents of organic matter) compared to the waste composition of the material deposited throughout the last few years (i.e. C&D2 and C&D3 are relatively young landfills). The differences in leachate pollution levels illustrate the importance of the quality of the deposited material for the emission behavior of the landfill. Monitoring data on sulfate concentrations are available only for C&D1 and C&D3 and range from around 500 mg/l (first years of waste deposition at C&D3) to around 2000 mg/l at both sites (cf. Figure 3-16). Concentrations of AOX are all the time below 0.2 mg/l at the younger sites and follow a decreasing trend at C&D1 from initially 0.4-0.6 mg/l to values around 0.2 mg/l after the average waste age increased to 3 years and above. However, except for AOX and potentially zinc, there are no apparent trends visible in the monitoring data. On the one hand, data series are still relatively short for C&D2 and

[6] The total amount of nitrogen in the leachate is used as a parameter, because of leachate sampling from open leachate ponds at the sites C&D2 and C&D3. As some aeration of the leachate is expected after discharge to the pond at these sites, the nitrogen species (NH_4-N, NO_3-N, and NO_2-N) in the leachate are aggregated to allow for a valid comparison between the individual landfills.

C&D3 and on the other hand, all of the sites are still in operation and waste is continuously deposited, potentially blurring the evolution of emission characteristics from older waste parts.

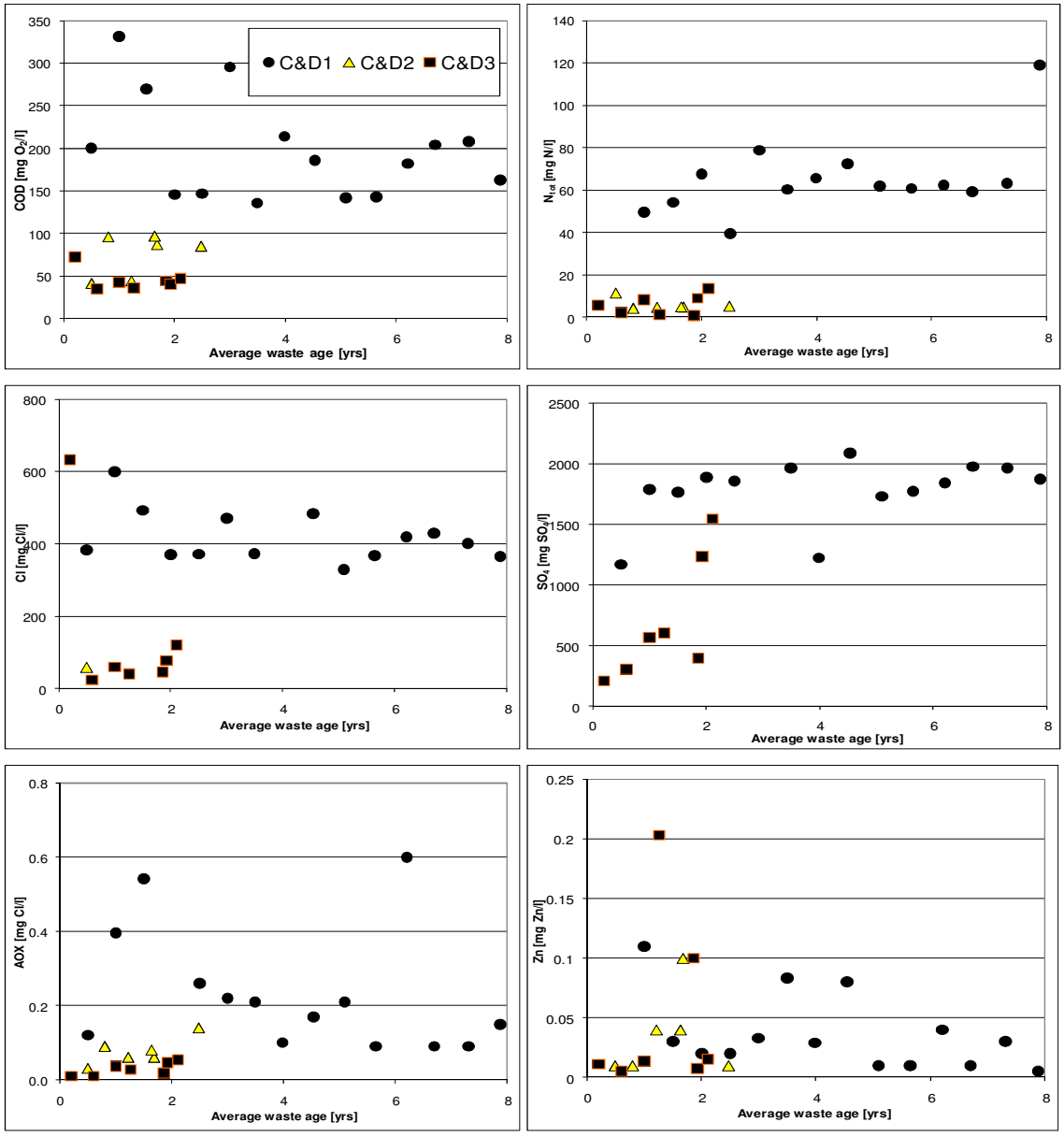


Figure 3-16: *Concentrations of COD (top left), N_{tot} (top right), Cl (center left), SO_4 (center right), AOX (bottom left), and Zn (bottom right) plotted against the average waste age at landfill C&D1, C&D2, and C&D3*

Based on the analysis in Figure 3-16 it may be expected that organic leachate pollution and nitrogen levels in the leachate are more problematic at older C&D waste landfills with higher contents of organic matter contained in the originally deposited waste than at younger deposits, where more stringent waste acceptance criteria have been applied. Information on the evolution of leachate characteristics at C&D waste landfills after closure, can be currently obtained primarily from column leaching or lysimeter experiments (cf. Susset and Leuchs 2008), but will become increasingly available at full-scale sites when more and more engineered (i.e. with containment system) C&D waste landfills are turned into aftercare.

In general, the quality of monitoring data needs to be critically evaluated, even more so at C&D waste landfills, as leachate samples are often taken from leachate ponds (effects on leachate quality!) and complementary data such as leachate generation rates during sampling are often missing. Landfill emissions need to be evaluated in view of site specific aspects, as the quality of the deposited material and characteristics at the site (e.g. climate, deposition rate, and thickness) vary from one site to another.

The multiplication of average annual concentrations in the leachate with the annual amount of leachate generated results in annual substance loads in the leachate of the investigated C&D waste landfills (see Figure 3-17 and Figure 3-18). The quality of the underlying data has been evaluated and uncertainty ranges have been estimated based on specific criteria (e.g. completeness of the time series, documentation of sampling and leachate analysis methods, method to measure or estimate the leachate generation rate, etc.). The relatively large uncertainty ranges (expand over one to two orders of magnitude) shown in Figure 3-17 and Figure 3-18 are primarily due to the poor quality of data on the annual amounts of leachate, as direct discharge measurements were not carried out and data series are incomplete.

The annual loads of chemical oxygen demand in the leachate (Figure 3-17, left) show a decreasing trend with increasing average waste age at the investigated landfills. At C&D2 and C&D3, the initial loads of approximately 30 g of COD per ton of dry waste and year decrease to less than 3 g of COD per ton of dry waste and year at an average waste age of 2 years. At C&D1, the initial load of 55 g COD per ton of dry matter and year decreased to around 7 g of COD per ton of dry waste after an average waste age of 5 years had been reached. Similar trends, however at lower levels, can be observed for the annual nitrogen loads in the leachates (Figure 3-17, right). For both parameters, COD and N_{tot}, the reason for the decreasing annual loads with increasing waste age is not a decrease in concentrations (cf. Figure 3-16, top), but decreasing amounts of leachate generated at the sites per ton of waste (e.g. increasing waste thickness through ongoing waste deposition and constant area-specific infiltration rates result in less leachate generated per ton of waste deposited at the site).

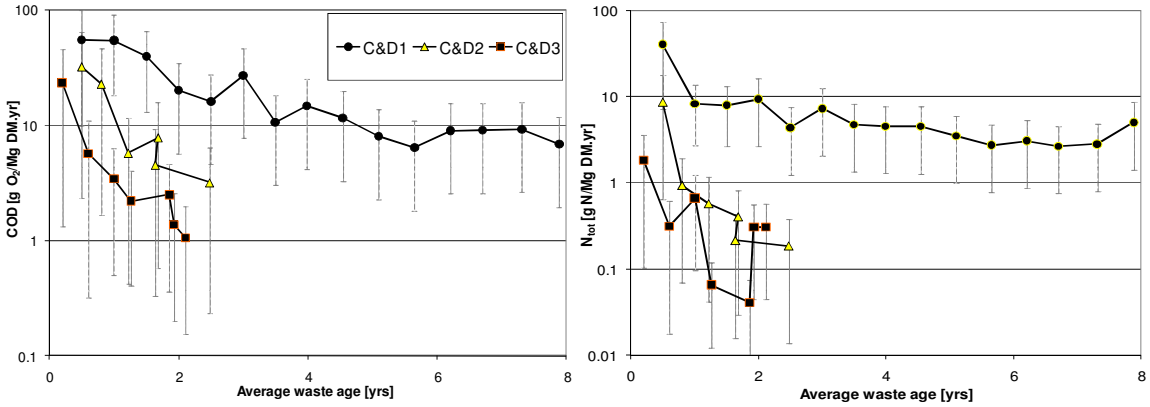


Figure 3-17: Specific annual substance loads of COD (left) and N_{tot} (right) in the leachate of C&D1, C&D2, and C&D3 plotted against average waste age at the sites

The annual loads of chloride and sulfate in the leachate of the C&D waste landfills are shown in Figure 3-18. In case of landfill C&D2, there is only one measurement available for the concentration of Cl and SO_4 has not been measured in the leachate at all (cf. Figure 3-16, center). At the

other sites the substance loads in the leachate decrease for both parameters with increasing average waste age. In the leachate of C&D1 chloride loads decrease to approximately 15 g per ton of dry waste and year and sulfate loads decrease to approx. 100 g per ton of dry waste and year after an average waste age of 5 years has been reached. At C&D3 the annual loads of Cl and SO_4 in the leachate decrease reach levels of 5 g Cl per ton of dry waste and year and of 40 g SO_4 per ton of dry waste and year at an average waste age of 2 years. Again, the decreasing loads with increasing average waste age are not due to a decrease in concentration levels (cf. Figure 3-16, center), but due to decreasing amounts of landfill leachate relative to the deposited waste mass. Although, the loads of AOX and Zn are not shown, it should be noted that a slight decrease of annual loads with increasing average waste age can be observed also for these parameters due to same phenomenon.

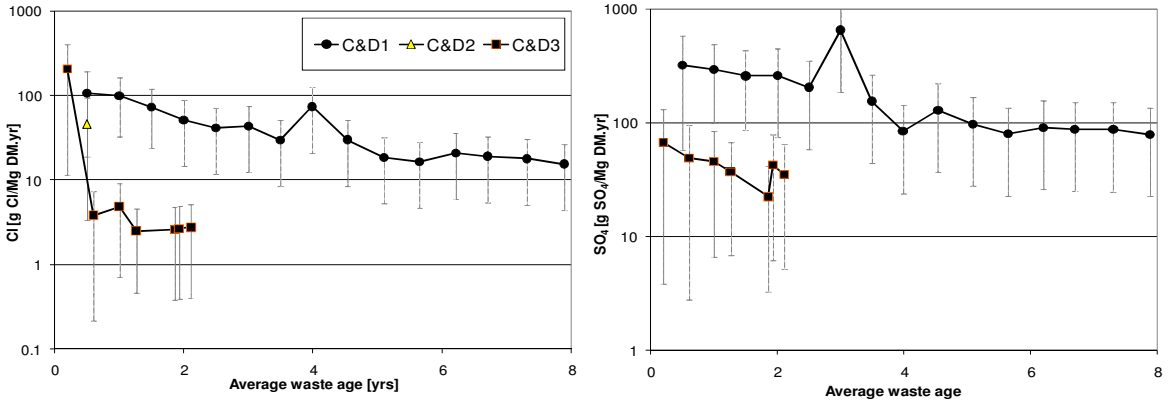


Figure 3-18: Specific annual substance loads of Cl (left) and SO_4 (right) in the leachate of C&D1, C&D2, and C&D3 plotted against average waste age at the sites

The emission loads in the leachate of the C&D waste landfills are not used to calculate mass balances, because of the lack of data on the initial waste composition and the limited significance of mass balances after rather short periods of landfill existence and for sites with ongoing waste deposition. However, the estimated substance loads can be used as a basis to evaluate the actual threat to groundwater posed by the leachate from these landfills, as potential effects of leachate releases on groundwater quality are determined by the load of pollutants in the leachate.

The analysis of leachate emission data from C&D waste landfills has shown that emission characteristics are related to the quality of the deposited material and thus, should be evaluated on an individual basis. Nevertheless, such parameters as the concentrations of soluble salts, organic leachate compounds (COD, DOC, AOX), and ammonium-nitrogen (or total nitrogen) are of general significance in the leachate of C&D waste landfills. Although decreasing trends can be observed for specific annual leachate emission loads with increasing average waste age, monitoring data from closed C&D waste landfills are necessary to evaluate the emission behavior after the end of waste deposition. As monitoring data from closed sites are not yet (or hardly) available in good quality, estimates on the evolution of leachate characteristics at C&D waste sites are typically based on laboratory experiments (e.g. shaking leaching tests, column leaching tests, lysimeter studies). Models to evaluate the future emission behavior of C&D waste landfills are discussed in the next section.

3.3.3 Evaluation of mid- to long-term emissions

The above discussion of emission data from C&D waste landfills highlights that monitoring data are available for a relatively short period at landfills of this type (i.e. lack of containment systems to control emissions at older sites) with waste deposition typically still taking place at these sites. The evolution of leachate characteristics in the mid- to long-term can therefore be assessed primarily on the basis of leaching experiments at different scales. Up to date, efforts to characterize and predict leachate emission levels from C&D wastes have been made primarily to assess the risk of recycling these wastes as fill or construction materials (e.g. van der Sloot 1996; Susset and Leuchs 2008; Susset and Grathwohl 2011). However, the results of these studies can be also used to address the emission characteristics of formal C&D waste landfills, as they represent deposits of the same materials.

In general, two different types of source term behavior, with transitions in between, can be distinguished. The two types were referred to as "finite sources" and "virtually infinite sources" by Susset and Leuchs (2008) or "availability-controlled substances" (= finite source) and "solubility-controlled substances" (= virtually infinite source) by van der Sloot (1996). Salts and mobile metal fractions are typically finite sources (see concentrations of chloride and chromium shown in Figure 3-19). The leachate concentrations of such substances decrease rapidly with increasing liquid-to-solid ratio often following exponential decrease functions. The virtually unlimited type of release is characteristic for substances showing an initial period of concentration decrease before an extended period of constant concentrations in the leachate. This behavior is typical for some heavy metals. The first period of decrease may be missing for organic compounds strongly sorbed to solids (e.g. PAHs), as they exhibit constant concentration levels in the leachate from the beginning. However, the release of PAHs from the waste can be strongly affected by biochemical degradation processes under aerobic conditions, which could be observed during experiments accounting for such processes (see PAHs concentrations in the leachate of column and lysimeter experiments in Figure 3-19).

The knowledge about the type of release for a substance contained in the C&D waste is a prerequisite for the selection of appropriate experimental settings and the respective prediction of emissions in view of potentially changing conditions (e.g. pH). In general, the transferability of results from leaching experiments to real-scale landfills is dependent on the substance under consideration, the material quality, the water distribution and preferential flow in the waste, the redox conditions, and mineral solution and aging effects (Susset and Leuchs 2008). The comparison between results from leaching experiments at column- and lysimeter-scale in Figure 3-19 illustrates, that some substances are observed at very similar levels at both scales (e.g. Cl and Cr), whereas the concentrations of other substances may not be directly transferable from the column-scale to the field-scale. For example, the concentrations of PAHs in the column leachate represent an upper boundary of what can be expected at the field-scale, because biochemical degradation processes are not taking place during the column leaching test, but potentially lower the concentration levels in the leachate under field conditions, if regions with aerobic conditions regions exist in the landfill (cf. Figure 3-19). For some substances, such as sulfate, no relationship between concentrations observed during a column leaching test and leachate concentrations observed during field-scale experiments could be established, due to the complex interaction of different processes. In case of sulfate,

varying pH conditions at the micro-scale, the infiltration of air into the waste, and the heterogeneous distribution of minerals, were identified as the main causes for the complexity of transferring leachate concentration levels from one scale to another (cf. Susset and Leuchs 2008).

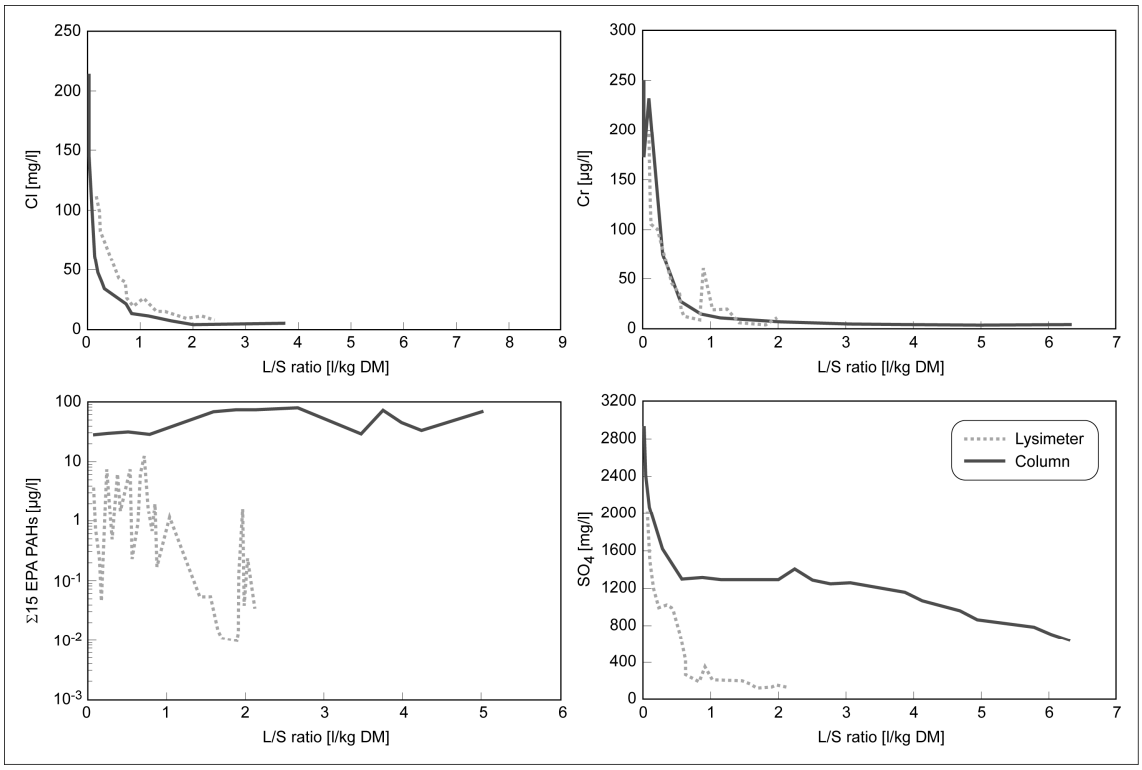


Figure 3-19: Comparison of substance concentrations in the leachate of columns and lysimeters filled with C&D waste plotted against the liquid-to-solid ratio (adapted from Susset and Leuchs 2008)

In conclusion, the suitability of column leaching experiments as a basis to estimate future emission characteristics at C&D waste landfills is different from one type of substance to another. For easily soluble salts and mobile metal fractions, the concentrations in the column leachate may be transferred directly to the field-scale based on equivalent liquid-to-solid ratios. However, aspects of waste heterogeneity (i.e. representative sampling of C&D waste) and water flow heterogeneity at different scales (cf. Fellner et al. 2009a) are to be considered in such an endeavor. For PAHs, the concentrations in the column leachate can be used as an estimate of maximum concentration levels in the landfill leachate. For the type of substances without direct relationships between lab-scale and field-scale data (e.g. SO_4), emission models need to account for many different factors relevant to the rate and extent of substance release from the waste. However, to evaluate emission characteristics during the landfill post-closure period, monitoring data gathered at closed C&D waste landfills will be most valuable. Direct observations at the closed site can be used to adapt an emission model to measured data and to validate (or adapt) model predictions based on continuous monitoring. Thus, although results from leaching experiments prove to be useful for establishing emission models of C&D waste landfills, appropriate monitoring data at the landfill site are necessary to calibrate and validate model estimates.

3.4 Key parameters to evaluate landfill emission behavior

The main aspects of the emission behavior of landfills containing different types of waste have been analyzed in the previous sections. Based on the quality of the deposited waste the emission characteristics differ from one type of landfill to another.

The emission characteristics of MSW landfills are strongly associated with the biochemical degradation of organic matter contained in the waste, which results in the production of landfill gas (i.e. CH_4 and CO_2) and organic leachate pollution. In addition, the elution of soluble salts and ammonium represent major substance release processes. The most problematic parameters in the leachate of closed MSW landfills with respect to discharge limits (based on a statistical analysis of emission characteristics) are NH_4, TOC (or COD), Cl, and AOX. Although decreasing trends can be observed with increasing waste age for the concentrations of most leachate parameters, the rate of decrease is very slow for some substances (e.g. ammonium). That is why ammonium is regarded as the most problematic long-term leachate constituent at MSW landfills (see chapter 3.1.2 or Kjeldsen et al. (2002)). Heavy metals concentrations in the leachate are typically low, but may exceed quality standards for discharge to surface water at individual sites or for individual measurements. With respect to landfill gas emissions methane is of primary importance. However, trace gases may be relevant based on the conditions at the site (e.g. odors) and the composition of the emission source (e.g. CFC emissions from foam insulations contained in the waste).

The emissions of MSWI bottom ash landfills are dominated by the discharge of soluble substances during operation and during the first decades after closure in the landfill leachate. The high concentrations of soluble salts, as well as the concentrations of TOC or NH_4 may be problematic with respect to leachate discharge into surface waters. Organic leachate pollution is due to the incomplete combustion of (degradable) organic matter contained in the fresh MSW and the subsequent degradation of the remaining fraction in the landfill. The concentrations of heavy metals are observed at very low levels in the leachate of MSWI bottom ash landfills, but may increase in the long term (millennia) as pH levels are speculated to decrease.

For C&D waste landfills emissions of SO_4 in the leachate are of primary interest, but also TOC, N_{tot} and Cl may be present at significant levels in the leachate. This is particularly the case if the original purity or the degree of sorting of the deposited C&D waste has been low. Metals (e.g. zinc, iron) or specific organic pollutants (e.g. PAHs) may be of sporadic relevance in the leachate of C&D waste landfills depending on the quality of the deposited material at a site.

The short overview of emission characteristics of different types of waste deposits has illustrated typical emission profiles for these landfills. Nevertheless, an evaluation of the emission behavior of a certain landfill is essentially a site-specific task. Emission models are to be established on an individual basis and need to account for such factors as the quality of deposited waste, the waste heterogeneity, the water flow regime and preferential flow, and the conditions at the site (e.g. water infiltration rates, waste thickness, deposition technique, extraction systems, etc.).

The parameters listed in Table 3-14 address the required information to evaluate landfill emission behavior. In this context, aspects of data quality management during the collection and analysis of landfill emission data should be emphasized. In general, the quality of measurements (e.g. documentation of leachate discharge rates during sampling, existence of sampling and analysis

protocols, representative sampling location and procedure) is at least as important as an adequate measurement frequency. A monitoring protocol at closed landfill sites could be based on more extensive characterization of the leachate (and landfill gas) at longer intervals, for example screening analysis of a broad range of parameters during periods with low leachate discharge rates to avoid leachate dilution with preferential flow once every few (e.g. 5-10) years, combined with routine monitoring of only a few key emission parameters (e.g. electric conductivity, ammonium) (cf. Table 3-14). Thereby the overall intensity and costs of emission monitoring could be decreased under stable conditions at the landfill site without a significant loss of information to assess the landfill emission behavior.

Table 3-14: Key parameters to evaluate the emission behavior of (closed) landfills as a basis to establish appropriate emission models

Leachate – primary parameters	**Leachate – secondary parameters**

- Leachate quality
  o Ammonia (NH_4), Nitrate (NO_3), Nitrite (NO_2)
  o COD and/or TOC, DOC, and BOD_5
  o Cl, SO_4, Metals (Fe, Zn, Cr, etc.)
- Leachate generation
  o Leachate generation rate (continuous measurement, pumping documentation, water balances, etc.)
	- EC (electric conductivity)
	- Leachate discharge during sampling
	- pH value, Redox (Eh), Temperature
	- BOD_5/COD ratio (< 0,1 typical for methanogenic phase leachate)*
	- Trace organic compounds (BTEX, AOX, hydrocarbons)
	- Other leachate constituents

Landfill gas – primary parameters*	**Landfill gas – secondary parameters***
- Landfill gas generation and collection rate (measured or calculated) - Landfill gas quality: CH_4, CO_2, O_2, N_2 (in % of volume)	- CH_4/CO_2 ratio - Trace gas compounds (e.g. H_2S, CFCs)
Deposited waste – physical factors	**Deposited waste – quality**

- Waste materials (material heterogeneity, hydraulic conductivity)
- Liquid-to-Solid ratio
- Heterogeneity of water flow
- Settlements
- Bulk density
	- Waste composition (chemical), substance balances
	- Investigations:
	  o Soluble waste fractions (function of pH value),
	  o Biochemical availability of organic waste components*

*Typically relevant only for MSW landfills.

Note:

Primary parameters are the most important emission parameters.

Secondary parameters comprise parameters which are valuable for process and/or system understanding (e.g. Redox conditions, BOD_5/COD ratio) and emission parameters of less significance, which should be included in screening analysis.

4 Barriers – technical and natural systems

The composition of a landfill is different from the surrounding environment, which causes flows to balance the differences between the systems. Apart from the actual waste composition ("first barrier"), the impact of substances released from the waste depends on the performance of technical barriers at the landfill ("second barrier", i.e. the containment system), on the one hand, and the ability of the natural environment to mitigate potential negative effects of released substances on environmental media ("third barrier", i.e. natural attenuation potential), on the other hand. These two barrier systems may prevent adverse effects on human health and the environment. Thus, they need to be included in an evaluation of landfill environmental compatibility. In this chapter, basic aspects of the long-term functioning of these systems are presented and discussed. Based on the discussion, specific approaches to include the effect of technical and natural barriers within the evaluation methodology are presented in chapter 5.

4.1 Technical barriers

Technical barriers are used to limit and control the interaction between the deposited waste and the surrounding environment. Potentially hazardous emissions can be contained (e.g. low permeability layers), collected (e.g. drainage layers), and potentially treated before they are released to the environment (cf. Table 4-1). Therefore, technical barriers are essential to prevent negative effects on the environment from modern landfills, provided the deposited waste itself is not environmentally compatible without containment (see Figure 2-2). As man-made buildings in general and technical barriers in particular, are subject to aging and material deterioration, it is of crucial importance to evaluate the long-term performance of a technical barrier system and its potential effect on the substance flows between the landfill and the surrounding environment. The analysis in this section focuses on the level of isolation and control technical barriers can provide over different time periods. With respect to the environmental compatibility of closed landfills, the main question is: How effective are technical barriers to limit interaction and for how long are they effective in the absence of care?

Table 4-1: Components and functional mechanisms of engineered barrier systems (from National Research Council 2007)

Barrier orientation	Typical component	Functional mechanism
Top barriers	Low-permeability soil layers, geosynthetic clay liners, geomembranes, asphalt-cement-liners;	Resistance
	Evapotranspirative soil layers (vegetated)	Capacitance
	Blanket drainage layers, gas vents	Extraction
Bottom barriers	Low-permeability soil layers, geosynthetic clay liners, geomembranes, asphalt-cement-liners;	Resistance
	Compacted soil attenuation layers	Capacitance
	Blanket leachate collection layers, hydraulic control layers	Extraction and/or injection
Lateral barriers	Soil-bentonite and cement-bentonite walls, cement walls, jet-grouted walls, sheet pile walls, vertical geomembranes;	Resistance
	Low-permeability treatment walls	Capacitance
	Vertical wells and trenches	Extraction and/or injection

Different components of technical barriers and their functional mechanisms are shown in Table 4-1. Lateral barriers are of primary importance at sites which have been constructed without a bottom barrier and which have been enclosed due to groundwater pollution later on. The functions of barrier components are based on different mechanisms, which are typically combined to achieve a specific performance level of the barrier system. Components which resist diffusive/convective flow are of central importance for technical barriers. In addition to resistance, compacted clay layers also employ the functional mechanism of capacitance, as they may retard the breakthrough of pollutants due to their cation exchange capacity. However, if capacitive components are not combined with other mechanisms, for example with extraction: evapotranspirative barriers store the infiltrating water before the flow gradient is inversed and the stored water is extracted due to evaporation and transpiration, they will slow down pollutant migration but not prevent it. Typical components of technical barrier systems employing extraction as a functional mechanism are gas vents or drainage layers, where the fluid is removed due to a hydraulic gradient or pumping (cf. National Research Council 2007).

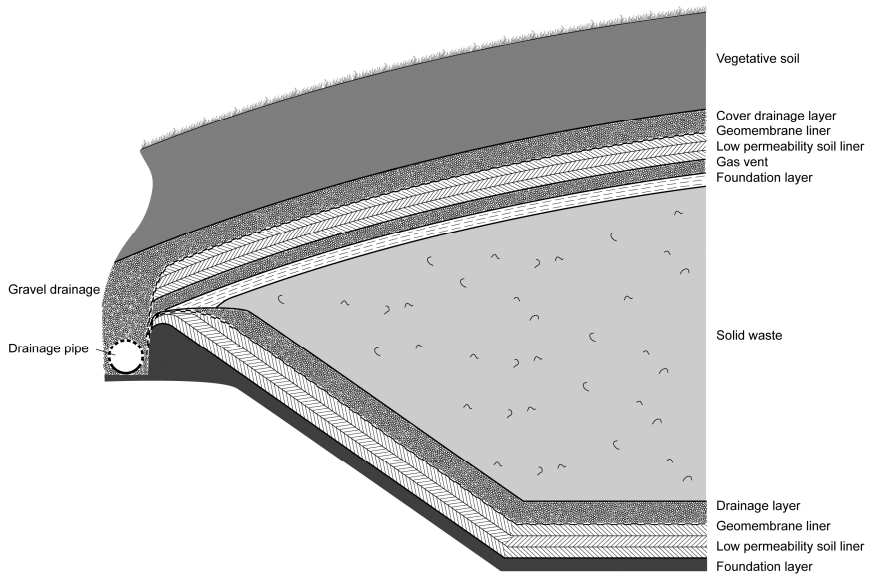


Figure 4-1: Schematic illustration of a landfill containment system based on the requirements of the Austrian landfill directive for landfills containing pre-treated MSW (cf. Figure 4-2 and Figure 4-4)

A technical barrier system according to the minimum requirements of the Austrian landfill directive (MoE 2008) for landfills containing pre-treated MSW is schematically shown in Figure 4-1. The illustrated barrier system is composed of composite liner systems with low permeability soil liners and geomembrane liners at the top and at the bottom of the landfill. According to Austrian regulations alternative barrier systems can be permitted if they perform equally to the standard systems (see also Figure 4-2 and Figure 4-4). Therefore, and due to different regulations over time and across landfill types, a large variety of technical barrier systems relying on different functional mechanisms and components exists at landfills. Numerous studies have been carried out to analyze and compare different barrier systems with respect to their performance and reliability. Extensive work on the long-term performance of containment systems was presented for instance by Rowe (2005), the National Research Council (2007), and Inyang (2004). A comprehensive

investigation based on a large data set on the performance of engineered containment systems at field scale has been reported by Bonaparte et al. (2002). These studies are, among others, an important basis for the review of the long-term performance of technical barrier systems in this section.

4.1.1 Components of technical barriers

Technical barrier systems comprise several components fulfilling different functions. Although the effectiveness of a barrier component depends on the other system components and on local conditions (e.g. climate), properties and functional mechanisms of barrier components are discussed individually below, to illustrate their reliability and use as elements of technical barrier systems. An overview of different barrier components is given in Table 4-2 together with major performance concerns on the short, medium, and long term.

Table 4-2: Performance concerns for barrier system components on the short, medium, and long term (based on National Research Council 2007)

Barrier system component	Service period	
	Short-term aspects	Medium- to long-term aspects
Earthen barriers	Defective materials; inadequate compaction (density and/or moisture); slope stability	Cracking due to shrink/swell, freeze/thaw, root penetration, differential settlement, desiccation; chemical incompatibility; waste and slope stability
Geomembranes	Defective materials; physical damage due to construction; defective seams	Puncture; global and local stability; degradation
Geosynthetic clay liners (GCLs)	Defective material; seam separation	Cracking due to shrink/swell, freeze/thaw, root penetration, differential settlement, etc.; chemical incompatibility; local and global stability; reinforcement degradation (needle-punched reinforced GCLs); inadequate hydration (encapsulated GCLs)
Asphaltic cement barriers	Defective material	Cracking due to shrinkage or deformation, degradation of the asphalt binder or supplemental material (e.g. crumb rubber)
Drainage layers (granular and geosynthetic)	Inadequate capacity	Clogging due to soil infiltration, biological action, and mineral precipitation; geosynthetic drainage layers are susceptible to soil and geosynthetics penetration and creep of the geonet core
Evapotranspirative barriers	Defective material; inadequate thickness; inability to establish vegetation	Inadequate storage capacity for infiltration; inability to sustain vegetation; cracking and development of other secondary permeability features; erosion; penetration by vegetation or animals

4.1.1.1 Low permeability earthen barriers

Low permeability soil liners are the best investigated components of barrier systems (Inyang 2004). The main properties of low permeability earthen barriers with respect to pollutant transport are hydraulic conductivity, chemical robustness, chemical transport parameters (e.g. cation exchange capacity), and mechanical aspects (e.g. vulnerability to erosion or cracking). The major mechanisms of pollutant transport through the low permeability soil layer are advection, under saturated conditions described by Darcy's law (Darcy 1856), and diffusion, described by Fick's law (Fick 1855)[7]. In general, the lower the hydraulic conductivity of a soil layer, the more important the

[7] Darcy's law: $v_f = k_f * i$ (v_f… filter velocity [m/s], k_f… hydraulic conductivity [m/s], i… hydraulic gradient [m/m]); Fick's law: $J_d = D_0 * dc/dx$ (J_d… diffusive flux [mol/(m².s)], D_0… diffusion coefficient [m²/s], dc/dx… concentration gradient against diffusive flux [mol/m^4])

diffusive flux may be for pollutant transport processes. A more detailed description of pollutant transport processes in soil is provided in section 4.2.2.

Hydraulic conductivities below 10^{-9} m/s are often required for low permeability earthen barriers and can be achieved by appropriate design and construction. Nevertheless, several factors potentially cause the hydraulic conductivity to increase directly after construction or in the medium to long term (cf. Table 4-2). Field tests (e.g. double ring infiltrometer experiment) should be used to demonstrate the hydraulic conductivities after construction, as laboratory-scale experiments do not necessarily reflect the hydraulic conductivity in the field scale (Benson et al. 1999). A decrease of barrier performance after construction, may be caused by crack formation (shrink/swell, freeze/thaw, root penetration, differential settlement, desiccation), chemical or mechanical instability and can typically not be detected directly due to the lack of direct monitoring (cf. BAM 2000). Once cracked, the hydraulic conductivity of the low permeability soil layer remains elevated even if the crack disappears again (cf. Gray 1989; Stoffregen et al. 1999). While a significant increase in hydraulic conductivity is not expected unless the crack reaches over more than 80 % of the soil layer thickness, hydraulic conductivity may increase by three orders of magnitude in case of continuous cracks through the whole liner (Gray 1989).

Despite inadequate design of the barrier system, e.g. insufficient thickness of protective layers on the top of the landfill to prevent desiccation (cf. Bonaparte et al. 2002; Henken-Mellies 2006), temperature-induced water flow may be an important factor for the desiccation of low permeability soil liners (cf. Stoffregen et al. 1999). At the landfill base the primary factors to evaluate the probability of temperature-induced water transport causing desiccation are the distance to groundwater and the temperature gradient (positive correlation) as well as the thickness of the liner and the heat conductivity (negative correlation). At the top, temperature-induced water transport and desiccation are dependent mainly on the temperature in the waste and the climatic conditions. Due to the complex interaction of different factors, the risk of desiccation has to be evaluated on an individual basis, whereby crack formation tendency decreases at the landfill base with increasing overburden, i.e. waste thickness.

The chemical incompatibility of clay-rich soil layers with a liquid may be directly associated with an increase in hydraulic conductivity (e.g. highly concentrated (> 75 vol%) organic solvents[8] (cf. Farquhar and Parker 1989)) or cause chemically induced material deterioration and a subsequent increase of hydraulic conductivity. In general, clay-rich soil layers are more susceptible to chemical incompatibility with liquids exhibiting higher concentrations and/or charges of dissolved ions. Clay layers with high plasticity indices (e.g. increasing content of bentonite) are more susceptible to chemical incompatibility than earthen barriers with low plasticity (Benson et al. 1999).

Apart from resistance to advective and/or diffusive flux of pollutants through low-permeability soil layers, clay-rich soil layers have the capacity to retard or prevent the breakthrough of several pollutants (e.g. Griffin and Shimp 1975; Farquhar and Parker 1989). While anions (e.g. Cl, Br) and organic solvents are barely retained in clay-rich soil layers, the retention of heavy metals (e.g. Pb, Zn, Cd) and other cations is potentially high. Non polar organic compounds are primarily sorbed to

[8] However, note that typical concentrations of organic solvents in the leachate of MSW landfills are in the range of parts per million.

organic carbon in the soil, making the total organic carbon content of the low permeability soil layer the dominant indicator to evaluate the sorption of non polar organic compounds.

In summary, low permeability soil layers are an effective component of technical barrier systems over geologic time periods, provided appropriate design and construction. Advantages of these components are relatively low construction costs, their attenuation capacity with respect to specific pollutants, and their robustness in view of mechanical damage. In addition, the multi-layered construction of such barrier components decreases their vulnerability to cracking and defective materials. The main risks with respect to their effectiveness relate to potential desiccation or damage during construction, on the one hand, and secondary desiccation processes and consequent cracking in the medium and long term, on the other hand (cf. Table 4-2).

4.1.1.2 Geomembranes

Modern geomembranes consist of high density polyethylene (HDPE) and have been used at landfills since the 1970s. Initially other materials such as polyvinylchloride (PVC) or low density polyethylene (LDPE) have been also used in geomembranes, but HDPE geomembranes[9] became increasingly popular due to their higher robustness in comparison to alternative materials, as for example softeners are leached from PVC and membranes get brittle, LDPE is more susceptible to mechanical damage than HDPE (cf. Bräcker 2002). Without holes, geomembranes do not allow for advective flow, roots cannot penetrate, and they are not susceptible to damage by rodents.

On the short term, geomembranes may not reach their design performance level due to poor material quality, mechanical damage during construction and waste placement, or defective seams (cf. Table 4-2). On the medium and long term, a decrease in performance may occur primarily due to differential settlements and gradual material deterioration (cf. Table 4-3). The number of holes in a geomembrane is largely dependent on the stringency of quality assurance and control (QA&C) programs during construction. It has been demonstrated that the frequency of holes in geomembranes decreases significantly if QA&C programs have been in place during construction (e.g. Phaneuf and Peggs 2001; Forget et al. 2005). For instance, Forget et al. (2005) reported 0.5 holes per hectare at sites with QA&C programs during construction and 16 holes per hectare at sites without appropriate QA&C programs. This is consistent with investigations by Rowe et al. (2004), who reported average hole frequencies at HDPE geomembranes between 0.7 and 11 holes per hectare, whereby 30% of the investigated liners did not have any holes and 50% had less than 5 holes per hectare. The magnitude of advective flow due to holes is dependent on the quality of contact between the geomembrane and the low permeability soil layer in composite liners (cf. Rowe 2005). In addition to the quality of contact between HDPE membrane and soil liner, the presence/non-presence of wrinkles (originating from construction practice and/or thermal expansion processes) is significant for the magnitude of advective flow through holes at geomembranes. The higher the number of wrinkles (Rowe et al. (2004) reported a case with 1200 wrinkles per hectare), the more pronounced the effect of holes on the release of leachate and pollutants is. Different analytical (e.g. Giroud and Bonaparte 1989) and empirical (e.g. Rowe and

[9] While the thickness of HDPE geomembranes is typically 2.5 mm in Austria or Germany, 2 mm thick HDPE liners are widely used in the USA and Canada.

Booker 2000) equations are available to describe leachate flow through geomembranes with holes. A description and discussion of these approaches is provided in Rowe (2005). However, at undamaged HDPE geomembranes without holes, not advective but diffusive flux of pollutants (gaseous and liquid) is of primary importance, particularly for volatile organic compounds (cf. National Research Council 2007).

Technical lifetimes of geomembrane liners have been estimated based on laboratory data, as field observations are not available over a sufficient time period (Rowe 2005). From Table 4-3 it is apparent that the degradation of the liner material is strongly dependent on the temperatures at the geomembrane, as it may take from a few decades to several thousands of years until the polymers are deteriorated. The deterioration process can be divided into a first phase of antioxidant depletion, a second phase of initial polymer degradation, and a third phase of polymer degradation. The technical lifetime (=period during which design performance is reached) of HDPE geomembranes at MSW landfills has been assumed to range from around 100 years to several hundreds of years (e.g. Bräcker 2002; Simon and Müller 2004; Rowe 2005; Henken-Mellies 2006; Ministry of the Environment 2010).

Table 4-3: Service lives for HDPE liners (2 mm thickness) estimated via calculations based on laboratory experiments (Source: Rowe 2005)

Temperature	Stage 1*	Stage 2**	Stage 3**	Total
[C°]	[Years]	[Years]	[Years]	[Years]
10	280	30 – 50	1380 – 2445	1690 – 2775
20	115	10 – 15	440 – 765	565 – 900
30	50	4 – 6	150 – 260	205 – 315
35	35	2 – 4	90 – 155	130 – 190
40	25	1 – 2	55 – 95	80 – 120
50	10	0.6 – 1	20 – 35	35 – 50
60	6	0.3 – 0.4	9 – 15	15 – 20

* Based on simulated liner antioxidant depletion tests

**higher values relate to experiments with water-air-interface, lower values are adjusted for the possible effect of leachate on the rate of degradation

In summary, HDPE geomembranes without holes/defects do not allow for advective flow and are robust components of technical barrier systems. As proper construction of the liner and of the protective layers is of high significance, appropriate QA&C programs at a site are crucial to achieve design performance specifications. Technical lifetimes of HDPE geomembrane liners are typically assumed to be more than 100 years, if high temperatures and extreme external stresses are not expected to affect the geomembrane.

4.1.1.3 Geosynthetic clay liners (GCLs)

Similarly to low permeability earthen barriers, geosynthetic clay liners, i.e. geotextiles with a bentonite filling as liner, can be functional barrier components for geologic timeframes. The major performance concerns are similar to low permeability earthen liners with the additional aspects of the degradation of the reinforcing geotextile and the inadequate hydration of encapsulated GCLs (cf. Table 4-2). The permeability of a non-defective GCL is equal to the permeability of a low per-

meability soil liner with a thickness of 0.5 m and a hydraulic conductivity k_f= 5*10^{-9} m/s (cf. Bräcker 2002).

The likelihood of chemical incompatibilities of GCLs with liquids increases with increasing concentrations of divalent ions (e.g. Ca^{2+} or Mg^{2+}) in a liquid, as these cations replace the Na^{+} contained in bentonite and thereby increase the hydraulic conductivity of the GCL potentially by several orders of magnitude (cf. Kolstad et al. 2004). Apart from the increase in hydraulic conductivity, the substitution of sodium by higher valent cations reduces the self-healing ability of bentonite, as the extent of clay swelling decreases. Desiccation poses a threat to the performance of GCLs in general. In top cover systems GCLs may be also affected by root penetration, which should be excluded via protective layers of sufficient thickness.

GCLs, similar to clay-rich low permeability soil liners, have a potential to retain pollutants and attenuate negative effects of leachate percolating through the liner at the landfill base. However, as GCL thickness is approximately one tenth of a compacted soil liner's thickness, their potential to attenuate pollutants is consequently lower.

4.1.1.4 Asphalt and asphaltic cement barriers

Asphalt and asphaltic cement liners are composed of bitumen binder and sand as mineral additive. Typically, these liners are used as an alternative to geomembrane liners in barrier systems. They do not allow for advective flow through the liner with technical service lives in a range of hundreds of years (similar to geomembrane liners (cf. Bräcker 2002)). Asphalt-based liners are robust towards mechanical and biological stresses (e.g. roots, rodents) and not vulnerable to desiccation. Negative effects on liner performance can be caused by material incompatibilities (e.g. chemical attack of asphalt by hydrocarbons (Simon and Müller 2004)) or vertical displacements of the liner. Nevertheless, if properly constructed, asphalt and asphaltic cement liners represent a practically impermeable barrier component for a long period of time.

4.1.1.5 Drainage layers

Drainage layers are used to extract fluids from the barrier system due to the higher conductivity of these layers compared to other barrier components (cf. Figure 4-1). The layers can be built from mineral material (e.g. gravel) as well as from geotextiles (e.g. polyethylene or polypropylene meshes). Drainage layers are important in a barrier system to avoid excessive head on liners and their function is based on high hydraulic conductivities and sufficient layer thicknesses.

On the short term, the main reasons for poor performance of a drainage layer may be insufficient hydraulic conductivity or thickness (i.e. design), poor filter stability, or a geotechnically instable drainage layer. In the medium to long-term, the primary cause of drainage problems is the clogging of the drainage layer due to fine particles leached from other barrier components, biological growth in the layer (biofilms), or the precipitation of dissolved substances, like $CaCO_3$ (cf. Table 4-2 and Rowe (2005)). Clogging may result in a decrease of hydraulic conductivity by up to eight orders of magnitude and thus, represents a critical process with respect to long-term drainage layer performance. In the case of granular drainage layers, the probability of clogging is lower for layers with large grain sizes and uniform grain size distributions. Consequently, sandy drainage layers are more likely to clog (e.g. it was observed within a few years at MSW landfills) than gravel drainage

layers (cf. Rowe 2005). In the case of geotextiles, the compression of the geotextile due to increasing overburden, the degradation of polymers, and geotextile creep can also cause performance problems. Advantages of geotextile drainage layers are the industrial production and quality control, the lower required thickness of the layer, and the easy handling during construction (Simon and Müller 2004). In general, if drainage layers are properly designed, constructed, and maintained, technical lifetimes of several decades can be expected (cf. Rowe 2005). However, as the importance and interaction of clogging mechanisms is not fully understood, confident estimates on the medium- to long-term performance of drainage layers are hardly possible.

4.1.1.6 Vegetative cover soil

The vegetative cover soil (=recultivation layer) is a part of the top cover to protect drainage layers and liners against different impacts (e.g. roots, freezing) and represents the face of the closed landfill in the landscape. Austrian regulations require a minimum thickness of the vegetative soil of 0.5 meters (cf. Figure 4-2). Based on long-term considerations, e.g. typical vegetation at the site, thicknesses of the vegetative cover soil from 1.5 to 3.0 meters have been put forward (cf. Bräcker 2002). As a long-term component of the barrier the vegetative cover soil should have significant water storage capacity to maintain a water balance with low water percolation rates to the lower layers of the barrier system (Henken-Mellies and Schweizer 2011). However, top soil layer design and required performance levels may vary from one site to another based on the aftercare and after-use concept and the local conditions (e.g. climate, vegetation).

4.1.1.7 Further barrier system components

Various components are used in barrier systems to provide specific functions. Geotextiles are used in numerous applications as nets, meshes, or sheets. Capillary barrier layers can be used on the side slopes of landfill top covers and consist of a capillary layer (fine grained material) above a capillary block (larger grain sizes). Under unsaturated conditions the capillary layer has a higher hydraulic conductivity as the capillary block and therefore drains the water to the side without releasing it to the capillary block (cf. Simon and Müller 2004). However, although robust to desiccation, capillary barriers are difficult to design, which makes the demonstration of field performance inevitable.
A special type of low permeability earthen liner is commercially available under the name Trisoplast and consists of 90% sand, 10% bentonite, and some polymer additives (<0.2%). The liner has a low permeability (0.1 m of Trisoplast perform equally to 0.5 m compacted clay) and is insensitive to desiccation as well as to differential settlements (Bräcker 2002). Apart from that, the critical issues described for low permeability soil layers and geosynthetic clay layers are of potential significance also for the performance of Trisoplast.

The list of barrier components in this section is not complete, but comprises the most widely applied elements of technical barrier systems. In general, it should be emphasized that performance criteria for individual barrier components are derived from the specifications of the whole system and its aspired performance. Thus, the performance of the whole barrier system is of primary interest in an evaluation and not the performance of single components.

4.1.2 Performance of landfill top cover systems

The main purposes of the barrier system at the top of a landfill are to control/minimize water infiltration into the waste, to prevent the uncontrolled release of landfill gas to the atmosphere (in case of MSW landfills), and to allow for an appropriate after-use of the landfill site. In addition, the top cover makes the waste inaccessible to birds and rodents, prevents wind-blown dispersal of waste, and reduces unpleasant odors from the waste. However, as the latter factors are not of primary importance for closed landfills, these will not be addressed in more detail below.

The minimum requirements of the Austrian landfill directive (MoE 2008) for top cover systems at construction and demolition waste landfills (left) and landfills for pre-treated MSW (right) are shown Figure 4-2. Although different systems can be used as long as equal performance with the systems in Figure 4-2 can be demonstrated, the discussion in this section will mainly focus on these "standard" top cover designs.

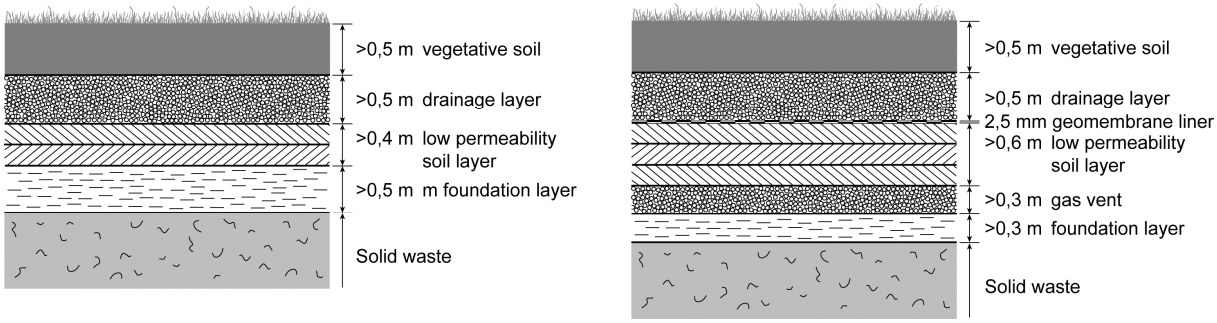


Figure 4-2: Minimum top cover barrier system according to the Austrian landfill directive (MoE 2008) for C&D waste landfills (left) and landfills for pretreated MSW (right)

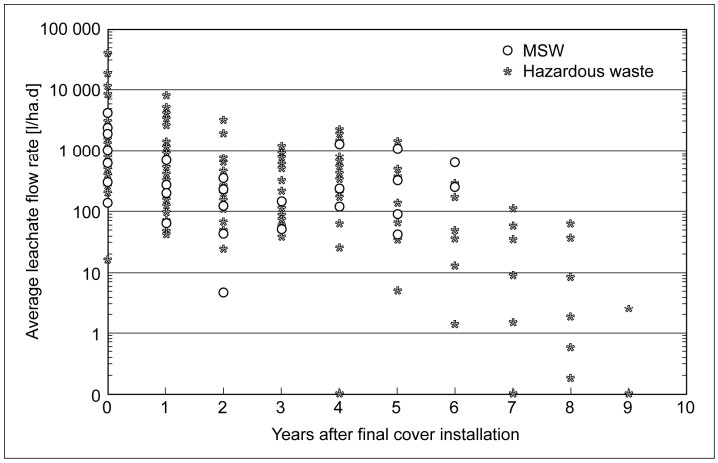


Figure 4-3: Average leachate generation rates after closure for 11 MSW cells and 22 cells containing hazardous waste (Source:Bonaparte et al. 2002)

The performance of top cover systems is typically evaluated based on the amount of water percolating through the top cover into the waste body. Often the corresponding criterion is the amount of leachate generated at a landfill as a fraction of the annual precipitation at the site. For instance, the Austrian landfill directive (MoE 2008) requires that final cover systems do not allow for leachate generation rates above 5% of the average annual precipitation. This performance criterion needs to

be reached at the latest 5 years after the installation of the final cover and demonstrated throughout the whole period of aftercare. In Figure 4-3 the average leachate flow rate at 33 landfill cells is shown after a final cover system had been installed at these sites (cf. Bonaparte et al. 2002). Especially at hazardous waste landfills (with a composite liner system) a rapid reduction of leachate flow rates could be observed, as the flow rates decreased by two orders of magnitude within the first decade after cover installation. Thus, composite liner systems appear to be highly efficient systems to minimize water infiltration into the waste, provided proper construction and design.

Mineral liner systems at the top of landfills often exhibit decreasing performance within a few years after installation (e.g. Stoffregen et al. 1999; Benson 2001; Bonaparte et al. 2002; Melchior et al. 2002; Henken-Mellies and Schweizer 2011). For instance, Huber et al. (2002) analyzed data on leachate flow rates at 17 closed landfills in Bavaria (Germany) with mineral liner systems at the top, with thicknesses of at least 0.5 m and specified hydraulic conductivies $<10^{-8}$ m/s. They found that the average leachate flow rates at the sites amounted for 10 to 20% of the average annual precipitation. Thus, the investigated mineral liner systems were generally not able to minimize water infiltration for more than a few years. Similarly, Bonaparte et al. (2002) and Melchior et al. (2002) found that top cover systems based on compacted clay layer as the sole resistive layer allowed for annual water infiltration rates above 10% of precipitation already a few years (two to six years) after cover installation. In most cases the reason for the poor performance of single mineral liner systems at the landfill top is the desiccation of the low permeabliliy soil liner and the subsequent formation of cracks. The desiccation of the liner is often due to insufficient thickness of the vegetative soil layer, clay liners with high plasticity indices, and/or root penetration and consequent water extration by plants. In addition, differential settlements (e.g. due to significant organic degradation processes still ongoing in the waste) may cause the formation of cracks (cf. Simon and Müller 2004). Consequently, mineral liner systems are not suited to guarantee for a lasting minimization of leachate generation due to their susceptibiliy to desiccation and cracking. As cracks are preferential flow paths for liquid and gaseous emissions, they are problematic also with respect to the collection and treatment of potentially generated landfill gas (cf. Bonaparte et al. 2002). Therefore, if a top cover system employs a sole mineral liner, it is important to protect the liner against desiccation (i.e. through appropriate thickness of vegetative soil layer) and to assure that settlements of the landfill surface have been largely completed.

Contrarily to mineral liner systems, composite liner systems (i.e. HDPE geomembrane above a low permeability soil liner) have been demonstrated to be effective barriers at the top of landfills at many sites with different climatic conditions (see Figure 4-3 and Benson 2001; Bonaparte et al. 2002). The two liners function independent from one another and, provided appropriate design and construction, represent a system to effectively minimize interaction between waste and the surrounding environment. The effect of holes and fissures of the geomembrane is less pronounced if the contact between geomembrane and low permeability soil layer is intimate (i.e. water infiltrates only locally into the low permeability soil liner). As long as the geomembrane liner is functional, the main threat of soil liner desiccation is posed by temperature induced water transport processes (cf. Stoffregen et al. 1999). The potential of such processes to dry out the soil liner has to be evaluated individually and is dependent on several factors, such as waste temperatures, fluctuations of atmospheric temperatures, and vegetative soil layer thickness. However, after the

technical lifetime of the geomembrane is expired (often a few hundred years are assumed, see section 4.1.1), the mineral liner will be the controlling resistive barrier at the landfill top. Thus, the long-term functioning of this liner must not rely on a fully functional geomembrane liner for protection (e.g. the geomembrane shall not be designed to protect the low permeability soil liner against root penetration) if minimal water infiltration shall be maintained over long timeframes (cf. Stoffregen et al. 1999; Henken-Mellies and Schweizer 2011).

Table 4-4: Examples of typical top cover barrier systems using composite liners (adapted from Henken-Mellies 2006)

Composite liner systems	Components (excluding gas vent and foundation layer)
Low permeability soil liner and geomembrane liner (cf. Figure 4-2)	> 0.5m vegetative soil, > 0.5 m drainage layer, 2.5 mm geomembrane liner, 0.6 m low permeability soil liner (construction in 3 layers)
Geosynthetic clay liner (GCL) and geomembrane liner	Same as above, but with GCL instead of low permeability soil liner
Trisoplast* and geomembrane liner	2.0 m vegetative soil, 0.3 m drainage layer, 2.5 mm geomembrane liner, 0.07 m Trisoplast*
Capillary barrier and geomembrane liner	2.0 m vegetative soil, 0.4 m capillary layer, 0.25 m capillary block, 2.5 mm geomembrane liner
Low permeability soil liner and asphalt liner	1.5 m vegetative soil, 0.3 m drainage layer, 0.075 m asphalt liner, 0.2 m soil foundation layer, 0.2 m capillary layer, 0.1 m capillary block
Low permeability soil liner and capillary barrier**	2.0 m vegetative soil, 0.15 m drainage layer, 0.6 m low permeability soil layer, 0.3 m capillary layer, 0.15 m capillary block

*Trisoplast is a commercially available liner and consists of sand (90%), bentonite (10%) and polymer additives (<0.2%) (see also section 4.1.1).

** In a lysimeter study this combined liner system became inefficient within a few years.

Beside the standard top cover systems, numerous alternatives exist to control/minimize the infiltration of water into the waste. Some designs of composite liner systems, which have been investigated in the field-scale (lysimeter studies or full scale landfill covers), are listed in Table 4-4. Apart from one system (the combination of a low permeability soil liner with a capillary barrier underneath), the observations demonstrate the appropriate performance of the top covers. However, conclusions with respect to the medium- and log-term performance of these systems cannot be drawn from the available data yet, as observations typically exist for a few years only.

An alternative to the minimization of water infiltration into the waste via resistive barrier components (i.e. single and composite liner systems) are landfill top covers designed as evapotranspirative barriers (cf. Wimmer and Reichenauer 2006). The functioning of such systems relies on the storage of rainwater and subsequent extraction (of a large part of the water) due to evaporation and transpiration (roots) processes. Major advantages of these systems are low maintenance requirements and long-term functionality, provided a design appropriate for the conditions at the site. Evapotranspirative barriers allow for some level of water infiltration, which may be comparable to the local groundwater recharge rates of natural soils at the site.

The performance of top cover systems is typically evaluated based on leachate collection rates, because direct monitoring data are not available for the barrier system. For instance, a direct monitoring system would be the (area-wide) measurement of the electrical resistance in the liner to

infer changes of the water content from changes in electrical resistance (cf. BAM 2000). Based on such data, measures against decreasing barrier efficiencies could be taken before unwanted effects of water infiltration (e.g. landfill gas and leachate generation) occur.

The performance of barrier systems at the top of landfills depends on several factors such as the climatic conditions, the properties of the landfill body (i.e. waste characteristics and ongoing processes), and the constitution of the barrier system. Top covers employing solely a low permeability soil liner as a resistive component often do not perform according to design criteria due to desiccation and subsequent cracking of the liner (leachate flow rates amounted for 10-20% of annual precipitation a few years after cover installation). In particular, systems with mineral liners built from fine grained material and with a protective vegetative soil layer of less than 2 meters thickness are susceptible to desiccation cracking (cf. Henken-Mellies and Schweizer 2011). Top covers with composite liner system minimize the infiltration of water into the waste effectively, provided that they are properly designed and constructed. While the performance of these systems can probably rely on the combined function of the geomembrane and the mineral liner for a few hundred years, the long-term performance of the system will be controlled by the mineral liner. The reason is the gradual degradation of the geomembrane liner without maintenance (100-300 years have been assumed) and the potential functionality of low permeability soil liners over geologic timeframes.

4.1.3 Performance of technical barriers at the landfill bottom

Engineered barriers at the landfill bottom have been constructed for several decades and changed from simple systems (e.g. peat layers) to more sophisticated systems (e.g. composite liner systems) over time due to increasingly stringent landfill regulations (Krümpelbeck 2000). The minimum design requirements for bottom barrier systems according to the Austrian landfill directive are schematically illustrated in Figure 4-4 for C&D waste landfills (left) and for landfills containing pretreated MSW (right). Though alternative systems exist and may be permitted (provided that equal performance with the respective system in Figure 4-4 can be demonstrated), the standard bottom barrier systems in Figure 4-4 are most widely applied and in the focus of this section.

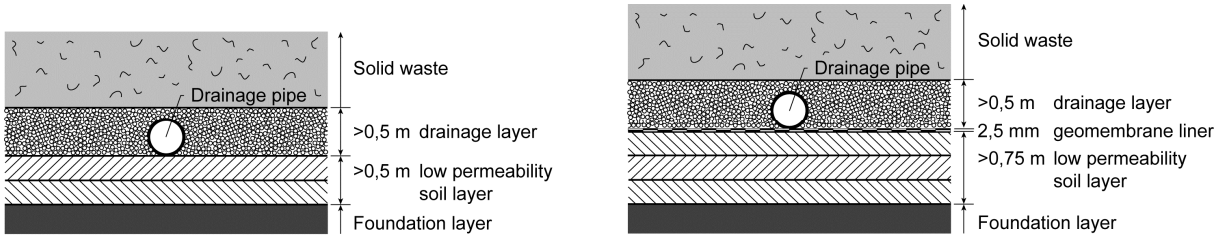


Figure 4-4: Minimum bottom barrier system according to the Austrian landfill directive (MoE 2008) for C&D waste landfills (left) and landfills for pretreated MSW (right)

Bottom barrier systems with a low permeability soil liner are not impermeable to advective flow, but reduce water flow through the barrier due to low hydraulic conductivities (typically $\leq 10^{-9}$ m/s) and the prevention of high hydraulic heads on the liners (i.e. drainage layer). Therefore, diffusion may be a significant process with respect to pollutant migration through low permeability soil liners (National Research Council 2007). For instance, Rowe (2005) reported diffusive transport dis-

tances for chloride of 0.75 m within 4.5 years and of 1.0 m within 15 years at a landfill with a mineral liner system, respectively. He estimated typical diffusion coefficients for chloride in low permeability soil layers to be 0.0063-0.019 m²/yr, which is in the same range as diffusion coefficients for natural clay-rich soils under saturated conditions.

In general, the combination of two liners in a barrier system is more effective to prevent the release of pollutants than a single liner system. For instance, Bonaparte et al. (2002) investigated the leachate flow rates in secondary collection systems below the primary bottom barrier system at different landfills[10] and found that composite liner systems were more effective than single liner systems with a sole geomembrane liner. The leachate percolation rate through the geomembrane liner was by factor two higher than at composite liner systems with a geomembrane and a low permeability soil layer and by factor 400 higher than at composite liner systems with a geomembrane and a GCL (Note: A drainage layer below a low permeability soil liner increases the risk of liner desiccation due to temperature-induced water transport processes (cf. Stoffregen et al. 1999)). Another investigation of leachate percolation rates at landfills with primary and secondary bottom liner systems was reported by Barlaz et al. (2002). The leachate collection efficiencies of the primary liner system ranged from 97.1 to 99.9% (see Table 4-5), which highlights the ability of composite liner systems to effectively contain generated leachate.

Table 4-5: Efficiency of primary bottom liner systems at 17 landfills in the years 1998 and 1999 (Source: Barlaz et al. 2002)

Year	Value	Leachate collection rate [l/(ha*d)]	Leachate collection after primary liner [l/(ha*d)]	Liner efficiency [%]
1998	Average	159.1	1.0	99.2
	Maximum	372.5	4.0	99.9
	Minimum	27.4	0.1	98.0
1999	Average	117.4	0.8	99.1
	Maximum	290.8	2.6	99.9
	Minimum	16.3	0.1	97.1

Note: Data from 17 landfills in the state of New York (landfills are in operation)

In a composite liner system the effect of holes or defects of the geomembrane is limited to a local release of leachate into the mineral liner if the two layers are in intimate contact. Poor contact, for instance due to wrinkles, may have a strong effect on the permeability of the whole system to advective flow. Depending on the number and the location (with respect to wrinkles) of holes in the geomembrane, the hydraulic conductivity may increase by several orders of magnitude compared to composite liners with good contact between the geomembrane and the low permeability soil liner (cf. Rowe 2005). In addition to the robustness against holes in the geomembrane, synergetic effects of the combination of two liners occur with respect to diffusive transport processes through the bottom barrier system. While geomembranes are practically impermeable to polar molecules (e.g. salts, metals, organic acids), they have a low resistance against non polar molecules (e.g. vola-

[10] The investigated sites are equipped with a primary and a secondary bottom liner system. The primary liner system lies above the secondary liner system and includes the leachate collection system. The secondary liner system provides data on the amount of leachate percolating through the primary liner, as the percolate is collected in a drainage layer.

tile carbon compounds). As the latter can be retained by polar soil particles in the mineral liner their concentration increases at the interface of the geomembrane and the low permeability soil liner. Consequently, the concentration gradient along the barrier system decreases and diffuse transport is slowed down significantly. However, as concentrations of non polar substances are typically low in landfill leachate, the importance of diffusive transport may be rather limited for composite liner systems with a functional geomembrane liner (National Research Council 2007).

A decrease in the performance of the low permeability soil liner after the construction of the composite liner system can be caused by temperature-induced water transport processes. The higher the temperature at the landfill base, the more water is transported downwards from the liner interface. Thus, if the liner can drain to a layer with higher hydraulic conductivity, the low permeability soil layer may desiccate and cracks can form (Stoffregen et al. 1999). Whether temperature-induced water transport poses a threat to a mineral liner depends on several factors (e.g. overburden, heat conductivity of the liner, distance to groundwater, etc.) and is difficult to evaluate.

Apart from liners, drainage layers represent important components for the functioning of bottom barrier systems. Drainage layers are used to avoid the build-up of excessive hydraulic heads on liners by collecting and extracting the leachate at the bottom of the landfill. The leachate typically drains into collection pipes directing it to a storage tank. The performance of a drainage layer depends on appropriate dimensions (i.e. thickness) and sufficient hydraulic conductivity. Several processes can contribute to the deterioration of a drainage system, eventually causing the build-up of significant hydraulic heads in the landfill. Technical lifetimes of drainage systems vary from one system to another, but are supposed to be less than 100 years in the absence of maintenance (cf. Rowe 2005; Ministry of the Environment 2010).

In summary, composite liner systems represent an effective technical barrier to prevent the release of pollutants to the subsurface at landfills. Synergetic effects of the liner combination are the increased robustness of the system with respect to geomembrane defects (holes, fissures) and the complementary barrier effects against diffusive transport of pollutants. Provided appropriate design and construction, technical lifetimes of such systems may expand over long time periods, with the long-term performance controlled by the low permeability soil liner or the geosynthetic clay liner, respectively. Though, models of different complexity are available to describe pollutant transport processes (i.e. empirical, analytical, and numerical models, see Rowe 2005) through composite liner system, the interaction of various processes, and the potential change of material properties represent unresolved issues with respect to modeling. In addition, data to calibrate and validate such models are hardly available, as direct monitoring (i.e. primary and secondary liner systems) is often not possible at bottom barrier systems. Typically, the performance of bottom barrier systems is monitored indirectly via groundwater wells downstream of the landfill. However, as observation wells may potentially miss an existing pollutant plume and/or pollutant travel times (time period between actual release of a pollutant and the detection at the monitoring well) may be very long (cf. Inyang 2004), the direct monitoring of bottom barrier system performance represents the best way to increase system understanding based on field-scale monitoring data.

4.1.4 Long-term performance of technical barriers

4.1.4.1 Performance evaluation and modeling

The performance of technical barriers can be evaluated based on different criteria. Top covers are often evaluated based on the amount of water infiltrating into the waste approximated by the flow rate in the leachate collection system. Direct performance evaluations of bottom barrier systems are not possible at many sites, as monitoring often focuses on potentially affected environmental media and not on the barrier system itself (e.g. concentrations of pollutants detected at groundwater monitoring wells). In addition to performance indicators of the whole system, criteria can be defined also for single components. Such criteria are, for instance, the hydraulic conductivity of a low permeability soil liner or the thickness of a geomembrane liner or the maximum build-up of hydraulic head in a drainage layer. It should be noted though that these criteria can be addressed primarily during and immediately after construction and therefore, secondary effects on performance levels can typically not be evaluated directly any more.

Table 4-6: Observed problems at landfill containment systems, their significance, and the dominant human factor contributing to the problem (based on Bonaparte et al. 2002)

Containment system (component)	Significance*	Dominant human factor		
	(%)	Design**	Construction**	Operation**
Bottom liner system – construction	18	1	11	2
Bottom liner system – degradation	9	3	3	1
Bottom liner system – stability	15	9	0	3
Bottom liner system – displacement	5	4	0	0
Leachate drainage system – construction	8	0	6	0
Leachate drainage system – degradation	6	2	3	0
Leachate drainage system – malfunction	5	2	0	2
Leachate drainage system – operation	5	1	0	3
Top cover system – construction	2	0	2	0
Top cover system – degradation	2	2	0	0
Top cover system – stability	23	16	2	0
Top cover system – displacement	2	1	1	0

* Overall 69 landfills have been investigated – percentages relate to the frequency of the problems occurred at these sites

** In total 80 problems have been reported at the 69 landfills

Table 4-6 shows some results of an investigation by Bonaparte et al. (2002) including 69 landfills in the USA, where problems concerning the containment system were reported. The reported problems were distinguished based on the barrier system elements (top cover, leachate collection, and bottom liner), and on the dominant human factor contributing to the failure of the containment system. Based on the data in Table 4-6, 47% of the reported problems related to the bottom liner system, 29% related to the top cover, and 24% related to the leachate collection system. The most important failure mechanism was the loss of geotechnical stability, which accounted for 38% of all the reported problems, followed by inappropriate construction, responsible for 28% of the reported problems. Among the human factors poor design was the prior reason for containment failure in more than half of cases, 35% of the problems were due to poor construction quality and control, and 14% of the problems were caused by improper operation of the landfill. This analysis highlights the importance of stringent quality assurance mechanisms during the design and

construction of technical barriers. However, the analysis is based exclusively on reported (i.e. detected) problems and the age of the landfills is rather low compared to the time period with significant emissions potentially expected from the waste.

Models describing the efficiency of a technical barrier system are based on the behavior of the barrier components and their interaction within the system. A widely applied model describing the performance of top cover systems is the HELP model (Schroeder et al. 1984), which is based on a layer concept and calculates water flow from one layer of the top cover system to the next layer in the downwards direction. The simplicity of the model does not allow for a mechanistic description of the complex hydrodynamic processes in a top cover system. The HELP model is used primarily as a screening assessment tool to estimate leachate generation (cf. National Research Council 2007). More complex models, such as Hydrus-1D (Šimůnek et al. 1998), use the Richards equation (Richards 1931) and water retention curves to express the relationship between the water content and the soil water potential to describe water flow in top cover systems (e.g. Mallants et al. 1999; Scanlon et al. 2002). The application of such models requires more and better input data, more calculation capacity, and expertise in using numerical simulation models. Nevertheless, also these models face limitations with respect to the consideration of preferential flow in top cover systems or in view of changing material properties over time (e.g. material degradation, biological processes, external stresses, etc.).

The modeling of water flow through bottom barrier systems with a sole low permeability soil liner employs the same models used in soil physics to describe the water flow through saturated (Darcy equation) and unsaturated (Richards equation) media (see also section 4.2.2). Pollutant transport can be described via advection-diffusion equations, which allow for various transport processes such as advective transport, diffusive transport, sorption, and degradation or transformation. Though one dimensional models (e.g. Hydrus 1D) may be sufficient to describe pollutant transport through bottom liners, multi-dimensional models are widely applied, too (Rowe 1998). While well established and widely applied models exist to describe water and pollutant flow through single mineral liner systems at the landfill base, the modeling of composite liner systems is more nebulous. As advective flow is bound to the presence of holes or defects of the geomembrane liner, corresponding models are based on assumptions about hole frequency, the presence of wrinkles, and the quality of liner contact (cf. Rowe and Booker 2000; Rowe 2005). Based on a review of leakage rates at U.S. landfills, the National Research Council (2007) concluded that the observed flow rates cannot be explained in a plausible way without assuming wrinkles and increased head on the geomembrane liner. The lack of direct observations (e.g. measurements of the number and size of holes or the location and extent of wrinkles) introduces high model uncertainties and makes the estimation of advective flow through geomembrane liners a challenging task. In addition to the incomplete knowledge about the condition of the current system, the change of system properties over time, such as decreasing effectiveness of drainage system and increased build-up of hydraulic head, polymer degradation of the geomembrane liner, and local or global geotechnical instability due to changing mechanical properties, cannot be accounted for within the transport models.

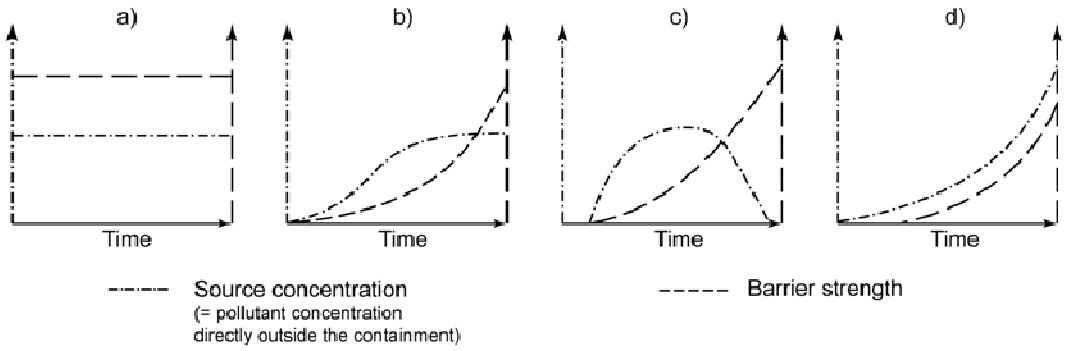


Figure 4-5: *Models of deterioration patterns at landfill containment systems and resulting source concentrations (adapted from Inyang(2004)): a) constant pollutant release from the waste and constant barrier strength; b) constant pollutant release from the waste and exponentially decreasing barrier strength; c) limited pollutant release from the waste and exponentially decreasing barrier strength after some time; d) unlimited pollutant release from the waste and exponentially decreasing barrier strength after some time;*

An approach to tackle the problem of changing properties of the technical barrier system over time is the definition of different service periods, associated with a defined condition of the system during the respective service period. Based on this concept, Inyang (2004) distinguished between an initial period (=technical barrier is mostly intact), an intermediate period (=flaws are generated), and an extended period (=behavior of the barrier is dominated by flaws). For each period, different models and/or model parameters are appropriate to describe water and pollutant transport. Subsequently, the period-specific models can be aligned along the time axis to derive a continuous estimate of technical barrier performance. Apart from the assumptions during model development, the level of complexity in such estimates is exacerbated by the interrelationship between the performance of the top cover system and the stresses (i.e. leachate generation) working on the bottom barrier system. Although this procedure allows for a consideration of changing system properties over time, the uncertainty inherent in such estimates needs to be emphasized. The effect of different barrier deterioration models and different source characteristics over time on the source concentration (=pollutant concentration directly outside the containment) is schematically illustrated in Figure 4-5. The curves in Figure 4-5a) show the model assumption of constant barrier strength and constant pollutant release from the waste resulting in constant source strength. The assumption concerning barrier strength could be justified for continuously maintained and repaired systems, but is incorrect for barrier systems in the absence of maintenance. Figure 4-5b) illustrates the case of exponential barrier deterioration with a constantly high release of pollutants from the waste resulting in high source concentrations after an initial increase in pollution. The effect of exponential deterioration of the barrier in combination with limited substance release from the waste on source concentrations is shown in Figure 4-5c). Finally, Figure 4-5d) illustrates the case of exponential barrier deterioration after an initial lag phase (e.g. maintenance period) with an inexhaustible release of pollutants from the waste resulting in increasing source concentrations. Inyang (2004) concluded, based on the analysis of the effect of different model assumptions, that the choice of deterioration model is of crucial importance for the evaluation of future pollutant release from landfills and thus, model choice and application need to be scientifically based and critically discussed.

The discussion above highlights that statements about the long-term efficiency of technical barrier systems are associated with substantial uncertainties. On the one hand, field data on the performance of these systems have been collected for several decades, which is a rather short time period compared to the time period over which many landfills contain a significant pollution potential. On the other hand, monitoring data are typically not collected to gain an increased system understanding of the technical barriers but to evaluate the effect of a landfill on potentially affected systems (e.g. groundwater monitoring). To gain better insight into the processes and mechanisms significant for the performance of technical barriers, more direct monitoring data need to be collected and made available (cf. National Research Council 2007), and estimates about the development of system performance need to be based on longer observation periods at the field scale. As the latter cannot be achieved in a short time, the statement made by Farquhar and Parker (1989) more than 20 years ago is still valid today: *„Yet we must work with systems whose performance we do not fully understand and cannot adequately predict. This dilemma requires that we be conservative in our judgments and vigilant in assessing their impact"*.

4.1.4.2 Expert survey to evaluate long-term performance of technical barriers

The large uncertainties inherent in the evaluation of long-term barrier performance are primarily due to a lack of appropriate field performance data (monitoring periods and data quality), the complex interaction of different processes contributing to a decrease in barrier performance, and the inability to determine future conditions encountered in and around the landfill. As estimates on future barrier performance cannot be verified, they are often based on assumptions which have a significant effect on the outcome (Inyang 2004). Therefore, experts were invited to participate in a survey to evaluate the significance of different factors for the performance of technical barrier systems typically installed at Austrian landfills (cf. Figure 4-2 and Figure 4-4). 23 experts in landfill geotechnics from Austria and Germany were asked to create a list of relevant factors in a first step and subsequently evaluate the importance of these factors for different time periods (= service periods) and barrier systems in a second step (see also appendix 2 and Laner et al. 2011c). 13 of the invited experts (approximately 57%) participated in the survey (see Table A-4 of appendix 2). Based on a suggestion by the author, the experts agreed on a list of factors potentially important to evaluate long-term system performance (see Table 4-7). The factors address the resistance built into the barrier system (e.g. thickness of the vegetative cover) and the stresses induced on the barrier system (e.g. climatic conditions, heat production in the waste). Then, the experts assigned scores to each of the factors for different service periods (see appendix 2: Table A-5 – Table A-8), based on the importance of this factor for the performance of the barrier system. The evaluation scores ranged from 1 (insignificant) to 5 (highly significance). For instance, a score of 5 for the factor "observed barrier performance" in service period 1 means that the system performance at the time of evaluation is of very high significance for the expected system performance during the first service period. Based on the expert evaluation, factors of high significance for barrier performance during a specific time period can be identified (cf. appendix 2 and Table 4-8) and the expected importance of different factors over time can be analyzed (see Figure 4-6 and Figure 4-7). Figure 4-6 and Figure 4-7 illustrate some trends in factor importance over time for composite liner systems at the top and at the bottom of a landfill. For top covers (cf.

Figure 4-6), the importance of "heat production in the waste", "observed barrier performance", "expected settlements", "drainage system", and "construction and design quality assurance program" decreases with increasing service period, while the importance of "climate" and "vegetation & after-use" increases with time. For bottom barriers (cf. Figure 4-7), the importance of factors such as "heat production in the waste", "observed barrier performance" or "leachate drainage system" decreases strongly with time, while the significance of factors such as "landfill geometry" or "distance to groundwater level" increases or remains similar over time. Note that the mean evaluation scores in Figure 4-6 and Figure 4-7 are normalized. Therefore, they do not reveal the absolute significance assigned to a factor by the experts, but illustrate the individual trends of expected factor importance over time (service periods: several decades, around a century, several centuries). An overview about the score assignments by the experts is provided in appendix 2.

Table 4-7: List of significant factors for the performance of engineered barrier systems

#	Bottom barrier system	Top cover barrier system
1	Observed barrier performance	Observed barrier performance
2	Construction & design quality assurance program	Construction & design quality assurance program
3	Climate (e.g. temperature, precipitation)	Recultivation layer (vegetative cover thickness)
4	Heat production in the waste	Heat production in the waste
5	Distance to groundwater surface level	Climate (e.g. temperature, precipitation)
6	Overburden pressure	Vegetation and projected after-use
7	Leachate quality	Relief
8	Monitoring (e.g. groundwater monitoring)	Expected settlements
9	Leachate drainage system	Lining control systems (direct monitoring)
10	Stability of foundation	Drainage system
11	Landfill geometry (heap vs. cavity)	

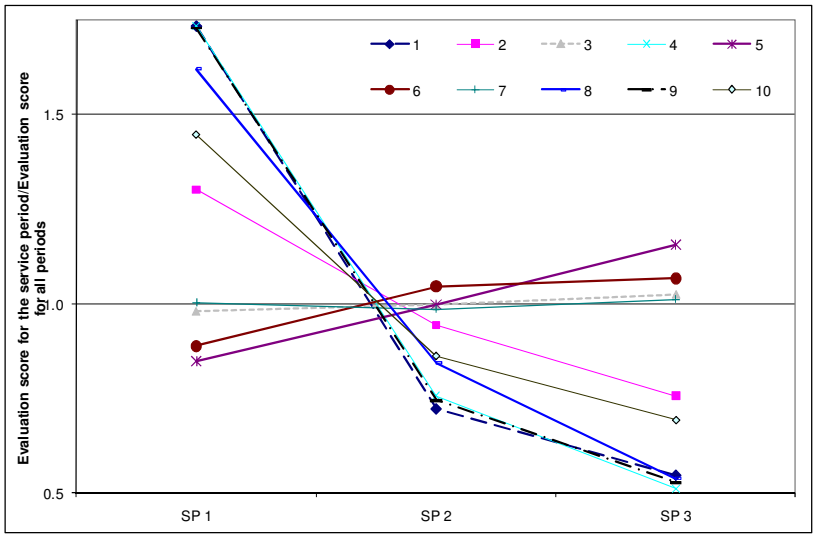


Figure 4-6: Normalized evaluation scores (mean evaluation score for the service period/mean evaluation score for all service periods) for the factors in Table 4-5 to illustrate the trend of factor significance over time (i.e. over the service periods) for a top cover system with a composite liner

The most important factors (i.e. number of rankings among the top three factors by an expert) are shown in Table 4-8 for each period and barrier system. It is apparent that construction & design

quality assurance is regarded as a very important factor for barriers at the top and at the bottom of a landfill in general and in particular during the first service period. On the longer view, vegetation and after-use, the thickness of the vegetative soil layer, and climate become the dominant factors for top cover performance and landfill geometry as well as foundation stability get increasingly important for bottom barrier performance.

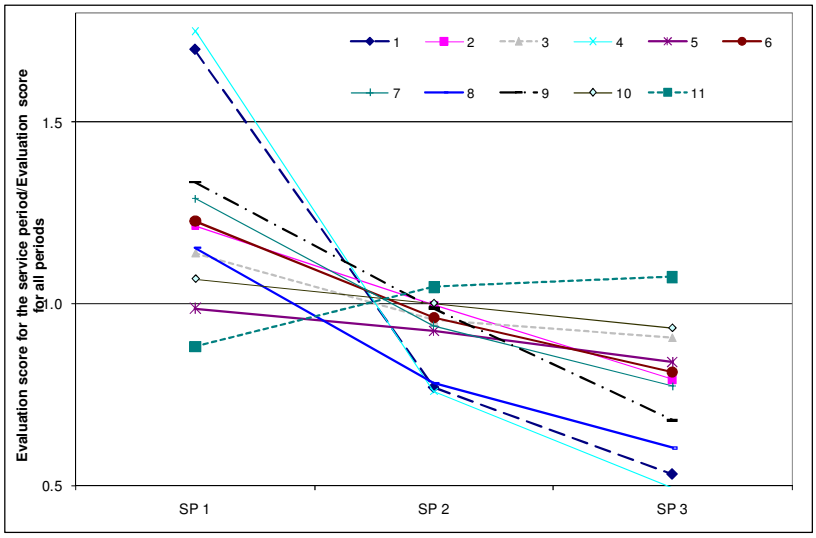


Figure 4-7: *Normalized evaluation scores for the factors in Table 4-5 to illustrate the trend of factor significance over time for a bottom barrier system with a composite liner*

Table 4-8: *List of the top factors influencing the performance of the barrier systems throughout different time periods (in brackets: number of experts ranking the factor among the top three factors during the service period)*

Barrier system	Service period 1 (< 50 yrs)	Service period 2 (50 - >100 yrs)	Service period 3 (several centuries)
Top cover: Composite liner	Construction quality assurance (12) Local settlements (11) Vegetative soil layer (8) Observed performance (7*)	Vegetative soil layer (10) Vegetation & after-use (10) Construction quality assurance (10)	Vegetation & after-use (12) Vegetative soil layer (11) Climate (10)
Top cover: Low permeability soil liner	Local settlements (13) Construction quality assurance (12) Vegetative soil layer (11)	Vegetative soil layer (11) Vegetation & after-use (9) Climate (9) Construction quality assurance (6*)	Vegetative soil layer (10) Climate (9) Vegetation & after-use (9) Construction quality assurance ((7*)
Bottom barrier: Composite liner	Construction quality assurance (13) Leachate drainage (13) Observed performance (8)	Construction quality assurance (11) Foundation stability (9) Landfill geometry (8)	Landfill geometry (9) Construction quality assurance (9) Foundation stability (8)
Bottom barrier: Low permeability soil liner	Construction quality assurance (12) Leachate drainage (11) Observed performance (9)	Construction quality assurance (11) Foundation stability (11) Landfill geometry (8) Leachate drainage (8*)	Landfill geometry (9) Foundation stability (9) Construction quality assurance (8)

Notes:
*A fourth factor is included in the list, if the experts ranked the factor on the top as often as the third factor. For instance, for composite liner system at the top the factor "Observed performance" was ranked 6 times on top, which is equal to the number of top rankings for "Vegetative soil layer", with "vegetative soil layer being ranked more often on the 2nd or 3rd spot.

Although subjective, the expert evaluations provide a basis to rank the importance of different factors with respect to their (expected) effect on barrier system performance. The expert evaluations are subsequently used to determine weighting factors and develop a scenario of gradually decreasing barrier performance (see chapter 5.2), which forms the basis to illustrate the effect of decreasing barrier performance on the release of substances from the landfill. The different scenarios to address long-term barrier performance are described in chapter 5.2 as a part of the methodology for evaluating the environmental compatibility of closed landfills.

4.1.5 Summary and discussion

Technical barriers control and limit the interaction between the deposited waste and the surrounding environment. Technical barriers typically consist of several components with low-permeability mineral layers (e.g. clay liners), geomembranes (e.g. HDPE liners), geosynthetic clay layers, drainage layers, and vegetative soil layers being among the most important ones. The long-term performance of different barrier components (in the absence of maintenance and repair) is estimated to range from several decades for leachate drainage systems to geologic time periods (thousands of years) for low permeability mineral liners. Although several models exist to describe barrier performance, the lack of appropriate field data to calibrate and validate models, the complex interaction of different processes relevant to water and pollutant transport, and the potential change of material properties over time introduce substantial uncertainties into the modeling. As subjective assumptions may dominate model outcomes (cf. Inyang 2004), experts were invited to assess the importance of different factors for the performance of barrier systems over specific time periods (=service periods). Existing studies and the expert evaluations were used to outline basic indicators for developing models of long-term barrier performance (see Table 4-9). The indicators listed in Table 4-9 relate to the resistance built into a system (e.g. components of the barrier system, quality assurance), the stresses induced on the system (e.g. settlements, climate, natural hazards), and the currently (i.e. during the aftercare period) observed performance of the system. This information is used to estimate future barrier performance as a part of the evaluation of landfill environmental compatibility. In this work a scenario-based approach is used, which is described in chapter 5.2 and illustrated in the case studies presented in chapter 6.

Table 4-9: Basic information to evaluate long-term barrier performance at closed landfills

Technical barrier system
- Design: Layers, extraction systems, design specifications, similar systems
- Construction: Quality assurance (protocols, field testing, documentation), local and global proof of geotechnical stability
- Observed performance: Monitoring data (leachate generation rate, efficiency of base lining system)
Waste properties
- Settlements: Overall settlements and differential settlements
- Temperatures: Areas of different temperatures in the waste body
- Emissions: Leachate composition (e.g. substances increasing the risk of drainage system clogging)
Landfill site
- Climate and water balance: Precipitation, temperatures, groundwater recharge, prevailing winds
- After-use: Proposed after-use, projected vegetation cover/typical vegetation at the site
- Other factors: Hydrogeology, potential exposure to natural hazards (e.g. flood plain)

4.2 Natural systems attenuating landfill emissions

4.2.1 Overview

Until 50 years ago, landfills were built with the belief that the buffering capacity of the local environmental system (i.e. the geological conditions at the site) is sufficient to prevent negative effects on human health and the environment from landfill emissions. The potential of naturally occurring processes (e.g. degradation, transformation, retardation, dispersion, and dilution) to mitigate pollution from landfills was highlighted in several studies (e.g. Gray et al. 1974; Bagchi 1983). The concept of "natural attenuation landfills" was often applied at sites with inappropriate geologic and hydrologic conditions, so groundwater pollution became a problem at many of these landfill sites (e.g. Zanoni 1972; Christensen et al. 1994; Kjeldsen and Christophersen 2001). As a consequence, modern landfills are typically equipped with a containment system (=technical barrier system, see previous section) to separate the deposited waste from the environment and limit/control two-way substance flows (cf. European landfill directive (EC 1999)). The technical barrier system at the bottom of the landfills is constructed before waste placement and a final cover, typically with low permeability, is installed some time after the end of waste deposition (cf. Figure 2-1). Thus, the waste is largely isolated from the surrounding environment implying minimal substance flows into and out of the landfill (=containment period in Figure 4-8). Due to the limited resistance built into the containment system and the continuous stresses induced on the system, the containment deteriorates and the waste may increasingly interact with the surrounding environment (=deterioration period in Figure 4-8). The landfill evolution illustrated in Figure 4-8 can be expected in the absence of maintenance and repair/replacement of the containment system, but note that also a functional containment system allows for some residual level of interaction between the waste and the environment. Therefore, the evaluation of the environment's capability to mitigate negative effects of pollutants released from a landfill is a major element of the assessment of the environmental compatibility of a closed landfill, particularly in the long term.

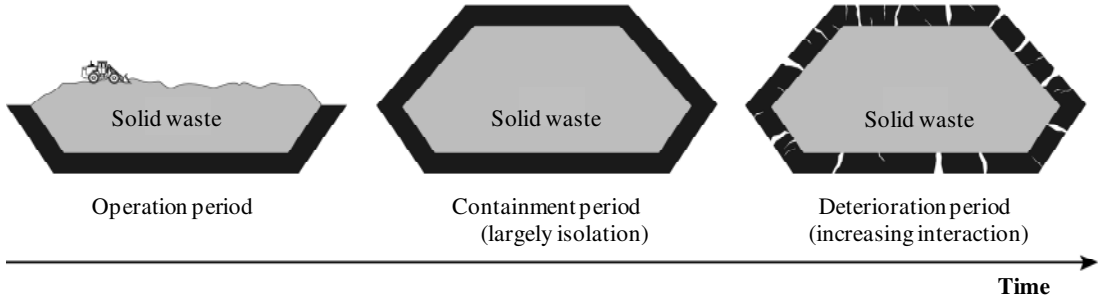


Figure 4-8: Interaction period at a containment-type landfill with decreasing performance of the technical barriers in the absence of maintenance and repair

The main focus of this chapter is on the release of leachate to the subsurface and the subsequent migration of leachate constituents in the environment, as landfill leachate is the most significant long-term emission (cf. chapter 3 or Krümpelbeck 2000) and because of the potential pathway associated with limited or decreasing technical barrier performance (cf. Figure 4-8). Apart from leachate migration in the subsurface, the migration of landfill gas through top cover systems and the discharge of collected leachate to surface waters are briefly addressed below.

4.2.1.1 Landfill gas migration and methane oxidation

Modern top cover systems at MSW landfills include a landfill gas collection system, but many old landfills are not equipped with such a system. In either case a (potentially small) fraction of the landfill gas will migrate through the top cover. During the passage of the top layers of the cover, methane can be oxidized under the presence of oxygen (at least 2% of volume) to carbon dioxide and water (Huber-Humer et al. 2008). Methane oxidation has been demonstrated at landfills with and without gas collection systems as well as for cover soils designed as biologically active covers (i.e. compost amended soils) and for conventional vegetative top soil layers (e.g. Oonk and Boom 1995; Barlaz et al. 2004; Chanton et al. 2010; Chanton et al. 2011). While biologically active covers oxidize a higher fraction of the methane entering the cover system in general (cf. Barlaz et al. 2004; Huber-Humer 2004; Chanton et al. 2010), the actual rate of methane oxidation in a cover is dependent on numerous factors. Important factors are the availability of oxygen, the methane flux (=flow rate per area), the temperature, the water content, the presence of inhibitors (e.g. ammonium), the content of humified organic matter, the root penetration, and the aging (=duration of methane application) of the biologically active soil layer. The optimum temperatures for methane oxidation processes are between 25 and 38°C (cf. Scheutz and Kjeldsen 2004), thus fluctuations of atmospheric temperature throughout the year may also have a significant effect on methane oxidation rates in a cover. Although higher methane fluxes into the cover system have been demonstrated to increase the absolute rate of methane oxidation, the fraction of oxidized methane tends to decrease with increasing flux (cf. Chanton et al. 2011; Rachor et al. 2011). Methane oxidation rates in landfill covers have been reported between 0 and more than 100 g $CH_4/m^2.d$, corresponding to fractions of oxidized methane between 0 and 99% (e.g. Huber-Humer et al. 2008; Kühle-Weidemeier and Bogon 2008; Chanton et al. 2010; Scheutz et al. 2011). The highest oxidation rates have been observed at biologically active covers with mature compost[11] and adapted microbiology. For instance, Huber-Humer et al. (2008) showed that optimized cover systems could completely oxidize methane fluxes into the top cover of up to 4 l CH_4/m^2.hour. For low methane flux rates (below 2 l CH_4/m^2.hour) high oxidation rates can be expected also at covers not specifically designed to optimize methane oxidation (Kühle-Weidemeier and Bogon 2008).

The defininition of a constant fraction of the input flux for estimating methane oxidation does not account for the effect of various factors such as CH_4 flux into the cover or climatic conditions on methane oxidation rates (cf. Chanton et al. 2011). Nevertheless, constant values are often suggested as a first screening evaluation of the fraction of methane which can be oxidized in a cover system. While the suggestion of 10% for the average annual fraction of oxidized methane by the IPCC (2007) is rather conservative and very simplistic, the suggestions in Table 4-10 consider different cover types and methane fluxes. It should be emphasized that methane oxidation processes are influenced by many factors and that, particularly high, methane oxidation rates need to be demonstrated in the field-scale over an extended period of time before they are used to assess methane emissions to the atmosphere.

[11] Compost with a significant fraction of readily biodegradable organics will reduce oxygen availability for methane oxidation, cf. Scheutz et al. 2011.

Table 4-10: Fraction of oxidized methane during the migration through the top cover (oxidized fraction relates to the methane released into the cover) for different cover systems and methane fluxes (based on Kühle-Weidemeier and Bogon 2008)

Top cover system	Methane flux > 2 l CH_4/m^2.hour	Methane flux < 2 l CH_4/m^2.hour
Cohesive soil (silty sand or sandy silt)	**0.10**	**0.15**
Vegetative soil rich in humus	**0.15**	**0.30**
Biologically active methane oxidation layer (during aftercare with monitoring/maintenance)	**0.60**	**0.70**
Biologically active methane oxidation layer (long-term without monitoring)	**0.40**	**0.50**

4.2.1.2 Leachate discharge to surface waters

In Austria the controlled discharge of leachate to surface waters is regulated by emission-related ordinances[12] (leachate ordinance (MoE 2003) and waste water discharge ordinance (MoE 1996a), see Hefler (2003) for further details). If the quality of landfill leachate does not comply with these regulations at the point of leachate discharge (i.e. the outlet to the surface water body), the leachate has to be treated before discharge. Though the authorities can specify differing criteria based on the vulnerability of the potentially affected water body, this has been hardly exercised yet. Consequently, as generic limit values are used to evaluate the release of collected leachate to surface waters, the effect of these (controlled) emissions on environmental media is not addressed in more detail.

Apart from controlled discharge of leachate, leachate-polluted groundwater could have an impact on water quality in affected surface waters. This emission pathway is included in the analysis of pollutant migration in the subsurface. Another possible way of uncontrolled leachate-induced pollution of surface water bodies is the flooding of landfills, as a significant proportion of landfills in Austria are located in flood-prone areas (cf. Laner et al. 2009). The erosion of deposited waste and/or the leaching of pollutants contained in the waste are potential sources of surface water pollution. However, due to the large dilution potential during a flood event, a flooded landfill will be typically one among many sources of pollution of similar magnitude during a flood event (Laner et al. 2009).

4.2.2 Leachate migration in the subsurface

The migration of pollutants in the subsurface has been intensively investigated (e.g. Christensen et al. 1994; Christensen et al. 2000; National Research Council 2000; Luckner 2010) with the aim to evaluate the potential of natural environments to mitigate negative effects of a release of landfill leachate. Naturally occurring physical, chemical, or biological processes that, under favorable conditions, reduce the mass, toxicity, mobility, volume, or concentration of pollutants in soil or groundwater are called “natural attenuation processes”. They include biodegradation, dispersion, dilution, sorption, volatilization, radioactive decay, and chemical or biological stabilization, transformation, or destruction of pollutants (U.S. Environmental Protection Agency 1997). It has

[12] The discharge quality standards for several landfill leachate parameters are shown in Table 3-3, Table 3-10, and Table 3-13

been suggested that only retardation (i.e. transformation of substances from mobile, liquid phases to immobile phases) and degradation (i.e. transformation of substances into less harmful or harmless substances) should be considered in remediation strategies for contaminated sites (cf. Luckner et al. 2008; Luckner 2010). However, processes causing a sole decrease in pollutant concentrations (i.e. dispersion, dilution) naturally take place in the subsurface environment and are subsequently considered in the evaluation of environmental impacts due to leachate emissions.

4.2.2.1 Landfills as a threat to groundwater

Landfill leachate emissions are a significant threat to the groundwater quality in Europe (cf. Europäische Umweltagentur 2004). In the USA landfills are reported to be one of the most important sources of groundwater pollution (3rd rank among reported sources of groundwater pollution in Figure 4-9). In the past, landfills have often been constructed with inadequate technical barrier systems at sites where the geology and hydrogeology could not prevent groundwater pollution. Although modern landfills are typically equipped with technical barrier systems, they represent a potential source of groundwater pollution, particularly on the long term (cf. Figure 4-8).

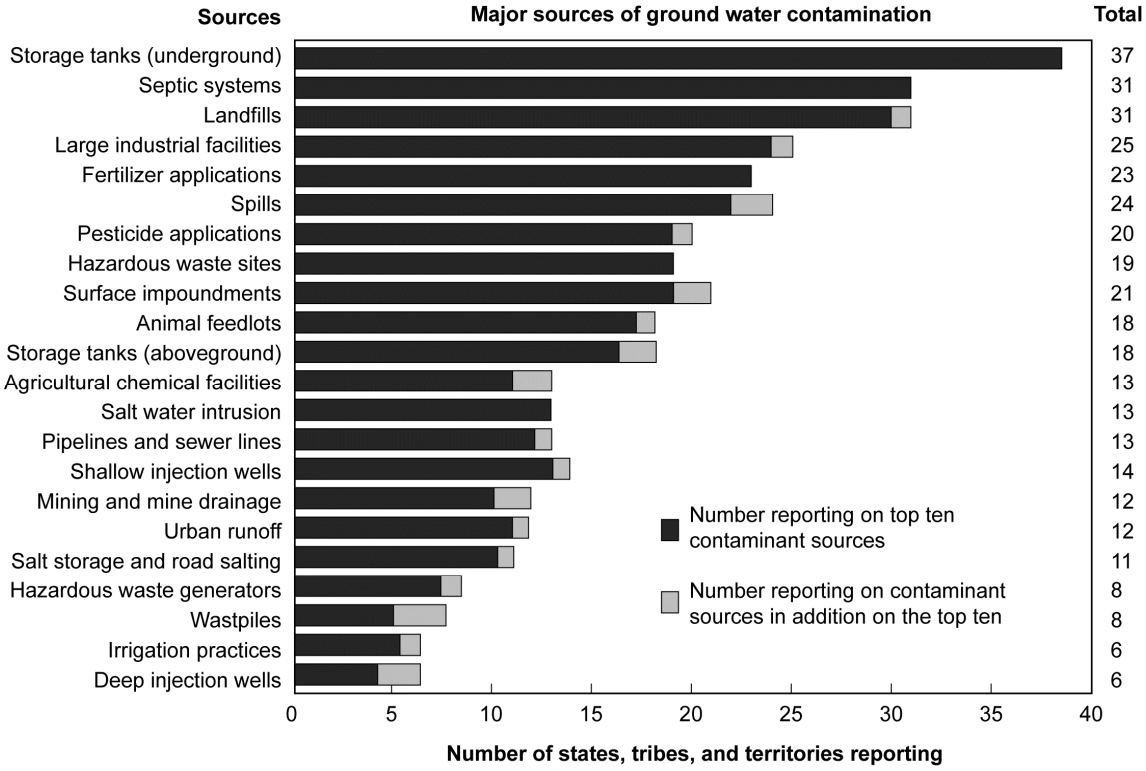


Figure 4-9: Major sources of groundwater pollution in the USA (Source: U.S. EPA 2000)

Despite the fact that indivdual landfills differ with respect to waste composition and local conditions, typical impacts of released landfill leachate on the groundwater quality can be illustrated via statistical analysis of measurements in contaminated and uncontaminated groundwater (cf. Arneth et al. 1989; Kerndorff et al. 1993; Voigt and Wippermann 1998). In Table 4-11 and Table 4-12 measurements of groundwater quality at 126 drinking water extraction plants (=uncontaminated groundwater) and at 205 leachate plumes from old landfills (=affected by waste deposit) are analyzed and compared. The mean and median values of each data set in Table 4-11

and Table 4-12 show that pollution by landfill leachates is typically associated with increased concentrations of substances in the groundwater[13]. To evaluate the importance of indivdual pollutants, two indices are calculated: The emission detection frequency (EDF) is the percentage of the "affected by waste deposit" data set above the mean value calculated for the "uncontaminated groundwater" data set. The influence factor (IF) expresses the strength of pollution by multiplying the EDF with the quotient of the mean values of the two data sets (see Table 4-11 and Table 4-12). Table 4-11 and Table 4-12 also show the marginal effect levels (MEL) for different substances (=threshold levels to identify groundwater damage (cf. LAWA 2004)) and the Austrian drinking water quality standards (MoE 2001) for comparisons.

Among inorganic groundwater constituents, the concentrations of ammonium, arsenic, chromium, boron, and chloride differ the most between the contaminated and the uncontaminated groundwater data set, based on the calculated influence factors (see Table 4-11). Emission detection frequencies above 75% can be observed for hydrogen carbonate, boron, sodium, chloride, and magnesium. Boron has been suggested as an indicator for landfill-borne groundwater pollution, due to its low geogenic concentrations and its typical presence in groundwater leachate plumes from landfills (cf. Voigt and Wippermann 1998). However, in terms of leachate quality it should be emphasized, that a substance may well be present in the landfill leachate but not be detected in the groundwater, due to high retardation or low persistence (i.e. short degradation half-life). Therefore, the substance concentrations of a groundwater leachate plume depend on the composition of the released leachate, the geochemical mobility and persistence of the leachate constituents, the redox conditions, and the range of parameters measured in the groundwater.

The mean values of most of the organic groundwater constituents listed in Table 4-12 are below the respective detection limits for the uncontaminated groundwater data set. While most of the organic compounds in landfill leachate plumes are (rather harmless) organic acids (e.g. humic acids), more hazardous organic compounds are present, too (cf. Öman and Junestedt 2008; Weber et al. 2011). Among the most important are BTEX, halogenated hydrocarbons and various pesticides (cf. Table 4-12; Baun et al. 2004; Europäische Umweltagentur 2004). However, the extent of leachate plumes with organic pollutants in the groundwater has been observed to be quite limited at many landfills due to relatively fast degradation and dilution/dispersion of the pollutants in the groundwater (cf. Christensen et al. 1994). This is illustrated by Table 4-13, as reported lengths of groundwater plumes downstream of old MSW landfills are typically in the range of several hundreds of meters. However, pollutant plumes in coarse sand aquifers were reported to impact the groundwater over several kilometers, particularly with respect to conservative, inorganic pollutants such as chloride (cf. Table 4-13).

[13] Some substance concentrations could be also reduced due to the influence of landfill leachate (e.g. sulfate reduction in an anaerobic pollutant plume, cf. Figure 4-10).

Table 4-11: Comparison of inorganic groundwater constituent concentrations in groundwater affected by waste deposits and uncontaminated groundwater (based on Voigt and Wippermann 1998; Luckner et al. 2008)

Parameter		Affected by waste deposit[1]		Uncontaminated groundwater[1]		EDF[2]	DW limit[3]	MEL[4]	IF[5]
	Unit	Mean	Median	Mean	Median	[%]			[-]
Major anions									
Hydrogen carbonate	mg/l	519	408	83	78.7	**85.8%**			5.4
Chloride	mg/l	202	74.2	14	11.9	**79.2%**	200	250	**11.4**
Sulfate	mg/l	194	122	18	12.7	64.5%	250	240	7.0
Nitrate	mg/l	25.6	2.5	3	2.2	50.7%	50	-	4.3
Major cations									
Calcium	mg/l	175	156	83	78.7	74.5%	-	-	1.6
Magnesium	mg/l	38.1	22	14	11.9	**76.0%**	-	-	2.1
Sodium	mg/l	128	45.6	18	12.7	**83.4%**	200	-	5.9
Ammonium	mg/l	11	0.4	0.17	0.07	53.6%	0.5	-	**34.7**
Iron	mg/l	9.6	2	2	0.93	40.6%	0.2	-	1.9
Manganese	mg/l	1.5	0.43	0.18	0.12	62.2%	0.05	-	5.2
Trace components									
Aluminum	µg/l	596	80	50	<10	33.9%	200	-	4.0
Lead	µg/l	6.3	<0.05	1.4	<0.5	38.3%	10	7	1.7
Cadmium	µg/l	1.5	<0.05	0.13	<0.1	14.9%	5	0.5	1.7
Chromium (total)	µg/l	21.2	<10	0.58	<0.5	56.3%	50	7	**20.6**
Copper	µg/l	24.5	<10	7	4.5	52.4%	2000	14	1.8
Nickel	µg/l	22.9	13	2.8	<1	64.3%	20	14	5.3
Mercury	µg/l	<0.2	<0.2	<0.1	<0.1	14.7%	1	0.2	-
Strontium	µg/l	580	399	240	200	74.1%	-	-	1.8
Zinc	µg/l	410	64.5	117	30.2	41.2%	-	58	1.4
Arsenic	µg/l	45.4	1.3	1.2	<0.5	61.3%	10	10	**23.2**
Nitrite	µg/l	0.49	<0.1	<0.01	<0.01	35.4%	100	-	-
Selenium	µg/l	0.1	<0.2	0.55	<0.1	22.3%	10	7	0.0
Boron (total)	µg/l	832	181	34.8	26	**85.7%**	1000	740	**20.5**
Fluoride	µg/l	<0.5	<0.5	<0.1	<0.1	52.6%	1.5	0.75	-
Cyanide (total)	µg/l	-	-	<1	<1	-	50	5 (50)	-
Phosphate (total)	µg/l	1.8	0.1	0.14	0.08	41.5%	-	-	5.3

[1] Measured data at 205 groundwater wells downstream of waste deposits (=affected groundwater) and data from water quality monitoring at 126 drinking water extraction plants (=uncontaminated groundwater) (cf. Kerndorff et al. 1993)

[2] EDF = Emission detection frequency (=fraction of measurements in the affected groundwater above the mean value of the uncontaminated groundwater data set) based on Voigt and Wippermann (1998)

[3] Quality standards for drinking water (Austrian drinking water quality directive, MoE (2001))

[4] Marginal effect levels (MEL) according to LAWA (2004)

[5] Influence factor (IF): IF= (mean value$_{affectedt}$/mean value$_{uncontaminated}$)*emission detection frequency

Table 4-12: Comparison of organic groundwater constituent concentrations in groundwater affected by waste deposits and uncontaminated groundwater (based on Voigt and Wippermann 1998; Luckner et al. 2008)

Parameter		Affected by waste deposit[1]		Uncontaminated groundwater[1]		EDF[2]	DW limit[3]	MEL[4]	IF[5]
	Unit	Mean	Median	Mean	Median	[%]			
Aggregate parameters									
DOC	mg/l	-	-	1.6	1.2	-	-		-
AOX	µg/l	-	-	2.7	<2	-	-		-
Volatile halogenated hydrocarbons								20	
Trichloromethane	µg/l	76.2	1.4	0.12	<0.1	22.0%	-	-	139.9
1.2-Dichloroethane	µg/l	107	<5	-	-	18.8%	3	2	-
cis-1.2-Dichloroethylene	µg/l	22100	166	-	-	30.1%	-		-
Chloroethylene	µg/l	1690	99.5	-	-	17.7%	0.5	0.5	-
Trichloroethylene	µg/l	11.5	2.3	0.21	<0.1	40.8%	10*	10*	22.3
Tetrachloroethylene	µg/l	56.1	1.4	0.34	<0.1	46.2%			76.2
BTEX									
Benzene	µg/l	141	13.5	<0.1	<0.1	29.1%	1	1	-
Toluene	µg/l	4.1	3.5	<0.1	<0.1	16.5%	-	-	-
o-Xylene	µg/l	13.8	4.7	<0.1	<0.1	9.5%	-	-	-
m/p-Xylene	µg/l	3.4	2.8	<0.1	<0.1	22.8%	-	-	-
Ethylbenzene	µg/l	6.6	4.4	<0.1	<0.1	11.3%	-	-	-
PAH							0.1**	0.2**	-
Naphtaline	µg/l	1.5	1.1	-	-	12.1%	-	1	-

[1] Measured data at 205 groundwater wells downstream of waste deposits (=affected groundwater) and data from water quality monitoring at 126 drinking water extraction plants (=uncontaminated groundwater) (cf. Kerndorff et al. 1993)

[2] EDF = Emission detection frequency (=fraction of measurements in the affected groundwater above the mean value of the uncontaminated groundwater data set) based on Voigt and Wippermann (1998)

[3] Quality standards for drinking water (Austrian drinking water quality directive, MoE (2001))

[4] Marginal effect levels (MEL) according to LAWA (2004)

[5] Influence factor (IF): IF= (mean $value_{affectedt}$/mean $value_{uncontaminated}$)*emission detection frequency

* Values relate to the sum of tetrachloroethylene and trichloroethylene

** Values relate to the sum of PAH without naphtaline

Table 4-13: Dimensions of pollutant plumes downstream of old MSW landfills (based on Christensen et al. 2000 and Bjerg et al. 2009)

Landfill	Area [m²]	Dimensions (longitudinal/lateral) [m]	Geological stratum	Investigated parameters*
Borden, CAN	45,000	700 \ 600	Sand	1, 2, 3
North Bay, CAN	280,000	400 \ 100	Silt	1, 2, 3, 4
Woolwich, CAN	35,000	600 \ -	-	1, 3, 4
Grindsted, DEN	100,000	400 \ < 400	Sand	1, 2, 3, 4
Vejen, DEN	60,000	400 \ 100	Sand	1, 2, 3, 4, 5
Unknown, GER	150,000	3000 \ 500	Coarse sand	1, 3
Noordwijk, NED	60,000	- \ -	Coarse sand	1, 4
Army Creek, USA	240,000	700 \ -	Sand	1, 2, 3, 4
Babylon, USA	100,000	3000 \ 600	Coarse sand	1
Islip, USA	70,000	1500 \ 400	Coarse sand	1
KL Avenue, USA	270,000	1600 \ -	Sand	1, 2, 3, 4
Sjoelund, DEN	6,300	100	Fine sand	2, 3, 4

*Parameters: 1… Inorganic parameters, 2… Redox pairs, 3… DOC, 4… XOC, 5… Heavy metals

Due to the organic pollution of leachate from old MSW landfills, typical redox zones can be observed in groundwater pollution plumes downstream of such landfills (cf. Figure 4-10). The sequence of redox zones is based on the available electron acceptors during the microbial degradation of organic matter and the amount of chemical energy released during the reactions (cf. Zobrist 1999). Ranked by the amount of chemical energy released, the sequence of reactions is:

1.) Aerobic respiration: $C_{org} + O_2 \rightarrow CO_2 + H_2O$
2.) Denitrification: $C_{org} + NO_3^- \rightarrow N_2 + HCO_3^- + CO_2$
3.) Manganese reduction: $C_{org} + MnO_2 \rightarrow Mn_2^+ + HCO_3^-$
4.) Iron reduction: $C_{org} + FeOOH \rightarrow Fe^{2+} + HCO_3^-$
5.) Sulfate reduction: $C_{org} + SO_4^{2-} \rightarrow HS^- + HCO_3^- + CO_2$
6.) Methane generation: $C_{org} + CO_2 \rightarrow CH_4 + CO_2$

At a site where leachate with strong organic pollution is released to the groundwater, the zone of methane generation below the landfill is surrounded by a sulfate reduction zone, provided that some sulfate is present in the leachate (see Figure 4-10). Further downstream, oxides of iron and manganese are still available as electron acceptors and corresponding zones can be identified in the leachate plume. In addition, nitrate is often leached to the groundwater from the surface (e.g. agricultural activities) and a denitrification zone forms at the fringe of the pollutant plume. Also at the end of the organic pollutant plume nitrate is available as an oxidant and denitrification takes place before the groundwater turns aerobic again further downstream (see Figure 4-10). A detailed discussion of the biogeochemistry of landfill leachate plumes in the groundwater can be found for instance in Christensen et al. (2001) or Fetter (2008).

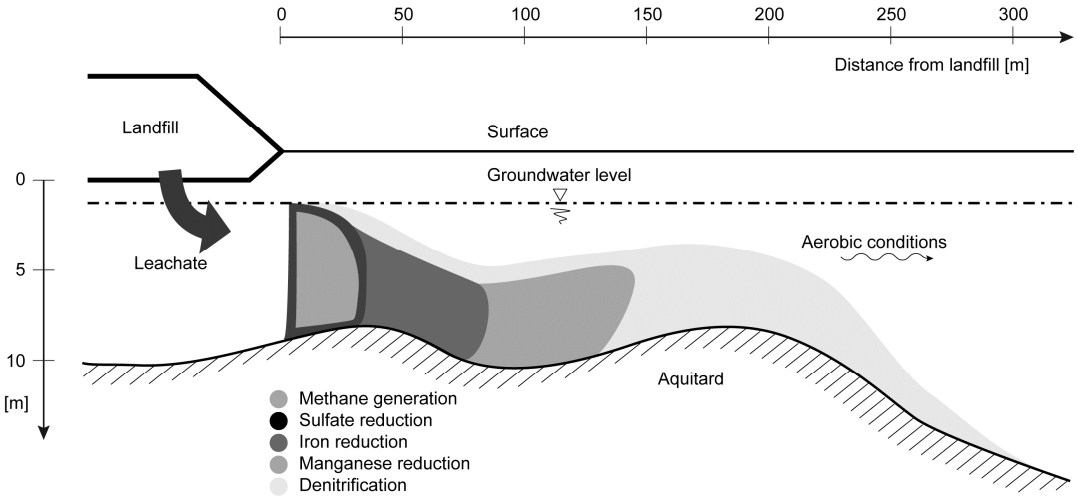


Figure 4-10: Redox zonation in groundwater polluted by landfill leachate based on investigations at the landfill „Grindstet“ by Ludvigsen et al. (1998)

While there is a general understanding of pollutant migration in the subsurface, an evaluation of substance transport is possible only in view of the local environmental conditions and for specific pollutants. Major attenuation mechanisms of different pollutants in a clay-rich soil are listed in Table 4-14 and are meant to qualitatively illustrate the most important processes potentially mitigating negative effects of the pollutants on the groundwater. Evaluations of landfill-borne pollutant migration for a specific system are typically associated with substantial uncertainties due to various reasons. Firstly, investigations of natural attenuation potentials have focused mainly on

hydrocarbons (i.e. petroleum hydrocarbons) and chlorinated solvents in the past, and not on typical landfill leachate constituents (e.g. National Research Council 2000). As a consequence, marginal effect levels have not been proposed for typical leachate constituents such as ammonium (cf. Table 4-11 and Table 4-12). Secondly, leachate is a complex mixture of different substances mutually influencing each others' migration behavior. Thirdly, the heterogeneity of the emission source (heterogeneous waste body, preferential water flow) and the dimensions of the source area (pollutant fluxes, points of release, groundwater flow directions) make the evaluation of natural attenuation difficult. Finally, natural variations in environmental system properties (e.g. soil properties) and incomplete system understanding also represent a source of uncertainty.

Table 4-14: Major attenuation mechanisms for landfill leachate constituents in clayey soil (Source: Bagchi 1987)

Leachate parameter	Major attenuation mechanisms	Mobility in clayey soil
Ammonium	Exchange, biological uptake	Moderate
Arsenic	Precipitation, adsorption	Moderate
Boron	Adsorption, precipitation	High
Cadmium	Precipitation, adsorption	Moderate
Calcium	Precipitation, exchange	High
COD	Biological uptake, filtration	Moderate
Chloride	Dilution	High
Chromium	Precipitation, exchange, adsorption	Low (Cr^{3+}), high (Cr^{6-})
Cyanide	Biotransformation, Adsorption	High
Iron	Precipitation, exchange, adsorption	Moderate
Lead, zinc, copper	Adsorption, exchange, precipitation	Low
Magnesium	Exchange, precipitation	Moderate
Manganese	Precipitation, exchange	High
Mercury	Adsorption, precipitation	High
Nickel	Adsorption, precipitation	Moderate
Nitrate	Biological uptake, dilution	High
Sodium	Exchange, dilution	Low to high
Sulfate	Exchange, dilution	High

Substance migration in the soil and groundwater zone is dependent on the mixture and concentration of the substances in the leachate as well as on the local environmental setting. To evaluate the impact of released landfill leachate on the subsurface environment, dominant substance transport mechanisms are incorporated into mathematical models, which can be used to determine substance concentrations along the transport pathway. The mathematical descriptions of important substance transport processes and selected transport models are presented below.

4.2.2.2 Models of substance transport in the subsurface

The subsurface environment under investigation comprises the vadose zone above the groundwater (provided the landfill bottom is not located in the groundwater zone) and the potentially affected groundwater zone. The landfill represents the pollutant source with the release rate and the composition of the leachate as the major parameters of interest. The dominant substance migration processes are advection, dispersion and diffusion, as well as sorption and transformation (degradation and volatilization). While these processes are only addressed briefly below, more elaborate descriptions of the individual migration processes can be found for instance in Luckner and Schestakow (1991), Hillel (1998) or Fetter (2008).

Description of major migration processes

Advection is the transport of a solute due to the movement of a liquid in the subsurface. If a liquid is in motion, advection[14] takes place. The movement of water in an unsaturated porous media is described by Richards equation (Richards 1931), which is based on Darcy's equation (Darcy 1856) and the continuity equation (law of mass conservation), see equation 4-1. Consequently, to describe advective transport, information about the porous medion (e.g. hydraulic conductivity, effective porosity) and about the hydraulic gradient causing water flow is necessary under water saturated conditions. Under unsaturated conditions, additional knowledge of the water content and the water retention curve is needed to describe advective transport.

(4-1) $$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left[k(\theta) \cdot \left(\frac{\partial h}{\partial z} + 1 \right) \right] - S(z,t)$$

where θ is the water content [m³/m³], h is the hydraulic head or hydraulic potential [m], k(θ) is the hydraulic conductivity as a function of the water content [m/s], t is the time [s], z is the depth [m], and S is the source/sink term [s^{-1}].

Hydrodynamic dispersion causes solute transport relative to the advective flow of the fluid. Hydrodynamic dispersion is dependent on advective flow (no dispersion without advection). The extent of hydrodynamic dispersion is related to the scale of investigation, as the dominant factors causing hydrodynamic dispersion differ from micro- to macro-scale. At the micro-scale, hydrodynamic dispersion is primarily caused by the variability of flow velocities across pore cross-sections, varying pore widths causing different flow velocities, diverse flow paths due to pore structure (tortuosity), and transversal pollutant spreading due to pore ramifications (cf. Luckner and Schestakow 1991). At the large scale, factors such as geologic stratification of the subsurface or the presence of local layers with different properties (e.g. soil lenses with low permeability in a gravel aquifer) have a dominant effect on hydrodynamic dispersion.

Molecular diffusion is mainly caused by the Brownian motion of particles, but also due to osmotic forces, heat diffusion, and electro-osmosis. Diffusion is described by Fick's law (Fick 1855), which establishes a direct relationship between the substance flux in static liquids and the concentration gradient of a substance in the liquids, see equation 4-2. In the unsaturated zone, the diffusion coefficient is strongly dependent on the water content and the tortuosity (cf. Hillel 1998). Although molecular diffusion is included in the dispersion coefficient in many substance migration models, it should be emphasized that molecular diffusion is related to another physical mechanism than hydrodynamic dispersion and it would also occur without advection, which is not the case for hydrodynamic dispersion.

(4-2) $$J_d = -D_0 \cdot dc / dx$$

where J_d is the diffusive flux [mol/(m².s)], D_0 is the diffusion coefficient [m²/s], and dc/dx is the concentration gradient against the direction of diffusive flux [mol/m^4].

The substance transport processes described above are combined to the advection-dispersion equation, which is given by equation 4-3 for unsaturated conditions.

[14] Advection is also referred to as convection.

(4-3) $$\frac{\partial(B_d \cdot C^*)}{\partial t} + \frac{\partial(\theta \cdot C)}{\partial t} = \frac{\partial}{\partial z}\left(D_s \cdot \theta \cdot \frac{\partial C}{\partial z} - q \cdot C\right) + \sum_i Y_i$$

where B_d is the bulk density of the soil [kg/m³], C* is the substance concentration bound to the solid phase [kg/kg], t is the time [s], θ is the water content [m³/m³], C is the substance concentration in the liquid phase [kg/m³], D_s is the substance-specific dispersion coefficient (including diffusion) [m²/s], z is the spatial coordinate in the flow direction [m], q is the advective water flow velocity [m/s], and Y_i is the source/sink term to consider transformation, release, and extraction processes [s^{-1}]. If the variables B_d, D_s, θ and q in equation 4-3 are constant, the equation resembles the advection-dispersion equation for the groundwater zone.

The first part of equation 4-3 describes the distribution of a substance between the liquid and the solid phase. Equilibrium reactions are often assumed, which means that thermodynamic equilibrium is reached in a short time between the substance concentration in the liquid and in the solid phase. Sorption isotherms adhering to this concept are the linear sorption isotherm, the Freundlich sorption isotherm, or the Langmuir sorption isotherm. The linear sorption isotherm is widely applied due to its simplicity ($C^* = k_d \cdot C$; with k_d being the distribution coefficient in kg/m³). But it is based on the assumption of unlimited sorption capacity of the soil, which is unrealistic in general and problematic in case of high substance loads in the liquid phase. Sorption processes are considered reversible, which means that substances accumulated in the solid phase can be released to the liquid phase again after a decrease of the concentrations in the liquid phase.

The last term of the advection-dispersion equation (4-3) considers source or sink processes taking place in the subsurface (cf. Luckner et al. 2008). Sinks for substances are, for instance, bioaccumulation processes (e.g. substance is metabolized for microbial growth), transformation processes, and substance extraction via roots. Transformation processes can be also a source process for daughter products from the degradation of pollutants originally present in the leachate (e.g. trichloroethane degradation is typically associated with the accumulation of vinyl chloride). However, the immobilization and transformation of pollutants is the product of many different processes and the modeling of these processes is confronted with limited knowledge and complex interactions. Therefore, while advection, hydrodynamic dispersion and diffusion are well understood, the description of substance immobilization and transformation is more uncertain (cf. National Research Council 2000).

The effects of different substance transport processes on the break-through curves in a column flow experiment are schematically illustrated in Figure 4-11. Ideal plug flow could be observed if substance transport was solely based on advection (i.e. the substance traveled at the same velocity as the liquid). Dispersion processes cause a stretching of the break-through curve (some solutes travel faster and others slower relative to advective flow), but do not decrease the overall substance concentration (in case of continuous application at the top). Adsorption and dispersion processes cause a retardation of substance break through, but only a minor reduction in the final substance concentration at the outlet. Thus, the only process significantly reducing the substance concentrations at the outlet of the experimental column is biodegradation, as a part of the applied substances is transformed into other substances and mass reduction of the applied substance takes place.

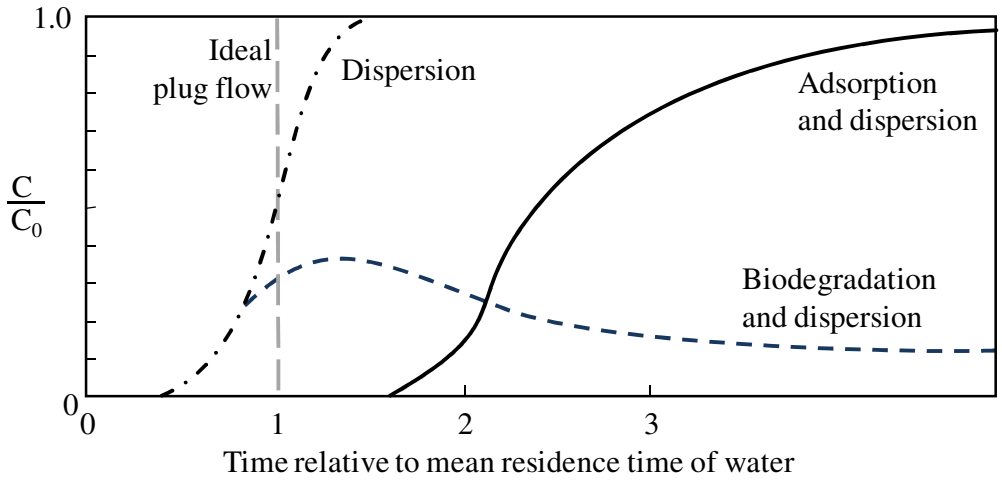


Figure 4-11: Schematic tracer break through curves to illustrate the effect of dispersion, sorption, and degradation processes on solute transport (according to Fetter 2008 based on Roberts et al. (1980))

Analytical and numerical transport models

The complexity of models to describe substance migration in the subsurface varies from simple screening tables to sophisticated numerical models. In general, the model choice depends on the processes of interest and on the available data. It should be emphasized that the most sophisticated model will not be available to describe processes realistically without appropriate input data. In many cases, a simple model based on conservative assumptions may be a good starting point to derive first estimates and potentially establish a more complex model with higher data and resource requirements in a subsequent step. In any case, the model complexity should directly relate to the quality and quantity of data to calibrate and validate the model.

Several analytical and numerical models to estimate substance transport in the vadose and/or groundwater zone are discussed briefly below. While there are numerous transport models available, the models presented here shall provide an overview about different model complexities and capabilities.

Analytical models

Analytical models are based on mathematical equations which are continuous in space and time and can be solved without iterations. They typically include assumptions on regular geometries and homogenous properties. The major advantages of such models are (cf. Foster Wheeler Env. Corp. 1998) that

- they are quick to set up and easy to use,
- they may be used to validate numerical models for a defined set of input data,
- they deliver stable solutions,
- they may be used to quickly derive conservative estimates before more complex models are used, and
- they can partly consider anisotropic conditions.

Apart from the advantages of ease of use, the simplifications and assumptions present in analytical models may cause the models to ignore significant migration processes occurring in the real system, for example due to the ignorance of heterogeneous soil properties or of the temporal variabil-

ity of boundary conditions. As examples, two analytical models for the unsaturated zone (AF model and Altex-1D) and one analytical model for the saturated zone (DOMENICO) are briefly discussed below.

Altex-1D

The model Altex-1D (LABO 2008) is based on the analytical solution of the advection-dispersion equation (cf. equation 4-3) for the one-dimensional case according to van Genuchten (1982). The model assumes homogeneous soil properties (equivalent soil properties can be calculated for a soil profile with different layers based on an empirical formula) and a stationary, one-dimensional downward flow field (transversal dispersion can be considered in the model making it three-dimensional (symmetric)). Two functions are available in the model to consider the pollutant source: a) constant source term strength for a limited period of time or b) exponential decrease of source term strength. The Altex-1D model assumes no pre-contamination of the transport pathway. Sorption processes are described via linear sorption isotherms. Degradation processes can be considered at a first order rate, i.e. the degradation rate is directly proportional to the concentration of the substance, in the model. The Altex-1D model has been developed for forecasting leachate concentrations from waste-derived material deposits in and on the soil. Case studies to illustrate the application of Altex-1D, including a description of the spread sheet calculations, are presented in the report by LABO (2008).

AF model

The AF model (Schneider and Stöfen 2004 and 2006) is also based on the analytical solution of the advection-dispersion equation (cf. equation 4-3) for the one-dimensional case (van Genuchten 1982). A continuous, stationary and downward flow field in the unsaturated zone is assumed without initial pollution of the transport pathway. Processes considered in the model include advection, dispersion, diffusion in soil liquid and gas, linear sorption, and degradation at a first order rate. Multiple soil layers can be specified in the transport model. Two source term functions are available in the model: The release of pollutants for a finite time and at a constant level on the one hand, and the release of pollutants at an exponentially decreasing rate on the other hand. Based on the comparison of average substance residence times in the unsaturated zone and the decreasing/limited source term strength, an attenuation factor is calculated. The relationships of substance residence times, finite or decreasing source term strength, and the resulting attenuation factors are expressed in nomograms (cf. Figure 5-12 and Figure 5-13), which were derived from numerous solutions of the advection-dispersion equation for a range of input parameters (Schneider and Stöfen 2004). The AF model does not determine a pollutant concentration over time, but calculates the maximum pollutant concentration above the groundwater surface. Like Altex-1D, the AF model has been developed to estimate the impact of pollutants from deposited waste-derived materials and it has been demonstrated to be a robust and conservative approach for estimating substance migration in the vadose zone (Schneider and Stöfen 2006).

DOMENICO

The DOMENICO model provides an analytical solution of the advection-dispersion equation under water saturated conditions (Domenico 1987). The solution is approximate. Further simplifications are the assumption of substance transport along the direct line between the source and the point of interest, homogenous concentrations in the source zone, and no pre-contamination of the aquifer.

Pollutant migration is three dimensional (symmetric around the flow path) due to the consideration of transversal dispersion. Apart from dispersion, further processes potentially decreasing substance concentrations in the model are sorption (linear sorption isotherms) and substance degradation at a first order rate. Input parameters to establish the model are the source concentration of the investigated substance, the dimensions of the source, the advective groundwater flow, and the dispersivity. Due to its simplicity and ease of use the DOMENICO model is frequently implemented in groundwater models to consider solute transport (e.g. BIOSCREEN).

Numerical models

Partial differential equations are used within numerical models to describe substance transport discretely in time and space. The system of differential equations can be solved iteratively (method of finite differences or method of finite elements) based on the definition of boundary conditions. Numerical models can simulate substance transport in heterogeneous systems with varying system properties over space and time. Major advantages of numerical models are (cf. Foster Wheeler Env. Corp. 1998) that

- they can describe the complex interaction of different processes,
- they allow for the simulation of multidimensional systems,
- they can consider different boundary conditions as well as the spatial variability of system properties,
- they can simulate stationary and dynamic flow conditions,
- they deliver temporally and spatially discrete results for the whole system.

The advantages of numerical models listed above are confronted by increased requirements with respect to the quality and quantity of input data, the higher resource-demand to establish and apply such models, and the possibility of numerical instabilities in the calculation. If appropriate data are available to apply numerical models, they represent a powerful tool to describe substance migration processes and their interaction in the subsurface. Out of the numerical models briefly described below, Hydrus and BIOF&T2D can simulate substance transport in variably saturated media and MODFLOW has been developed to model transport processes in the saturated zone.

Hydrus (1D, HP1, 2D/3D)

Hydrus 1D (Šimůnek et al. 2008) can simulate vertical water, heat and solute transport in variably saturated porous media. Water flow is modeled based on the Richards equation (cf. equation 4-1) with the van Genuchten soil parameters. Different hydraulic boundary conditions can be specified and potentially vary over time. While Hydrus 1D is restricted to one-dimensional transport processes, the Hydrus 2D/3D (Šimůnek et al. 2007) model allows for modeling the migration of water, heat, and substances in variably saturated porous media in two or three dimensions.

Substance transport in Hydrus is based on the advection-dispersion equation (cf. equation 4-3). The model can consider up to six substances, which may migrate independently from each other or can be interreleated by chemical reactions. Transformation processes can occur at a first or zero order rate, whereby reaction rates can be defined explicitly for the solid, liquid, and gas phase. Equilibrium and non-equilibrium sorption processes can be considered in the model. In addition, Hydrus 1D has been coupled with PHREEQC-2 within the HP1 model (cf. Jacques and Šimůnek 2005) to geochemically model reactive substance transport in the subsurface (interaction of the soil matrix with the liquid phase).

The various Hydrus models can be applied to a variety of problems relating to migration processes in porous media. While the detailed modeling of water flow has been the primary focus of the Hydrus models, the integrated HP1 model also allows geochemical modeling at a highly sophisticated level. Hydrus represents a powerful tool to model migration processes in the subsurface.

BIOF&T2D

The bio flow and transport (BIOF&T) model was developed as a two-dimensional (2D) model to evaluate the effect of remediation activities at contaminated sites (Resources & Systems International 1995-2000). Water flow in variably saturated media is described by the Richards equation (cf. equation 4-1) with limited possibilities of defining boundary conditions in the model. Substance migration processes are expressed via the advection-dispersion equation (cf. equation 4-3) and based on a dual porosity approach (cf. van Genuchten and Wierenga 1976), which assumes advective flow in larger pores and sole diffusive substance transport in small pores. The focus of BIOF&T2D is on biological degradation processes, which can be described by aerobic or anaerobic degradation following a first order rate or Monod kinetics. In case of anaerobic degradation, the sequential degradation of a substance and the formation of daughter products can also be considered. However, to realistically model the degradation of organic compounds in the subsurface, the model needs to be based on empirically determined reaction rates under site-specific conditions (e.g. leachate composition, soil properties, redox conditions, etc.).

MODFLOW

MODFLOW was developed by the geological survey of the USA to simulate three-dimensional groundwater transport (McDonald and Harbaugh 1988). The numerical solution of the water flow equations is based on the method of finite differences. Stationary as well as non-stationary water flow through heterogeneous, anisotropic porous media can be modeled. While the initial focus of MODFLOW was primarily on the description of groundwater flow, additional modules have been continuously integrated into the model to consider various migration processes. Apart from substance transport, density flow or aquifer compression, there are also modules allowing to model water flow in variably saturated media.

Evaluation of transport models

Apart from analytical or numerical models, simple screening tables or mass balance approaches can provide a basis to evaluate pollutant migration in the subsurface, too. Screening tables may be useful for a first assessment of the situation at a site, but cannot be adapted to site-specific conditions to a high degree and are not able to consider the effect of various mechanisms on substance migration. Mass balance approaches, without the consideration of processes reducing substance concentrations apart from dilution (i.e. assuming plug flow), can provide a quick, conservative estimate on substance migration. In general, the choice of a model to describe substance migration processes in the subsurface, depends on several factors:

- the type and quality of information which is expected from the model (e.g. screening assessment vs. detailed risk assessment)
- the migration processes and pathways of interest
- the complexity of different models and data requirements to establish the models
- the availability of data to establish and apply a model
- the expected level of detail in the modeling outcomes

- the experience of modelers and decision-makers with a specific model

In summary, the complexity of the modeling approach should reflect the quality and availability of data on the investigated system, the expertise available for establishing and adapting the model, and the required level of detail in the results to provide decision-support. As the source term function (i.e. quality and quantity of landfill leachate) is associated with substantial uncertainty and the amount and quality of site-specific data to establish transport models is typically limited (e.g. hydrogeologic investigations before/during landfill construction, data from groundwater monitoring wells), simpler models are often more appropriate to describe substance transport below closed landfills. However, as simple models are based on various assumptions, the limitations of the models need to be discussed in view of the accuracy and usefulness of model outcomes. Monte Carlo simulations can be used to outline inherent uncertainties and the sensitivity of the results with respect to the variation of specific parameters by assigning probability density functions to uncertain input parameters. Such an approach has been implemented in the groundwater risk assessment software "LandSim", which was developed to evaluate the impact of a landfill on groundwater quality (Environment Agency 2004). The software combines a source term function (exponential decrease of pollutant concentrations), estimates on the effectiveness of the containment system, and pollutant transport models for the unsaturated and saturated zone to evaluate the impact of a landfill on groundwater quality (cf. Hall et al. 2003). While the software is not flexible with respect to different source term functions or other transport models, it allows for a probabilistic evaluation of the modeling results, which is valuable to understand the reliability of model outcomes.

Although complex models enable the detailed simulation of substance migration in the subsurface, simple models are more appropriate to evaluate substance migration at most landfills due to reduced input data requirements, easier handling and application of the models, and the possibility of using simple models as a basis to adapt a higher tier model in a consecutive evaluation. Therefore, the AF model is suggested for the modeling of transport processes in the vadose zone and a mass balance approach is suggested to describe substance migration in the groundwater within the evaluation methodology. The models and their application within the evaluation methodology are described in chapter 5.2.

4.2.3 Summary and discussion

The potential of natural systems to mitigate negative effects of released substances on environmental media is based on naturally occurring processes reducing the mass, toxicity, extent, mobility or concentration of pollutants in the environment. Landfill-borne pollutants may migrate through a top cover in the landfill gas, may be released to surface waters in collected leachate or surface runoff, or may be released to the subsurface via leachate migrating through the landfill bottom. The latter was in the focus of this section, due to landfill leachate as a potential long-term threat to groundwater quality. Pollutant migration in the subsurface can be mathematically described via the advection-dispersion equation, addressing the processes of advection, hydrodynamic dispersion, diffusion, sorption, and transformation. While advective transport and dispersion/diffusion processes are well understood, knowledge of substance immobilization and transfer in the subsurface is still incomplete and respective models are associated with substantial uncertainty. As the

quality and quantity of data to establish substance migration models is typically rather limited at landfills, a simple modeling approach (combining the AF model in the vadose zone with a mass balance approach in the groundwater zone) is suggested to evaluate transport processes in the subsurface. The data requirements to establish and adapt the chosen, relatively simple, approach at a landfill are listed in Table 4-15. As the required information to model substance migration shown in Table 4-15 probably exceeds actual data availability at many closed landfills, complementary data from the literature and/or from empirical estimation formulas may be necessary to establish the models. Nevertheless, at landfills where more and better data are available or can be collected, the evaluation of substance migration processes in the subsurface may build on more sophisticated models (e.g. numerical models).

Table 4-15: Basic information to establish a simple model on the subsurface migration of leachate constituents

Source term (scenario-based emission estimates)
- Flow rate: Leachate release rate to the subsurface
- Pollutant levels: Concentrations of critical leachate constituents (current and future development)
- Geometry: Contaminated area, dimensions in relation to the groundwater flow direction
Unsaturated (vadose) zone
- Layers: Soil layers (including low permeability soil layer at the landfill base) For each layer: Thickness and soil properties (grain size distribution, bulk density, C_{org}, hydraulic conductivity, etc.)
- Pollutant-specific transport parameters: Diffusion coefficients (water and air), degradation rates (half lives) For each layer: Sorption coefficients
Groundwater
- Flow characteristics: Flow direction and hydraulic gradient
- Aquifer characteristics: Hydraulic conductivity, thickness, geologic strata
- Groundwater quality: Concentrations up- and downstream (existing pollution, dissolved oxygen, Redox conditions)

5 Evaluation method to determine aftercare completion criteria

The evaluation methodology consists of four elements: a) models on emission characteristics, b) models on the performance of the containment system, c) substance migration models in the environment (i.e. subsurface environment), and d) points of compliance (PoC), where certain criteria (e.g. groundwater quality criteria) have to be met, which are combined to derive site-specific aftercare completion criteria. The scenario layout underlying these completion criteria may imply additional criteria to be considered during the evaluation of landfill environmental compatibility (e.g. geotechnical stability). The procedure to derive completion criteria for a closed landfill is schematically illustrated in Figure 5-1.

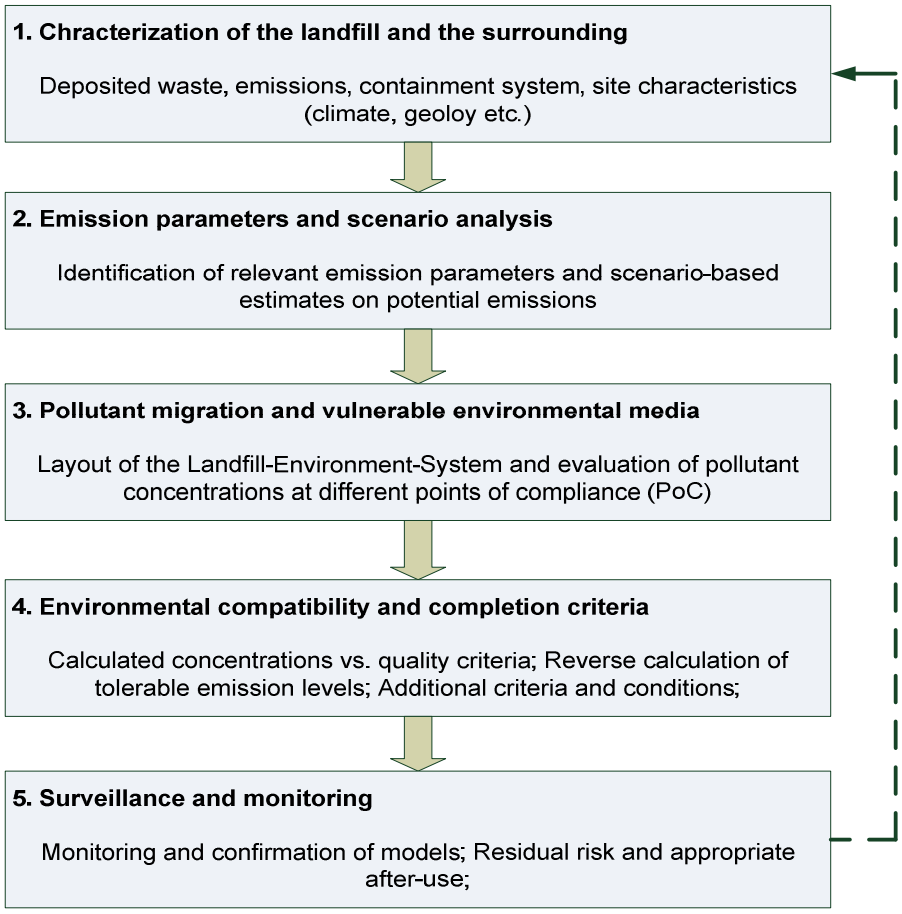


Figure 5-1: Schematic illustration of the procedure to evaluate landfill environmental compatibility and determine post-closure care completion criteria

To evaluate any system, data need to be collected on the system itself and the boundary conditions relevant to the evaluation. Hence, as a first step, data on the landfilled waste (e.g. amount and composition of the deposited material over time, state of waste degradation, hydraulic regime), the containment system (e.g. design and construction, field tests and performance data), the emission monitoring (e.g. quality and amount of landfill gas and landfill leachate over time), and the site characteristics (e.g. climatic conditions, geologic and hydrogeologic conditions, vulnerable uses in the vicinity, natural hazards) need to be collected. Subsequently, based on an analysis of this data

(especially with respect to landfill monitoring), relevant emission parameters are identified and emission scenarios are developed to estimate potential emission levels of the landfill associated with specific landfill conditions. The scenario-based emission estimates form the input for modeling pollutant migration in the surrounding environment during step 3 (see Figure 1). The effect of released landfill pollutants is evaluated at defined PoCs located along the migration pathways. In the fourth step, the calculated concentrations at the PoCs are compared to quality criteria. If the effect of the scenario emissions complies with the quality criteria applied at the PoCs, the scenario can be considered environmentally tolerable. If not, the tolerable scenario emission levels at the source can be determined based on the acceptable quality criteria at critical PoCs and the attenuation factors derived from the transport modeling. The reverse calculation starting from the PoC in the direction of the source (landfill) results in a tolerable emission level for a specific substance and a specific scenario. The tolerable emission levels at the source are calculated for the most probable long-term emission scenario and the critical point of compliance. In addition, there might be criteria addressing geotechnical stability, containment system performance, after-use, further care, etc. In any case, a period of monitoring will be necessary at the site after the evaluation in order to validate the underlying models and to confirm/increase the reliability of the evaluation results (cf. Figure 5-1). Specific elements of the evaluation procedure are described in more detail below.

5.1 Evaluation of the state of the landfill and the site

The analysis of the landfill state includes current and historic emission levels, a characterization of the waste body (direct or indirect), information about the performance of the landfill containment system (i.e. top cover and base lining system), and data addressing the climatic, hydrogeologic, and topographic conditions at the site.

5.1.1 General data on the conditions at the site

The climatic conditions at the site are a major factor to estimate the (potential) infiltration of water into the waste and to evaluate the influence of temperature fluctuations on the landfill containment system. Some landfills are equipped with monitoring stations, whereas for others data will be available only from nearby measurement stations operated, for instance, by the meteorological service. The current use, post-closure care strategy and planned after-use are of interest, as the use of the site needs to be assessed in view of its impact on the landfill conditions. Information about the geology at the site, properties adjacent to the landfill, potentially affected uses in the vicinity of the site (e.g. groundwater resources, residential areas), and potential exposure to natural hazards can be obtained from landfill application documents, official notifications and permits, plans on the design and construction of the site, investigation of site geology and hydrogeology prior to and during landfill construction, etc. The rate of waste deposition (quantity and quality over time) should be available from annual reports on landfill operations and topographic documentations of the used landfill space. For sites with incomplete data, aerial photographs taken at different times or interviews with neighbors might provide an indication of the deposition history (Ehrig and Brinkmann 1999). In general, collected data should be verified by involved personnel (e.g. landfill staff, public authorities) and site inspections should be carried out by the evaluator. If data are

missing for the site, it may be possible to gather this information from regional information services (e.g. flood risk zoning and other hydrologic data, geologic maps and surveys, maps of sensitive ecosystems, etc.).

5.1.2 Emission data

5.1.2.1 Leachate

Before monitoring data can be used to describe the state of the landfill, they should be subject to critical review with respect to accuracy and representativeness. This includes the identification and appropriate treatment of outliers (e.g. transcription errors, data coding, calibration errors, etc.), the evaluation of potential interferences at the sampling location (e.g. leachate collection pipe vs. leachate tank), or the check of plausibility via related parameters (e.g. comparison of ammonium concentrations to nitrate concentrations in anaerobic leachate). If the data suggest the presence of outliers after a visual inspection, statistical tests can be applied to identify outliers and potentially remove them from the monitoring data set. Dixon's Test (Dixon 1953) has been recommended by Gibbons and Coleman (2001) to test for outliers in data sets addressing environmental contamination. However, cross checks with other parameters will be necessary to assess outliers in a leachate parameter data set, as factors such as the leachate discharge rate might have a strong effect on substance concentrations in the leachate (cf. Johnson et al. 1999; Zweifel et al. 1999). Consequently, the measurement of leachate discharge rates, when leachate samples are taken, might prove valuable to understand natural variations in the monitoring data set of some parameters (e.g. soluble salts). After the evaluation of the data quality, statistical analysis can be used to estimate representative current and maximum values, establish trends evident from the emission data, or determine levels of confidence for exceeding given standard values (cf. Gibbons and Bull 2006). A visual assessment might indicate a trend, which can be statistically tested. With respect to landfill leachate, Sen's Test (Sen 1968) has been recommended, because it is a rather simple, non-parametric test which can be used to test for trend hypotheses as well as to determine the average slope of a trend (Gibbons and Bull 2006). In addition, statistical relationships between different emission parameters can be used to retain information about parameters with a reduced number of measurements (cf. Kylefors 2003). For instance, the electric conductivity of leachate might be strongly dependent on the actual discharge rate and well correlated with the concentration of chloride in the effluent. In such a case, the availability of continuous electric conductivity measurements would also allow for statements on chloride concentrations in the same temporal resolution without specific chloride measurements (cf. Laner et al. 2011b).

As hydrologic boundary conditions of the waste body are generally altered in the post-closure phase of a landfill (due to top cover installation), it is useful to relate emission data to the liquid-to-solid ratio (L/S), which is the amount of water (in liters) percolating through a given waste mass (in kg dry matter of waste). The L/S can be used to compare field-scale data to results from laboratory experiments (cf. Kylefors et al. 2003) and it can provide an indication of the remaining pollution potential in a landfill (cf. Allgaier and Stegmann 2003). Water flow patterns in landfills are highly heterogeneous and thus, the remaining stock of mobilizable substances in the landfill cannot be assessed solely on the basis of observed emission levels (cf. Fellner et al. 2009a). Emission data

will provide information restricted to the fraction of the landfill body which is in contact with water, where leaching and organic degradation processes take place (cf. Huber et al. 2004). Therefore, an evaluation of the remaining emission potential within a landfill needs to include hydraulic heterogeneity of a landfill, e.g. wet as well as dry landfill parts, the persistence of dry regions in the landfill, and preferential pathways for leachates (Rosqvist et al. 2005).

5.1.2.2 Landfill gas

The discussion of representativeness and reliability of leachate data is, in general, equally valid for landfill gas data. However, collection rates are typically much lower for landfill gas than for leachate prior to final capping. Depending on the duration of landfill operation, the landfilled waste, the intermediate and final covers, and the gas extraction system, the collected fraction of landfill gas may be between a few percent and 2/3 of the total amount of produced landfill gas (cf. Krümpelbeck 2000). In general, the collection efficiency will increase after a cover is installed at the site and might well be 85% or above at sites with low-permeability final cover systems (cf. Spokas et al. 2006). Landfill gas generation can be estimated based on theoretical gas generation models (cf. chapter 3.1.1.1) or measurements of fugitive emissions at the landfill surface can be used to complement gas collection data. Theoretical gas generation models typically overestimate the amount of landfill gas, as the conditions within the waste body deviate from model assumptions. Heterogeneous waste composition, different micro-climates, non-availability of water, presence of inhibitors, etc. cause actual gas generation to be below model estimates (Krümpelbeck 2000). Based on the analysis of monitoring data sets (including the measurement of fugitive emissions) at various MSW landfills, Fellner et al. (2004) concluded that annual methane emission loads estimated by theoretic gas generation models are on average 100 % higher than the actual loads. Fugitive gas monitoring techniques (e.g. box or chamber systems, tracer gas methods) could provide site-specific data to complement gas flow rates in a collection system and estimate gas generation trends. In addition, trend analysis can provide insights with respect to landfill gas generation, as the collected fraction of generated landfill gas is typically nearly constant after final capping. However, as landfill gas data are of poor quality at many sites landfill (e.g. short period of gas collection, suction of air into the system, no measurements of fugitive emissions), statements based thereon should be made with caution.

5.1.3 Deposited waste

Generally, the waste body of a MSW landfill is highly heterogeneous. On the one hand, this is due to diverse compositions of the deposited material, waste placement practices (stratified waste layers), and operation of landfill compartments (e.g. leachate re-infiltration). On the other hand, this is due to different micro-climates established within the waste body, most notably because of the presence or non-presence of mobile water (Rosqvist and Bendz 1999). The latter is of primary importance, as substances in landfill zones not taking part in the water flow regime remain largely unaltered and therefore, could be responsible for increased emission levels in the future if water flow and moisture increase due to a change in water distribution. Apart from tracer experiments (e.g. Rosqvist and Bendz 1999; Fellner 2004), geophysical investigation methods might be suitable to map the distribution of water and its dynamics in a landfill body (e.g. Bernstone et al. 2000;

Hoekstra et al. 2009). Such non intrusive characterization methods can provide cost-effective tools to obtain information about the hydraulic characteristics of the landfill. However, the achievable resolution of measured data and the need of additional information to interpret the measurements limit the informative value of such investigations. Therefore, in most cases a detailed investigation of the landfill body will include intrusive techniques to characterize the landfilled waste. Sampling of deposited waste is a difficult task due to initial waste heterogeneity and different states of degradation, so a large number of samples is necessary to characterize the landfill body. Nevertheless, high resolution sampling of waste bodies was proven to be useful to characterize the landfilled waste and its state of biochemical degradation (e.g. Sormunen et al. 2008). As the emission characteristics of a MSW landfill are largely determined by the biochemical degradation of organic matter, numerous methods have been developed to investigate the biological reactivity of deposited waste samples (see Table 5-1). The amount of material used for the analysis is usually smaller than the amount extracted. Hence, in addition to the original heterogeneity of the waste body, the necessary reduction of sample mass for analysis introduces further errors. The smaller the sample size for analysis, the more care has to be taken to maintain the representativeness of the reduced sample (cf. Gy 2004).

Table 5-1: Methods for determining the extent of biological degradation of organic matter contained in waste samples

Method of analysis	Parameter	Unit	Sample size
Waste sorting	Paper and cardboard content	kg/kg	kg - Mg
Landfill simulation reactor (LSR)	Leachate and gas emissions, settlements, temperature, analysis of solid samples (before – after)	various	kg - Mg
Biological tests	Biochemical methane potential (anaerobic), respiration activity (aerobic), self-heating tests	various	g
Leaching tests	Leached concentrations (e.g. NH_4-N, DOC)	kg/m³	g
Fourier transform infrared spectroscopy (FT-IR)	Presence or absence of specific bands is used to gain information about the decomposition status	cm^{-1} (wavenumber)	mg – g
Chemical analysis	e.g. fiber analysis (ratio of cellulose (+ hemicellulose) to lignin), cellulose content, TOC, N, loss on ignition	kg/kg	mg - g

In Table 5-1 methods to determine the state of biodegradation are shown together with sample sizes typically used to measure the parameters of interest. The largest amount of material can be analyzed by waste sorting, as all the extracted waste can theoretically be sorted, with the content of paper and cardboard often used to characterize the state of waste decomposition (e.g. Sormunen et al. 2008). However, as the initial composition of the extracted waste is generally unknown, in most cases the paper and cardboard content would be one among several indicators to describe the extent of waste transformation. Another option to determine the state of degradation of waste samples is to observe their behavior in landfill simulation reactors (LSR) (e.g. Kruse 1994; Fellner et al. 2009a). At a smaller scale, biological tests like the biochemical methane potential test might be used to quantify the biochemical stability of landfilled waste. Apart from the test duration (e.g. 100 days), the results of these experiments vary due to different inoculates, container volumes, and sample sizes (e.g. Kelly et al. 2006). Waste samples can also be used in leaching experiments to determine mobilizable substance fractions. However, as these experiments cannot take into account biochemical degradation processes, they are of limited value for estimating remaining emission

potentials related to the degradation of organic matter (Kylefors et al. 2003). Further methods to investigate the degree of waste decomposition are infrared spectroscopy (e.g. Smidt et al. 2002), fiber analysis (cellulose, hemicelluloses, and lignin) of waste samples (e.g. Kelly et al. 2006), or direct chemical analysis of solid waste samples (e.g. content of total organic carbon). Provided that the limitations of the individual approaches are considered, the analysis of waste samples represents a valuable source of information about the characteristics of the landfill body. However, apart from scientific endeavors, intensive waste sampling and analysis has not been used to monitor and evaluate the waste quality in existing landfills yet.

5.1.4 Containment system

Landfill containment systems serve the purpose of separating the waste body from the surrounding environment and controlling the interaction between these compartments. A review of the state of knowledge about the performance of different containment systems and single elements is provided in chapter 4.1. Functional mechanisms of the components making up the barrier system are resistance (e.g. clay liners, geomembranes), capacitance (e.g. evapotranspirative soil layers, attenuation layers), and extraction/injection (e.g. drainage layers, gas vents) (National Research Council 2007). The performance of barrier systems is commonly expressed as acceptable mass flow rates through top covers or bottom liners. The "status quo" can be evaluated based on direct monitoring data (e.g. leachate generation), investigations of the integrity of single barrier components (e.g. visual inspection of leachate collection pipes), and indirect monitoring data (e.g. observation of groundwater wells, settlement measurements). A survey by Bonaparte et al. (2002) on reported failures of landfill containment systems showed that in 86 % of the cases the principal factor causing a system to fail was barrier design or construction. This is an indication of the crucial importance of appropriate design and the enforcement of construction quality standards in order to achieve specified barrier performance. Due the limited observation periods (a few decades at the maximum), the importance of other factors might increase with ongoing deterioration of the barriers. With respect to contaminant migration through the base liner, groundwater monitoring does not necessarily provide a reliable and timely indication of leachate emissions (cf. Christensen et al. 2000; Inyang 2004).

Barrier system components have limited technical life times and are likely to fail at different times. For instance, the service life of a HDPE geomembrane in a MSW landfill with a temperature of 35°C at the liner has been estimated to be about 160 years (cf. Rowe 2005 and Table 4-3). However, as the duration of service periods and respective efficiencies are dependent on the actual state of the barriers (=resistance built into the system) and the stresses induced on the system components (cf. Inyang 2004), a consequent evaluation has to be site specific. In any case, as long-term monitoring data on the performance of waste containment systems do not exist and predictions cannot be verified yet, future barrier performance should be judged with caution, but also in view of synergetic effects occurring in composite liner systems (cf. chapter 4.1).

5.1.5 Geology and hydrogeology at the site

Information about the geology and the hydrogeology can be obtained from investigations at the site associated with the permit or the construction of the landfill. Typically a landfill permit includes the operation of monitoring wells upstream and downstream of the landfill, to evaluate potential adverse effects on the local groundwater quality. In addition, data on the groundwater situation as well as the geologic strata are available from official maps and potentially from online platforms of central information services[15]. Although, site-specific information is preferred to regional information, the latter is useful to check plausibility of site data, describe the landfill surrounding, and as a basis to evaluate potential adverse effects of landfill emissions off the site.

5.2 Emission models and scenario-based predictions

The method to evaluate potential future emission levels builds on a detailed analysis of monitoring data, investigations and predictions of the behavior of the deposited waste, an evaluation of the long-term functionality of landfill facilities (i.e. containment systems), and an evaluation of site-specific factors such as climate or potential exposure to extreme events. The procedure to derive site-specific emission estimates is schematically illustrated in Figure 5-2. Emission levels are predicted by using an emission model under the assumption of constant release mechanisms (i.e. water flow pattern and mobilizable waste fraction) on the one hand, and potentially, adapted models for scenarios investigating a change of water flow pattern or a change of dominant release mechanisms, on the other hand. In any case, future emission characteristics are related to the change of the liquid-to-solid ratio of the deposited waste and the mobilizable waste fraction. Previous models on future leachate emissions (e.g. Belevi and Baccini 1989; Kruse 1994; Krümpelbeck 2000) were generally based on the assumption of stable boundary conditions at the landfill site and did not take into account the heterogeneity of water flow through the landfill body. Due to the investigation of different emission scenarios (i.e. analyzing different site conditions, such as levels of barrier performance, and emission models, for example a change of water flow paths) this methodology allows for a more comprehensive analysis of potential emission levels at a landfill (cf. Figure 5-2). Scenarios are established based on the current situation at the site and estimates of changes of specific conditions in the future. The scenario-based emission estimates are subsequently used to model pollutant migration in the surrounding environment and evaluate the impact of landfill emissions at defined points of compliance (PoC).

[15] For example, the Austrian hydrographic service: http://gis.lebensministerium.at/eHYD/ or the geological survey of Austria: http://geomap.geolba.ac.at/GEO/

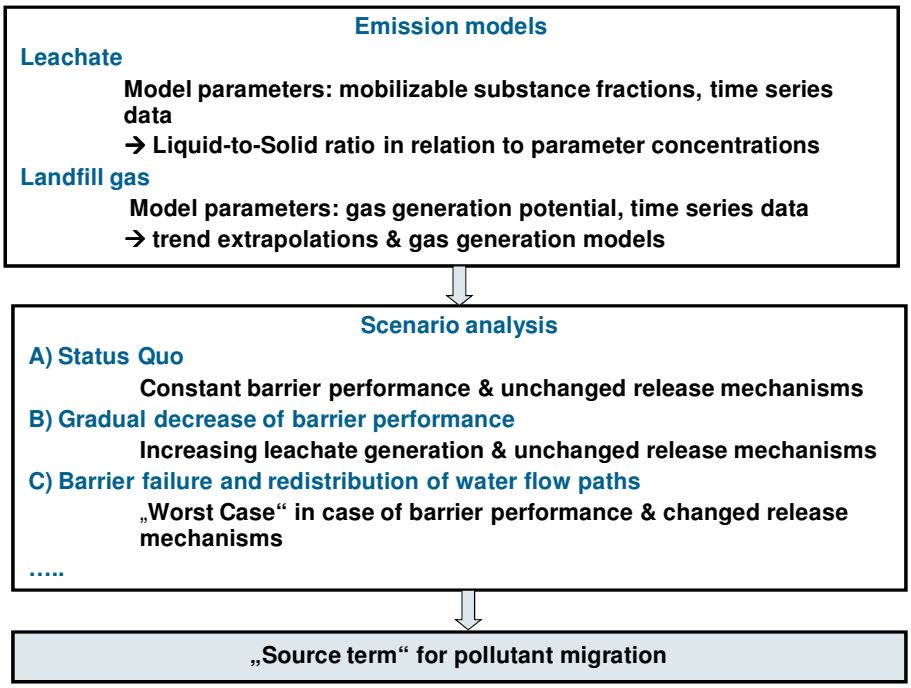


Figure 5-2: Schematic procedure to derive estimates on potential future emissions at the landfill

5.2.1 Emission models

5.2.1.1 Leachate emission models

Models to estimate future leachate emissions can be based on the extrapolation of observed emission behavior, both in full- and lab-scale (e.g. Krümpelbeck 2000), simple mathematical models (e.g. Belevi and Baccini 1989), complex biogeochemical models considering various processes in a spatially distributed way (e.g. White et al. 2004). As the former are not based on physical parameters relating to the landfill and the latter have not been able to cope with the complexity of processes occurring simultaneously and interacting with each other in a landfill yet, a simple "grey-box" model is introduced to estimate future leachate emissions. The model is developed for the period after gas generation levels have dropped significantly and a decreasing concentration trend can be observed in the leachate (i.e. typically some time after landfill closure). Future leachate characteristics are estimated based on the mobilizable substance potential of the waste, the heterogeneity of water flow, and the observed emission levels at the site. The model is built under the assumption that the landfill conditions after closure remain constant. This means no change of the water flow pattern will occur within the landfill and the dominant mechanisms of substance mobilization will remain the same (e.g. stable pH, anaerobic environment). Hence, provided that a statistically significant trend is observable in the post-closure emission data, which has been reported for many parameters in the leachate of MSW landfills after site closure (cf. Krümpelbeck 2000; Gibbons et al. 2007; Laner et al. 2010), this trend can be extrapolated to the future. The adaptation of an emission model to a data set on leachate concentrations is schematically illustrated in Figure 5-3. As a first approximation, an exponential decrease with a first order rate has been used for several leachate constituents (e.g. Belevi and Baccini 1989; Krümpelbeck 2000; Kruse 1994), assuming that leachate concentration depends on the mobilizable substance concentration of

the waste. The emission model described below addresses leachate from MSW landfills, but could be applied also to other landfill types if appropriate data are available and release mechanisms are comparable. A discussion of emission profiles for different landfill types (i.e. municipal solid waste incineration ash and construction & demolition waste) and possible approaches to predict future emissions is provided in chapter 3.

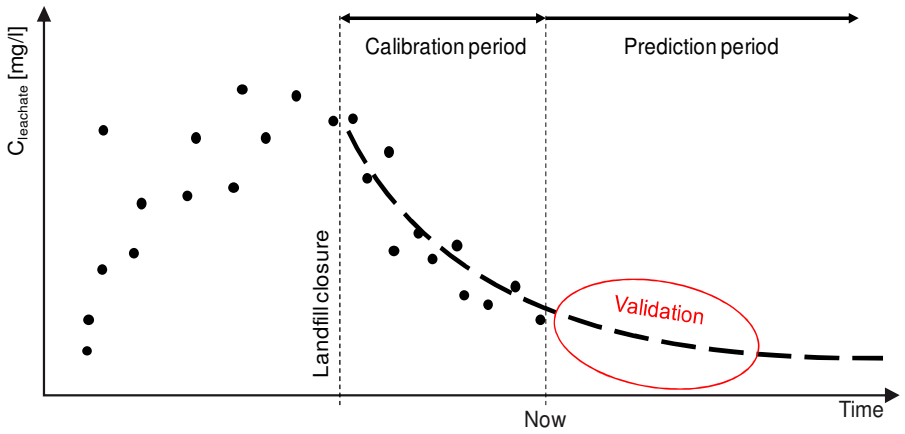


Figure 5-3: Adaptation of the emission model to the post-closure monitoring data at the site (including model validation)

The basic model formulation to estimate the mean leachate concentration of a specific substance as a function of time after landfill closure (c(t) in mg/l) is given in equation (5-1). The formulation is based on the model suggested by Belevi and Baccini (1989), but modified as follows: heterogeneity of the water flow regime is included via a heterogeneity factor (h) and different release rates are considered for processes dominated by leaching mechanisms (first term) as well as for processes associated with the degradation of slowly degradable organic matter are taken into account (second term):

(5-1)
$$c(t) = c_{0,leach} \bullet e^{-(\frac{c_{0,leach}}{m_{0,leach}} \bullet \Delta \frac{L}{S} \bullet h) \bullet t} + c_{0,org} \bullet e^{-(\frac{c_{0,org}}{m_{0,org}} \bullet \Delta \frac{L}{S} \bullet h) \bullet t}$$

In equation 5-1, $c_{0,leach}$ denotes the substance concentration readily leachable after the intensive reactor phase (mg/l). $m_{0,leach}$ is the leachable fraction of mobilizable substances (mg/kg DM). h is a heterogeneity factor accounting for the fraction of the landfill taking part in the water flow regime (dimensionless). ΔL/S is the change of the deposited waste's liquid-to-solid ratio per year (l/kg DM and year), and t is the time after intensive reactor phase, i.e. when gas generation has dropped significantly and leachate concentrations exhibit a downward trend (cf. Belevi and Baccini 1989), in years. The second term in equation 5-1 is used to describe the slow release of substances due to the degradation of slowly degradable organic matter. $c_{0,org}$ is the concentration of the substance made available for leaching via slow organic degradation processes directly after the intensive reactor phase (mg/l). $m_{0,org}$ is the fraction of mobilizable substances which is still being made available via organic degradation processes (mg/kg DM). This term is negligible for readily soluble salts (e.g. chloride) or the soluble fraction of metals contained in the waste, whereas it needs to be considered for substances associated with the degradation of organic matter (e.g. organic leachate pollution, ammonium). The heterogeneity factor h is calculated as the total volume of the waste

body divided by the volume taking part in water flow and contributing to leachate emissions. Typical values for h range from two to six (cf. Döberl et al. 2005; Rosqvist and Bendz 1999), but may vary significantly between individual landfills. As a first guess for the fraction of mobilizable substances (m_0), values can be taken from the literature. Some literature data on the potential of mobilizable substances in deposited MSW are given in Table 5-2. However, site investigations should confirm the parameters on mobilizable substances to establish a reliable emission model. The potentially mobilizable fraction is split into readily available substances ($m_{0,leach}$) and substances which are made available via slow organic degradation processes ($m_{0,org}$). The distinction between these fractions can be based on different experimental settings, investigating readily leachable substance concentrations (e.g. leaching tests), on the one hand, and slowly releasable waste fractions (e.g. landfill simulation reactors) associated with organic degradation processes, on the other hand. However, model parameters need to be validated using observed emission characteristics (landfill monitoring data) and should be checked for plausibility via literature data (see Table 5-2). The initial leachate concentration due to slow organic degradation processes ($c_{0,org}$) can be estimated using decay rates of the slowly degradable organic matter published for landfill gas generation models (cf. Krümpelbeck 2000). Based on the estimated half-life, the decay constant is calculated. This equals the value of the power term in brackets (equation 1, second term) and thus allows to estimate $c_{0,org}$. For instance, a decay constant of 0.0173 (corresponds to a half-life of 40 years of the organic matter), a change of liquid-to-solid-ratio of 0.01 l/kg DM per year, a water flow heterogeneity factor of 2, and a $m_{0,org}$ of 400 mg/kg DM result in a $c_{0,org}$ of 346.5 mg/l[16]. The readily leachable substance concentration after the intensive reactor phase ($c_{0,leach}$) results from subtracting $c_{0,org}$ from concentrations at the site observed after the intensive reactor phase. Monitoring data is necessary to check the plausibility of the model parameters and fit the model estimates to the observed leachate characteristics.

The model includes several parameters for which a range of probable values should be assigned based on site investigations. The effect of the variation of mobilizable substance fractions of chloride in the waste between 1.5 g/kg DM at the minimum and 3.0 g/kg DM at the maximum (cf. Table 5-2) and a range of heterogeneity factors between 2 and 6 result in the minimum model curve (=m_0→min, h→max) and the maximum model curve (=m_0→max, h→min) shown in Figure 5-4. The minimum and maximum estimates are compared to all the chloride measurements at 30 MSW landfills after closure (see chapter 3.1.2 and appendix 1, Table A-1). The average annual change of the liquid-to-solid ratio was calculated from the data of the 30 sites and the initial chloride concentrations (c_0) of the model curves were chosen based on the measured minimum and maximum concentrations. The comparison of model curves and chloride measurements during the aftercare period, illustrates that the range of parameters allows for a description of the observations at the sites. The large bandwidth of potential model estimates highlights the need for site-specific data to calibrate and subsequently validate the emission model.

[16] Compare to equation 5-1, second term: $\frac{c_{0,org}}{m_{0,org}} \bullet \Delta \frac{L}{S} \bullet h = 0.0173 \rightarrow c_{0,org} = \frac{400 \bullet 0.0173}{0.01 \bullet 2} \rightarrow c_{0,org} = 346.5\ mg/l$

Table 5-2: Data on mobilizable substance fractions of landfilled MSW via leachate from various literature sources, in mg/kg dry matter (DM) of MSW

Parameter [mg/kg DM]	Extrapolation of landfill monitoring data [1]	Leaching tests with landfilled waste A (age 11-14 yrs) [2]	Extrapolation based on LSR experiments with MSW [3]	LSR experiments with landfilled waste (age 18 yrs) [4]
CSB	2500 - 11200	-	3000	1538-1885
BSB_5	-	-	-	90 – 150
TOC	-	2100 – 7100	-	515 – 616
Cl	900 – 3800	1000 - 1500	2100 - 2500	1563 – 1818
SO_4	-	-	-	3.1 – 9.4
NH_4-N	1400 – 3400	200 – 310	2200	794 – 991
Fe	-	20 – 39	-	5.1 – 8.8
Pb	-	0.1 – 2.5	-	-
Cu	-	1.0 – 6.7	-	-
AOX	2 – 22	-	0.6 – 1.2	-
Cd	0.002 – 0.006	0.06 – 0.22	-	-
Zn	0.2 – 16	14 – 98	-	-

Notes: LSR… Landfill Simulation Reactor

Sources: [1] Krümpelbeck 2000, [2] Belevi and Baccini 1989, [3] Kruse 1994, [4] Döberl et al. 2005

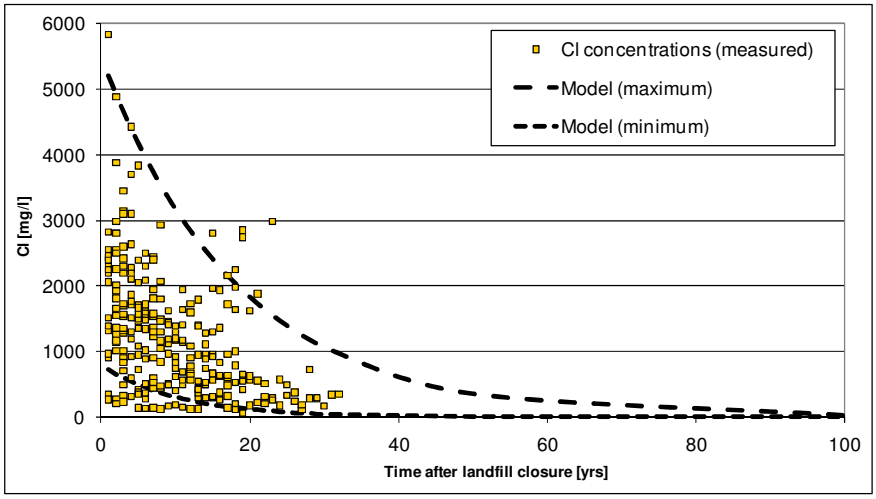


Figure 5-4: Minimum and maximum model estimates for chloride based on the variation of the model parameters m_0 and h in a plausible range in comparison to chloride measurements at 30 closed MSW landfills

In similarity to chloride, the minimum and maximum model curves are estimated for ammonium-nitrogen based on a plausible range of mobilizable substance fractions between 1.0 and 4.0 g/kg DM (the fraction of slowly mobilizable nitrogen is assumed between 20 and 90%) and for a water flow heterogeneity factor from 2 to 6. The minimum and maximum model curves are compared to the NH_4-N measurements at 30 closed MSW landfills (cf. chapter 3.1.2 and appendix 1, Table A-1) in Figure 5-5. Again, the model curves envelop the measured concentrations, which underlines the ability of the range of model parameters to describe NH_4-N concentrations in the leachate of closed MSW landfills. However, the large margin between minimum and maximum curves also highlights

the need for appropriate data. Apart from monitoring data (model calibration and validation), direct investigations of mobilizable substance potentials or water flow heterogeneity may be necessary to derive reliable model estimates for a specific landfill. In any case, it should be noted that the mobilizable fraction used in the model is not static, but will vary with the period of investigation and the considered substance mobilization processes. However, under the assumption of stable conditions (=no change of water flow paths, constant dominant release mechanisms) the parameters are to be specified in a way that the model fits the emission data at the site. The model estimates need to be verified by future monitoring to gain confidence into the model. Without validation, the model remains more or less speculative, depending on the amount of site-specific data (e.g. water flow heterogeneity, mobilizable substance fractions) available to build the model.

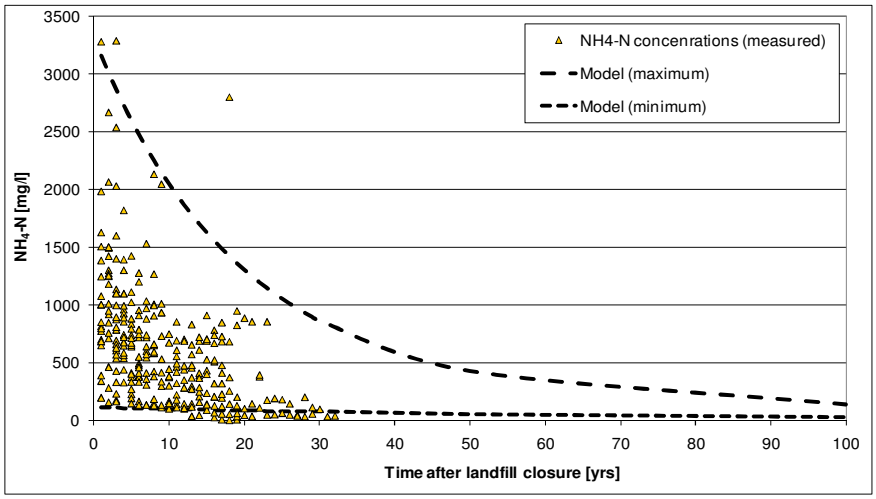


Figure 5-5: *Minimum and maximum model estimates for ammonium-nitrogen based on the variation of the model parameters m_0 and h in a plausible range in comparison to NH_4-N measurements at 30 closed MSW landfills*

The emission model described above is based on the assumption that the mechanisms of substance release from the waste and the water flow distribution in the waste are stable. If an emission scenario involves a change of these processes, a different emission model will be needed as a basis for such a scenario. For instance, a redistribution of the water flow paths in the waste will most likely be associated with an increase in leachate concentrations due to the leaching of substances from previously dry areas (cf. Laner et al. 2011b). The new water flow pattern might include some parts of the waste body which were not previously taking part in the water flow regime. Thus, in these previously "dry" zones, biochemical processes might be restored and leachable substances will concentrate in the leachate. Such processes might occur as a consequence of human intrusion into the landfill or because of the water saturation of the waste body (e.g. flooding or pooling of leachate in the landfill).

Based on the extent of change of the hydraulic flow pattern assumed within the scenario and the original heterogeneity of water flow (parameter h in equation 5-1: the higher it is, the larger is the fraction of mobilizable substances in previously dry zones), the mobilizable waste fraction of the adapted emission model can be calculated. For instance, if a scenario occurs after the mobilizable substances in the waste originally taking part in the water flow regime have been leached, the remaining mobilizable substance potential for the scenario emission model would be $m_{0,scenario} = m_0$

- m_0/h (cf. equation *5*-1). Neglecting a potentially occurring initial "reactor phase" and defining the water flow heterogeneity in the scenario, the emission model for this scenario can be estimated (c_0 can be calculated using a constant ratio of c_0/m_0 from the calibrated emission model during post-closure monitoring). With all the assumptions stated above, the model formulation of equation *5*-1 can be used to estimate leachate characteristics during the scenario. A comparison of the emission model based on unchanged conditions and an emission model according to the description above (complete redistribution of water flow paths) is schematically illustrated in Figure 5-6. The concentrations of the emission model based on stable conditions are lower in Figure 5-6, as they have been decreasing already during the post-closure period until the time of evaluation. From the discussion above it is apparent that scenario emission models are often highly speculative and that they could be established in many different ways. For instance, a possible emission model for a scenario to illustrate the effect of a redistribution of water flow in the waste on the emissions of a MSW landfill has been described in Laner et al. (2011a). Again, for other scenarios different models might be appropriate. Accordingly, the formulation of the emission model has to be described in the scenario layout and be consistent with the investigated effects. In general, the emission model based on the post-closure monitoring data may provide a valuable basis for the establishment of scenario emission models, too.

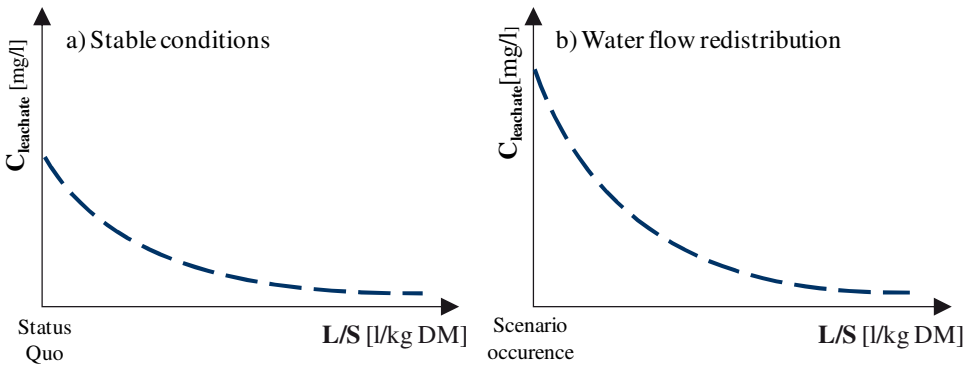


Figure 5-6: Schematic illustration of different emission models (concentration of a substance in the leachate as a function of the waste's liquid-to-solid ratio) based on scenario conditions

5.2.1.2 Landfill gas generation models

It is of crucial importance that landfill gas generation models are applied and validated with site-specific data (cf. Barlaz et al. 1990). Validation is imperative, since actual gas generation is dependent on waste characteristics, moisture content and distribution, temperature, pH, microbial communities, nutrient availability, and the presence of inhibitors. Due to the assumption of optimal conditions many landfill gas generation models drastically overestimate the amount of gas produced (Fellner et al. 2004). Depending on the investigated landfill conditions, different models might be appropriate to estimate landfill gas generation. Under stable conditions at the site, the extrapolation of observed gas generation trends or the adaptation of a specific model (cf. Table 3-1) to observed generation rates may be used to estimate future gas generation rates. At closed MSW landfills low levels of landfill gas generation have been observed already a few decades after landfill closure. For instance, Krümpelbeck (2000) reported gas collection rates at German MSW land-

fills below 2 liters per kg of dry waste matter and year already 10 to 15 years after closure. Nevertheless, within specific scenarios, a landfill gas generation model could be used to estimate an increase of landfill gas generation even at old MSW landfills due to a reactivation of biochemical processes based on the remaining fraction of degradable organics in the waste.

Several models have been suggested to estimate gas generation rates at MSW landfills. Models differ with respect to their complexity and with respect to basic assumptions on the biodegradability of organic matter (i.e. degradation rate or half life values for refuse components). For instance, models which have been widely applied are the LandGEM model by the U.S. EPA (2005), the GasSim model by Golder Associates (e.g. Attenborough et al. 2002), or the model by Tabasaran and Rettenberger (1987). However, the resulting gas generation rates and yields are rough estimates and should be treated accordingly.

5.2.2 Scenario layouts

Scenarios are used within the evaluation methodology to illustrate the effect of specific landfill conditions on emission levels. The layout and the number of investigated emission scenarios is dependent on the current conditions, the potential future conditions (e.g. site in a flood plain), and the long-term management strategy (e.g. whether human intrusion is to be included in a scenario analysis) at the site. Comprehensible scenario analysis is based on the description of scenario design and underlying emission models, the discussion of sensitive assumptions, and reasoning for the inclusion/exclusion of specific scenarios. A basic procedure to derive scenarios addressing potential future conditions has been suggested by Schwartz (1991) and includes six steps of scenario creation (adapted to long-term landfill management):

1. Identify critical decision criterion
   Why are the scenarios created and which questions shall be answered based on scenario analysis?
2. Significant landfill processes and mechanisms for the emission behavior
   Identify and describe dominant processes with respect to the release of substances from the landfill.
3. Stress induced on the landfill system at the site
   Connect site-specific conditions to processes associated with the release of substances from the landfill.
4. Rank different factors (stresses) based on importance and uncertainty
   Identify the most important factors and those with the highest uncertainties for further investigation.
5. Describe scenario logics
   Define the relationship between stresses (site-specific conditions) and processes in the landfill leading to emissions.
6. Establish scenarios for analysis
   Define scenarios to evaluate the effect of different site-specific conditions on landfill emissions.

The procedure described above represents a general guidance to establish consistent scenarios. With respect to long-term landfill management, substantial uncertainties due to the long time frames and the limited knowledge about the current and future landfill system have to be tackled. To foster understanding of the potential significance of an event with respect to possible emissions, it is recommended to analyze best and worst case scenarios associated with specific conditions (e.g. level of containment), before more complex scenarios are investigated.

Apart from the persistence of the status quo, continuous changes of landfill conditions (e.g. deterioration of technical barriers, intrusion of air into the waste body), and extraordinary events, like earthquakes or flooding, might be relevant for scenario development. Flooding has been shown to be of potential significance for many older landfill sites in Austria with consequences for landfill emissions in the short and in the long-term (cf. Laner et al. 2009). The intrusion of air into the waste body and consequent changes of redox-conditions and pH may be associated with a significant release of heavy metals. However, the aeration of a MSW landfill is expected not to occur naturally within many centuries, for an unsaturated, uncovered landfill, to several hundred thousands of years, for a covered, water-saturated landfill (cf. Bozkurt et al., 1999). Consequently, such a scenario may be of interest primarily for long-term considerations and subsequent emission rates might be rather low due to the extended duration of the aeration period in most cases.

5.2.2.1 Scenarios addressing containment system performance

Although existing observations confirm that modern landfill containment systems are principally able to separate the waste body from the environment and control landfill emissions (cf. National Research Council 2007), these data relate to a few decades only. The technical life times of barrier systems in field-scale applications are largely unknown. Models of deterioration patterns range from constant barrier strength (or no barrier) to exponential deterioration of the barrier, with the choice of model having a dominant impact on predicted emission levels. Despite the assumption of constant strength being incorrect without maintenance, it can be used to describe the lack of an engineered barrier (constantly zero) or a system which is fully serviceable and forever maintained (Inyang 2004). Without maintenance, an exponential deterioration pattern has been suggested to be the most suitable to assess long-term performance of containment systems. Such degradation is associated with a gradual increase of percolation rates through the barrier systems. As field data to evaluate the long-term performance of containment systems is missing, a set of scenarios is used to illustrate the effect of different containment system performance levels on potential landfill emissions. The persistence of the status quo (constant conditions at the landfill including unchanged performance of the containment system) is used as a “best case scenario” within the scenario analysis. Further scenarios might address the slow deterioration of the technical barriers associated with a gradual decrease of barrier performance levels, and a “worst case scenario” may assume a total failure of the containment system. The specification of containment system performance levels for these scenarios is described below. A description of the different containment systems and the basics to evaluate long-term performance is provided in chapter 4.1.

Constant performance – persistence of the status quo

As the scenarios are used to evaluate the effect of the termination of post-closure care at the landfill, constant performance levels of the containment system resemble the "best case" with respect to landfill isolation. The best performance level is based on the final layout of the landfill. Hence, at landfills, where the final cover has not been installed yet, performance levels of the final system might be estimated from monitoring of similar final cover systems at comparable sites (i.e. climatic conditions), as site-specific data are not yet available. Austrian regulations require a final cover to limit infiltration into the landfill to less than 5% of annual precipitation (Austrian landfill directive, MoE (2008)). However, at most closed landfills the final cover will be in place at the time of evaluation and leachate generation rates can be used as a basis to assess the top cover performance after gravity drainage of the waste body.

The performance of the base lining system is harder to assess, because direct monitoring data on leachate percolation rates (i.e. secondary liner system) are typically not available. Performance data for composite liner systems at landfills show that leachate collection efficiencies greater than 99% can be achieved (e.g. Barlaz et al. 2002), especially if a rigid quality assurance program has been followed during construction. However, collection efficiencies as low as 90% of the generated leachate have been reported for geomembrane bottom liners at landfills (Bonaparte et al., 2002), too. In some cases groundwater monitoring data can give an indication of containment system performance, but leachate plumes might not be detected at all (spatial resolution) or with a substantial time delay due to long travel times from the landfill to the monitoring well (cf. Inyang 2004).

Complete failure of the containment system

The complete failure of the barrier system is the "worst case" with respect to the interaction level between the waste body and the surrounding environment. The failure of the top cover system (i.e. the low permeability elements of the cover are ineffective) is typically assumed to result in infiltration rates similar to the local groundwater recharge rate. However, higher infiltration rates might be justified under specific conditions. For example, the failure of a top cover with a low top soil layer thickness (< 0.5 m) or a low fraction of surface runoff (e.g. depression or insignificant slope) may be associated with infiltration rates above the natural groundwater recharge rate at the site. The efficiency of the base lining system within this scenario is assumed to be zero. As no leachate is collected, all the generated leachate is released to the subsurface below the landfill during the worst case of complete containment failure.

Gradual decrease of containment system performance

Factors and mechanisms relevant for the performance of barrier systems or single components are described in chapter 4.1. In general, the future performance of the containment system is a function of the resistance built into the system and the stresses induced on the system. Long-term performance predictions are speculative in nature, with future performance levels expected to be between the current service level (at best) and the complete inefficiency of the containment (at worst). The evaluation of long-term performance is based on the analysis of the current state of the system and the expected impacts on the system at the site. To determine the rate and extent of a gradual decrease in containment system performance the modeling period is split into three service

periods and performance levels are estimated for each period. Performance is delimited by the current system performance (the best possible service level provided there is no future maintenance) and the complete inefficiency of the liner elements (the worst service level). The predicted service levels of the containment system are based on the evaluation of different performance factors and their importance in a specific service period. The list of factors to assess system performance is shown in Table 5-3 and Table 5-4 for a top cover and a base lining system including a composite liner (for more details on barrier system evaluation see chapter 4.1). An evaluation score between 1 (good) to 3 (bad) is assigned to each factor in Table 5-4 based on the criteria in Table 5-3, which refer to the conditions at the landfill site. Table 5-3 provides some suggestions for criteria in consideration of Austrian conditions, which may be modified within the evaluation process to consider different conditions or different sensitivities of the barrier system at other sites. For instance, a score of one for "barrier performance at the time of evaluation" means that the barrier has performed according to its design specifications, whereas a score of three means that it has not achieved its intended service level at the time of evaluation. After the scores have been assigned by the evaluator, they are multiplied with specified weighting factors and a total is calculated for each service period (cf. Table 5-4). Because the weighting factors for a specific service period add up to two, the evaluation result is between two (i.e. if all factor scores were one) and six (i.e. if all factor scores were three). A score of two is equivalent to the best service level in the service period (i.e. the system performance at the time of evaluation or during the previous service period) and a score of six is equivalent to the worst service level in this period (i.e. complete inefficiency). The overall evaluation results (between two and six) are linearly translated into respective performance levels. The corresponding procedure is illustrated in Figure 5-7 for top cover performance during a modeling period of 300 years. The resulting infiltration levels are also shown in Table 5-4. Within these illustrative calculations a local groundwater recharge rate of 158 mm/yr is assumed (worst case performance) and the top cover currently allows for a water infiltration of 3.2 mm/yr (initial performance level). For the bottom lining system an evaluation result of two equals a barrier efficiency of 99% (actual and best future performance) and a result of six equals a barrier efficiency of 0% (worst performance level).

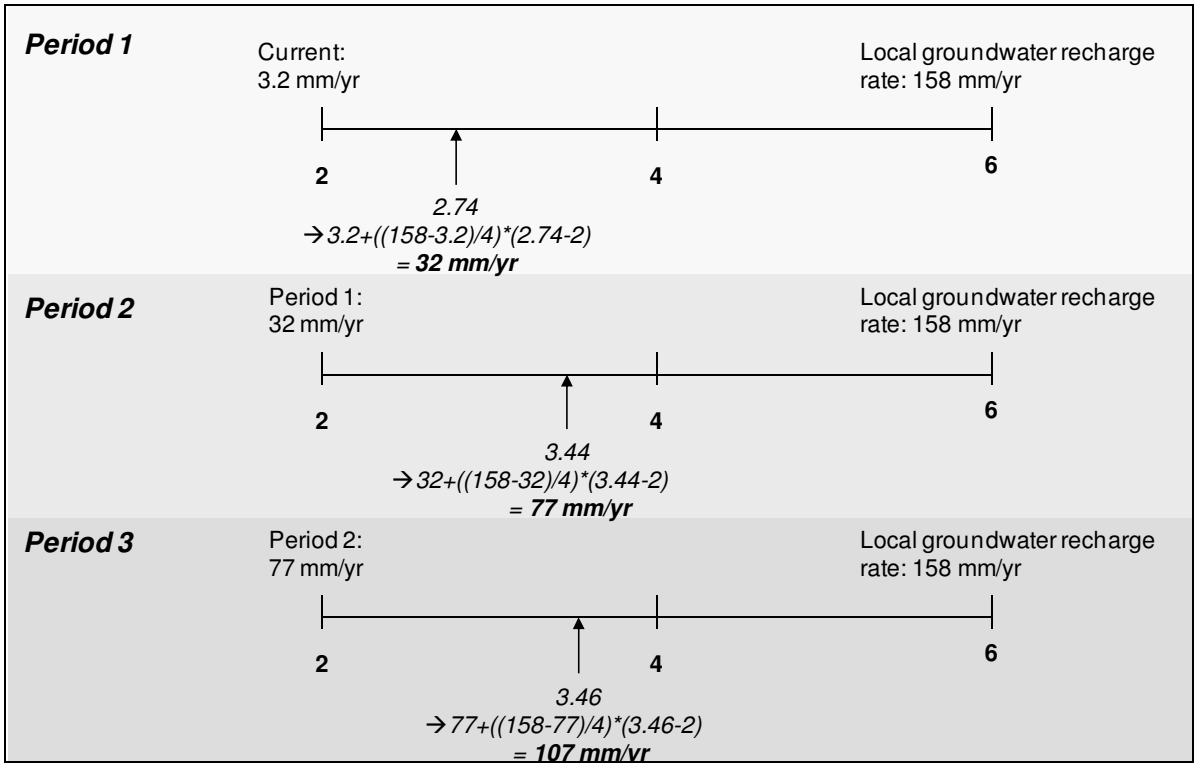


Figure 5-7: *Schematic illustration of the procedure to translate the evaluation results in Table 5-3 into corresponding infiltration rates for the top cover system (annual precipitation at this site is 630 mm/yr)*

The whole procedure to estimate a gradual decrease of barrier performance levels is highly speculative. It should be emphasized that these are assumptions within a scenario to illustrate the effect of decreasing barrier performance on landfill emissions and not deterministic predictions of future barrier performance levels. The performance levels estimated for each service period are reached at the end of the respective period. Hence, for the example in Table 5-4 the rate of water infiltration into the waste would have increased from 3.2 mm initially to 32 mm/yr after 100 years.

Table 5-3: Basic criteria for the evaluation of relevant factors to estimate future barrier performance levels suggested for Austrian conditions (see Table 5-4)

SCORE	GOOD (1)	MEDIUM (2)	POOR (3)
Top cover (composite liner)			
Barrier performance at time of evaluation	Monitoring verifies design specifications	Monitoring data is not available (yet), but design specifications are probably met	Minimum design requirements are not met
Construction quality program	Appropriate documentation of quality assurance is available	Quality assurance has been reported, but protocols/test results do not exist	No quality assurance program during landfill construction
Recultivation layer (cover)	Thickness: >1.5 m (in any case above higher than root or frost penetration depths)	Thickness: 0.5 - 1.5 m (recultivation + drainage > 1 m)	Thickness: > 0.5 m
Heat production within waste	Low reaction levels in the waste (< 35 °C)	Waste temperatures measured/expected: 35 - 60 °C	Strongly exothermic reactions (i.e. MSWI landfills with > 60 °C)
Climate	Moderate Austrian climate	Moderate climate with very cold winter or high annual precipitation (> 1400 mm/yr)	Extreme climate with very high fluctuations in temperature
Typical vegetation and projected after use	Planned after-use does not threaten top cover	Vegtation with high root penetration depths (compare to thickness of recultivation layer)	Plants at the site typically have root penetration depths above 1.5 m
Relief	Heap with moderate slopes	Landfill in a depression, but slopes in place to guarentee stormwater runoff	Flat surface or depression not facilitating surface runoff
Expected settlements	Low settlement expected: differential settlements within 1.5 m radius below 0.1 m	Some settlments expected, but at least 90 % of total settlement has been completed	Stronger settlements have to be expected with differential settlements above 0.3 m
Lining control systems (direct monitoring)	Direct monitoring system exists	Only indirect monitoring of performance	No monitoring
Drainage system	Appropriate design and materials	Drainage layer from alternative materials without proof of equival performance to standards	No drainage layer
Base liner (composite liner)			
Barrier performance at time of evaluation	Direct or indirect monitoring verifies design specifications	No monitoring, but design specifications are probably met	Minimum design requirements are not met
Construction quality program	Appropriate documentation of quality assurance is available	Quality assurance has been reported, but protocols/test results do not exist	No quality assurance progrm during landfill construction
Climate	Moderate Austrian climate	Moderate climate with high annual precipitation (> 1400 mm/yr)	Extreme climate with high annual precipitation and frequent high intensity rainfalls
Heat production within waste	Leachate temperature is typically below < 30 °C (at the outflow)	Leachate temperature between 30 °C and 40 °C	Leachate with temperatures above 40 °C
Distance to groundwater	Groundwater surface (max.) is more than 1 m below the landfill base	Groundwater surface (max.) is just below the landfill base	Groundwater surface might rise into the landfill
Overburden pressure	Maximum landfill thickness (incl. base liner and top cover system) > 20 m	Maximum landfill thickness (incl. base liner and top cover system) 10 - 20 m	Maximum landfill thickness (incl. base liner and top cover system) < 10 m
Leachate quality	Low load of organic substances and of potentially precipitating constitutents	Leachate with higher loads (organic, precipitating), but decreasing trend	Leachate with high loads (organic, precipitating) and chemically aggressive
Monitoring (e.g. groundwater monitoring)	Direct monitoring system exists	Only indirect monitoring of performance	No monitoring
Drainage system	Appropriate design and materials	Drainage layer from alternative materials without proof of equivalent performance to standards	No drainage layer
Stability of foundation	Geotechnical stability has been proven (global and local)	Geotechnical stability might decrease (e.g. slope stability)	Landfill is not geotechnically stable or there is a high risk of extreme events (e.g. earthquakes)
Landfill geometry (heap vs. cavity)	Heap with gravitational drainage	Lanfill in a depression, but with gravitational drainage	Landfill in a depression without gravitational drainage

Table 5-4: Example of an evaluation chart to determine future barrier performance levels of a top cover and base lining system at a specific site

Factors – top cover (composite liner)	**Evaluation score**	**Weighting factors***		
	1_{good}–3_{bad}	0–100 yrs	100–200 yrs	200–300 yrs
Barrier performance at time of evaluation	**1**	0.36	0.08	0.065
Construction quality program	**1**	0.38	0.44	0.12
Recultivation layer (cover)	**3**	0.33	0.48	0.487
Heat production within waste	**1**	0.09	0.06	0.04
Climate	**1**	0.08	0.12	0.487
Typical vegetation and projected after-use	**2**	0.09	0.48	0.487
Relief	**1**	0.08	0.1	0.12
Expected settlements	**1**	0.38	0.09	0.065
Lining control systems (direct monitoring)	**1**	0.1	0.06	0.05
Drainage system	**1**	0.1	0.09	0.08
	Evaluation results:	**2.74**	**3.44**	**3.46**
	Leachate generation rates [mm/yr]:	**32**	**77**	**107**
Factors – bottom liner (composite liner)				
Barrier performance at time of evaluation	**1**	0.44	0.08	0.07
Construction quality program	**2**	0.46	0.49	0.44
Climate	**1**	0.07	0.08	0.09
Heat production within waste	**1**	0.09	0.06	0.05
Distance to groundwater	**1**	0.08	0.08	0.09
Overburden pressure	**1**	0.08	0.08	0.08
Leachate quality	**2**	0.08	0.08	0.08
Monitoring (e.g. groundwater monitoring)	**3**	0.08	0.07	0.07
Drainage system	**2**	0.46	0.11	0.10
Stability of foundation	**1**	0.10	0.44	0.46
Landfill geometry (heap vs. cavity)	**1**	0.08	0.43	0.48
	Evaluation results:	**3.18**	**2.82**	**2.77**
	Leachate percolating through landfill base [mm/yr]:	**10**	**34**	**59**

* Weighting factors were derived based on expert interviews:

The survey involved 23 experts from Austria and Germany and was conducted in two steps. In the first step, 23 experts were invited to create a list of relevant factors for barrier performance based on a suggestion by the author. 13 of the 23 invited experts accepted the invitation and participated in the list of relevant factors (see above). In the second step, the 13 experts were asked to evaluate the importance of each factor for two standard (according to the Austrian Landfilling Directive) top cover and base lining systems at a landfill site with average Austrian climate (e.g. precipitation around 1000 mm/yr). For three different time periods each factor had to be evaluated with scores ranging from 1 to 5 (1 being insignificant and 5 being very important for the performance of the barrier system). Based on the evaluations relative weights are calculated of each factor and time period (average score for a specific factor divided by the sum of average scores for all factors). In addition, the factors most often ranked among the top three factors in a specific period by the experts are given additional weights (the sum of additional weights of the top factors is equal to the sum of the weights of all factors). The final weighting factors for a composite lining system at the top and the bottom of a landfill are shown above (see also chapter 4.1 and appendix 2).

5.2.2.2 Scenarios addressing a change of emission behavior

In many cases the persistence of anaerobic conditions and the stability of water flow patterns in the landfill will be a valid assumption to build emission scenarios. For such scenarios the emission model based on post-closure monitoring data can be used. These scenarios will be useful primarily for modeling timeframes of several decades to a few centuries. However, for long-term modeling (e.g. natural aeration of MSW landfills, dissolution of mineral phases releasing heavy metals from MSW incineration ash landfills due to a decrease of pH) or for externally induced, sudden changes of landfill conditions (e.g. flooding, human intrusion), a change of emission behavior can be expected.

The persistence of preferential water flow paths has not been studied in landfills. However, such investigations have been carried out for natural soils with little dissolution and degradation processes (compared to waste), where preferential flow paths appear to be stable at least for many decades to centuries (Bundt et al. 2000; Hagedorn and Bundt 2002). The invasion into the waste body or the remodeling of the waste body will have an impact on the water flow pattern and consequently change the emission characteristics of the landfill. The latter has been observed during the installation of a final top cover system at a MSW landfill (cf. Laner et al. 2011b), where the reorganization of top waste layers to guarantee an appropriate cover slope, caused pollutant levels in the leachate (e.g. NH_4-N, Cl) to increase substantially. The potential increase of emission levels due to a redistribution of water flow paths is directly correlated with the heterogeneity of water flow in the landfill and the total amount of mobilizable substances remaining within the waste body. These relationships have been illustrated in Figure 5-8 for the amount of degradable organic matter in a landfill and the hydraulic regimes in the waste body (see van Vossen 2010). Consequently, the smaller the fraction of waste taking part in the water flow regime, the higher is the remaining emission potential within the landfill (as long as the water flow pattern does not change).

The flooding of a landfill has a substantial impact on landfill emission behavior (cf. Laner et al. 2009). Given the long time period a landfill may present a potential threat to the humans or the environment, and the practice of locating landfills close to rivers in the past, landfill flooding could be a significant emission scenario for many closed landfills. For instance, in Austria around one third of closed MSW landfills are located in flood risk zones (Laner et al. 2009). Scenarios to estimate landfill emissions due to flooding can range from a complete erosion of the landfill to a reactivation of biochemical degradation processes after the flood event. The associated risks have to be evaluated on a site-specific basis and in view of other potential sources of pollution adjacent to the river (Laner et al. 2009).

In the long view, the aeration of MSW landfills could be associated with changing emission characteristics (cf. Kjeldsen et al. 2002). As this is speculated to occur over an extended time period of thousands of years for most landfills (Bozkurt et al. 1999), the associated mobilization processes, for example with respect to metals mobility, will occur at a slow rate. Due to the very long view of such a scenario, predictions are difficult and the importance of different factors such as pH value, redox conditions, humic substances, or the sorption capacity of the waste is hard to evaluate.

Another long-term scenario at MSW incineration ash landfills might look at the release of metals due to a decrease of pH within the landfill after the acid buffering capacity has been consumed, which is expected to occur over many thousands of years (cf. Förstner and Hirschmann 1997). Geochemical modeling has been suggested to evaluate the effect of mineral dissolution on the concentrations of heavy metals in the landfill leachate (cf. Hellweg 2000; Dijkstra et al. 2006). Based on equilibrium modeling, Hellweg (2000) presented an approach to estimate the sequential dissolution of different mineral phases and associated metal concentrations in the leachate over time (see also chapter 3.2). As modeling periods are typically in the area of a hundred thousand years, practical consequences can hardly be expected for the management of closed landfills from such modeling.

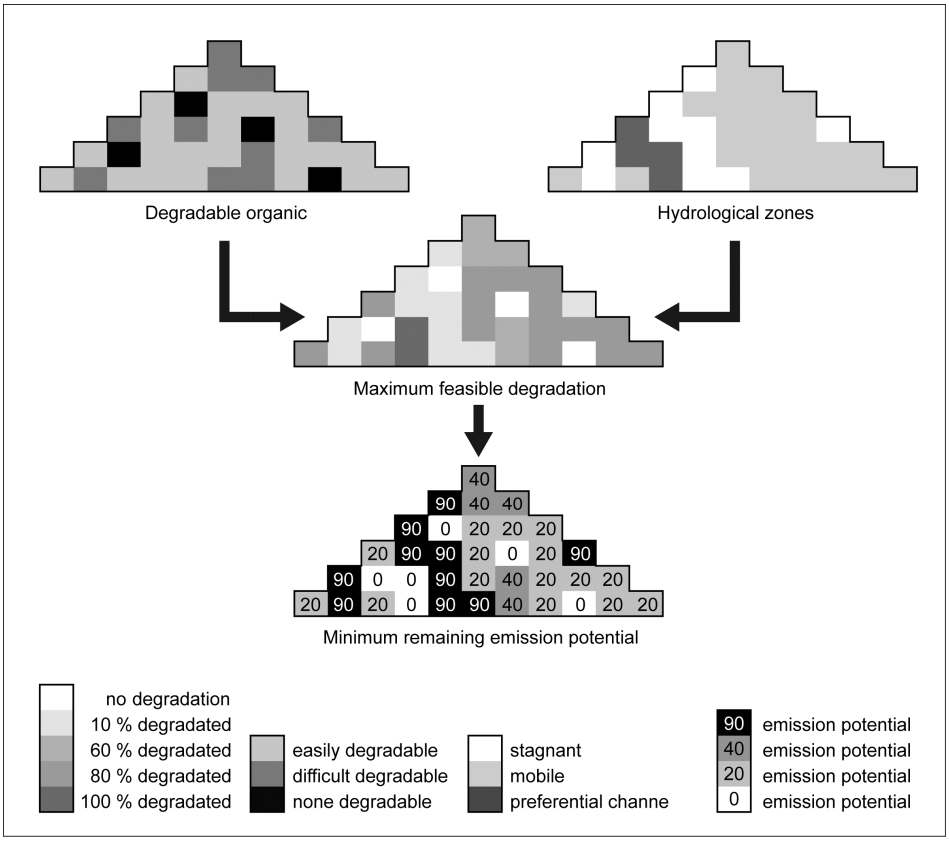


Figure 5-8: Conceptual illustration of the distribution of degradable organic matter and of water flow regimes within a MSW landfill with respect to the effect on the remaining emission potential (Source: van Vossen 2010)

The discussion about a change of landfill emission behavior illustrates the speculative nature of such scenarios, as they are either associated with an extreme, specific change of conditions at the site (e.g. flooding, human intrusion) or refer to very long time periods (e.g. release of heavy metals due to aeration or mineral dissolution). In general, scenario layouts will differ from one site to another. The range of scenarios considered at a specific site depends on the modeling timeframe of the analysis and assumptions on potential future conditions at the site. Relevant aspects are the location of the site with respect to nearby rivers (and potentially flood protection measures), the planned after-use at the landfill, and the danger of invasive activities at the site.

5.2.2.3 Scenario selection

The choice and layout of emission scenarios is based on potential conditions at the site and the purpose of scenario analysis.

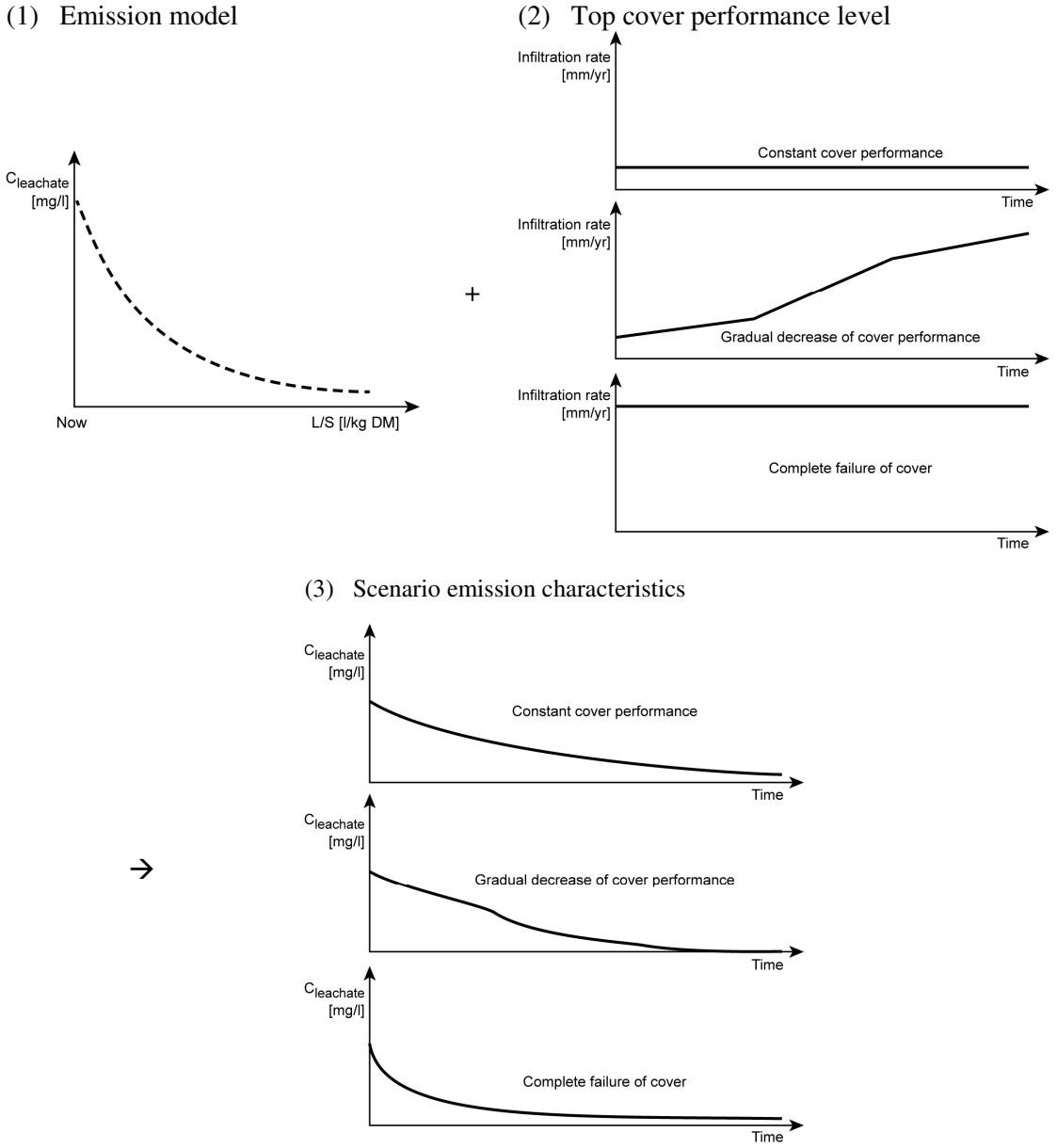


Figure 5-9: Composition of emission scenarios to investigate different top cover performance levels assuming stable conditions for the emission model

Although the set of scenarios might differ from one landfill to another, the persistence of the "status quo" will typically be among the investigated scenarios, as this scenario reflects the final state of the landfill during post-closure care. The scenario can be used to evaluate the current environmental impact of the landfill and, with respect to long-term management, represents a scenario of perpetual care at the site. Additional scenarios based on an emission model assuming stable conditions within the landfill (i.e. anaerobic conditions and no change of water flow pattern) could address the effect of decreasing barrier efficiency on emission levels. If the end of aftercare is associated with the end of maintenance and control of the containment system, deterioration of the

containment system over time is expected. Consequently, an evaluation of the associated emissions will be necessary to analyze long-term environmental risks. A schematic illustration of different elements and their combination within scenarios to investigate the effect of different top cover performance levels is given in Figure 5-9. Although Figure 5-9 refers to scenarios on containment system performance, the combination of the basic elements (a) an emission model and b) an estimate on the level of interaction between the landfill and the surrounding environment) would be similar for any other emission scenario.

In the case of long-term landfill management, different scenarios are established to support decisions on aftercare and to evaluate the associated environmental risk. As thousands of years are hardly a manageable time period, scenario modeling will typically extend over centuries. However, for scenarios investigating the effect of a sudden, extreme change of conditions on emissions the time of occurrence is not of primary importance anyway, provided that the remaining emission potential is conserved within the landfill after final closure (cf. Allen 2001; Laner et al. 2011a). Consequently, whether to include such scenarios in the analysis or not, depends on the probability of their occurrence and not on the time when they occur.

5.3 Pollutant migration modeling and ambient concentrations

The emission scenarios established for a specific landfill form the input for pollutant migration modeling. In general, several pollutant pathways might be relevant within an evaluation of landfill environmental compatibility. If direct exposure to the deposited waste is excluded (i.e. a top cover has been installed at the site), pollutants migrate as constituents of landfill gas or contaminated waters. Landfill gas can migrate through the top cover and in the subsurface. Surface runoff waters might be contaminated if erosion has led to the exposure of deposited waste. The direct discharge of collected leachate may affect water quality in receiving surface waters and leachate may percolate through the landfill base and release pollutants to the subsurface below the landfill.

The migration of pollutants in the landfill environment has been discussed in chapter 4.2 with the focus on the migration of leachate constituents in the subsurface. In this section, only the migration of leachate in the subsurface will be addressed as a part of the evaluation methodology. Nevertheless, it should be noted, that each of the pathways mentioned above could represent an environmental hazard.

5.3.1 Pollutant migration model in the subsurface

5.3.1.1 Overview

Leachate seeping from a landfill enters the zone below the landfill and potentially contaminates the groundwater underneath. Different processes occurring during the passage of these environmental media might, to a certain degree, mitigate negative effects of a release of leachate. Although, it has been shown that natural attenuation of many pollutants present in landfill leachate can be substantial, there is no easy-to-use protocol available on how to assess attenuation potentials for a specific environment (Christensen et al. 2000).

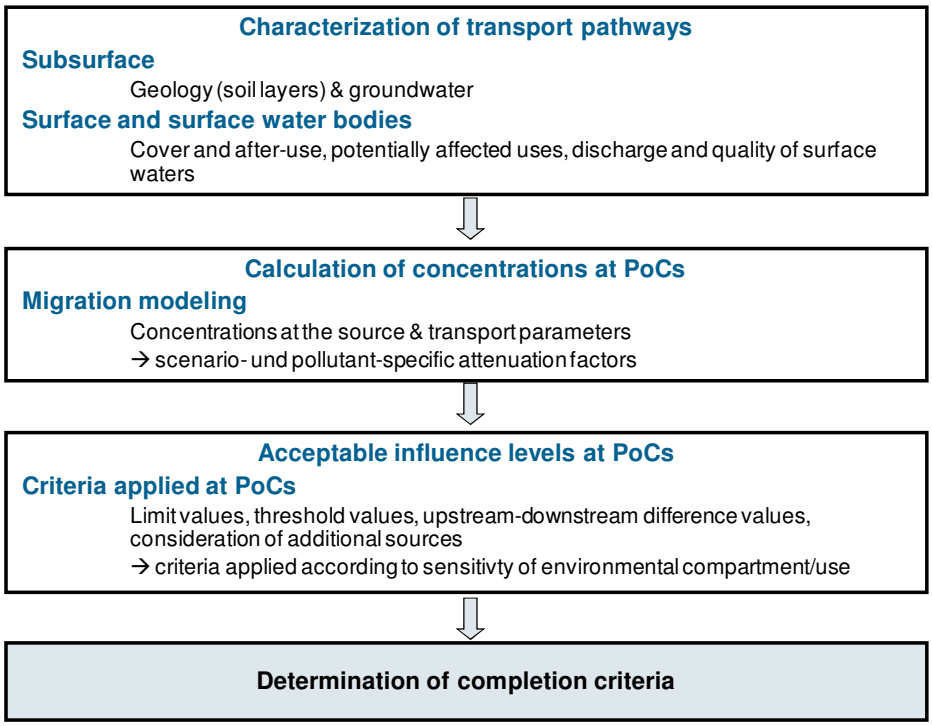


Figure 5-10: Schematic procedure to evaluate pollutant migration in the subsurface due to the release of leachate from the landfill

The procedure of evaluating the migration of leachate constituents in the subsurface is illustrated in Figure 5-10. The landfill-environment system is defined based on site characteristics (containment system, geology, hydrogeology, climate, etc.). The properties of the environmental system surrounding the landfill are outlined and points of compliance are defined for the evaluation. The characterization of transport pathways (cf. Figure 5-10 and Table 4-15) in the subsurface will typically comprise information about

- the low permeability soil layer at the landfill base (e.g. bulk density, grain size distribution, hydraulic conductivity, etc.),
- the soil layers in the vadose zone below the landfill (e.g. bulk density, grain size distribution, content of organic matter, hydraulic conductivity, field capacity, etc.),
- the position of the groundwater surface, the groundwater flow direction and velocity, the quality of the groundwater,
- the properties of the aquifer (e.g. hydraulic conductivity, thickness, etc.),
- and potentially affected surface waters (e.g. water quality, habitats, water discharge, etc.).

The data requirements of the transport model need to take into account the amount and quality of data available for the site. For simple, empirically-based models the data available from geological surveys before and during landfill construction and from existing monitoring sites may be sufficient to set up a model. Whereas for the application of more complex, spatially discretized models, it will be typically necessary to gather additional information about the site based on field- (e.g. groundwater pumping testes, additional monitoring wells, drillings, etc.) and lab-scale (e.g. cation-exchange-capacity tests, hydraulic conductivity tests, etc.) investigations.

After an appropriate conceptual model of the site has been built and a model to estimate pollutant migration has been established, pollutant concentrations can be calculated at the points of compliance. For each emission scenario (source concentration and release rates) and for each pollutant of potential concern (e.g. leachate constituents which have been observed above drinking water quality standards) the concentrations are determined. The attenuation factors are defined as the maximum concentration at the source divided by the maximum concentration at the respective point of compliance ($AF = C_{max,source}/C_{max,\ PoC}$). Hence, based on the transport modeling for each pollutant, scenario, and PoC, the attenuation factors can be calculated.

Finally, the maximum concentration at the PoC is compared to environmental quality standards (cf. Figure 5-10). The standards can refer to drinking water quality (e.g. Drinking water directive (MoE 2001), marginal effect levels (e.g. LAWA 2004), background concentrations, or any other accepted level of impact. In any case the standards should be assigned in consideration of the vulnerability of the environmental system and the potential use of these resources (e.g. groundwater as a source of drinking water). Based on the assigned standard, which has to be met at the PoC, and the attenuation factors calculated from the transport modeling, scenario- and pollutant-specific completion criteria for landfill aftercare can be derived.

5.3.1.2 Approach for migration modeling

To establish an appropriate reactive transport model for the unsaturated and saturated zone below the landfill, substance- and site-specific data need to be gathered. The complexity of the model should reflect the quantity and quality of data, which are available for establishing and applying the model. The more one needs to rely on literature data instead of site-specific measurements, the simpler and more conservative the modeling approach should be. Hence, within the evaluation methodology rather simple and robust models are used to describe pollutant migration processes in the landfill surrounding. On the one hand, this is in accordance with the data generally available at closed landfill sites and on the other hand, the simpler model can be replaced by a more complex model in a higher tier evaluation, provided that additional data are collected at the site.

Modeling migration processes in the vadose zone

The pollutant migration in the vadose zone below the landfill is evaluated with the so called "AF model" (Schneider and Stöfen, 2004), which is a one dimensional model based on the analytical solution of the advection-dispersion-equation (cf. equation 4-3). The model assumes stationary, one-dimensional downward flow and no pre-contamination of the transport pathway. Processes which can be considered within the migration model are advection, dispersion and diffusion, linear sorption, and first order degradation. Based on the comparison of average pollutant travel times in the vadose zone and the decreasing/limited source term strength, an attenuation factor is calculated. The model can consider the effect of multiple soil layers, but instead of a concentration over time as an output, it produces an estimate on the maximum concentration above the groundwater surface. The AF model has been shown to be a robust and conservative approach to describe potential attenuation in the vadose zone based on comparative analysis with the more complex Hydrus-1D model (Schneider and Stöfen 2006). Within the evaluation methodology only longitudinal dispersion processes (i.e. the decrease in concentrations above the groundwater is due

to decreasing source term strength) are considered, as the model might be sensitive with respect to the inclusion of transversal dispersion processes.

The AF model has been designed for constant leachate generation rates. Therefore, for scenarios with varying leachate generation rates (e.g. gradual decrease in barrier performance), a mass balance approach is used in addition to the AF model. The mass balance calculations address the time when the maximum load is released to the subsurface. Note that with a decreasing source strength but increasing leachate generation rates, this is not necessarily the time of maximum concentrations at the source: Load(t) [g/yr]= $C_{leachate}$(t) [g/l] * Leachate generation(t) [l/yr]. The concentration in the leachate at the time of maximum load is consequently used as the concentration above the groundwater table within these calculations. The concentration is compared to the result of the AF model (with an averaged leachate generation over the emission period) and the more conservative estimate is used to quantify pollutant migration. A mass balance approach is also applied to describe the mixing of leachate with the groundwater and pollutant migration in the groundwater within the evaluation methodology.

As described above, the AF model builds on the analytical solution of the advection-dispersion-equation (see chapter 4.2, equation 4-3). As linear sorption of pollutants is assumed in the AF model, the advection-dispersion-equation can be written as

(5-2) $$R + \frac{\partial C}{\partial t} = D_s \cdot \frac{\partial^2 C}{\partial z^2} - v \cdot \frac{\partial C}{\partial z} - \lambda \cdot C$$

where R is the retardation factor [-], C is the concentration in the liquid phase [kg/m³], t is time [years], D_s is the longitudinal hydrodynamic dispersion coefficient [m²/year], z is the depth [m], v is the seepage velocity [m/year], and λ is the first order degradation coefficient [1/year].

Some of the expressions used in equation 5-2 are converted to the formulations in equations 5-3 to 5-5. Whereby q_s is the leachate release rate [m/year], θ_w is the water content [m³/m³], ρ_d is the dry bulk density of the soil [kg/m³], k_d is the distribution coefficient between liquid and solid phase [m³/kg], α_L is the longitudinal dispersivity [m], D_w is the diffusion coefficient in water [m²/year], τ_w is the tortuosity for soil water [-], H is Henry's constant [-], D_g is the diffusion coefficient in gas [m²/year], θ_g is the air content [m³/m³], and τ_g is the tortuosity for soil air [-]. For some of these parameters empirical relationships have been established to allow for their estimation from other soil properties (cf. Schneider and Stöfen 2004).

(5-3) $$v = \frac{q}{\theta_w}$$ (5-4) $$R = 1 + \frac{\rho_d \cdot k_d}{\theta_w}$$ (5-5) $$D_s = \alpha_L \cdot v + D_w \cdot \theta_w \cdot \tau_w + \frac{H}{\theta_w} \cdot D_g \cdot \theta_g \cdot \tau_g$$

Based on the equations above (5-2 to 5-5) an estimate on the average travel time of a pollutant through the vadose zone can be derived. Main input data requirements to establish and run the AF model are briefly illustrated below.

a) Transport pathway

The transport pathway in the vadose zone includes the low permeability soil layer of the landfill containment system (provided its existence) and the soil layers above the groundwater surface. For each soil layer data on its thickness, field capacity, k_d values of the pollutants, and dry bulk density are required. Site-specific data is preferred over literature data or empirical relationships to estimate soil properties based on other soil characteristics (e.g. pedotransfer functions (ptf)). However, if there is no directly measured data, estimates based on soil classification (i.e. grain size distribution) may be used. Provided the grain size distribution of subsurface layers is known, the guidance on soil mapping and characterization (KA5) provides some empirically derived relationships to determine specific soil properties (AG Boden 2005). Figure 5-11 shows the basic soil classification diagram of the guidance and Table 5-5 provides estimates on field and air capacities of different soil types with dry bulk densities of 1.5 Mg/m³.

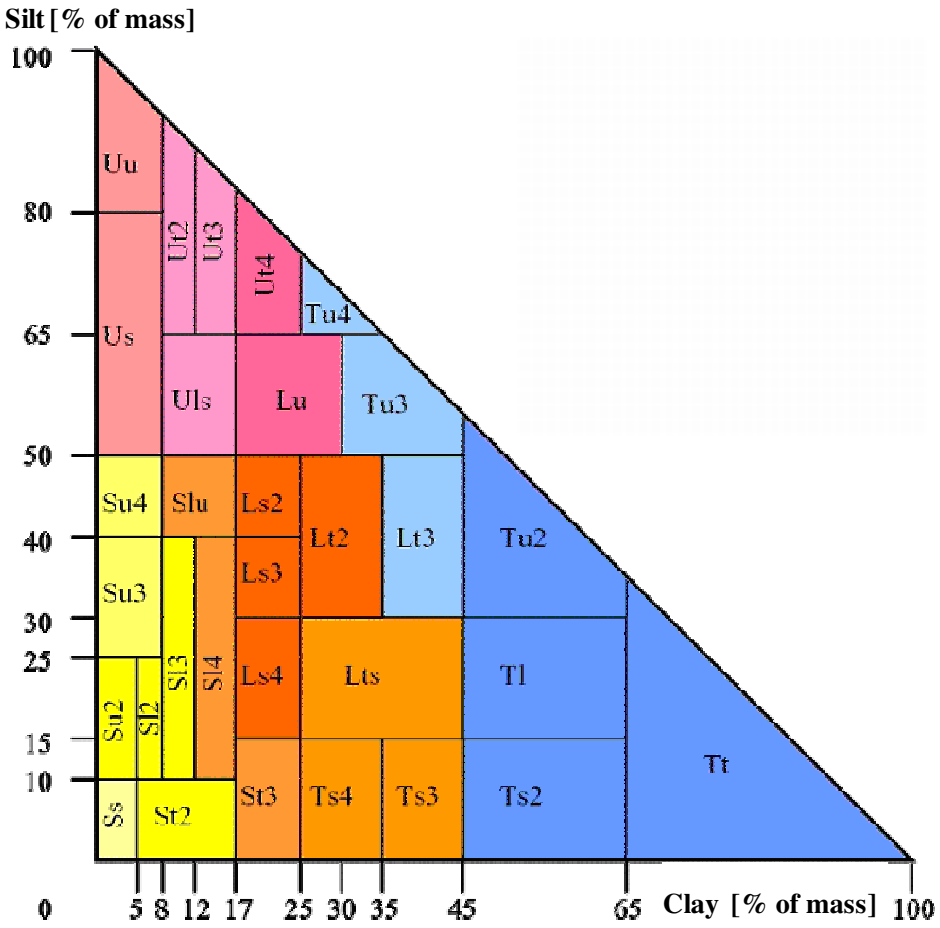


Figure 5-11: Soil type classification according to KA5 based on grain size distribution (Source: AG Boden 2005)

Table 5-5: Empirical estimates on the field and air capacity of different soil types (see Figure 5-11) according to the soil classification KA 5 (AG Boden 2005)

Soil type	Dry density [Mg/m³]	Field capacity* [% vol]	Air capacity* [% vol]	Soil type	Dry density [Mg/m³]	Field capacity* [% vol]	Air capacity* [% vol]
Ss	1.5	11	32	Uls	1.5	35	8
Sl2	1.5	25	18	Us	1.5	35	9
Sl3	1.5	27	15	Ut2	1.5	37	6
Sl4	1.5	30	12	Ut3	1.5	37	6
Slu	1.5	33	10	Ut4	1.5	37	7
St2	1.5	22	20	Tt	1.5	43	3
St3	1.5	30	14	Tl	1.5	41	4
Su2	1.5	23	21	Tu2	1.5	42	4
Su3	1.5	29	14	Tu3	1.5	38	6
Su4	1.5	32	11	Tu4	1.5	37	6
Ls2	1.5	34	9	Ts2	1.5	39	4
Ls3	1.5	33	9	Ts3	1.5	37	6
Ls4	1.5	32	11	Ts4	1.5	32	10
Lt2	1.5	36	7	Sand			
Lt3	1.5	39	5				
Lts	1.5	37	6	fS, fSms, fSgs	1.5	14	31
Lu	1.5	36	7	mS, mSfs, mSgs	1.5	10	32
Uu	1.5	38	7	gS	1.5	8	33

*Estimated values are based on the dry density given in the table.

Correction factors can be used to consider coarse soil fractions and the organic content of the (Table 15 and Table 72 of KA55)

b) Leachate constituents and specific soil parameters

The pollutants of concern in the landfill leachate may differ from one site to another. Knowledge about the migration behavior in the subsurface is still scarce for many landfill pollutants (Luckner et al. 2008), especially because pollutants are mixed with other substances in the leachate mutually influencing each others migration behavior. Hence, data should be gathered based on site-specific investigations with local soil types and representative leachates. If such data are not available, literature data reported for comparable conditions can be used. Some reported values for substance specific parameters are shown in Table 5-6 for selected leachate parameters. The sorption of ammonium to the soil matrix should be only considered in low-permeability soils with some clay content. For low-permeability soil liners a k_d value of 0.5 is regarded as a conservative estimate (cf. Buss et al. 2004). The degradation of pollutants is primarily relevant for well degradable organic compounds (e.g. petroleum hydrocarbons) and can be considered within the AF model at a first order rate. Again, half lives should be determined specifically for the conditions at the site, but are also available in various databases on subsurface pollutant degradation (e.g. CalTOX), which can be used to provide a first estimate.

Table 5-6: Examples of substance-specific transport parameters for selected leachate constituents

Parameter	Linear sorption k_d [l/kg]	Reference soil	Diffusion in water [2] D_w [m²/yr]
Ammonium[1]	0 – 5 (0.5)	Low permeability soil liner	0.0378
Chloride	0	-	0.0315
Sulfate	0	-	0.0189
Organic pollutants	Parameters depend strongly on the type of organic substances and the content of organic matter in the soil ($K_d = C_{org} * K_{OC}$))		
Heavy metals	Varies for different metals and soils. Approaches to estimate metal sorption in different soils are described in Utermann et al. 2005.		

Source: [1] Buss et al. 2004; [2] Furrer and Wehrli 1996

c) Seepage velocity and hydrodynamic dispersion

The seepage velocity is calculated from the leachate release rate (emission scenario output) and the volumetric water content of the soil layer (cf. equation 5-3). Within the model calculations the field capacity (=the volumetric water content a porous media can hold against gravitation) is used for the determination of the seepage velocity.

The formula to calculate longitudinal hydrodynamic dispersion is shown in equation 5-5. In case of non-volatile substances longitudinal dispersion causes a decrease of the maximum pollutant concentrations if source strength is decreasing with time or finite. The decrease of maximum concentration is dependent on the average travel time of the pollutant in relation to the emission strength of the source term. Hydrodynamic dispersion is strongly dependent on the scale (see chapter 4.2). In the AF model longitudinal dispersivity is by default set to one tenth of the travel distance.

d) Source term behavior

Pollutant migration modeling builds on the outcomes of the emission scenarios. Hence, the source term is defined by the results of the emission scenarios during the previous step of the evaluation method. The AF model distinguishes two basic cases of source term strength: a) constant source term strength (pollutant concentration) for a limited time period, $C(t)= C_0$ for $t < t_{source}$ (cf. Figure 5-12); b) exponential decrease of source strength, $C(t)= C_0 \exp(-k_s.t)$ (cf. Figure 5-13). The emission levels estimated within the scenarios are approximated by one of these basic cases and the emission period is determined. For exponentially decreasing concentrations, the emission period is calculated as the inverse of the first order rate of decrease ($=k_s$), $t_{source}= 1/k_s$.

The basic cases of source term behavior within the AF model assume constant leachate release rates. However, for some emission scenarios this assumption will not be appropriate (e.g. decreasing barrier performance). Although an average leachate release rate could be used for the emission time period, it is hard to approximate the varying rate of concentration decrease at the source within such an approach. Therefore, it is suggested to estimate maximum concentrations at the groundwater surface also based on a mass balance approach for the time the maximum load is released from the landfill. Within the latter approach, no decrease of the pollutant concentrations in the leachate at the time of maximum load is considered while it is traveling through the vadose zone. Hence, the leachate concentration at the source at the time of maximum load is the same as

the leachate concentration arriving at the groundwater surface. For reasons of conservativeness, the approach (AF model or mass balance at the time of maximum emission load) which results in higher maximum concentrations at the groundwater surface will be used to evaluate pollutant migration.

e) Determination of attenuation factors

The attenuation factor (AF) is the quotient of the maximum concentration at the source (C_0) and the maximum concentration at the bottom of the vadose zone ($C_{max,vz}$): $AF = C_0/C_{max,vz}$. AF is calculated based on the pollutant travel time in the vadose zone and the lifetime of the source (t_{source}). The AF increases with increasing travel time and decreasing lifetime of the source, respectively. To determine the relationship between travel time and source lifetime so called "nomograms" have been developed (Schneider and Stöfen 2004). The nomograms (see Figure 5-12 and Figure 5-13) are based on analytical solutions of the advection-dispersion equation and can be used to visually determine the AF for a given pollutant travel time (see equation 5-6) and a specific lifetime of the source. The effective dispersivity has been set to 10 % of the transport distance within the calculations. However, different dispersivities could be considered via a correction factor to calculate the equivalent travel time (see equation 5-7). The same correction factor can be used to consider first order pollutant degradation in the model. However, if degradation is considered, effective dispersivity is fixed at the value of 0.1 times the travel distance. For a more elaborate discussion on correction factors and the analytical solutions see Schneider and Stöfen (2004).

(5-6)
$$t_t = \frac{\sum_{i=1}^{N} M^i \cdot \theta^i{}_{fieldc} \cdot R^i}{q}$$

The formula to determine the average travel time of a pollutant in the vadose zone is given in equation 5-6. Whereby t_t is the average travel time of the pollutant [years], N is the number of soil layers [-], M^i is the thickness of soil layer i [m], $\theta^i{}_{fieldc}$ is the water content of layer i at field capacity [m³/m³], R^i is the retardation factor of soil layer i [-], and q is the leachate release rate. The equivalent travel (ta_{ae}) time is calculated after equation 5-7 to potentially consider a correction factor (t_c) due to different longitudinal dispersivity or pollutant degradation processes together with the calculated average pollutant travel time (t_t).

(5-7)
$$\log t_{ae} = \log t_t + \log t_c$$

The nomograms in Figure 5-12 (basic case: limited constant source strength) and Figure 5-13 (basic case: exponential decrease of source strength) can be used to determine the attenuation factor after the equivalent travel time and the lifetime of the source have been determined. In case of an AF above 10, the formulas in equation 5-8 can be used to calculate the AF directly without the aid of nomograms.

(5-8)
$$AF = \frac{t_{ae}}{t_{source}} \quad \text{or} \quad AF = \frac{t_{ae}}{1/k_s}$$

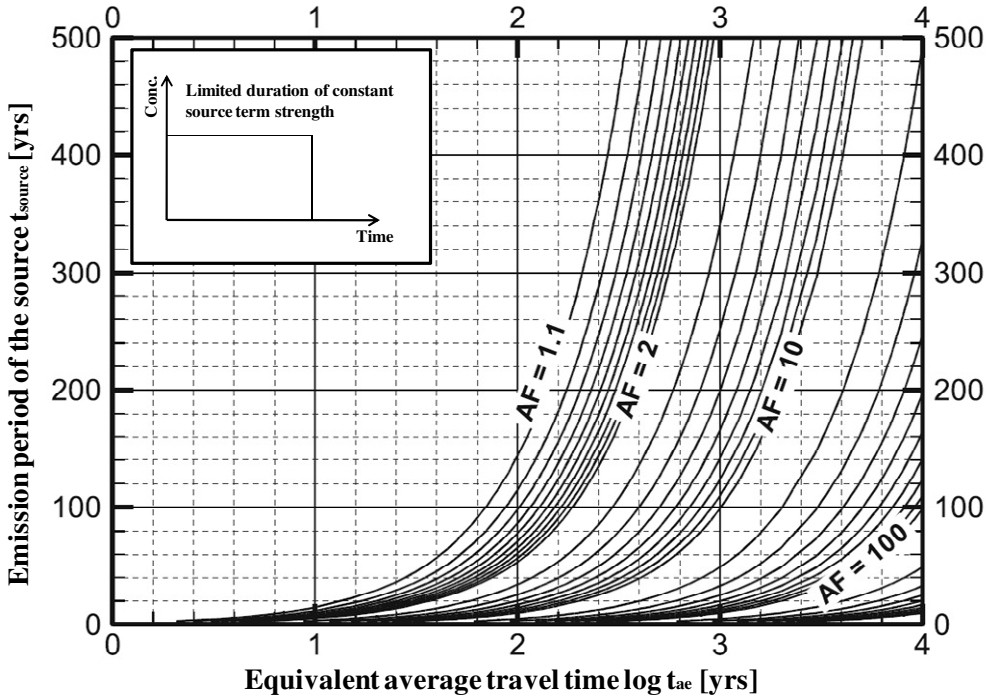


Figure 5-12: Nomogram to determine the AF value for the case of constant source term strength for a limited emission period (Source: Schneider and Stöfen 2004)

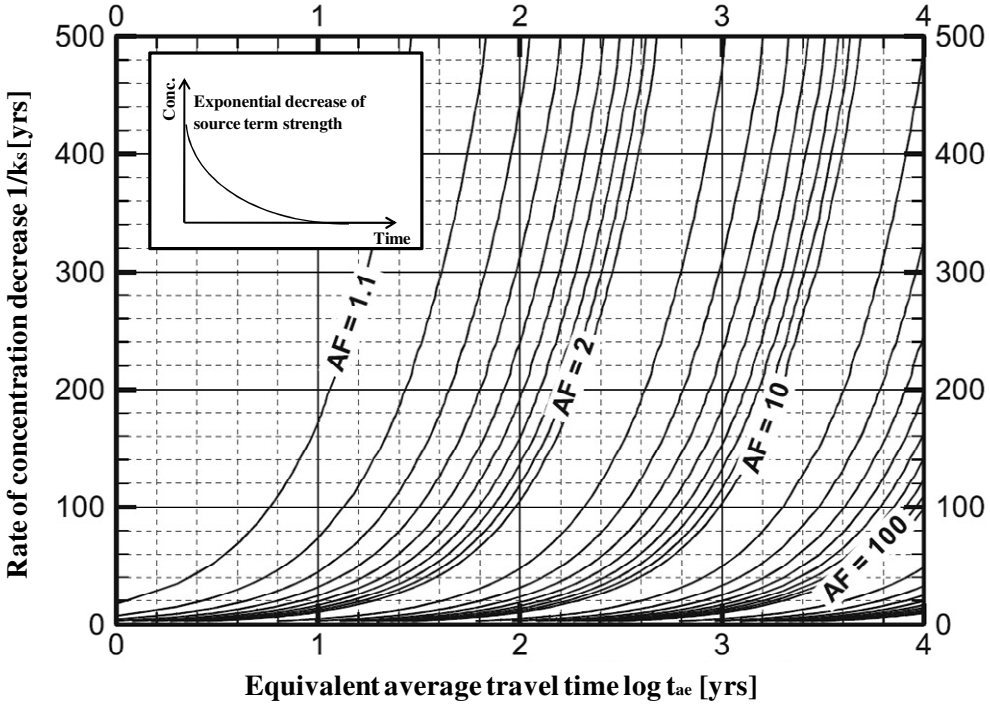


Figure 5-13: Nomogram to determine the AF value for the case of exponentially decreasing source term strength (Source: Schneider and Stöfen 2004)

Pollutant migration in the groundwater

The mixing of leachate and groundwater causes a decrease of pollutant concentrations (i.e. provided that background groundwater levels are below the levels in the leachate). Sophisticated models can be applied to describe pollutant migration in groundwater systems, but within the

evaluation methodology a simple and conservative model based on mass balance considerations is suggested. The application of the model is outlined below.

The resulting concentrations above the groundwater surface from the vadose zone model form the input for the model calculations. The presented approach addresses leachate induced concentrations and does not consider background concentration levels in the groundwater. However, for the determination of total pollutant concentrations at a specific point of compliance, background levels or other sources have to be taken into account. The attenuation factor due to the mixing of leachate (AF_{GW}) and groundwater is calculated from the leachate discharge rate into the groundwater (Q_L) in m³/year and the groundwater flow in the mixing zone in m³/year (thickness m_{aq} in Figure 5-14) according to equation 5-9. The underlying calculations to derive Q_L and the groundwater discharge rate downstream of the landfill (Q_D in m³/year) are shown in equation 5-10. Whereby Q_U is the groundwater discharge rate in the mixing zone upstream of the landfill [m³/year], k_f is the hydraulic conductivity [m/year], W_c is the contaminated width perpendicular to the groundwater flow direction [m], m_{aq} is the thickness of the mixing zone [m], I is the hydraulic gradient [m/m], q is the leachate release rate [m/year], and L_c is the contaminated length in groundwater flow direction [m].

(5-9) $$AF_{GW} = \frac{Q_L}{Q_D}$$

(5-10) a) $Q_D = Q_U + Q_L$ b) $Q_U = k_f \cdot W_c \cdot m_{aq} \cdot I$ c) $Q_L = q \cdot W_c \cdot L_c$

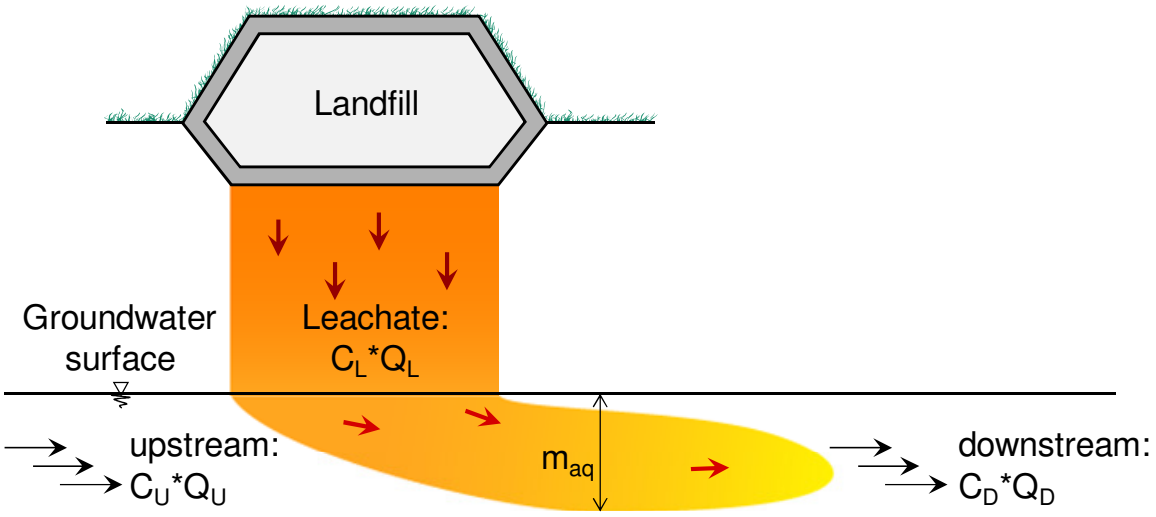


Figure 5-14: Mass balance approach to describe mixing of leachate and groundwater

In the groundwater further processes can cause pollutant concentrations to decrease. In order to account for hydrodynamic dispersion and other more complex transport phenomena (e.g. degradation, sorption), the advection-dispersion equation can be used to model pollutant transport in the groundwater. However, as few data are generally available to establish more complex models, the presented approach does not specifically address these processes. A decrease of concentrations in the model occurs primarily due to the dilution of pollutants in the groundwater. In addition, the

simple approach represents a robust framework for reverse calculations to determine maximum acceptable concentrations at the source based on the attenuation factors.

5.3.1.3 Discussion

In general, the complexity of a modeling approach should reflect the amount and quality of data available to establish and calibrate a model. As available data at closed MSW landfills do typically not allow for sophisticated models to be adapted to the site in a meaningful way (without additional site investigations), simple approaches are suggested to describe pollutant migration in the unsaturated and saturated zone below the landfill. The AF model (Schneider and Stöfen 2004) is used to estimate pollutant transport in the vadose zone (complemented with a mass balance approach for emission scenarios with varying leachate release rates) and a robust mass balance approach is used to estimate pollutant spreading in the groundwater.

Although the limitations of these approaches were discussed before, some main aspects are highlighted below:

- Only linear distribution coefficients (k_d) can be considered within the AF model. They assume unlimited sorption capacity of the soil, which is unrealistic in general, but especially if high pollutant concentrations are encountered in the leachate. Sorption isotherms following a different behavior can be linearized (cf. Utermann et al. 2005), but should be cautiously handled in any case. Ideally, k_d values are based on specific experiments reflecting the actual conditions at the site (e.g. leachate composition and soil materials). For ammonium some suggestions for appropriate k_d values were presented by Buss et al. (2004).
- Homogeneous flow is assumed in the unsaturated zone in the AF model. Nevertheless, preferential flow could be considered by lowering the volumetric water content available for pollutant transport processes.
- Degradation of pollutants can be modeled only with a first order rate in the AF model together with a dispersivity of 0.1 times the transport distance. To describe these processes in different ways or combinations, more complex analytical or numerical need to be used.
- Varying leachate release rates can only be approximated (average rate over emission period) in the AF model. Therefore, a mass balance approach to complement the AF model is suggested to deal with such source terms.
- The flux of pollutants is assumed to be constant throughout the whole area of the landfill base. This assumption underestimates the seepage velocity in the vadose zone as the heterogeneity of the release pattern is neglected. If degradation and sorption processes are not considered for the transport evaluation, this does not have any effect on the load of pollutants reaching the groundwater. However, the travel time would be shorter due to the higher seepage velocity. Irregular release of leachate throughout the base could be considered by assuming a smaller contaminated area below the landfill than the area of the landfill base. Nevertheless, estimates on heterogeneous release patterns will be highly speculative, but probably closer to the real situation (e.g. base liners with a geomembrane layer will mainly release leachate at hot spots like holes or flaws).

In addition to the factors listed above, the input data and parameters are associated with uncertainties. Reasons are the natural variability of site characteristics, incomplete knowledge of current and future conditions, as well as wrong or inadequate measurements. To account for the effect of different parameters/inputs on the results of the modeling, the sensitivity of outcomes needs to be assessed. Monte Carlo analysis can be used to evaluate the robustness of the results with respect to the variation of input and parameter values.

5.4 Determination of completion criteria

In general, completion criteria are determined based on a critical point of compliance, tolerable concentration levels at this point, the choice of the most probable or most relevant long-term emission scenario, and the calculated attenuation factors for substances released to the surrounding environment during this scenario. With respect to landfill leachate site-specific completion criteria can be determined based on the combination of the models presented in this chapter. The critical point of compliance (PoC) is chosen in consideration of the vulnerability of different environmental media and the current, planned, and potential uses at the site, respectively. A quality standard is applied at the PoC to define the maximum acceptable level of influence due to a release of leachate from the landfill. A reverse calculation, based on the pollutant- and scenario-specific attenuation factors determined during the pollutant migration modeling, is subsequently used to define tolerable emission levels at the source (i.e. in the landfill leachate). Hence, from specifying maximum acceptable impacts on groundwater quality at a certain point corresponding leachate quality criteria for a specific emission scenario can be derived. For instance, a quality standard of 10 mg Cl/l at PoC3 (defined as the critical point of compliance) in Figure 5-15 would result in a tolerable concentration of Cl in the landfill leachate of 200 mg/l (= 10*2*2*5). Thus, if the Cl concentration in the leachate decreases to and remains below 200 mg/l, future leachate emissions will be environmentally tolerable, provided that the conditions of the underlying emission scenario are met. In this way, aftercare completion criteria for leachate are determined for a landfill based on a defined long-term emission scenario. As completion criteria are associated with an expected future behavior of the landfill and specific conditions at the site (i.e. conditions within the emission scenario), criteria relating to the underlying long-term emission scenario are to be included. For instance, as the emission model used within the emission scenario must be validated, confirmation monitoring to verify model assumptions will be required as a completion criterion. This could be similar to the monitoring and action matrices for natural attenuation at closed landfill sites, which have been suggested as an element of corresponding remediation strategies (cf. Luckner et al. 2008). Another example would be the assumption of an undisturbed waste body in the emission scenario, which would imply criteria to prevent invasive activities at the site. The assumption of optimal containment system performance in the underlying long-term emission scenario would include the maintenance and repair of the containment system as an aftercare completion criterion. However, one might doubt if authorities accepted the latter as a de-minimus care actvitity.

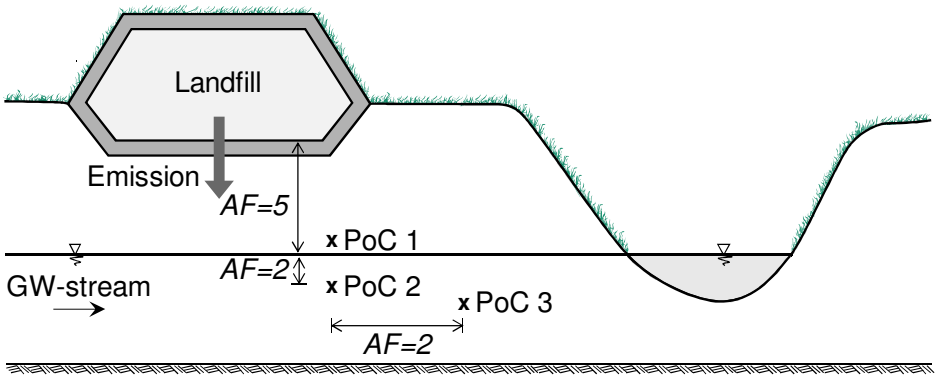


Figure 5-15: Scheme of the landfill-environment system with scenario- and substance-specific attenuation factors between the different PoCs

Aftercare completion criteria for landfill gas can be based on a minimum of cumulative gas generation demonstrated at the site complemented with target values for maximum gas emission rates specified in technical guidances (e.g. Fellner et al. 2008) or regulations (e.g. Wisconsin Department of Natural Resources 2007). Attenuation of methane concentrations in the landfill cover may be considered if it can be demonstrated that methane entering the cover is oxidized during cover passage. Criteria addressing subsurface migration of landfill gas and associated risks (e.g. explosion hazards in buildings) have not been included in the evaluation methodology. However, if there is a potential for gas migration at the site, landfill gas risk assessment to derive acceptable emission levels and surveillance monitoring in the landfill surrounding are necessary to evaluate the end of post-closure care.

In summary, completion criteria addressing landfill emissions are typically complemented by criteria associated with the emission scenario underlying the evaluation. Such additional criteria may comprise geotechnical aspects, the functionality of landfill elements (e.g. containment system), surveillance and maintenance activities at the site, restrictions of use, etc. The only emission scenario with no additional criteria (to completion criteria addressing emissions), would be one where the tolerable emission levels relate to a scenario of waste without any containment and fully exposed to external conditions at the site (e.g. precipitation). For all the other emission scenarios some additional completion criteria will be necessary to assure environmental compatibility of the closed landfill. Because the procedure of deriving completion criteria requires emission scenario conditions to be addressed as additional completion criteria, it is expected that the basic long-term management concept will be better understood by decision makers and that a consequent evaluation will be comprehensible to them.

Emission scenarios investigated but not used as a basis for completion criteria, provide a basis to evaluate residual environmental risks at the site. For instance, the potential effects of a complete failure of the containment system or the effects of landfill flooding on the environmental system can be evaluated if such scenarios have been included in the analysis. If the emission scenario "complete failure of the containment" is used as a basis to derive completion criteria, residual risks might be low, whereas if the emission scenario "constant containment level" is used to derive completion criteria, residual risks might be higher. In any case, the decision on the completion of aftercare will be one about acceptable risks, as long as there is some hazard associated with the deposited waste.

6 Case studies

To test the developed methodology with respect to applicability and practicality, it is applied at three landfills. Aftercare completion criteria are determined at each site and consequent aftercare periods are discussed. The selection of the case study landfills is based mainly on three criteria:

a) Sufficient quality and availability of data: Emission monitoring (time series data on quantity and quality), waste deposition (time series data on composition and amount of deposited waste), landfill engineering (deposition techniques, containment system), information about local climate, geology, hydrogeology, etc.
b) Site characteristics: Waste type and emission potential, level of interaction between waste and environment, vulnerability of the local environment, etc.
c) Cooperative attitude of the landfill owner or the one responsible for aftercare: Access to archives and support with data collection, interest in the evaluation of the site and the proposed aftercare strategy, etc.

The selection of landfills with different waste types, engineering systems and environmental conditions is intended to highlight the importance of site-specific factors with respect to the evaluation of landfill environmental compatibility and to illustrate the applicability of the methodology to landfill sites with varying characteristics.

Due to the environmental significance and the comparatively large number of closed MSW landfills in Austria[17], two of the three case study sites are closed MSW landfills. Based on the information and contacts during data collection to establish typical landfill emission profiles (cf. chapter 3), a pre-selection of suitable sites was made based on the criteria listed above. After additional inquiries and site visits, a large, closed MSW landfill (closed in 2005 with a waste volume of 1.7 million m³) in the centre of Austria with temperate, continental climate (precipitation: 960 mm/yr, potential evaporation: 630 mm/yr) was chosen (Landfill A) together with a smaller MSW landfill (Landfill B) in the West of Austria (closed in 2004 with a waste volume of 135,000 m³) at a site with humid, continental climate (precipitation: 1960 mm/yr; potential evaporation: 550 mm/yr). The third case study site (Landfill C) is situated in the East of Austria (arid climate with 610 mm/yr precipitation and 630 mm/yr potential evaporation). It was closed in 2008, and contains around 400,000 m³ of construction and demolition (C&D) waste. A C&D waste landfill is chosen because of the quantitative significance of this landfill type and because of the different emission potential of the waste relative to MSW. A landfill containing municipal solid waste incineration residues has not been included as a case study site, as there is no mono landfill of this kind in Austria (data presented in chapter 3.2 are from Swiss landfills). However, the methodology could be applied in a similar way to such a landfill site.

[17] Since 2004 the landfilling of MSW without pre-treatment is prohibited in Austria. Therefore, many MSW landfills have been closed during the last decade and are now in the post-closure care period.

6.1 Landfill A - closed MSW landfill

6.1.1 Site description and monitoring data

Landfill and site conditions

Municipal solid waste (MSW) was deposited at Landfill A between 1972 and 2005. The landfill is split into eight compartments (C1-C8), which were successively filled with MSW. The oldest compartment was closed in 1975 and the youngest was operated from 1998 to 2005 (cf. Table 6-1). In total around 2 million tons of MSW have been disposed at the site, resulting in a landfill volume of 1.7 million m³. The whole landfill covers an area of 114,200 m² with average deposition heights of the compartments between 6 and 26 meters, compartment 1 and compartment 5, respectively (cf. Table 6-1). The changes in Austrian landfill regulations over time can be observed from the different designs of barrier systems at the top and bottom of the landfill compartments shown in Table 6-1. For instance, at the base of the first compartment a low permeability earthen barrier has been installed as a sole liner, whereas the last compartments have been equipped with a composite lining system at the base, consisting of a three-layer clay liner and a HDPE membrane of 2.5 mm thickness above it. Due to the limited effectiveness of the initially installed base lining systems, a slurry wall encapsulation has been built around the landfill in the beginning of the 1990s. The encapsulation is used to extract the contaminated groundwater below the landfill and directly discharge the water into an adjacent river (cf. Figure 6-1).

Table 6-1: Operation and design of the 8 compartments of Landfill A

Comp.	Operation	Area [m²]	Volume [m³]	Landfill base	Landfill cover
C 1	1972-1975	14000	84000	Low permeability earthen barrier (>0.5 m with k_f <10^{-8} m/s) + drainage layer	Asphalt liner
C 2	1975-1994	9700	146000		Low permeability earthen barrier (2 layers (each 0.25 m), k_f <10^{-9} m/s) + drainage layer + recultivation layer
C 3	1980-1994	13300	200000		
C 4	1986-1994	20000	300000		
C 5	1990-2000	13000	348000	Clay liner (3 layers (each 0.2 m), k_f <10^{-9} m/s) + drainage layer	Geosynthetic clay liner (k_f<10^{-9}m/s) + drainage layer + recultivation layer
C 6	1992-2000	11900	209000		
C 7	1998-2005	21800	249000	Clay liner (3 layers (each 0.2 m), k_f <10^{-9} m/s) + HDPE membrane + drainage layer	Geosynthetic clay liner (k_f<10^{-9}m/s) + HDPE membrane + drainage layer + recultivation layer
C 8	1994-2002	10500	219000		Geosynthetic clay liner (k_f<10^{-9}m/s) + drainage layer + recultivation layer

The average annual precipitation at the site is 956 mm/yr. The landfill is located next to a river, which flows along the embankment southeast of the landfill. Northeast of the landfill is a former gravel pit turned into a small lake (or pond) primarily fed from groundwater. The groundwater surface lies at least 2 meters below the minimum elevation of the landfill base. The thickness of the aquifer is between 12 and 15 meters. The geological strata below the landfill consist of glacial gravel terraces in the vadose and groundwater zone and marl as the local bedrock at the bottom of the aquifer. The hydraulic conductivity of the aquifer ranges from $1*10^{-5}$ to $5*10^{-3}$ m/s, but locally there may be silt lenses or conglomerates with lower hydraulic conductivities. The general groundwater flow direction is from West to East (cf. Figure 6-1).

There are no residential buildings in the neighborhood of the landfill, but the industrial site Southwest of the landfill includes several office buildings.

Figure 6-1: Aerial photograph of Landfill A and surroundings

Monitoring data

The leachate collection system consists of the gravel drainage layers at the landfill base, a network of leachate collection pipes to transport the leachate to underground storage basins outside the landfill body, and a pumping system to transport the leachate from the basins to a storage tank on the surface, from where the leachate is transferred to a treatment plant. The composition of the leachate is analyzed separately for each compartment twice a year (see Table 6-2). The leachate generation rate is calculated from the extracted amount of leachate from the different basins. As the compartments 1 to 4 drain into the same basin, their specific leachate generation rates are estimated based on the relative surface area of each compartment and the cover systems of each individual compartment (e.g. open system, temporary cover, and final cover). Overall, leachate generation rates strongly decreased after compartment closure. For the older compartments, around 3% of the annual precipitation is currently collected as leachate at the landfill base. For the younger compartments, the amount of leachate equals approximately 20% of the annual precipitation a few years after closure. However, according to the design parameters of the installed top cover systems, future leachate generation rates should be much lower, as the gravity drainage of water stored in the landfilled waste is expected to become insignificant.

From Table 6-2 it is visible that, after their closure, the younger landfill compartments (e.g. C7 or C8) produce leachate, which is still strongly polluted with organic compounds, nitrogen and soluble salts (see Table 6-2). Leachate collected from older compartments (e.g. C1 or C2) exhibits significantly lower concentrations of degradable organics (BOD), organic compounds in general (COD), nitrogen (especially ammonia), and easily soluble salts (e.g. chloride). Nevertheless, even the latter concentrations are still above the maximum concentration levels for direct leachate discharge into rivers (cf. Table 6-2).

The most problematic leachate parameters in view of the limit values for direct discharge are the organic compounds (COD, BOD, AOX), ammonia-nitrogen, chloride, phosphorus, and the metals Fe and Cr_{tot} (cf. Table 6-2). Based on the available data and the observable leachate quality trends,

the parameters COD, NH_4-N, Cl, and Cr_{tot} are chosen as parameters for emission modeling. However, it should be noted that a screening analysis of organic leachate pollution would be necessary to identify hazardous components potentially present in the leachate at low levels.

Table 6-2: Mean values of selected leachate parameters measured after waste deposition has been completed at the respective landfill compartment (bold: mean values above discharge limits and chloride)

Parameter	C1	C2	C3	C4	C5	C6	C7	C8	Limit*
pH value	7,86	7,78	7,92	7,85	8,24	7,84	7,87	8,00	6,5–8,5
EC [µS/cm]	5434	4216	13444	11739	26170	14514	18000	10003	-
NH_4-N [mg/l]	**150**	**146**	**786**	**681**	**2133**	**1072**	**996**	**634**	**10**
NO_2 [mg/l]:	**5,6**	**2,0**	**9,2**	**8,7**	**2,5**	**0,9**	**0,5**	**17,7**	**2,0**
NO_3 [mg/l]:	89,9	45,4	9,9	17,8	20,0	12,9	<5	272,0	-
Fluorides [mg F/l]:	0,4	0,4	1,0	0,9	1,3	0,9	0,9	0,7	10
Cl [mg/l]:	**495**	**478**	**1625**	**1573**	**4037**	**1713**	**2020**	**1125**	**Toxicity**
$Cyanide_{ges}$ [mg/l]:	0,02	0,03	0,01	0,00	0,22	0,07	-	<0,05	-
SO_4 [mg/l]:	646	71	87	136	1011	339	87	380	-
P [mg/l]:	1,7	0,8	**5,9**	**5,6**	**11,5**	**7,1**	**7,7**	**3,7**	2,0
COD [mg O_2/l]:	**633**	**352**	**2437**	**2318**	**7680**	**3922**	**4470**	**1217**	**50**
BOD_5 [mg O_2/l]:	**82**	**80**	**251**	**286**	**902**	**744**	**560**	**145**	**10**
Hydrocarbons [mg/l]	0,04	0,03	0,11	0,62	0,14	0,21	0,20	0,17	5,0
BTEX [mg/l]:	0,01	0,02	0,05	0,02	0,04	0,05	0,00	0,01	0,1
AOX [mg Cl/l]:	0,25	0,31	**1,37**	**1,06**	**7,48**	**0,93**	**0,95**	0,45	**0,5**
Fe [mg/l]:	**3,04**	1,99	**4,84**	**4,95**	**2,22**	**3,17**	-	1,40	**2,0**
Mn [mg/l]:	0,12	0,31	0,21	0,31	0,67	0,52	-	0,30	-
Zn [mg/l]:	0,46	0,08	0,27	0,18	0,13	0,14	0,06	0,11	0,5
Ni [mg/l]:	0,08	0,02	0,16	0,15	0,39	0,17	0,07	0,12	0,5
Co [mg/l]:	0,01	<0,05	<0,05	<0,05	0,01	<0,05	-	<0,05	-
Cu [mg/l]:	0,03	<0,05	0,02	0,02	0,03	0,04	<0,05	0,01	0,5
Cd [mg/l]:	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01	<0,01	0,1
Sn [mg/l]:	0,02	0,01	0,15	0,19	0,10	0,05	0,00	0,00	2,0
Pb [mg/l]:	<0,05	<0,05	<0,05	<0,05	<0,05	<0,05	0,01	<0,05	0,51
Cr_{tot} [mg Cr/l]:	0,15	0,09	**0,64**	**0,60**	**2,50**	**0,86**	0,47	0,24	**0,5**
Al [mg/l]:	0,35	0,21	0,53	0,54	1,40	1,13	-	<0,2	2,0
Ag [mg/l]:	<0,02	<0,02	<0,02	<0,02	<0,02	<0,02	-	<0,02	0,1
As [mg/l]:	0,01	0,00	0,03	0,03	0,13	0,04	<0,01	0,02	0,1
Hg [mg/l]:	0,002	<0,001	<0,001	<0,001	0,0002	<0,001	<0,001	<0,001	0,01

*Limit values for direct discharge into rivers according to the Austrian directive on leachate discharge (MoE 2003) and the Austrian directive on waste water discharge (MoE 1996), respectively.

In contrast to emission data on leachate composition and amount, data on landfill gas emissions are available only to a limited degree. The gas collection system at the landfill was installed in 1994 with an extraction system for all landfill compartments. The amount and composition (measured continuously since 2000 for methane, carbon dioxide, and oxygen) of the captured landfill gas are shown in Figure 6-2. Based on the collected amount of landfill gas, the specific gas generation rates are 1.4 m³ per ton of dry matter and year directly after final closure of the landfill. However, because of the restricted gas collection efficiency in general and the belated installation of the gas collection system at the site, it can be assumed that the actual gas generation rate is higher than the one calculated from the collected amount of landfill gas (cf. Krümpelbeck 2000).

Methane is chosen as the parameter of interest for landfill gas emission modeling, as it is of major environmental significance (globally: 25-fold global warming potential of CO_2, locally: explosion hazards) and typically present at high concentrations in landfill gas.

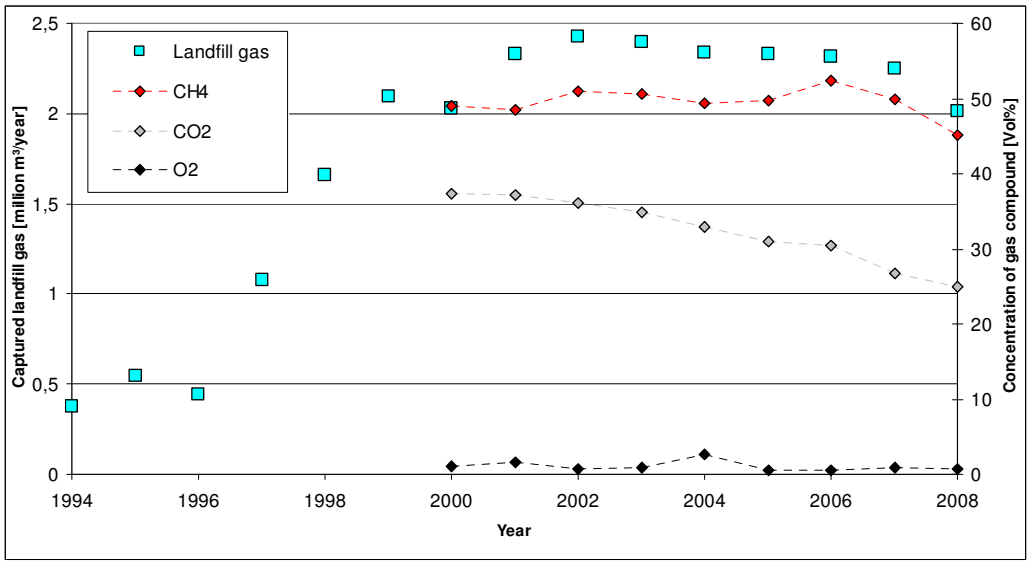


Figure 6-2: Collected amount of landfill gas and gas composition at Landfill A (common collection of gas for all landfill compartments)

Containment system

The containment system of Landfill A includes different designs at the compartment level (see Table 6-1). Information about the performance of the top cover systems is available from the leachate generation rates of the compartments. However, as the amount of leachate collected at the base of younger compartments is still rather high (approximately 20% of annual precipitation), the current cover performance will be measureable after the "bleeding out" of the waste body is completed. For now, the performance level of recently installed cover systems is estimated based on the design parameters of the systems and the performance levels observed at similar systems. Therefore, the fraction of precipitation infiltrating into the waste body is estimated to be 5 % for the compartments 1 to 4, 1 % for the compartments 5, 6, and 8, and 0.5 % for the compartment 7. These performance levels represent the best possible function of the covers, provided that the systems are not reconfigured in the future.

No direct observations of barrier performance are available for the systems at the landfill base. Although the contamination level of the extracted groundwater from the slurry wall encapsulation proves that there is leachate released to the subsurface below the landfill, it is not possible to quantify the performance of the bottom barrier systems with these data. The fraction of leachate percolating through the landfill base is estimated to be between 3 and 10 % for the compartments 1 to 4, 1 % for the compartments 5 and 6, and 0.5 % for the compartments 7 and 8. As these values are not based on direct measurements, they should be treated as assumptions rather than true system parameters.

6.1.2 Emission predictions

Emission models for leachate and landfill gas are used to estimate future substance releases from the landfill into the surrounding environment based on scenario analysis. The scenarios are built on the same emission model for each compartment (fraction of mobilizable substances, water flow heterogeneity, release mechanisms), but differ with respect to the level of interaction between the

landfilled waste and the surrounding environment (i.e. water infiltration rates and release rates to the subsurface). The scenarios are designed to illustrate potential emissions after post-closure care has been terminated. Hence, it is assumed that there is no repair or maintenance of the containment system taking place after the time of evaluation. The time of evaluation for Landfill A is the year 2007, as this is the last year with a complete set of monitoring data.

Table 6-3: Scenario layouts to investigate potential emission levels of Landfill A

Scenario	Technical barrier at the top	Technical barrier at the base
Status quo	Best performance	Best performance
Status quo*	Best performance	Ineffective
Scenario A	Gradual decrease of barrier performance	Gradual decrease of barrier performance
Scenario A*	Gradual decrease of barrier performance	Ineffective
Scenario B	Ineffective	Ineffective

Note:
"*" designates scenario variations of the status quo and Scenario A which assume a total release of the generated leachate to the subsurface.

The scenario layouts to investigate potential future emissions of the landfill are shown in Table 6-3. Within the status quo constant barrier function at the top and at the bottom of the landfill is assumed, showing the effect of continuous optimal barrier performance. Scenario A considers a gradual deterioration of the barrier systems and a consequent decrease of barrier function in the future. Scenario B investigates the complete failure of the technical barriers, thereby illustrating the "worst case" with respect to barrier performance. In addition, modified versions of the status quo and Scenario A are analyzed to account for the fact that there is no free drainage to remove the leachate from the leachate collection tanks. After the termination of post-closure care, leachate might not be extracted from the landfill anymore and percolate through the landfill base. Therefore, the complete release of leachate into the subsurface is investigated in the status quo* and in the Scenario A*. The modeling period for all scenarios is 300 years.

As described above, all scenarios assume constant release mechanisms and water flow patterns in the deposited waste and are designed to analyze the effect of changing hydraulic boundary conditions (infiltration at the top and percolation rates at the bottom) on landfill emissions. Other scenarios could be established to investigate the effect of landfill flooding (i.e. full water saturation of the waste body), of changing water flow patterns within the landfill, or of the aeration of the waste body on future emission levels. However, landfill flooding is improbable within the next centuries, as hydraulic simulations showed no inundation of the landfill area even in case of a 1000 year recurrence interval flood. A change of the water flow pattern in the waste is not investigated, because an abrupt change is not to be expected in an undisturbed landfill. As the aeration of the undisturbed waste body is expected to take place over several thousand years at a slow rate (cf. Bozkurt et al. 1999), such a scenario is not included in the analysis.

6.1.2.1 Leachate emissions

Emission model

For each landfill compartment an emission model is established based on the monitoring data at the site and data from the literature on mobilizable waste fractions (see Table 6-5 and Table 6-4). The

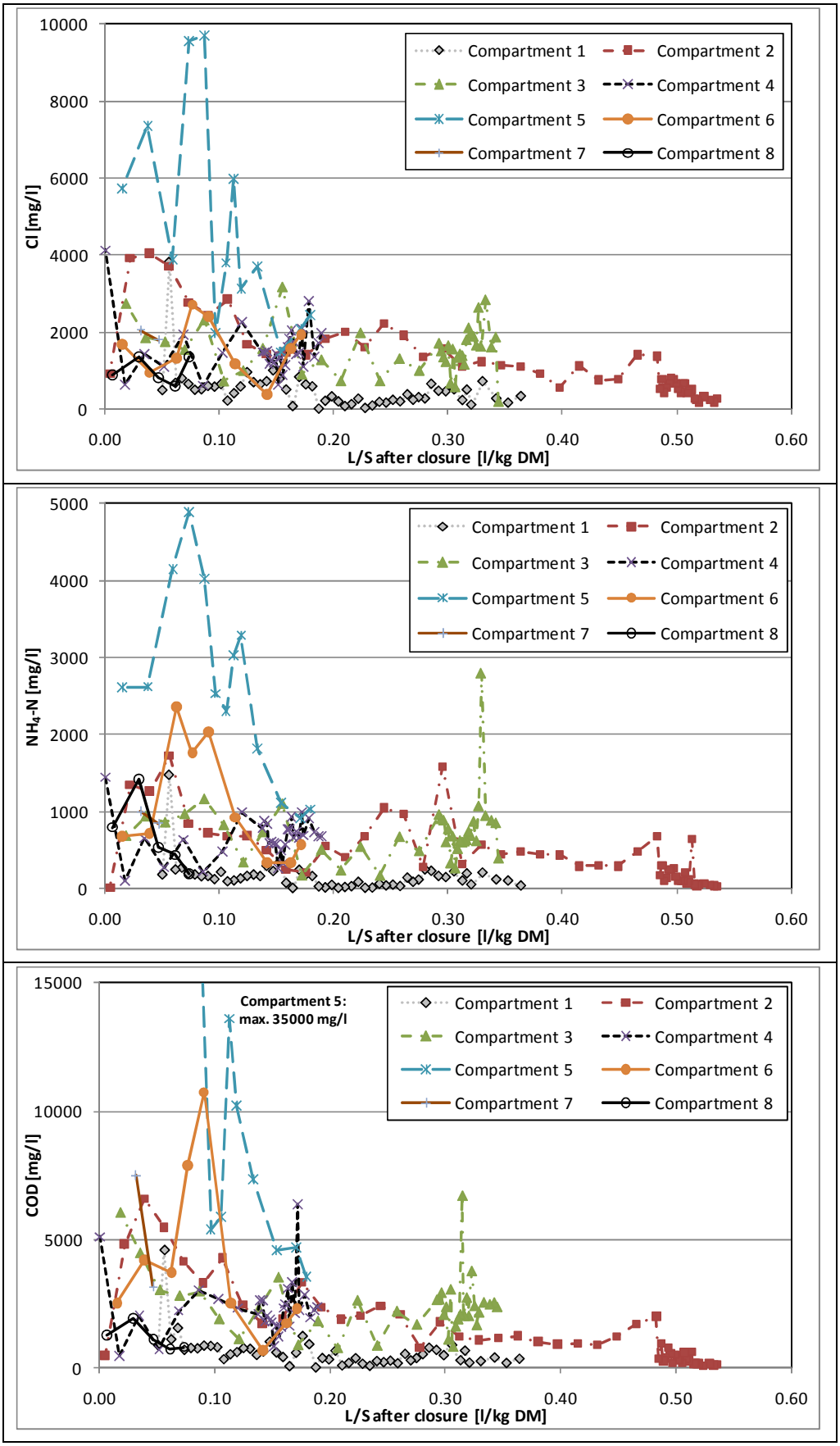
concentrations of Cl, NH_4-N, and COD in the leachate after compartment closure are shown in Table 6-5 as a function of the liquid-to-solid ratio of the waste. In general, a decreasing trend can be observed for leachate concentration levels, but for younger compartments additional monitoring data would be necessary to allow for reliable statements. The parameters of the calibrated emission models are shown in Table 6-4. The mobilizable waste fractions (m_s and m_{org}) as well as the water flow heterogeneity (h) are chosen in a plausible bandwidth (cf. Table 5-2) and the models are fitted to the measured leachate concentrations after compartment closure.

Table 6-4: Model parameters to estimate the concentrations of Cl, NH_4-N, and COD in the leachate of the landfill compartments (c_0 refers to the time of compartment closure)

Compart-ment	Parameter	c_0 [mg/l]	m_s [mg/kg]	h [-]	$c_{0,org}$ [mg/l]	m_{org} [mg/kg]
C1	Cl [mg/l]	1050	1500	5	-	-
	NH_4-N [mg/l]	800	300		76,7	500
	COD [mg/l]	1200	800		429	2800
C2	Cl [mg/l]	2500	1800	3	-	-
	NH_4-N [mg/l]	1000	500		125	600
	COD [mg/l]	4500	1600		416,7	2000
C3	Cl [mg/l]	1800	2000	2	-	-
	NH_4-N [mg/l]	500	500		402	1000
	COD [mg/l]	3000	2000		804	2000
C4	Cl [mg/l]	2000	1750	2	-	-
	NH_4-N [mg/l]	450	400		345	600
	COD [mg/l]	2600	800		1035	1800
C5	Cl [mg/l]	6000	1500	3	-	-
	NH_4-N [mg/l]	3500	600		429	1400
	COD [mg/l]	10000	3000		675	2200
C6	Cl [mg/l]	2000	1800	2	-	-
	NH_4-N [mg/l]	2000	400		230	800
	COD [mg/l]	10000	2200		632,5	2200
C7	Cl [mg/l]	2200	1800	3	-	-
	NH_4-N [mg/l]	1000	600		255,6	1000
	COD [mg/l]	8000	2000		511	2000
C8	Cl [mg/l]	1400	1500	3	-	-
	NH_4-N [mg/l]	1000	200		230	800
	COD [mg/l]	1500	500		287,6	1000

Model formulation: $$c(t) = c_0 \bullet e^{-(\frac{c_0}{m_s} \bullet \Delta \frac{L}{S} \bullet h) \bullet t} + c_{0,org} \bullet e^{-(\frac{c_{0,org}}{m_{org}} \bullet \Delta \frac{L}{S} \bullet h) \bullet t}$$

Table 6-5: Concentrations of Cl, NH_4-N, and COD in the leachate after compartment closure



The concentrations of Cr in the leachate are not included in Table 6-5 and Table 6-4, as they do not exhibit a significant trend and estimates on the mobilizable fractions of chromium in MSW are hardly available in the literature and highly uncertain. Therefore, the mean concentration values are extrapolated for the whole modeling period. The assumption of constant chromium levels in the leachate is conservative, as long as there is no increased release of chromium due to a change in ambient conditions in the waste or pH decrease.

Status quo

The status quo is based on the persistence of the best containment system performance at the site. In addition to the constant function of the technical barriers, the substance release mechanisms and water flow patterns in the waste remain unchanged. Depending on the investigated compartment, the top cover systems prevent between 95 and 99.5 % of the annual precipitation from infiltration into the waste and the efficiencies of the base lining systems (fraction of leachate captured within the landfill) lie between 90 and 99.5 %.

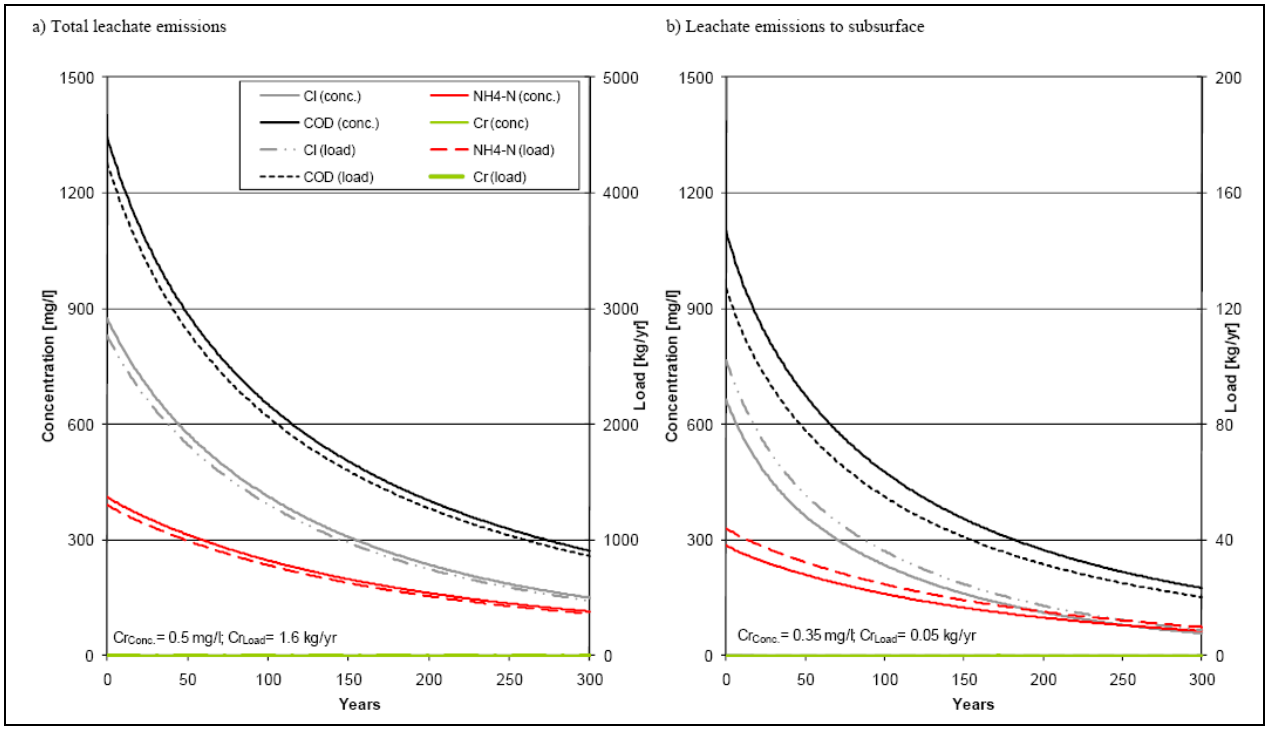


Figure 6-3: Leachate emissions for the Status quo (constant barrier performance) at Landfill A (left: total leachate emissions, right: leachate emission to subsurface)

The emission estimates for the status quo (see Figure 6-3) are based on the emission models in Table 6-4 and the barrier efficiencies described above. In Figure 6-3 the emissions of each compartment have been integrated for the whole landfill due to reasons of brevity. The concentrations of chloride, ammonia-nitrogen, and chemical oxygen demand (COD) in the leachate decrease slowly, because of the relatively low level of water infiltration into the waste body. Chloride concentrations fall from 870 mg/l to 150 mg/l, ammonia-nitrogen from 410 mg/l to 110 mg/l, and COD from 1350 mg/l to 270 mg/l within 300 years (see Figure 6-3, left). Similarly, the loads of Cl, NH_4-N, and COD decrease slowly from several thousand kg/yr to several hundred kg/yr after 300 years.

The concentrations and loads of chromium are constant during the whole modeling period at a level of 0.5 mg/l and 1.6 kg/yr, respectively. The concentrations in the leachate released to the subsurface (Figure 6-3, right) differ from the total leachate emissions (Figure 6-3, left), because of different efficiencies of the technical barriers at the landfill base on the compartment level. Concentrations in the released leachate are slightly lower, as the older compartments produce less polluted leachate and have lower barrier efficiencies. The annual emission loads of Cr_{tot} to the subsurface are lower by a factor 30 than the total loads in the leachate (see Figure 6-3, right).

The modified status quo* assumes the complete inefficiency of the technical barrier at the landfill base. Hence, the total leachate emissions in Figure 6-3 (left) and the emissions released to the subsurface are the same for this scenario.

Scenario A

This scenario is designed to investigate the effect of gradual barrier deterioration and decreasing barrier performance levels on landfill emissions. The conditions within the waste body remain stable (e.g. dominant release mechanisms, water flow pattern) and the hydraulic boundary conditions change over time (gradual increase of infiltration rates and percolation rates through the landfill base).

The future barrier performance is estimated for three service periods and performance levels are derived for each period (cf. Table 6-6). Barrier efficiency is delimited by the actual system performance (the best possible service level provided there is no future maintenance) and the complete inefficiency of landfill liners (the worst service level). The inefficiency of the final top cover is approximated by the local groundwater recharge rate of 27% of annual precipitation. Regarding bottom liner failure, all the generated leachate is released to the vadose zone below the landfill. The predicted service levels of the containment system are based on evaluating different performance factors and their importance in a specific service period. The evaluation scores and the translation of the scores into respective performance levels are shown in Table 6-6 for the technical barriers of compartment 1. The calculated performance levels in Table 6-6 (leachate generation rates and leachate percolation through landfill base) are reached at the end of each respective service period. The evaluation procedure is described in detail in chapter 5.2.

The calculated emissions for Scenario A are based on the emission models in Table 6-4 and the predictions on future barrier efficiencies of the compartments (e.g. Table 6-6). The total leachate emissions and the leachate emissions to the landfill subsurface are shown in Figure 6-4. The concentrations of chloride in the leachate decrease from 870 mg/l at the beginning to approximately 10 mg/l after 300 years, those of ammonia-nitrogen from 410 mg/l to 50 mg/l, and those of COD from 1350 mg/l to 100 mg/l (see Figure 6-4, left). The corresponding annual loads in the leachate peak after 30 to 60 years at 3420 kg Cl/yr, 1840 kg NH_4-N/yr, and 5350 kg COD/yr, respectively. Due to the constant chromium concentrations during the modeling period (0.6 mg/l)[18], the chromium loads increase steadily to a level of 9.1 kg Cr/yr after 300 years.

[18] Chromium concentrations are 0.6 mg/l for Scenario A relative to 0.5 mg/l for the staus quo, because of decreasing barrier performance at compartments with higher chromium concentrations in the leachate. The average concentration over the whole modeling period in the leachate is 0.6 mg/l and is kept constant (conservative assumption) for the leachte released to the subsurface.

Table 6-6: Evaluation of the containment system of compartment 1 of Landfill A to estimate future service levels within Scenario A

Factors – top cover	**Evaluation score***	**Weighting factors****		
	1_{good}–3_{bad}	0–100 yrs	100–200 yrs	200–300 yrs
Barrier performance at time of evaluation	**2**	0.36	0.08	0.065
Construction quality program	**1**	0.38	0.44	0.12
Recultivation layer (cover)	**2**	0.33	0.48	0.487
Heat production within waste	**1**	0.09	0.06	0.04
Climate	**1**	0.08	0.12	0.487
Typical vegetation and projection after use	**1**	0.09	0.48	0.487
Relief	**1**	0.08	0.1	0.12
Expected settlements	**1**	0.38	0.09	0.065
Lining control systems (direct monitoring)	**3**	0.1	0.06	0.05
Drainage system	**2**	0.1	0.09	0.08
	Evaluation results:	**2.98**	**2.77**	**2.73**
	Leachate generation rates [mm/yr]:	**98.6**	**128.7**	**151.9**
Factors – bottom lining system				
Barrier performance at time of evaluation	**2**	0.44	0.08	0.07
Construction quality program	**1**	0.46	0.49	0.44
Climate	**1**	0.07	0.08	0.09
Heat production within waste	**1**	0.09	0.06	0.05
Distance to groundwater	**1**	0.08	0.08	0.09
Overburden pressure	**2**	0.08	0.08	0.08
Leachate quality	**1**	0.08	0.08	0.08
Monitoring (e.g. groundwater monitoring)	**1**	0.08	0.07	0.07
Drainage system	**1**	0.46	0.11	0.10
Stability of foundation	**1**	0.10	0.44	0.46
Landfill geometry (heap vs. cavity)	**2**	0.08	0.43	0.48
	Evaluation results:	**2.62**	**2.59**	**2.64**
	Leachate percolating through landfill base [mm/yr]:	**23.6**	**45.3**	**69.2**

*Specific criteria are used to derive evaluation scores: For instance, a score of 1 for "barrier performance at the time of evaluation" means that the barrier has performed according to its design specifications, a score of 2 is awarded if there is no direct monitoring of the barrier function, but it is probably performing according to the design specifications, whereas a score of 3 means that it has not achieved its intended service level at the time of evaluation.

**Weighting factors were derived from expert evaluations during a survey on the importance of different factors for barrier performance during different service periods (cf. chapter 4.1.4, chapter 5.2, and appendix 2).

In the leachate released to the subsurface concentration levels of the emission parameters are lower than in the generated leachate at the beginning of the modeling period (except for Cr). However, these concentrations are increasing during the first decade due to the decreasing efficiency of the base lining system at compartments with strongly polluted leachate. After a few years the concentrations reach maxima of 664 mg Cl/l, 308 mg NH_4-N/l, and 996 mg COD/l (cf. Figure 6-4, right). The corresponding loads increase to levels of 330 kg Cl/yr after 130 years, 278 kg NH_4-N/yr after 225 years, and 578 kg COD/yr also after 225 years. Due to the constant chromium concentrations in the leachate, the chromium loads increase to 3.4 kg/yr at the end of the modeling period.

The modified scenario "Scenario A*" assumes the complete inefficiency of the technical barrier at the landfill base. Hence, the total leachate emissions in Figure 6-4 (left) and the emissions released to the subsurface are the same for Scenario A*.

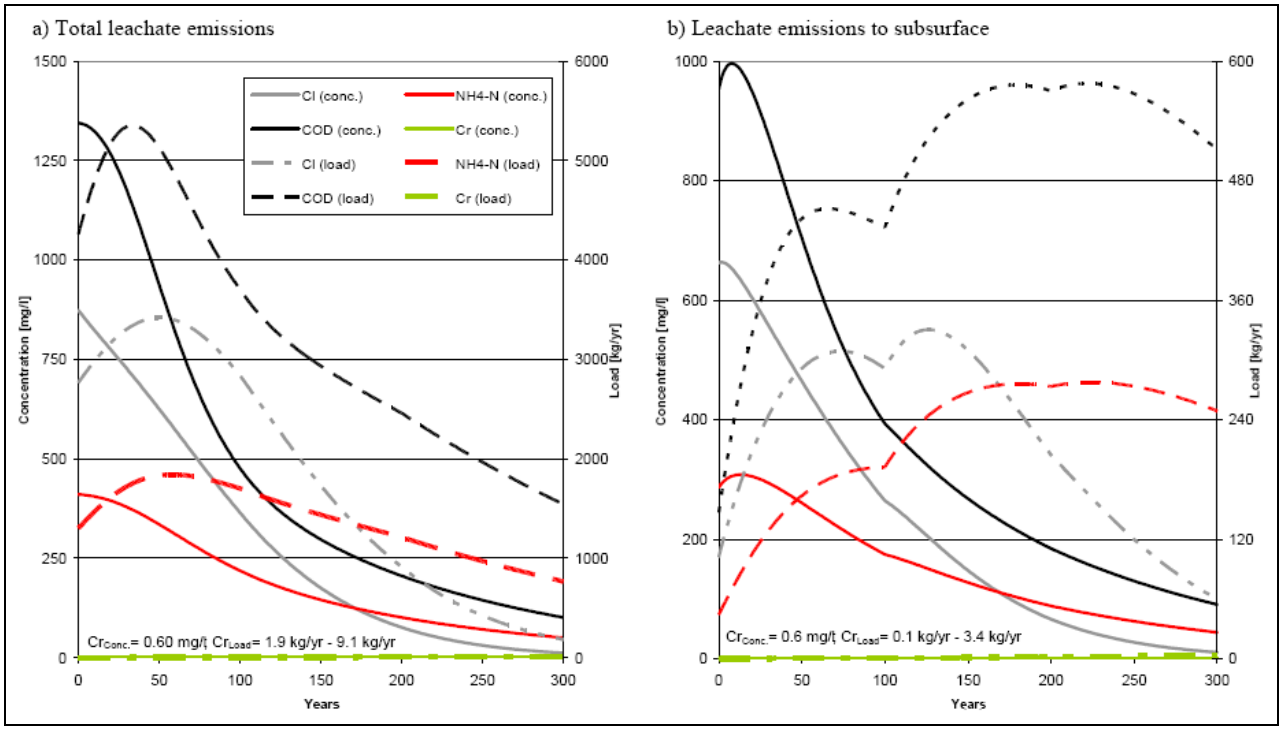


Figure 6-4: *Leachate emissions for Scenario A (gradual barrier deterioration) at Landfill A (left: total leachate emissions, right: leachate emissions to subsurface)*

Scenario B

Scenario B illustrates the effect of an ineffective containment system on the emissions of Landfill A. The worst performance level of the top cover is approximated by the local groundwater recharge rate (27% of annual precipitation). At the landfill base all the generated leachate is then released to the subsurface. The emission estimates for this scenario in Figure 6-5 are again based on the emission models shown in Table 6-4 and the inefficiency of the containment system, as described above. Within Scenario B the concentrations of chloride, ammonia-nitrogen, and chemical oxygen fall below a level of 200 mg/l within several decades, due to the high water infiltration rate (255 mm/yr) and the consequent wash-out of mobilizable substances (see Figure 6-5). The annual loads decrease by a factor 14 for Cl, 5 for NH_4-N, and 10 for COD to 2300 kg Cl/yr, 3200 kg NH_4-N/yr, and 6500 kg COD/yr within the first 50 years. After 100 years the concentrations of all the investigated leachate parameters are below 100 mg/l. For chromium the concentration is assumed constant at 0.69 mg/l[19] with corresponding annual loads of 20.1 kg Cr/yr.

[19] Concentration of Cr in the leachate is higher for Scenario B relative to the status quo and Scenario A due to the assumed total failure of the top cover system, as compartments with higher Cr concentrations in the leachate contribute to a larger degree to the total leachate generation during the scenario.

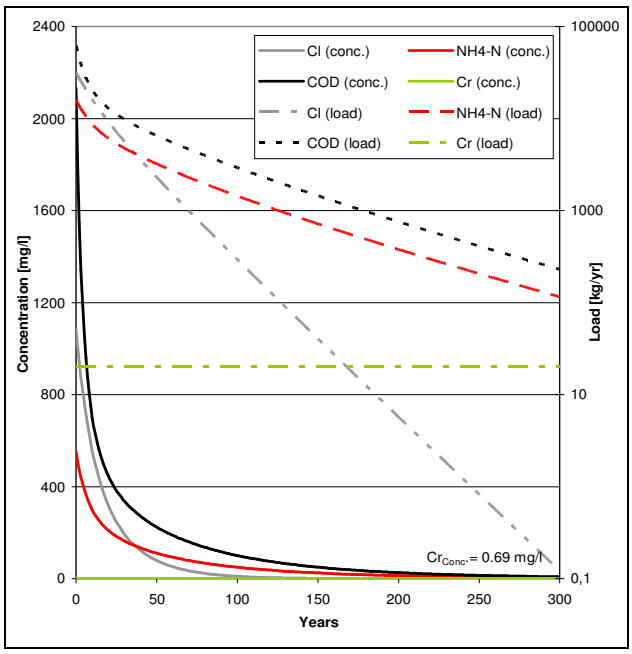


Figure 6-5: Leachate emissions for Scenario B (complete failure of barriers) at Landfill A

6.1.2.2 Landfill gas emissions

Due to the low quality of available landfill gas monitoring data, different models and model parameters are used to estimate potential landfill gas emissions. Thereby, a bandwidth of possible emission levels is illustrated. The landfill gas emission models are not adapted to the scenarios described above, but only extrapolated to the future based on the past conditions within the landfill. The reason for this is twofold: a) no relationship between the amount of water infiltration and the gas production rate at Landfill A can be established, and b) the large bandwidth of possible emission model parameters and resulting estimates do not allow for meaningful scenario analysis. In any case, it is probable that the scenarios investigating the effect of limited water infiltration into the waste body would be associated with relatively low gas generation rates, as the availability of mobile water is of primary importance for anaerobic degradation processes (cf. Klink and Ham 1982; Bogner and Spokas 1993; Christensen et al. 1996).

Emission model and predictions

In order to adjust a theoretical gas production model to the measured gas collection data at the site, several assumptions are necessary. First, the fraction of biodegradable organics contained within the deposited MSW must be estimated. Second, the fraction of landfill gas collected needs to be assumed. Based on data presented by Krümpelbeck (2000) for German landfills and the fact that gas collection was installed two decades after the operation of the landfill was started, a collection efficiency of 35 percent is assumed for Landfill A. The gas production rates calculated from the measured amount of collected landfill gas are shown in Figure 6-6. Two landfill gas production models are adjusted to the estimated (35 % collection efficiency) gas production rate. The model by Tabasaran and Rettenberger (Tabasaran and Rettenberger 1987) with a gas production potential of 187 m³ per ton of dry waste and the LandGEM model published by the U.S. EPA (U.S. EPA 2005)

with a gas production potential of 200 m³ per ton of dry waste (the moisture content of the deposited waste was 30% by weight). Typical degradation rates for conventional landfills (cf. Table 3-1) are chosen. For the Tabasaran & Rettenberger model the k-value is 0.035 per year and for the LandGEM model the k-value is 0.04 per year. The consequent gas production curves for these two models are shown in Figure 6-6. The Tabasaran & Rettenberger model calculates the highest gas generation rates, but also predicts a faster decrease of gas generation than the LandGEM model. Both models are rather conservative (overestimate actual gas generation) compared to the gas generation rates estimated from measured data with an assumed collection efficiency of 35% of total landfill gas. However, as the poor monitoring data and the large uncertainties (i.e. collection efficiency, degradable amount of organic matter) do not allow for more confident estimates on gas generation, the model estimates in Figure 6-6 are used to evaluate a bandwidth of potential future landfill emissions. The concentration of methane in the landfill gas is estimated to be 50 % of volume, which is confirmed by the monitoring data (cf. Figure 6-2).

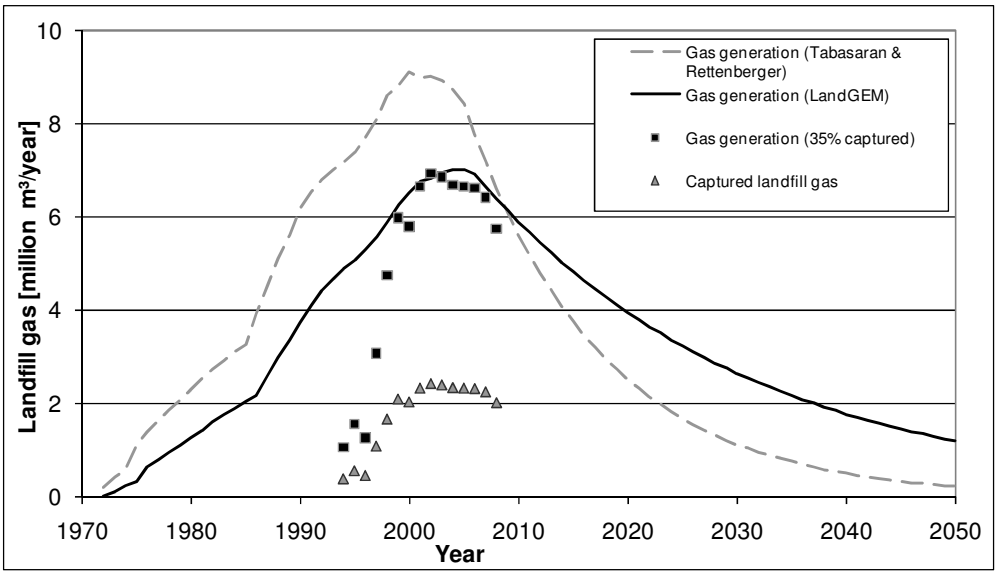


Figure 6-6: Comparison of measured data on captured landfill gas at Landfill A with estimates of different models about landfill gas generation

6.1.3 Pollutant migration

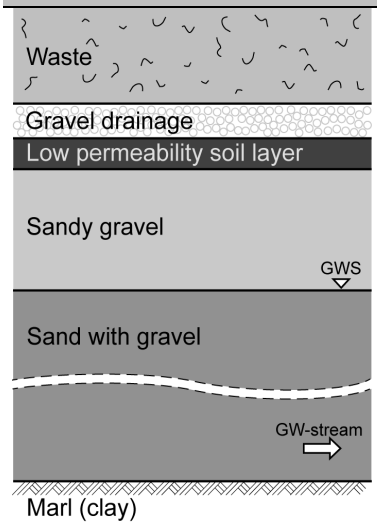
Based on the emission estimates developed above, the migration of released pollutants and their ambient concentrations in the affected environmental media (i.e. soil, groundwater, surface waters, and atmosphere) are evaluated.

Leachate migration

A scheme of the migration pathway through the vadose zone and the aquifer below Landfill A is shown in Table 6-7 together with the soil properties of the different layers. The design parameters of the landfill liner system of the compartments 1 to 4 are used for the whole landfill, although there might be thicker earthen barriers or composite liner systems at the base of other compartments (cf. Table 6-1). The earthen barrier is included in the pollutant migration modeling. The natural subsurface system is mainly built up from gravel with the groundwater surface approximately 2 meters below the lowest point of the landfill base. The slurry wall system around

the landfill and the consequent groundwater withdrawal are not included in the migration modeling, because these measures are not expected to be maintained after the end of aftercare. Hence, the investigated system corresponds to the situation without a slurry wall encapsulation.

Table 6-7: Layout of the subsurface below Landfill A and material properties of the soil layers

Layout of the subsurface below the Landfill A	**Thickness, grain size distribution and soil type** (according to AG Boden 2005)	**Hydraulic conductivity, field capacity, pore volume, and dry bulk density**
	**Low permeability soil layer:** Thickness: 0.5 m; 0 – 30 % sand, 50 – 65 % silt, 17 - 30 % clay (soil type: Lu) **Sandy gravel:** Thickness: 2.0 m; 70 – 95 % gravel, 5 – 30 % sand, 0 – 10 % silt, 0 – 2 % clay (soil type Gs) **Sand with gravel (aquifer):** Thickness: 15.0 m; See: sandy gravel **Marl** (aquitard, $k_f = 5*10^{-9}$ m/s)	 k_f < 10-8 m/s field capacity: 34 % total pore volume: 39 % dry bulk density: 2.1 g/cm³ $k_f = 1*10^{-5} – 5*10^{-3}$ m/s field capacity: 2 % total pore volume: 31 % dry bulk density: 2.0 g/cm³ $k_f = 1*10^{-5} – 5*10^{-3}$ m/s

Pollutant migration models are established for chloride, ammonia-nitrogen, chromium, and the chemical oxygen demand (COD). The COD migration is investigated as an aggregate parameter for organic leachate pollution due to the lack of substance-specific data and it is assumed that it travels conservatively (e.g. like Cl) in the leachate. Therefore, average travel times of COD in the unsaturated zone are assumed to be the same as those of chloride. Pollutant migration in the vadose zone is generally evaluated based on the AF model by Schneider and Stöfen (2004). However, for scenarios with gradually increasing percolation rates and for estimating COD migration, the concentrations are calculated based on the maximum emission loads into the subsurface (see chapter 5.3). Pollutant concentrations are calculated for the points of compliance (PoC) and are shown in Figure 6-7. In the migration model, the landfill emissions represent the source term and the PoCs are used to evaluate the concentration levels in different environmental media due to a release of pollutants from the landfill. The change in concentration from the source to PoC1 (cf. Figure 6-7) is due to the transport in the vadose zone. PoC2 is located in the groundwater plume below the landfill. A reduction in the concentration of a pollutant between PoC1 and PoC2 takes place only due to the dilution of the leachate while mixing into the groundwater. In the groundwater no spreading of the plume is assumed due to the short travel distances to the pond (North of the landfill) and the river (East of the landfill). PoC3 is in the river adjacent to the landfill, where the polluted groundwater is mixed with the river water. The leachate-induced pollutant levels in the small lake are assumed to equal the average pollutant concentration of the total groundwater stream (the pond is fed by the groundwater stream).

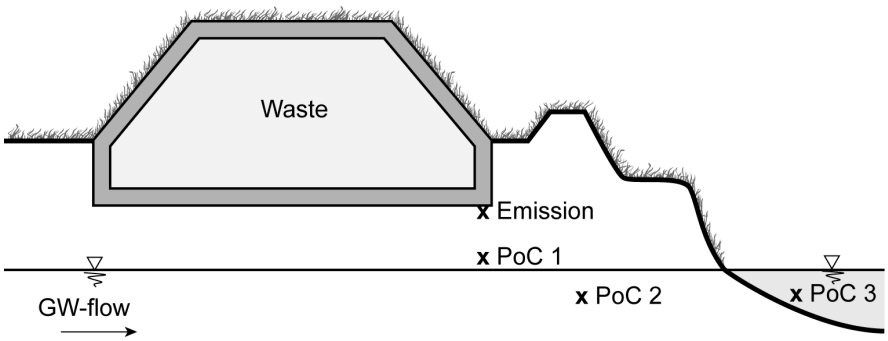


Figure 6-7: Scheme of the landfill-environment system and relevant points of compliance (PoC) at Landfill A (cross-section along the ground water flow direction)

Different transport parameters and assumptions within the pollutant migration modeling are listed in Table 6-8. As the distribution coefficients of the linear sorption isotherm (k_d) relate to a single substance and not to a complex mixture of substances like landfill leachate, the values in Table 6-8 were chosen conservatively (see also chapter 4.2)

Table 6-8: Parameters used for pollutant migration modeling in the subsurface below Landfill A

Vadose zone	Diffusion coefficient in water [m²/yr]	k_d-value [l/kg] (low perm. soil layer)	k_d-value [l/kg] (gravel & sand)	Comments
Cl	0.0315	0	0	No degradation
NH_4-N	0.0378	0.5	0	No degradation
Cr_{ges}	Unknown (0)	400	400	No degradation
Groundwater	Groundwater velocity [m/yr]	Mixing depth [m]	Contaminated width [m]	Comments
General parameters	150	0,5	250	No degradation*

* The dissolved oxygen content in the groundwater is used as an indicator to estimate the potential for aerobic degradation of organic substances and the nitrification of ammonia.

The attenuation factors in Figure 6-8 represent a first estimate of concentration reductions due to transport processes in the subsurface for the different emission scenarios. Some basic figures of the calculations to derive the attenuation factors for the selected pollutants and the different scenarios are given in Table 6-9 and Table 6-10.

The concentrations of Cl, NH_4-N, and Cr at the PoC1 are calculated with the AF model (Schneider and Stöfen 2004) for scenarios with constant leachate release rates (see Table 6-9). Migration processes considered within the model are longitudinal dispersion, diffusion and potentially sorption. The concentrations at PoC1 are then reduced due to the mixing with groundwater: dilution factor = (leachate flow rate [m³/yr] + groundwater flow in the mixing zone [m³/yr]) / leachate flow rate [m³/yr]. The concentrations in the groundwater remain unaltered, as there is no spreading of the plume, sorption, or degradation considered. Subsequently, the contaminated groundwater exfiltrates into the adjacent surface water bodies.

The concentrations of COD at PoC1 are calculated based on the released emission loads to the subsurface. In case of constant leachate release rates, the travel time of Cl (calculated in the AF model) is used to calculate the average COD load over this time period and the COD concentrations at PoC1. In case of varying leachate release rates, the leachate concentration at the

time when the maximum substance load is released to the subsurface is equal to the concentration at PoC1. This load-based approach is used for scenarios with varying leachate release rates, except in case of Cr, for which the concentrations are calculated according to the AF model with the average leachate release rate over the whole modeling period (see Table 6-10). The concentrations in the groundwater and in the surface waters are calculated based on a mass balance approach using maximum substance loads, too. The migration modeling represents a screening analysis with rather conservative assumptions about the environmental setup and the transport parameters. However, a more realistic pollutant migration modeling is only warranted, if more data are collected at the site to adapt more complex models to the conditions at Landfill A.

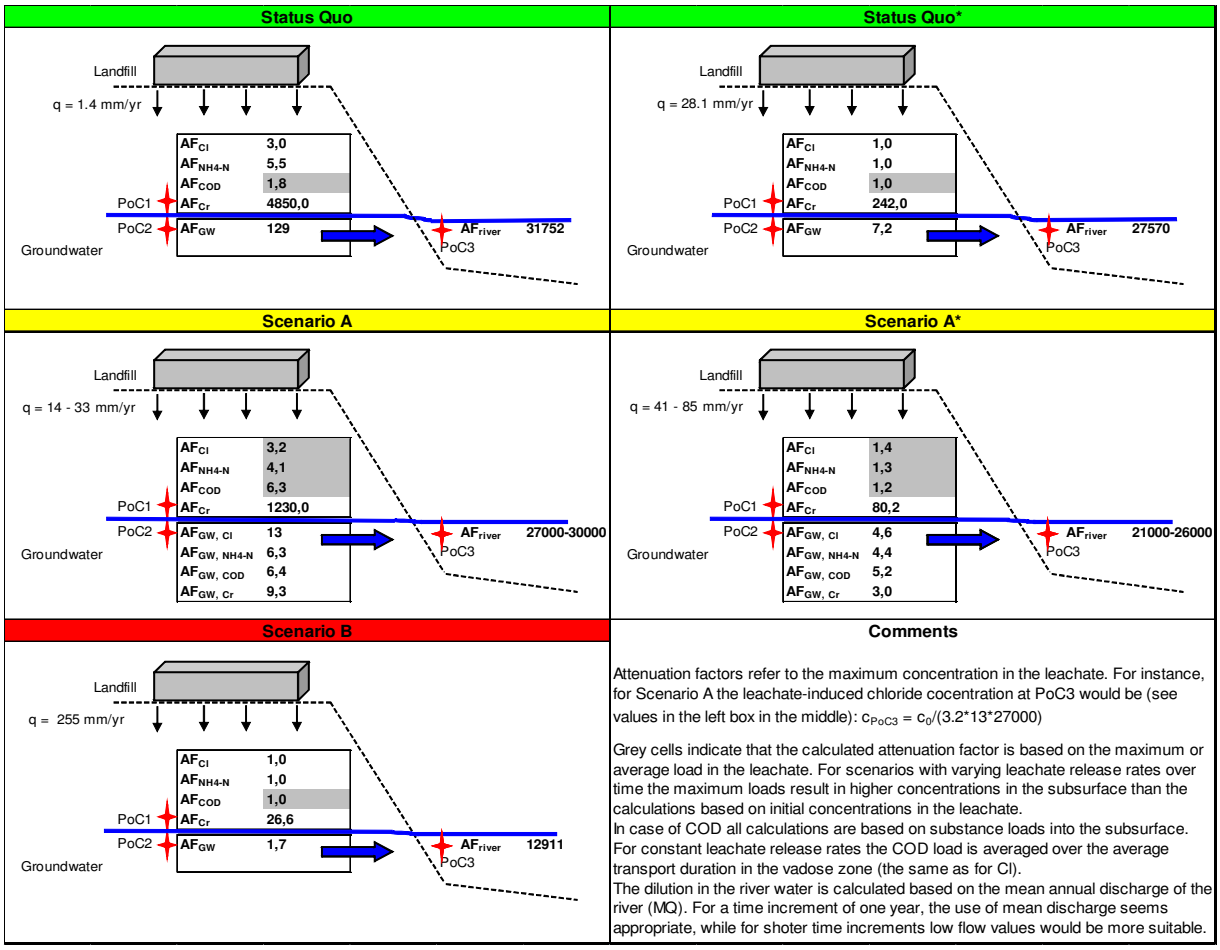


Figure 6-8: Attenuation factors to estimate the concentrations at relevant PoCs due to a release of leachate from Landfill A for the different scenarios

The pollutant migration modeling for Landfill A does not include any degradation processes which might take place in the soil and groundwater zone. However, the concentration of oxygen in the groundwater upstream of the landfill is around 7 – 9 mg/l, which results in an annual oxygen load in the groundwater mixing zone below the landfill of 140 kg O_2 per year. For the whole groundwater stream the oxygen load amounts to approximately 4200 kg O_2 per year. The oxygen demands (COD and nitrification of ammonia) induced on the groundwater during the emission scenarios status quo*, Scenario A*, and Scenario B exceed this load and therefore also affect the

oxygen levels in the adjacent surface waters (especially in the pond). However, reaeration will take place in the surface waters depending, among other factors, on the mixing rate.

Table 6-9: Data for the AF-model calculations to estimate concentrations at PoC1 and PoC2

Scenario	Parameter	$c_{0,max}$ [mg/l]	Release rate [mm/yr]	AF [-]	Travel time t_m [yrs]	c_{PoC1} [mg/l]	GW flow [m³/yr]	AF_{GW} [-]	c_{PoC2} [mg/l]
Status quo	Cl	662	1.4	3.0	150	221	19710	129	1.7
Status quo	NH_4-N	287	1.4	5.5	525	52	19710	129	0.4
Status quo	Cr	0.35	1.4	4850	> 1 Mio.	0.00	19710	129	0.0
Scenario B	Cl	1083	255.2	1.0	0.8	1083	19710	1.7	637.1
Scenario B	NH_4-N	551	255.2	1.0	2.8	551	19710	1.7	324.1
Scenario B	Cr	0.69	255.2	26.6	7900	0.03	19710	1.7	0.02
Status quo*	Cl	873	28.0	1.0	7.5	874	19710	7.2	120.9
Status quo*	NH_4-N	411	28.0	1.0	26.0	411	19710	7.2	56.9
Status quo*	Cr	0.5	28.0	242	71900	0.00	19710	7.2	0.0
Scenario A	Cr	0.6	20.7	1230	367000	0.0	19710	9.3	0.0
Scenario A*	Cr	0.6	84.5	80	23850	0.0	19710	3.0	0.0

Table 6-10: Data for the calculations based on maximum emission loads into the subsurface to estimate concentrations at PoC1 and PoC2

Scenario	Parameter	$C_{0,max}$ [mg/l]	Release rate [mm/yr]	Load [kg/yr]	Travel time t_m [yrs]	c_{PoC1} [mg/l]	AF [-]	AF_{GW} [-]	C_{PoC2} [mg/l]
Status quo	COD	953	1.4	81.0	150 [a]	526	1.8	129	4.1
Scenario A	Cl	662	14.3	329	14.0	204	3.2	13.0	15.7
Scenario A	NH_4-N	310	32.8	278	23.7	75	4.1	6.3	11.9
Scenario A	COD	1000	32.0	578	6.6	158	6.3	8.3	19.1
Scenario B	COD	2126	255.2	61950	0.8 [a]	2126	1.0	1.7	1251
Status quo*	COD	1345	28.0	4110	7.5 [a]	1298	1.0	7.2	180
Scenario A*	Cl	874	48.6	3420	4.0	625	1.4	4.6	137
Scenario A*	NH_4-N	411	51.2	1843	14.0	316	1.3	4.4	72
Scenario A*	COD	1345	41.3	5348	5.0[a]	1142	1.2	5.2	220

[a] Average travel time of COD in the vadose zone is assumed the same as for Cl.

The attenuation factors in Figure 6-8 can be used to determine the concentration levels at the PoCs due to the release of leachate from the landfill. Thus, the model outcomes do not reveal the total pollutant concentration in the groundwater, but the influence of landfill leachate on the groundwater. To calculate total concentrations, the pollutant concentrations in the groundwater upstream of Landfill A would have to be considered.

Landfill gas migration

The area-specific methane generation rates illustrated in Figure 6-9 are based on the landfill gas models for Landfill A shown in Figure 6-6. The generated methane is assumed to be released from the waste body and to migrate through the top cover, as landfill gas is not collected any more. The generated methane may be fully released to the atmosphere or some fraction may be subject to methane oxidation processes taking place in the recultivation layer of the top cover system (cf. chapter 4.2). A fraction of 15% of methane oxidation in the landfill top cover was suggested to

estimate methane emissions from sites with methane generation rates below 17 kg of CH_4 per m² and year (cf. Table 4-10). Without considering methane oxidation, methane emissions are expected to drop below 10 kg CH_4 per m² and year within 20 years and below 5 kg CH_4 per m² and year within 40 years.

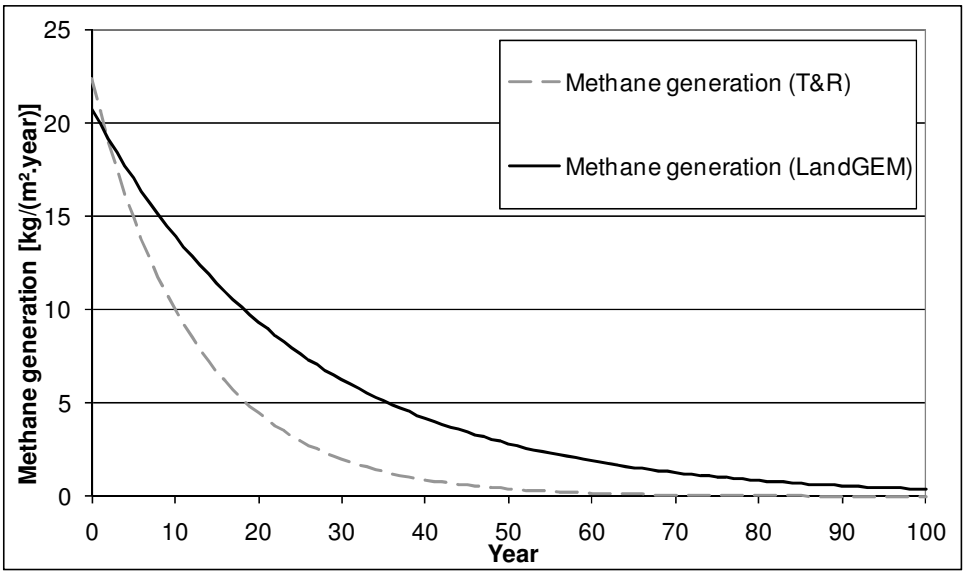


Figure 6-9: Methane generation rates per m² of landfill surface for the two gas generation models applied to Landfill A

The migration of landfill gas in the subsurface is not included in the analysis, as the landfill is hill shaped and no vulnerable uses (e.g. cellars of residential buildings) are located in the vicinity of the landfill.

6.1.4 Completion criteria and aftercare duration

Based on the developed emission scenarios, the pollutant migration modeling, and acceptable concentration levels at the points of compliance, tolerable emission levels at the source (Landfill A) are derived. The resulting scenario-specific substance concentrations in the leachate, at which aftercare can be terminated, are shown in Table 6-11. Because there is no gravity drainage of the leachate collected in the leachate basins, the status quo* and Scenario A* are considered as most relevant to assess the environmental risk for Landfill A. Due to the expected decrease in top cover performance after maintenance and repair activities are terminated, Scenario A* is used to assess tolerable emission levels, at which aftercare can be terminated. The maximum concentration levels in the leachate given in Table 6-11 allow for pollutant levels as high as the drinking water quality standards at PoC2 or in the pond, respectively. The limits applied for ammonia-nitrogen assume a nitrification of the ammonia to nitrate in the groundwater and COD levels at PoC2 are not evaluated. Hence, the limits set in Table 6-11 would allow for contamination of the groundwater below the landfill and assume this environmental compartment to be a kind of treatment filter for the released leachate. Nevertheless, the derived tolerable leachate concentrations of NH_4-N and COD would be below the current leachate concentrations. The acceptable concentration of Cl in the leachate would be in the range of current concentrations. The acceptable concentrations are determined without considering other potential pollutant sources or pollutant levels already present in the groundwater upstream of the landfill. Therefore, even if the groundwater is accepted as a

treatment filter and without looking at total concentration levels, the current levels of NH_4-N and COD are too high to meet the quality standards set in Table 6-11 for the groundwater and the pond.

Table 6-11: Tolerable leachate concentrations (c_0) at the source to comply with the specified limit values at PoC2 and in the pond

Parameter	PoC2	Status quo*	Scenario A*	Pond	Status quo*	Scenario A*
	c_{limit} [mg/l]	c_0 [mg/l]	c_0 [mg/l]	c_{limit} [mg/l]	c_0 [mg/l]	c_0 [mg/l]
Cl[a)]	200	1445	1286	200	43341	38584
NH_4-N[a)]	11.3	82	64	11.3	2449	1932
COD[a)]	-	-	-	5	1123	920
Cr[a)]	0.01	17	2	0.01	524	73

a) Limit values for leachate induced concentrations at PoC equal Austrian drinking water quality standards (MoE 2001). In case of ammonia-nitrogen limit values for nitrate are applied. COD limits are only applied to subsurface waters.

To estimate the time until acceptable leachate quality can be reached at Landfill A and the landfill can be released from aftercare, it is necessary to specify an aftercare strategy for the landfill. In Table 6-12 potential aftercare durations are given for each of the critical leachate parameters (NH_4-N and COD) based on three different (hypothetic) aftercare strategies. The first strategy is based on constantly low water infiltration rates (current performance level remains constant during the after-care period). The second strategy is based on a gradual increase of water infiltration rates (as described for top cover performance in Scenario A). The third strategy is based on the intensified infiltration of water into the waste body (infiltration rate equals the local groundwater recharge rate). The consequent aftercare durations are between 80 (intensive infiltration) and 450 (minimum infiltration) years with ammonia-nitrogen, as the critical parameter for the environmental compatibility of the leachate. Hence, even intensive water infiltration will not allow for the termination of aftercare at Landfill A within the next decades. The leachate management period required at Landfill A will be probably in the range of several centuries.

Table 6-12: Duration of aftercare for leachate emissions of Landfill A in consideration of different aftercare strategies

Parameter	Sealing at the top (constant performance)	Sealing at the top (decrease of performance)	Intensive water infiltration through the cover
NH_4-N	**450 yrs**	**270 yrs**	**80 yrs**
COD	100 yrs	50 yrs	7 yrs

If all the generated methane in the landfill is released to the atmosphere, it would take 60 to 110 years until the methane emissions drop below a value of 0.3 kg CH_4 per m² and year (see Figure 6-9), which was suggested as a threshold level for insignificant methane emissions (cf. Fellner et al. 2008). The oxidation of 15% of the generated methane in the top cover would slightly shorten the period of significant landfill gas emissions to 50 - 100 years. Thus, the aftercare duration at Landfill A is probably determined by the necessary leachate management periods (cf. Table 6-12) and not by landfill gas management. In addition, methane emissions may decrease at a faster rate in reality than predicted by the landfill gas models, due to the conservative assumptions in the models. Provided that monitoring data of sufficient quality are collected at the site in the future, more realistic predictions on gas generation ratesa at the site could be derived.

Although the presented evaluation is based on many (conservative) assumptions, inaccurate data, and rather simplistic models, it has been illustrated that the developed methodology can be used to define the desired state of the landfill at the end of aftercare and that different management strategies can be evaluated with respect to achieving this state of the landfill. The emission models, scenario layouts, and migration models used for the evaluation at Landfill A need to be verified with future monitoring data or potentially adjusted to correspond with the observed developments at the site. In any case, additional data need to be collected, as a basis for a more realistic evaluation of the environmental compatibility of Landfill A, which can be used to optimize long-term landfill management at the site.

6.2 Landfill B - closed MSW landfill

6.2.1 Site description and monitoring data

Landfill and site conditions

During the years 1988 and 2003 approximately 135,000 tons of municipal solid wastes were disposed at Landfill B. The waste was deposited in two sections, which were filled in succession. The waste deposition took place from 1992[20] to 2000 in the first section and from 2000 to 2003 in the second section. The two sections are only separated by a dam at the landfill base, the technical systems of both sections are identical, and emissions are collected for the whole landfill.

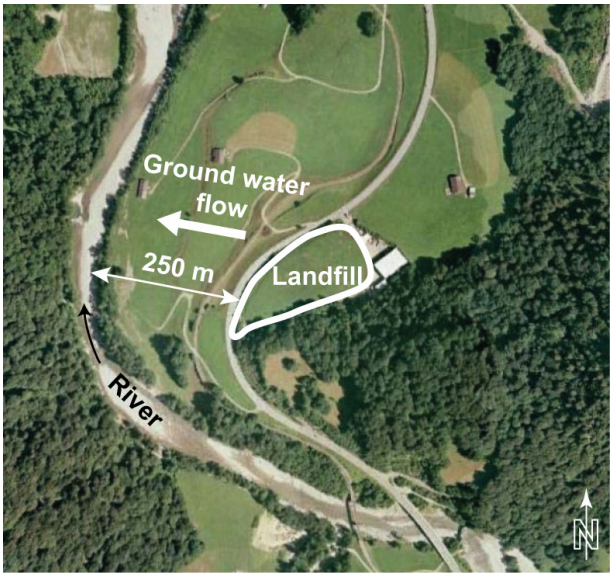


Figure 6-10: Aerial photograph of Landfill B and surroundings

The landfill is situated on a hillslope with a road passing nearby the Western boundary of the premises (see Figure 6-1). The direction of groundwater flow is West to Northwest. A river is located in the valley approximately 250 m from the landfill. The leachate collection tanks are situated

[20] Landfilling began at the site in 1988, but the waste disposed of between 1988 and 1992 was excavated in 1992 and then put into the upgraded landfill. Hence, although waste has been deposited since 1988 the landfill containment at the base was finished in 1992.

underground outside the landfill on the other side of the road, to enable gravity drainage and collection of the landfill leachate.

The climate at the landfill site is humid with an average annual precipitation of 1960 mm. The geologic strata below the landfill are composed of glacial gravel with high carbonate content. The fluctuation of groundwater surface levels below the landfill is high with a typically low thickness of the groundwater body. The distance between the lowest point of the landfill containment system and groundwater surface is not less than 1.2 m. The groundwater body in the valley is more voluminous than the groundwater along the hillslope below the landfill and has been investigated as a potential drinking water resource of a village located 2 km North of the landfill.

Monitoring data

The waste volumes deposited at Landfill B were rather low during the first 10 years of landfill operation. From 1988 to 1998 approximately 5000 tons of MSW have been disposed at the site annually. Afterwards the amount of MSW landfilled increased to peak at 32,700 tons per year in 2002. In consequence, the second landfill section was filled within 3 years although its volume is higher than the volume of the first section. In 2003, the landfill was closed and a temporary cover consisting of a soil layer and a compost layer at the top (cf. Table 6-14) has been installed. The temporary cover is planned as an aftercare measure until 2020, when a final cover shall be installed at the landfill site.

The generated leachate is collected at the bottom of the landfill in a gravel drainage layer with embedded leachate collection pipes. The pipes transport the leachate to the landfill bottom below the middle dam, which separates the two landfill sections. The collected leachate of both landfill sections is then directed to a leachate storage tank outside the landfill. The leachate generation rate is evaluated based on the removal of leachate from the storage tank for treatment in a nearby waste water treatment plant. The amount of leachate was continuously high at the landfill. During operation the annual leachate generation rate was 50 to 90% of the annual precipitation. After the installation of the temporary cover, the amount of leachate ranged between 60 and 80% of the average annual precipitation, which corresponds to 13,500 and 18,000 m³ per year, respectively.

The samples for leachate analysis are taken at the inflow of the leachate storage tank. Samples are taken twice a year and typically analyzed for 22 parameters. Some data for selected leachate constituents are presented in Table 6-13 for the period during landfill operation and the period after landfill closure. From the table it is evident that decreasing concentrations can be observed for many leachate parameters after closure, but that some parameters still range far above the specified quality standards for direct discharge of the leachate into a river. The mean values of post-closure measurements in the leachate for COD, BOD_5, TOC, NH_4-N, and Cl are above the limit values. The other leachate parameters generally comply with the discharge standards, with only the maximum concentrations of AOX and Cr being above the limits. Therefore, NH_4-N, COD, and Cl (as conservative migrant with elevated concentrations in the leachate) are selected as leachate parameters for the emission modeling at Landfill B.

Table 6-13: Statistical parameters of selected leachate properties of Landfill B during operation and after closure of the Landfill

Parameter	Operation period				Post closure				Limit*
	Mean	Median	Min.	Max.	Mean	Median	Min.	Max.	
pH	7.90	7.90	7.30	8.40	7.71	7.80	6.40	8.20	6.5-8.5
EC [µS/cm]	8542	7020	3220	16530	6308	5790	3560	12600	-
COD [mg O_2/l]	1986	1453	445	6450	965	720	220	2700	50
BOD [mg O_2/l]	445	327	43	13500	94.9	72.5	25.0	360.0	10
TOC [mg/l]	460	242	22	1500	459	445	160	1100	25
TNb [mg/l]	-	-	-	-	500	460	220	1100	
NH_4-N [mg/l]	629	427	38	1800	482	420	220	1100	10
NO_3-N [mg/l]	42.0	16.2	0.1	324.4	3.9	3.5	1.1	16.0	-
Sulfate [mg/l]	349.3	317.5	31.7	1230.0	148.4	120.0	25.0	480.0	-
Chloride [mg/l]	431	364	71	1060	301	210	84	940	Toxicity
AOX [µg/l]	594	570	1	1806	330	300	120	1100	500
Zn [mg/l]	0.22	0.13	0.05	0.85	0.08	0.04	0.02	0.28	0.50
Cr [mg/l]	-	-	-	-	0.19	0.12	0.06	0.60	0.50
Cu [mg/l]	-	-	-	-	0.05	0.02	0.01	0.49	0.50
Ni [mg/l]	-	-	-	-	0.04	0.04	0.02	0.11	0.50
Cd [mg/l]	-	-	-	-	<0.001	<0.001	<0.001	<0.001	0.10
Pb [mg/l]	-	-	-	-	<0.01	<0.01	<0.01	<0.01	0.50
Potassium [mg/l]	513	480	200	948	172	155	97	350	-
Sodium [mg/l]	1474	478	5	13800	212	190	140	420	-

*Limit values for direct discharge into rivers according to the Austrian directive on leachate discharge (2003) and the Austrian directive on waste water discharge (1996), respectively.

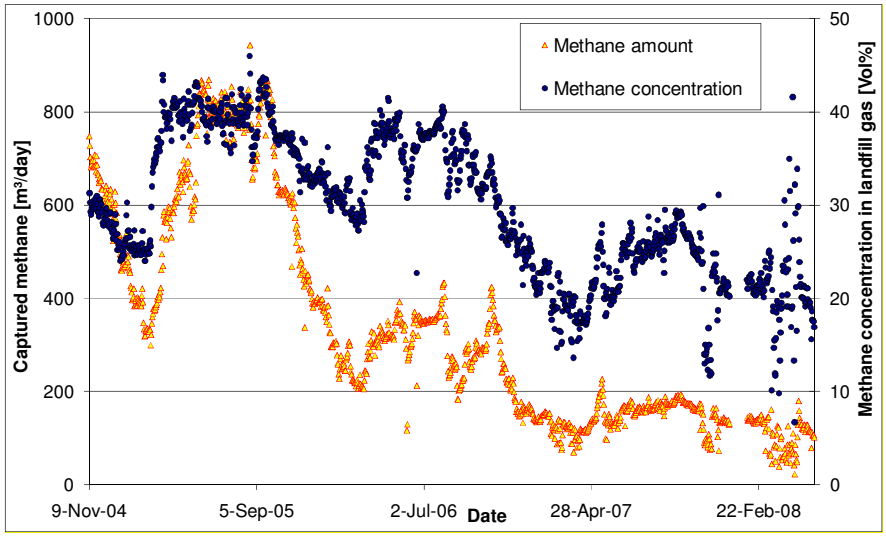


Figure 6-11: Landfill gas monitoring data for the amount of methane collected and the methane concentrations

In contrast to the good quality of available leachate data, landfill gas monitoring was carried out only over a comparatively short period and the collected data are associated with high uncertainties (e.g. fraction of landfill gas captured, amount of air sucked into the gas collection system). Although there has been a gas collection system with a flare in operation since the mid of the 1990s, data on the amount and composition of the landfill gas are available only for the period after the installation of the temporary cover. The measurements of the captured amount of methane and its concentration in the landfill gas are shown in Figure 6-11. A decreasing trend of the amount and

the concentration of methane in the landfill gas can be observed and the influence of different climatic conditions throughout the year is indicated by the presented landfill gas data in Figure 6-11. Nevertheless, the limited data availability makes it difficult to estimate the landfill gas generation rates for Landfill B.

Containment system

The landfill containment system is currently made up of the barrier system at the landfill base and the temporary cover system at the top of the landfill (see Table 6-14). The temporary cover allows on average 62% of the precipitation to infiltrate into the landfill body, which corresponds to an infiltration rate of 1221 mm per year. It is planned to install a final cover at the site in 2020, which includes a composite lining system (cf. Table 6-14) and is supposed to reduce the fraction of precipitation infiltrating into the waste to 1 % (observed performance of similar systems, see chapter 4.1). No data are available on the efficiency of the base lining system. However, as there are no indications of poor barrier performance at the monitoring wells downstream of the landfill, it is assumed that currently 99 % of the generated leachate is contained within the landfill, collected, and treated.

Table 6-14: Area and waste volume of Landfill B and design of the containment system

Area [m²]	Volume [m³]	Landfill base	Landfill cover
12200	135000	Clay liner (3 layers (each 0.2 m), k_f<10^{-9} m/s) + HDPE membrane (2 mm) + Geotextile + Gravel drainage layer (0.5 m) + Geotextile	**Temporary cover:** Soil layer (1.2 m) + Compost layer (0.25 m as recultivation layer) **Final cover:** Low permeability soil layer (k_f<10^{-9} m/s) + HDPE membrane + Drainage layer + Recultivation layer

6.2.2 Emission predictions

To estimate potential future emissions of landfill leachate and landfill gas, several scenarios are developed. The scenarios are used to illustrate the effect of different barrier performance levels after the end of landfill aftercare on landfill emission levels. The basic emission model (leachate quality as a function of liquid-to-solid ratio) is the same for all the investigated scenarios (i.e. no change of water flow pattern and constant release mechanisms). The scenario layouts are the same as for Landfill A (see Table 6-3), but with different infiltration rates corresponding to the described barrier performance levels at Landfill B. The status quo assumes constant barrier function at the top and at the bottom of the landfill. Scenario A considers the gradual decrease of barrier function at the top and bottom of the landfill due to the slow deterioration of the barriers. Scenario B is designed to illustrate the effect of complete barrier failure on landfill emission levels, which represents the "worst case" with respect to technical barrier performance. In addition, modified versions of the status quo and Scenario A are analyzed to investigate the effect of an inefficient technical barrier at the landfill base. Under the assumption that leachate collection is impeded (e.g. through clogging of drainage system) and liner systems fail, all the generated leachate is released to the subsurface below Landfill B. The corresponding scenarios are the status quo* and Scenario A* (see also Table 6-3). The modeling period for all scenarios is 300 years.

All the investigated scenarios assume constant release mechanisms and water flow patterns in the deposited waste and are designed to analyze emission levels for different infiltration rates at the top and percolation rates at the bottom of the landfill. Although the installation of a top cover might have an effect on the water flow pattern in the waste (cf. Laner et al. 2011b), this is not expected for Landfill B, as the final cover will be placed above the temporary cover without removing the latter. Obviously, further scenarios could be evaluated with respect to potential landfill emission levels. However, a flooding of the landfill can be ruled out due to its location on the hillside, an aeration of the landfill is not expected within many centuries, and unintentional invasion should be forestalled by appropriate after-use of the site.

The time of evaluation for Landfill B is the year 2010, as this is the last year for which monitoring data is available. Although the scenario-based emission estimates include the time of temporary cover (which is an aftercare measure), emission estimates are shown and discussed from year 10. Hence, the estimated emission levels correspond to the (planned) final state of the landfill, after the final top cover has been installed[21].

6.2.2.1 Leachate emissions

Emission model

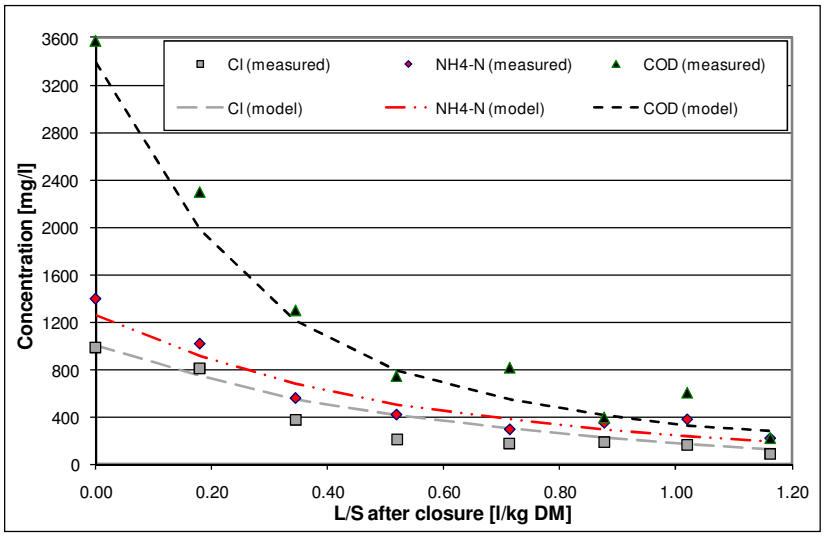


Figure 6-12: Concentrations of Cl, NH_4-N, and COD in the leachate of Landfill B after closure as a function of liquid-to-solid-ratio

Leachate emissions of ammonia-nitrogen, chemical oxygen demand, and chloride are estimated. The emission model is based on literature data for mobilizable waste fractions and initial concentrations as well as water flow heterogeneity estimated based on the available monitoring data after landfill closure (see Figure 6-12 and Table 6-15). Due to the high water throughput (L/S ratio increased by 1.15 liters per kg of dry waste within 5 years after closure), the concentrations in the leachate have been decreasing rapidly and are currently (2010) in the range of 100 mg/l for Cl

[21] The "bleeding out" of the landfill after final cover installation due to water stored in the waste and slow drainage is not considered in the modeling, as this of minor importance in view of the long modeling period.

and 220 mg/l for NH_4-N and COD. The model functions shown in Figure 6-12 are based on the model parameters listed in Table 6-15. The emission models form the basis for the emission scenarios discussed below.

Table 6-15: Model parameters to estimate concentrations of Cl, NH_4-N, and COD in the leachate of Landfill B (c_0 refers to the time of closure)

Parameter	c_0 [mg/l]	m_s [mg/kg]	h [-]	$c_{0,org}$ [mg/l]	m_{org} [mg/kg]
Cl [mg/l]	1000	1600		-	-
NH_4-N [mg/l]	1100	1400	3	154	1000
COD [mg/l]	3000	1800		385	2500

Model formulation: $$c(t) = c_0 \bullet e^{-(\frac{c_0}{m_s} \bullet \Delta \frac{L}{S} \bullet h) \bullet t} + c_{0,org} \bullet e^{-(\frac{c_{0,org}}{m_{org}} \bullet \Delta \frac{L}{S} \bullet h) \bullet t}$$

Status quo

Within the status quo the fraction of precipitation entering the waste body is 0.62 during the period of temporary cover and 0.01 after the final cover has been installed (after year 10). At the base of the landfill 99 % of the generated leachate is constantly removed via the collection system, the rest percolates through the technical barrier to the subsurface.

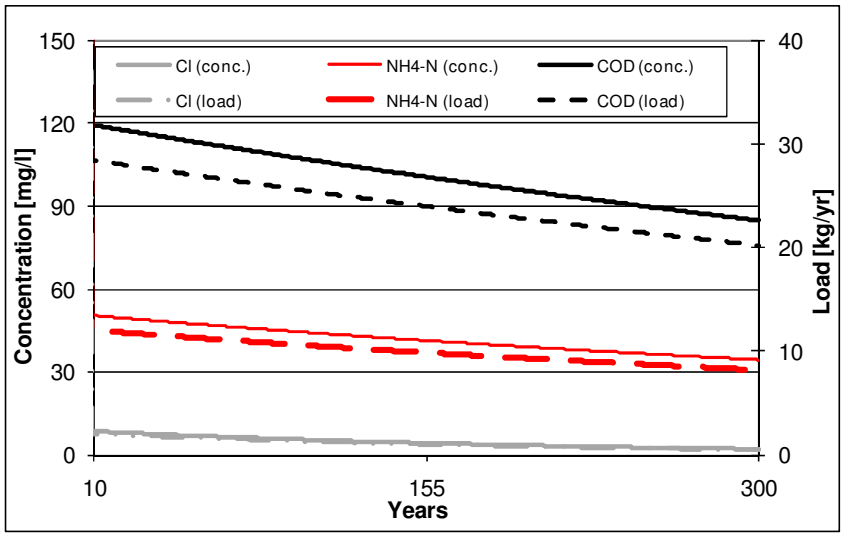


Figure 6-13: Leachate emissions for the status quo (constant barrier performance) at Landfill B

The total leachate emissions resulting from the models in Table 6-15 and the barrier performance levels described above, are shown in Figure 6-13. The emission projections start after year 10, as the period of temporary cover is not relevant with respect to the termination of aftercare (it is only a temporary measure). Because of the permanently low water infiltration rate, the concentrations of the leachate parameters decrease slowly and fall from 9 mg Cl/l, 50 mg NH_4-N/l, and 120 mg COD/l after 10 years to 2 mg Cl/l, 34 mg NH_4-N/l, and 85 mg COD/l at the end of the modeling period. Similarly, the emission loads decrease from 2 kg Cl/yr, 12 kg NH_4-N/yr, and 29 kg COD/yr to 0.5 kg Cl/yr, 8 kg NH_4-N/yr, and 20 kg COD/yr after 300 years. The emission levels to the sub-surface are constantly 1 % of these loads due to the assumption of constant barrier performance.

The emissions into the subsurface for the status quo* are the same as the total leachate emissions for the status quo in Figure 6-13, as the complete inefficiency of the barrier system at the landfill base is assumed during the status quo*.

Scenario A

Scenario A is based on the gradual deterioration of the technical barrier systems and the consequent decrease in performance levels. Nevertheless, during the first ten years (temporary cover) a constant infiltration rate of 1221 mm/yr and constant barrier efficiency at the landfill base are assumed. After the installation of the final cover, a slow decrease in barrier performance is assumed. The evaluation of the final containment system (see Table 6-14) is carried out based on the same criteria as for Landfill A, with evaluation scores according to the conditions at the site and the design of the final containment system shown in Table 6-16. The initial infiltration rate after final cover installation (year 10) is 19.57 mm/yr, then it increases to 100 mm/yr within 100 years and to 323 mm/yr after 300 years (the modeling period ends 290 years after top cover installation due to 10 years of temporary cover in the modeling period). The corresponding percolation rates at the landfill base are 0.2 mm/yr initially (after final cover installation), 5.2 mm/yr 100 years later, and 95 mm/yr after 300 years.

The leachate emissions associated with this evolution of the technical barrier performance levels are shown in Figure 6-14. Cl, NH_4-N, and COD concentrations decrease faster with increasing duration of the scenario due to the increasing water infiltration rates. The substance loads in the generated leachate reach the maximum levels after 60 years for Cl, after 110 years for NH_4-N, and after 125 years for COD (see Figure 6-14, left). With respect to the emission loads into the subsurface it takes 85 years for Cl and 240 years for NH_4-N and COD to reach the maximum levels (see Figure 6-14, right). The annual loads released to the subsurface peak at 0.16 kg Cl/yr, 6.1 kg NH_4-N/yr, and 15.2 kg COD/yr.

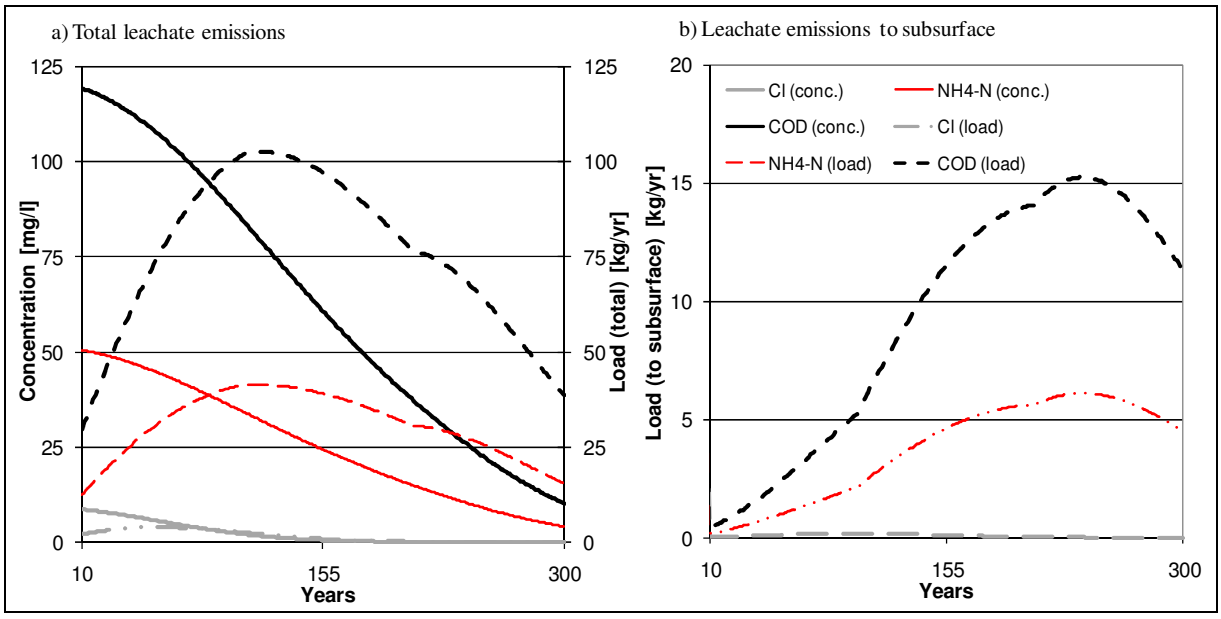


Figure 6-14: Leachate emissions for Scenario A (gradual barrier deterioration) at Landfill B (left: total leachate emissions, right: leachate emissions to subsurface)

Table 6-16: Evaluation of the containment system of Landfill B to estimate future service levels within Scenario A

Factors – top cover	**Evaluation score***	**Weighting factors**		
	1_{good}–3_{bad}	0–100 yrs	100–200 yrs	200–300 yrs
Barrier performance at time of evaluation	**1**	0.36	0.08	0.065
Construction quality program	**1**	0.38	0.44	0.12
Recultivation layer (cover)	**1**	0.33	0.48	0.487
Heat production within waste	**1**	0.09	0.06	0.04
Climate	**2**	0.08	0.12	0.487
Typical vegetation and projection after use	**1**	0.09	0.48	0.487
Relief	**1**	0.08	0.1	0.12
Expected settlements	**1**	0.38	0.09	0.065
Lining control systems (direct monitoring)	**3**	0.1	0.06	0.05
Drainage system	**1**	0.1	0.09	0.08
	Evaluation results:	**2.27**	**2.24**	**2.59**
	Leachate generation rates [mm/yr]:	**100.7**	**167.9**	**322.7**
Factors – bottom lining system				
Barrier performance at time of evaluation	**1**	0.44	0.08	0.07
Construction quality program	**1**	0.46	0.49	0.44
Climate	**2**	0.07	0.08	0.09
Heat production within waste	**1**	0.09	0.06	0.05
Distance to groundwater	**1**	0.08	0.08	0.09
Overburden pressure	**1**	0.08	0.08	0.08
Leachate quality	**1**	0.08	0.08	0.08
Monitoring (e.g. groundwater monitoring)	**1**	0.08	0.07	0.07
Drainage system	**1**	0.46	0.11	0.1
Stability of foundation	**1**	0.1	0.44	0.46
Landfill geometry (heap vs. cavity)	**2**	0.08	0.43	0.48
	Evaluation results:	**2.17**	**2.51**	**2.58**
	Leachate percolating through landfill base [mm/yr]:	**5.2**	**29.0**	**94.5**

*Specific criteria are used to derive evaluation scores: for instance, a score of 1 for "barrier performance at the time of evaluation" means that the barrier has performed according to its design specifications, a score of 2 is awarded if there is no direct monitoring of the barrier function, but it is probably performing according to the design specifications, whereas a score of 3 means that it has not achieved its intended service level at the time of evaluation.

**Weighting factors were derived from expert evaluations during a survey on the importance of different factors for barrier performance during different service periods (cf. chapter 4.1.4, chapter 5.2, and appendix 2).

In case of Scenario A*, the total leachate emissions of Scenario A, see Figure 6-14 on the left, equal the leachate emissions to the subsurface. Scenario A* is used to illustrate the effect of a gradual decrease of the cover performance together with the complete inefficiency of the technical barrier system at the landfill base on landfill emissions.

Scenario B

Within Scenario B the top lining system and the bottom lining system fail to provide containment of the waste. The "worst case" infiltration rate at the top is 1221 mm/yr, which is the rate observed during the period of temporary cover. At the bottom of the landfill all the leachate is released to the subsurface during this scenario.

The emission estimates for Scenario B are shown in Figure 6-15. Due to the high water throughput, the concentrations and annual loads of the leachate parameters decrease rapidly and all of them range below 10 mg/l already after less than 50 years. The modeling period is 300 years for this sce-

nario, but results are only shown until year 110, as the emissions decrease to very low levels within a few decades.

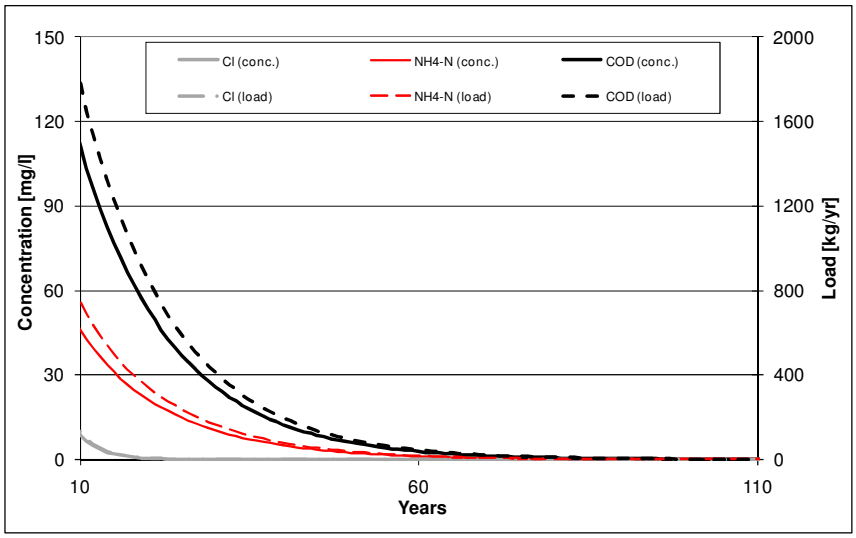


Figure 6-15: Leachate emissions for Scenario B (complete failure of barriers) at Landfill B

6.2.2.2 Landfill gas emissions

Emission model and predictions

Monitoring data on landfill gas emissions are available only for a short period of time and associated with large uncertainties due to the unknown collection efficiency at the site, the lack of data to estimate the amount of air sucked into the landfill gas collection system, and the lack of information about the amount of degradable organic matter in the waste. Thus, two gas generation models with different parameters are used to estimate future gas generation rates. Both models result in landfill gas generation rates, which are above the observed amounts of landfill gas, even if a collection efficiency of 35% is assumed at the site (see Figure 6-16). Hence, the model estimates are rather conservative, whereby the LandGEM model estimates are derived from typical parameters of wet landfills (k-value of 0.7 and gas generation potential 192 m³ per Mg of wet waste) and the model by Tabasaran and Rettenberger (1987) (=T&R model) is calculated based on a k-value of 0.1 and a potential gas generation potential of 196 m³ per Mg of wet waste (cf. Table 3-1). The future gas generation rate is supposed to range somewhere between the model estimates. Even after final cover installation, when water infiltration has been minimized, there should be sufficient water for continued gas generation in the landfill for some time. According to Figure 6-16, it can be expected that the gas generation rates at the landfill will drop to very low levels within a few decades. Based on the T&R model gas generation would be around 60,000 m³ of landfill gas/yr by the year 2020 and below 6,000 m³ of landfill gas/year by 2030.

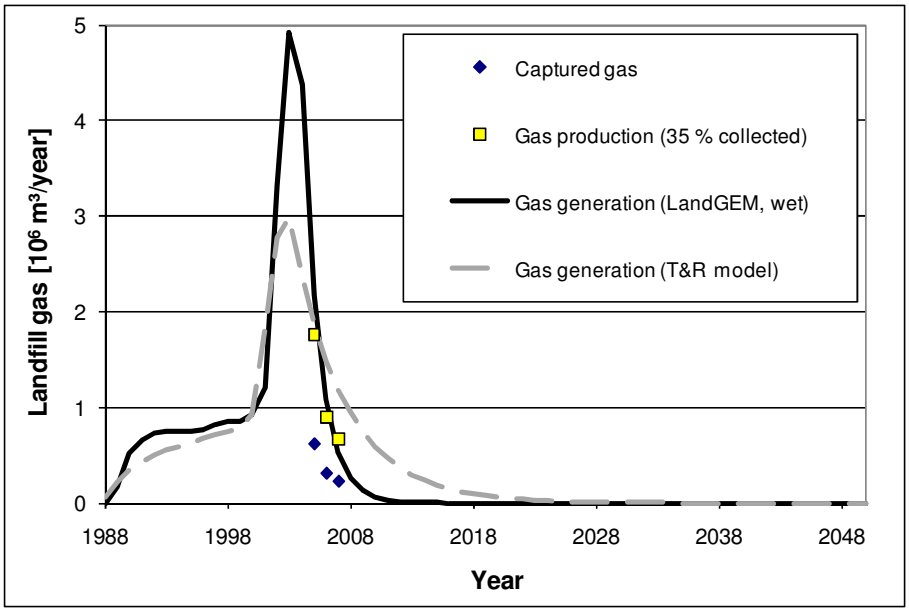


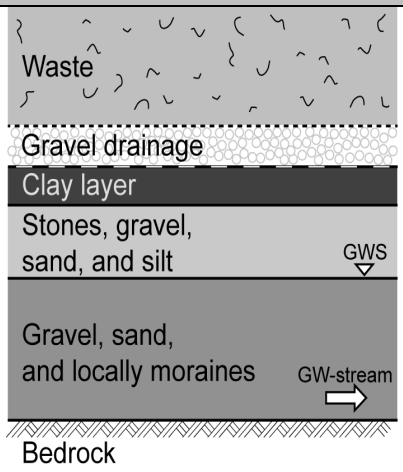
Figure 6-16: Captured landfill gas data at Landfill B and estimates on future landfill gas generation based on different gas generation models

6.2.3 Pollutant migration

Leachate migration

The investigated migration pathway for leachate in the subsurface includes the earthen barrier at the bottom of the landfill, the vadose zone below the landfill, the local groundwater, and the river, which is also fed with groundwater potentially affected by leachate emissions. An outline of the soil properties of the vadose zone and the aquifer below Landfill B is provided in Table 6-17. The soil below the landfill consists mainly of gravel, mixed with varying portions of fine particles. Stones and silty soil have been used to create a plain before the construction of the bottom liner at the site. As these materials have been mixed with the natural ground, the foundation layer is approximated as the layer "Stones, gravel, and sand" in Table 6-17. The low permeability soil layer of the base lining system is included in the substance migration modeling, because it is supposed to function over long time periods and may slow down the transport of pollutants in the subsurface. The geomembrane is not included in the transport modeling, as its function is evaluated in the emission scenarios. The minimum distance from the clay liner to the groundwater surface is 1.2 meters. However, due to the large fluctuations in groundwater surface levels, the thickness of the unsaturated zone will be higher at most times. The thickness of the groundwater below the landfill is only a few meters and the groundwater surface has a strong gradient downhill (see Figure 6-17). The average groundwater velocity is estimated to be 300 meters per year, which is probably at the lower range of realistic velocities due to the high hydraulic gradient of the groundwater (> 0.1 m/m). In the valley the groundwater thickness increases (PoC3 in Figure 6-17). This groundwater body is of potential significance as a source of drinking water for a nearby village and it is partially draining into the river located in the valley (PoC4 in Figure 6-17).

Table 6-17: Layout of the subsurface below Landfill B and material properties of the soil layers

Layout of the subsurface below the Landfill B	**Thickness, grain size distribution and soil type** (according to AG Boden 2005)	**Hydraulic conductivity, field capacity, pore volume, and dry bulk density**
	**Clay layer:** Thickness: 0.6 m; 40 – 60 % sand, 30 – 40% silt, 8 - 15% clay (soil type: Sl3)	$k_f < 10^{-10}$ m/s field capacity: 25 % total pore volume: 35 % dry bulk density: 2.1 g/cm³
	Stones, gravel, and sand: Thickness: 1.2 m; (soil type gS);	$k_f \approx 10^{-5}$ m/s field capacity: 6 % total pore volume: 35 % dry bulk density: 1.6 g/cm³
	Gravel and sand (aquifer): Thickness: < 5 m; Carbonatic material (lenses of fine grained material possible);	$k_f \approx 10^{-4}$ m/s

The points of compliance where the pollutant concentrations are determined, are shown in Figure 6-17. After the release of the leachate, the resulting maximum concentrations are calculated above the groundwater surface (PoC1), in the mixing zone of leachate and groundwater (PoC2) with an average thickness of 0.25 meters, in the groundwater body in the valley after a transport distance of 100 meters (PoC3), and in the affected river water (PoC4). The corresponding transport parameters for the leachate pollutants and some data on the groundwater conditions are shown in Table 6-18. The contaminated width of the groundwater body is assumed to equal the width of the landfill perpendicular to the groundwater stream. However, in reality a release of leachate will probably not occur at the same rate all over the landfill base, but there will be some "hot spots" were leachate escapes from the containment systems through cracks and flaws. Therefore, it is emphasized that the model calculations illustrate the effect of emissions on the environmental compartment "local groundwater". Clearly, higher concentrations could be encountered at specific points due to heterogeneous release patterns. For a screening evaluation, this approach seems justified, especially in view of the generally conservative assumptions within the migration modeling (e.g. vadose zone thickness and composition, groundwater velocity and transport distances, no degradation included in the modeling, etc.).

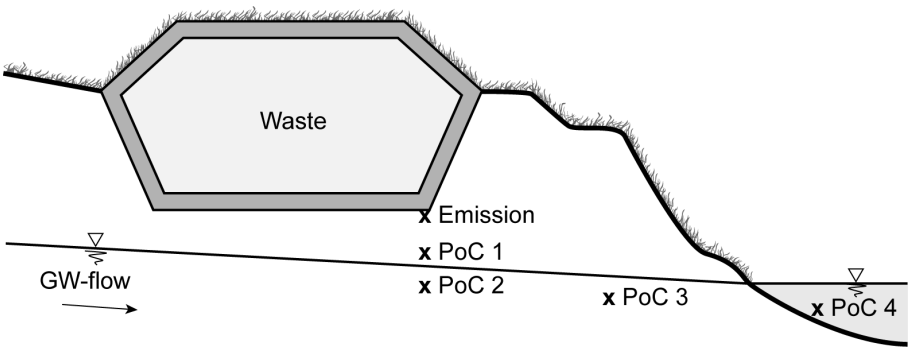


Figure 6-17: Scheme of the landfill-environment system and relevant points of compliance (PoC) at Landfill B (cross-section along the ground water flow direction)

Table 6-18: Parameters used for pollutant migration modeling in the subsurface of Landfill B

Vadose zone	Diffusion coefficient in water [m²/yr]	k_d-value [l/kg] (clay layer)	k_d-value [l/kg] (stones, gravel & sand)	Comments
Cl	0.0315	0	0	No degradation
NH_4-N	0.0378	0.5	0	No degradation
COD*	0.0315	0	0	No degradation
Groundwater	Groundwater velocity [m/yr]	Mixing depth [m]	Contaminated width [m]	Comments
General parameters	300	0.25	100	No degradation **

* Transport parameters for COD are the same as for Cl.

** The dissolved oxygen content in the groundwater is used as an indicator to estimate the potential for aerobic degradation of organic substances and for the nitrification of ammonia.

The emission scenario estimates are investigated after the installation of the final cover (starting from year 10). The polltuant migration modeling for Cl and NH_4-N is based on the AF model (Schneider and Stöfen 2004) for scenarios with constant leachate generation and release rates (see Table 6-19). For scenarios with varying leachate generation rates (Scenario A and A*), the maximum emission load to the subsurface is used to evaluate pollutant migration according to a mass balance apporach (see Table 6-20). The mass balance approach using maximum loads is also employed to estimate COD[22] migration, whereby the maximum loads into the subsurface are averaged over the estimated travel times in the vadose zone for scenarios with constant leachate generation rates. The concentrations in the leachate at the time of maximum emission loads equal the concentrations in the vadose zone above the groundwater surface (PoC1). In Table 6-19 and Table 6-20 maximum concentrations at PoC1 and PoC2 (groundwater mixing zone) are shown together with major calculation parameters. The migration modeling does not include degradation processes and considers sorption only to a very limited extent (ammonia cations are exchanged in the clay layer). Hence, concentration reductions are mainly caused by hydrodynamic dispersion and dilution processes.

Table 6-19: Data for the AF-model calculations to estimate concentrations at PoC1 and PoC2

Scenario (after final cover)	Parameter	$c_{0,max}$ [mg/l]	Release rate [mm/yr]	AF [-]	Travel time t_m [yrs]	c_{PoC1} [mg/l]	GW flow [m³/yr]	AF_{GW} [-]	c_{PoC2} [mg/l]
Status quo	Cl	9	0.2	8.0	1110	1.1	7500	3080	0.0
Status quo	NH_4-N	50.5	0.2	8.4	4260	6.0	7500	3080	0.0
Scenario B	Cl	9	1221	1.0	0.2	9.0	7500	1.5	6.0
Scenario B	NH_4-N	50.5	1221	1.0	0.7	50.5	7500	1.5	33.6
Status quo*	Cl	9	19.6	1.0	11.0	9.0	7500	32.5	0.3
Status quo*	NH_4-N	50.5	19.6	1.0	43.5	50.5	7500	32.5	1.6

[22] For more specific modeling of COD, migration data on the composition of organic leachate pollution would be necessary. As such information is not available, COD is modeled with the same parameters as Cl (it is conservatively assumed that COD transport is not retarded).

Table 6-20: Data for the calculations based on maximum emission loads into the subsurface to estimate concentrations at PoC1 and PoC2

Scenario (after final cover)	Parameter	$C_{0,max}$ [mg/l]	Release rate [mm/yr]	Load [kg/yr]	Travel time t_m [yrs]	c_{PoC1} [mg/l]	AF [-]	AF_{GW} [-]	C_{PoC2} [mg/l]
Status quo	COD	121.0	0.2	0.2	1110	100	1.2	3080	0.03
Scenario A	Cl	9.0	5.6	0.2	111	2.3	3.9	112	0.02
Scenario A	NH_4-N	50.5	43.6	6.1	25	11.5	4.4	15.1	0.8
Scenario A	COD	121.0	47.7	15.2	5.0	26.2	4.6	13.9	1.9
Scenario B	COD	121.0	1221	1800	0.2	121	1.0	1.5	80
Status quo*	COD	121.0	19.6	28.5	10	120	1.0	32.5	3.7
Scenario A*	Cl	9.0	65.4	3.9	3.4	4.9	1.8	10.4	0.5
Scenario A*	NH_4-N	50.5	102.7	41.5	7.8	33.2	1.5	7.0	4.8
Scenario A*	COD	121.0	110.8	102.7	2.0	76.1	1.6	6.6	11.6

[a] Average travel time of COD in the vadose zone is assumed the same as for Cl.

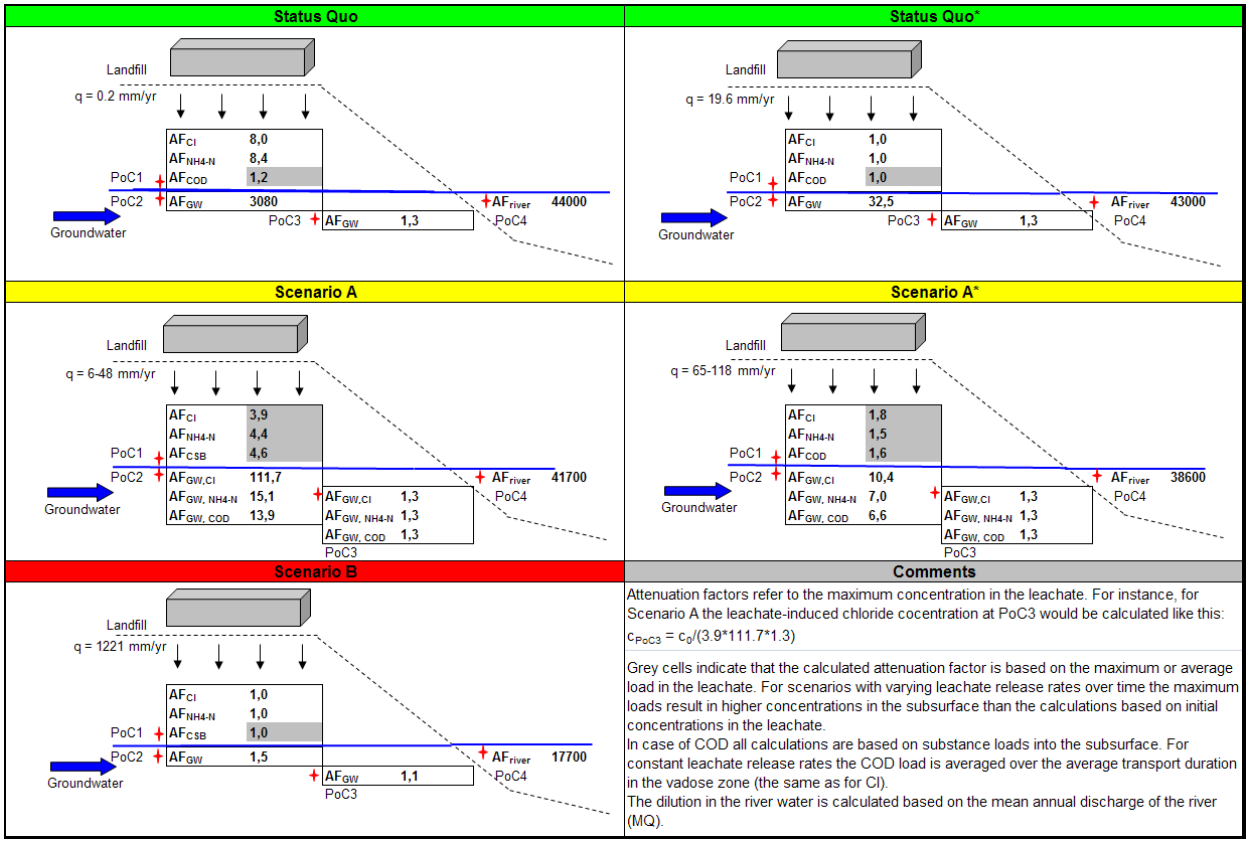


Figure 6-18: Attenuation factors to estimate the concentrations at relevant PoCs due to a release of leachate from Landfill B for the different scenarios

The calculations shown in Table 6-19 and Table 6-20 are used to estimate attenuation factors to determine the reduction of pollutant concentrations along the transport pathway. The attenuation factors to evaluate the migration of the leachate contaminants (Cl, NH_4-N, and COD) in the subsurface are shown in Figure 6-18. To calculate the concentrations at PoC3 (100 m downstream the landfill in the groundwater) the following assumptions are met: no spreading of the leachate plume (width 100 m and thickness 0.25 m) and dilution in the groundwater is caused solely by the groundwater recharge rate of 250 mm/year (the low recharge rate of 13% of the precipitation is

because of the surface slope and the partly sealed area). The concentration at PoC4 is calculated under the assumption that all the contaminated groundwater exfiltrates into the river and fully mixes with the river water (the calculations are based on the mean annual discharge). In case of varying leachate release rates, the lowest dilution factor is used as an attenuation factor in Figure 6-18.

Landfill gas migration

The predicted landfill gas emissions for Landfill B are associated with substantial uncertainties. Therefore, the bandwidth of potential emissions in Figure 6-16 is large. Based on the adapted landfill gas generation models, potential methane emissions are evaluated for the Landfill B. The resulting methane emission rates per m² of landfill surface (assuming 50 vol% methane content in the landfill gas and a complete release of all the generated methane through the top cover) are shown in Figure 6-19. According to these calculations, even the maximum estimates for methane emissions drop below 1 kg CH_4 per m² and year within 13 years or within 3 years after final cover installation, respectively. 8 years after final cover installation methane emissions will be below 0.3 kg CH_4/(m²*year). Hence, it is expected that methane emissions will decrease to insignificant levels within a few decades at the site even without considering any attenuating effects during gas migration through the top cover (methane oxidation can substantially reduce methane emissions, cf. Table 4-10). The migration of landfill gas in the subsurface is not evaluated, due to a lack of migration pathways (hillside terrain) and receptors (residential areas, basements) in the landfill surroundings.

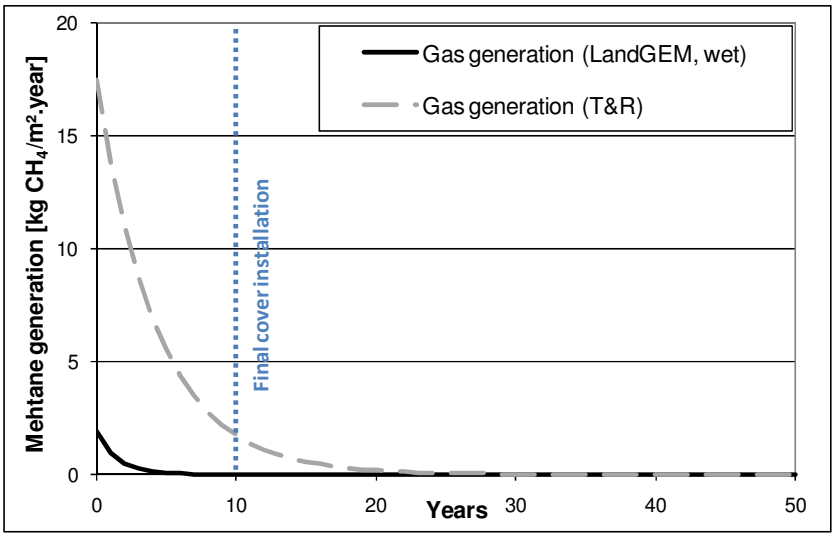


Figure 6-19: Methane generation rates per m² of landfill surface for the two gas generation models applied to Landfill B

6.2.4 Completion criteria and aftercare duration

The migration of leachate pollutants at Landfill B was evaluated with respect to the vadose zone below the site, the groundwater below and downstream of the landfill, and the river in the valley. To evaluate the environmental compatibility of the emissions the groundwater downstream of the landfill is chosen as the critical point of compliance, as the groundwater body in the valley has been identified as a potential source of drinking water for a nearby village. Therefore, the quality

standards for drinking water are applied as limit values for acceptable concentration levels at PoC3 in Table 6-21. It should be noted that the calculations relate to the release of leachate only and do not take into account any other pollution sources or contamination already present in the groundwater upstream of the landfill. However, such data would be necessary to evaluate the total extent of groundwater contamination and the suitability as a drinking water resource.

Table 6-21: Tolerable leachate concentrations (c0) at the source in order not to exceed the limit values specified at PoC3 for the emission scenarios of Landfill B

Parameter	PoC3	Status quo	Scenario A	Status quo*	Scenario A*	Scenario B
	C_{limit} [mg/l]	c_0 [mg/l]	c_0 [mg/l]	c_0 [mg/l]	c_0 [mg/l]	c_0 [mg/l]
Cl [a)]	200	6569744	115795	8591	4954	335
NH_4-N [a)]	0.5	17246	44	21	7	0.8
COD [a)]	5	25110	420	215	67	8.4

a) Limit values for leachate induced concentrations at PoC equal Austrian drinking water quality standards (MoE 2001). In case of ammonia-nitrogen limit values for ammonia are applied at PoC3. No nitrification of ammonia is assumed.

The reverse calculations of tolerable pollutant levels in the landfill leachate shown in Table 6-21 are based on the scenario-specific attenuation factors shown in Figure 6-18. Depending on the critical emission scenario to assess the risk of inacceptable pollution after post-closure care has been terminated, different concentration levels are determined, below which leachate emissions are tolerable in the absence of aftercare. Provided that leachate collection remains functional after landfill completion and a slow deterioration of the base lining system is assumed (Scenario A), the consequent tolerable concentration levels in the leachate would be 44 mg/l for NH_4-N and 362 mg/l for COD. In case of barrier deterioration at the top and inefficient base lining system after aftercare completion (Scenario A*), the corresponding concentrations in the leachate would be limited with 7 mg/l for NH_4-N and 64 mg/l for COD. The aftercare duration until these leachate emission levels can be reached are shown in Table 6-22 for three different aftercare strategies. One is sealing the landfill with constantly optimal barrier performance (as during the status quo), another one is installing a low permeability top cover and tolerating a slow decrease in performance (as during Scenario A), and the third one would be to extend the period of temporary cover with high water infiltration rates until leachate completion criteria can be met. From the necessary management durations estimated in Table 6-22 it is obvious that aftercare durations of a few decades are only realistic at Landfill B if intensive water infiltration is prolonged for a few more decades. The most critical parameter to reach acceptable leachate quality for aftercare termination is again ammonia-nitrogen. This can take from 11 years (intensive water infiltration during after care & Scenario A: slow deterioration of the top cover with functional leachate extraction at the bottom after completion) to more than 300 years (sealing with minimal interaction of waste & Scenario A*: slow deterioration of the top cover with ineffective base lining system after completion).

Whereas leachate emissions of Landfill B will require further aftercare at least for a few decades, the level of acceptable methane emissions of 0.3 kg per m² and year suggested by Fellner et al. (2008) can be reached within a decade. Hence, it can be expected that the leachate emissions will be of primary importance with respect to the termination of aftercare. To arrive at tolerable leachate emissions as soon as possible, the period of temporary cover should be extended at the site.

Depending on the underlying long-term emission scenario different activities will be necessary at the site after the end of aftercare. In any case, further extensive monitoring will be necessary at the landfill (e.g. geotechnical stability, diffuse gas emissions, recultivation layer, etc.) and in the potentially affected environmental media (e.g. groundwater wells) in order to verify model estimates and scenario assumptions.

Table 6-22: Duration of aftercare for leachate emissions of Landfill B in consideration of different aftercare strategies

Parameter	Sealing at the top (constant performance)	Sealing at the top (decrease of performance)	Temporary cover remains (high infiltration)
NH_4-N	**110 - >> 300 yrs**	**60 - 270 yrs**	**11 – 35 yrs**
COD	0 - > 300 yrs	0 – 60 yrs	0 – 20 yrs

In this case study many (conservative) assumptions have been made to evaluate the environmental compatibility of potential landfill emissions. Therefore, the evaluation was carried out on a screening level associated with large inherent uncertainties. This evaluation can be the basis for future monitoring efforts and investigations to fill existing data gaps. Monitoring data can be used to validate the predictions of the emission model and investigate the extent of preferential water flow for different hydraulic boundary conditions. Additional investigations of the mobilizable substance potentials present in the waste should be conducted to gain further confidence into the emission models (leachate and gas). In consequence, based on more reliable model estimates, after-care activities could be optimized to minimize actual and remaining environmental risks at the landfill site.

6.3 Landfill C - closed C&D waste landfill

6.3.1 Site description and monitoring data

Landfill and site conditions

At Landfill C waste was deposited between 1992 and 2008. The material disposed at the site was mainly construction and demolition (C&D) waste and, to a smaller extent, other mineral wastes and contaminated soils. Landfill operation was completed in 2009 and a final cover was installed at the site. The approximate waste volume is 400,000 m³ corresponding to 700,000 tons of waste (density around 1.8 g/cm³) with an average deposition height of 13 meters.

In the vicinity of the landfill are several other landfills including a closed non-organic waste landfill (e.g. deposition of mineral sludges and incineration residues), a closed landfill for contaminated soil, and an operating landfill receiving residues from mechanical biological treatment of MSW. A main road is passing the landfill property in the North and the river Danube is flowing several hundred meters further in the North (see Figure 6-20). The only buildings in the neighborhood of the landfill are located on the property next to the landfill site in the East.

Landfill C is located above a groundwater body of high thickness, which is strongly influenced by the hydraulic regime of the Danube. The groundwater flows in a Northeastern direction towards the Danube river and exhibits significant nitrate pollution levels (agricultural activities in the region).

The local climate is arid with an average annual precipitation of 613 mm/yr and a potential evaporation of 630 mm/yr. The geologic stratum below the landfill consists primarily of sand with varying fractions of gravel, silt, and clay.

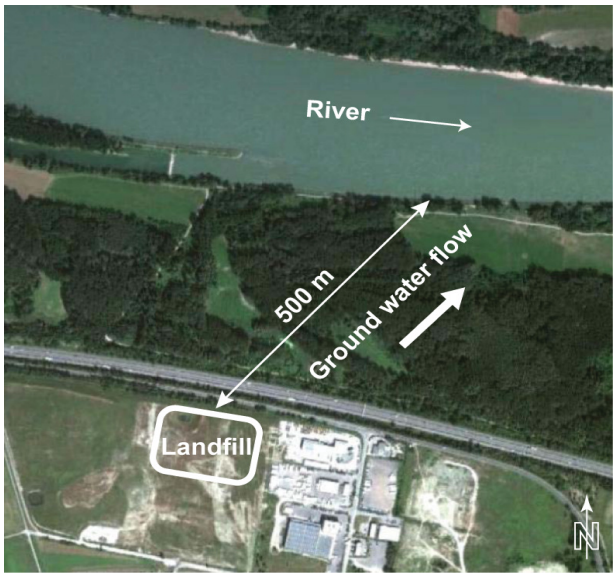


Figure 6-20: Aerial photograph of Landfill C and surroundings

Monitoring data

Landfill C was operated for 16 years. From 1992 to 2000 around 500,000 tons of construction and demolition waste were deposited at the site. In 2001 and 2002 there was almost no deposition of materials and afterwards the rate of landfilling increased again to 20,000 – 50,000 tons per year until 2008.

During the whole operation and after closure of the landfill, leachate was collected at the landfill base. The leachate is pumped out of storage tanks at the landfill bottom into a leachate collection basin on the surface next to the landfill. From the basin the leachate is directed to a leachate treatment facility. Leachate generation rates are estimated from data on pump operation times to pump the leachate from the storage tanks at the base to the collection basin on the surface. As there was leachate recirculated from the basin back into the waste body and as pumping data is not available for all the years, the data on leachate generation rates are associated with large uncertainties. Available data on the leachate levels in the basin (on the surface) are not reliable, because surface runoff could be collected in the basin and evaporation is an important factor. During landfill operation around 16 % of the annual precipitation has been collected as leachate at the landfill base. After closure the leachate generation dropped to approximately 1 % of annual precipitation, but due to the short period (1 year) and the uncertainties mentioned above (measurement of leachate levels in the basin on the surface) this value needs to be verified with future monitoring data.

Leachate samples for chemical analysis were taken from the open leachate basin at the surface once a year and analyzed for 21 parameters. Statistical data on the measurements at Landfill C are shown in Table 6-23. As the leachate quality in the basin might be affected by different factors (e.g. surface runoff diluting leachate in the basin, evaporation concentrating leachate constituents,

aerobic conditions enabling ammonia nitrification, etc.), the measured parameters may not be representative for the leachate quality at the bottom of the landfill. Consequently, the nitrogen compounds are analyzed in aggregate and it is assumed that all the nitrogen in the landfill leachate is present as ammonia-nitrogen in the landfill. Apart from representativeness issues, the monitoring data have also been corrected for outliers. In particular, one Cl measurement was excluded as concentrations were 10 times higher than Cl concentrations at all other times although electric conductivity of the same sample was normal to low. The data in Table 6-23 show that the mean values of all leachate parameters comply with the quality standards for direct discharge into surface waters. Only the maximum concentrations of COD and total nitrogen (sum of NH_4-N, NO_2-N, and NO_3-N) are above the limit values. Therefore, these two parameters will be included in the emission modeling together with sulfate[23] and chloride, as their concentrations are partly above drinking water quality standards, too.

Table 6-23: Statistical parameters for selected leachate properties of Landfill C

Parameter	Mean	Median	Min.	Max.	75-percentile	Limit*
pH [-]	8.04	7.96	7.10	9.20	8.46	6.5–8.5
EC [µS/cm]	2506	2830	100	3680	3510	-
NH4-N [mg/l]	0.2	0.1	0.02	1.2	0.2	10
NO_2-N [mg/l]	0.02	0.01	0.00	0.1	0.03	2.0
NO_3–N [mg/l]	3.8	1.2	>1	15.6	6.1	-
SO4 [mg/l]	784	860	137	1719	1327	-
Sulfide [mg/l]	<0.1	<0.1	<0.01	<0.5	<0.1	0.5
COD [mg O2/l]	26.2	29.5	18.0	93.0	44.0	50
TOC [mg/l]	8.3	9.4	2.1	16.9	14.7	25
Hydrocarbons [mg/l]	<0.01	<0.01	<0.01	0.25	0.25	5.0
PAH [mg/l]	0.00	0.02	0.02	0.02	0.02	
Fe [mg/l]	0.07	0.25	0.03	0.42	0.37	2.0
Mn [mg/l]	0.07	0.19	0.06	0.74	0.42	-
Zn [mg/l]	0.10	0.04	0.01	1.40	0.15	0.5
Ni [mg/l]	0.01	0.09	0.08	0.09	0.09	0.5
Cu [mg/l]	<0.01	<0.01	<0.01	0.01	0.01	0.5
Cd [mg/l]	<0.01	<0.01	<0.01	0.01	<0.01	0.1
Sn [mg/l]	<0.1	<0.1	<0.01	<0.5	<0.1	2.0
Pb [mg/l]	0.01	0.04	0.04	0.04	0.04	0.5
Al [mg/l]	0.4	0.1	0.1	0.8	0.2	2.0
Cl [mg/l]	86.6	115.2	4.0	211.0	209.0	Toxicity

*Limit values for direct discharge into rivers according to the Austrian directive on leachate discharge (2003) and the Austrian directive on waste water discharge (1996), respectively.

The landfill was not equipped with a gas collection system, because no significant gas generation is expected from the deposited waste (cf. chapter 3.3).

Containment system

The landfill containment system consists of a top cover with a low permeability soil liner and a technical barrier at the landfill base also with a low permeability soil liner (see Table 6-24). The current performance level of the top cover cannot be assessed, as there is no reliable monitoring

[23]The deposited waste has a high content of soluble sulfate. Sulfate is potentially highly mobile in the subsurface.

data available at present (see above). Therefore, the top cover system is assumed to allow for an infiltration rate of 5 % of the average annual precipitation (legal requirement), although current monitoring data indicate very low infiltration levels (around 1 % of annual precipitation). No data are available on the efficiency of the base lining system (no direct monitoring or monitoring wells). Hence, instead of measured data, the design performance of 99% leachate collection efficiency is used as the current (and the best) performance level of the base lining system. For both systems, at the top and the base of the landfill, further monitoring data are necessary to assess their actual performance. In particular for the top cover system, future data will show if a fast decrease of barrier performance occurs (e.g. cracks due to desiccation) or if the very low leachate generation rates will persist for a longer period.

Table 6-24: Design of Landfill C and its containment system

Area [m²]	Volume [m³]	Landfill base	Landfill cover
30000	400000	Loamy soil (thickness of 1 m) + low permeability earthen barrier (2 layers (each 0.2 m), $<10^{-9}$ m/S) + drainage layer (0.3 m)	Layer of excavated soil (thickness of 1 m) + low permeability soil layer (0.3 m) + recultivation layer (0.3 m)

6.3.2 Emission predictions

In case of Landfill C, potential future emissions are estimated for landfill leachate only, as there is no significant landfill gas generation at the site. Based on different scenarios, the effects of different interaction levels between the landfilled waste and the surrounding environment on leachate emissions are illustrated. The investigated scenario layouts are identical to those presented for Landfill A (see Table 6-3), but different performance levels of the technical barriers are applied according to the conditions at Landfill C. Therefore, all the scenarios are again based on one emission model (leachate quality as a function of liquid-to-solid ratio), which means that constant conditions are assumed within the landfill body (i.e. no change of water flow pattern and release mechanisms). The status quo is based on the persistence of optimal barrier performance at the top and at the bottom of the landfill. Scenario A assumes a gradual decrease of barrier performance at the top and at the bottom of the landfill due to continuous barrier deterioration. Scenario B is used to illustrate the effect of complete barrier failure on landfill emissions, representing the "worst case" with respect to technical barrier performance. In addition, modified versions of the status quo and Scenario A are investigated to illustrate the effect of an inefficient technical barrier at the landfill base on emission levels to the subsurface. As the leachate generated at Landfill C cannot gravity drain to the leachate collection basin, it is assumed that after aftercare completion (without further pumping) all the generated leachate is released to the subsurface. The corresponding scenarios are the status quo* and Scenario A* (see also Table 6-3). The modeling period for all scenarios is 300 years. The beginning of the modeling period is 2009, as this is the last year for which monitoring data is available.

Landfill flooding is improbable at the site due to area wide flood protection (designed for 1000 year recurrence interval flood), the water flow pattern of the waste body is expected to be stable (soil-like material) if the landfill remains undisturbed (i.e. no invasion), and a scenario of landfill erosion will not be significant within manageable post-closure care periods. Therefore, the analysis of

potential future leachate emissions is limited to different technical barrier performance levels at Landfill C.

Emission model

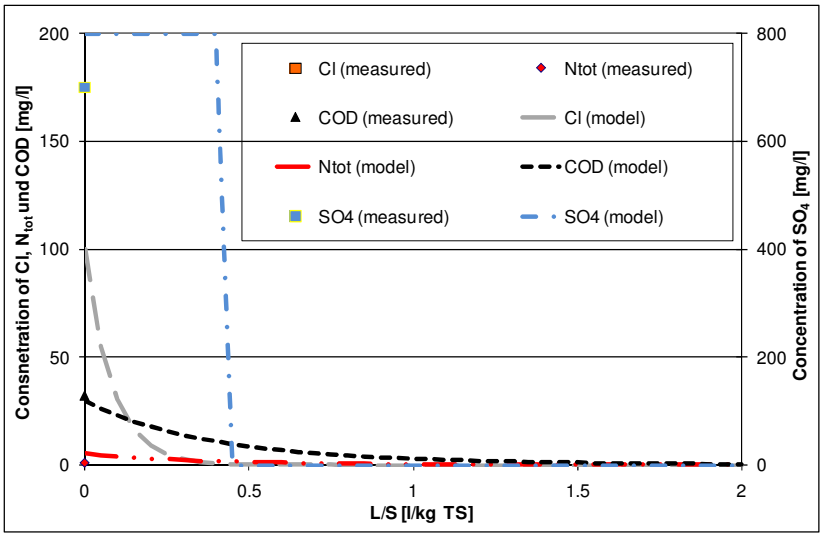


Figure 6-21: Concentrations of Cl, NH_4-N, COD, and SO_4 in the leachate of Landfill C after closure as a function of liquid-to-solid ratio

Table 6-25: Model parameters to estimate concentrations of Cl, NH_4-N, COD, and SO_4 in the leachate of Landfill C (c_0 refers to the time of closure)

Parameter	c_0 [mg/l]	m_s [mg/kg]	h [-]	$c_{0,org}$ [mg/l]	m_{org} [mg/kg]
Cl [mg/l]	100	50		-	-
N_{tot} [mg/l]	3	8	6	2.3	4
CSB [mg/l]	15	50		14.6	25
Model formulation: $c(t) = c_0 \bullet e^{-(\frac{c_0}{m_s} \bullet \Delta \frac{L}{S} \bullet h) \bullet t} + c_{0,org} \bullet e^{-(\frac{c_{0,org}}{m_{org}} \bullet \Delta \frac{L}{S} \bullet h) \bullet t}$					
SO_4 [mg/l]	800	2000	6	-	-
Model formulation: $c(t) = c_0$ for $t \bullet \Delta \frac{L}{S} \leq \frac{(\frac{m_s}{h})}{c_0}$; $c(t) = 0$ for $t \bullet \Delta \frac{L}{S} > \frac{(\frac{m_s}{h})}{c_0}$					

Potential future leachate emission levels of Landfill C are estimated for total nitrogen (modeled as ammonia-nitrogen), chemical oxygen demand, chloride, and sulfate. As there are no time series data for the concentrations of these parameters in the leachate after landfill closure (only one measurement), the initial concentrations of the emission models are assumed the same as the mean concentrations observed during landfill operation. The basic relationships between leachate concentration levels of the investigated parameters and the liquid-to-solid ratio (L/S ratio) are shown in Figure 6-21. The corresponding model parameters and the model formulation are presented in Table 6-25, whereby an exponential decrease in concentration is modeled for Cl, NH_4-N, and COD and constant concentrations are predicted for SO_4 until the soluble fraction of SO_4 in contact with

mobile water has been emitted. The mobilizable substance fractions in Table 6-25 have been derived from leaching tests with the deposited material at the site (waste acceptance procedures). The high heterogeneity of the water flow (h=6 in Table 6-25) could be due to the low hydraulic conductivity of the deposited waste and the highly heterogeneous waste structure (i.e. large variance of grain sizes). However, it should be noted that the heterogeneity of water flow is an assumption (in accordance with monitoring data during operation) and that the hydraulic regime in the waste should be investigated to collect site-specific data.

Status quo

The best performance level of the top cover system allows for an infiltration of 5 % of the average annual precipitation into the waste body. Thus, the current infiltration rate is around 31 mm/yr and constant throughout the whole modeling period, assuming no degradation of the technical barrier at the top. Similarly, the base lining system constantly retains 99 % of the generated leachate, which is directed to the leachate collection basin. Therefore, the leachate release rate to the subsurface is 0.31 mm/yr.

The emission levels at Landfill for the status quo are shown in Figure 6-22. The initial concentrations comply with the direct discharge quality standards (cf. Table 6-23) and are continuously decreasing for Cl, NH_4-N, and COD until the end of the modeling period. For SO_4, the concentrations remain at 800 mg/l until the mobilizable sulfate is emitted and then, after 295 years, the concentrations drop to 0 mg/l. The corresponding annual sulfate loads in the leachate are 756 kg/yr during most of the modeling period and 0 kg/yr during the last 5 years. The loads of the other leachate parameters are 92 kg Cl/yr, 5 kg N/yr, and 27 kg COD/yr at the beginning and decrease to 0.6 kg Cl/yr, 1.5 kg N/yr, and 9.4 kg COD/yr after 300 years. The leachate emissions to the subsurface during the status quo are the same as in Figure 6-22 in terms of concentrations, but only 1 % of the substance loads.

The status quo* is established to investigate the effect of the complete inefficiency of the base lining system on the emissions to the subsurface. Hence, the total emission levels for the status quo, shown in Figure 6-22, are identical to the emissions into the subsurface throughout the status quo*.

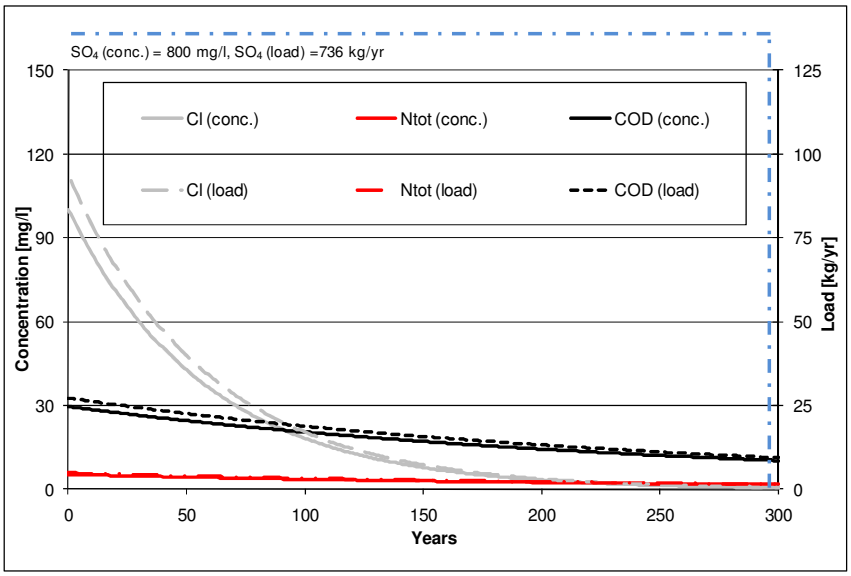


Figure 6-22: Leachate emissions for the status quo (constant barrier performance) at Landfill C

Scenario A

Scenario A illustrates the effect of gradually decreasing barrier performance on landfill emission levels. Barrier efficiency is delimited by the actual system performance (the best possible service level without future maintenance) and the complete inefficiency of landfill liners (the worst service level). Thus, the initial infiltration rate through the top cover is 30.6 mm/yr and is expected to increase within 100 years to 54.4 mm/yr, based on the evaluation of future barrier performance summarized in Table 6-26. At the end of the modeling period, the infiltration rate is estimated to be around 80 mm/yr, which is close to the worst possible barrier function of approximately 100 mm/yr. However, as top cover systems with a sole low permeability soil liner often show a rapid decrease of barrier performance within the first years after cover installation (e.g. Bonaparte et al. 2002; Henken-Mellies and Schweizer 2011 or chapter 4.2), future monitoring data will be essential to evaluate the rate and extent of barrier deterioration at Landfill C.

Table 6-26: Evaluation of the containment system of Landfill C to estimate future service levels within Scenario A

Factors – top cover	**Evaluation score***	**Weighting factors****		
	1_{good}–3_{bad}	0–100 yrs	100–200 yrs	200–300 yrs
Barrier performance at time of evaluation	**2**	0.36	0.08	0.07
Construction quality program	**1**	0.38	0.44	0.12
Recultivation layer (cover)	**3**	0.33	0.48	0.49
Heat production within waste	**1**	0.09	0.06	0.04
Climate	**1**	0.08	0.12	0.49
Typical vegetation and projection after use	**1**	0.09	0.48	0.49
Relief	**2**	0.08	0.10	0.12
Expected settlements	**1**	0.38	0.09	0.07
Lining control systems (direct monitoring)	**2**	0.10	0.06	0.05
Drainage system	**3**	0.10	0.09	0.08
	Evaluation results:	**3.39**	**3.38**	**3.37**
	Leachate generation rates [mm/yr]:	**54.4**	**69.8**	**79.8**
Factors – bottom lining system				
Barrier performance at time of evaluation	**2**	0.44	0.08	0.07
Construction quality program	**1**	0.46	0.49	0.44
Climate	**1**	0.07	0.08	0.09
Heat production within waste	**1**	0.09	0.06	0.05
Distance to groundwater	**1**	0.08	0.08	0.09
Overburden pressure	**1**	0.08	0.08	0.08
Leachate quality	**1**	0.08	0.08	0.08
Monitoring (e.g. groundwater monitoring)	**3**	0.08	0.07	0.07
Drainage system	**1**	0.46	0.11	0.1
Stability of foundation	**1**	0.1	0.44	0.46
Landfill geometry (heap vs. cavity)	**3**	0.08	0.43	0.48
	Evaluation results:	**2.78**	**3.08**	**3.18**
	Leachate percolating through landfill base [mm/yr]:	**11.0**	**29.2**	**47.1**

*Specific criteria are used to derive evaluation scores: for instance, a score of 1 for "barrier performance at the time of evaluation" means that the barrier has performed according to its design specifications, a score of 2 is awarded if there is no direct monitoring of the barrier function, but it is probably performing according to the design specifications, whereas a score of 3 means that it has not achieved its intended service level at the time of evaluation.

**Weighting factors were derived from expert evaluations during a survey on the importance of different factors for barrier performance during different service periods (cf. chapter 4.1.4, chapter 5.2, and appendix 2).

The evaluation of the base lining system results in an increase of the leachate release rate to the subsurface from 0.3 mm/yr initially to 11 mm/yr after 100 years, and 47 mm/yr at the end of the modeling period (see Table 6-26). Thus, the leachate collection efficiency will drop to around 41% after 300 years from 99% at the beginning.

The leachate emissions for Scenario A are based on the emission models in Table 6-25 and on the decrease of barrier performance illustrated in Table 6-26. The resulting emission estimates are shown in Figure 6-23. The concentrations of nitrogen and COD slowly decrease from 5.3 mg N/l and 30 mg COD/l to around 1 mg N/l and 6.4 mg COD/l after 300 years. Chloride concentration levels drop faster and halve within 70 years from 100 mg/l to 50 mg/l (see Figure 6-23, left). Sulfate concentrations remain at 800 mg/l for 180 years and then fall to 0 mg/l. The corresponding loads of sulfate in the leachate continuously increase to reach approximately 1590 kg/yr after 180 years and subsequently drop to zero. The loads of the other leachate parameters also increase from the beginning, but reach their respective maximum levels of 96 kg Cl/yr, 6.6 kg N/yr, and 37 kg COD/yr at an earlier time (after 20 years for Cl, after 90 years for N, and after 100 years for COD).

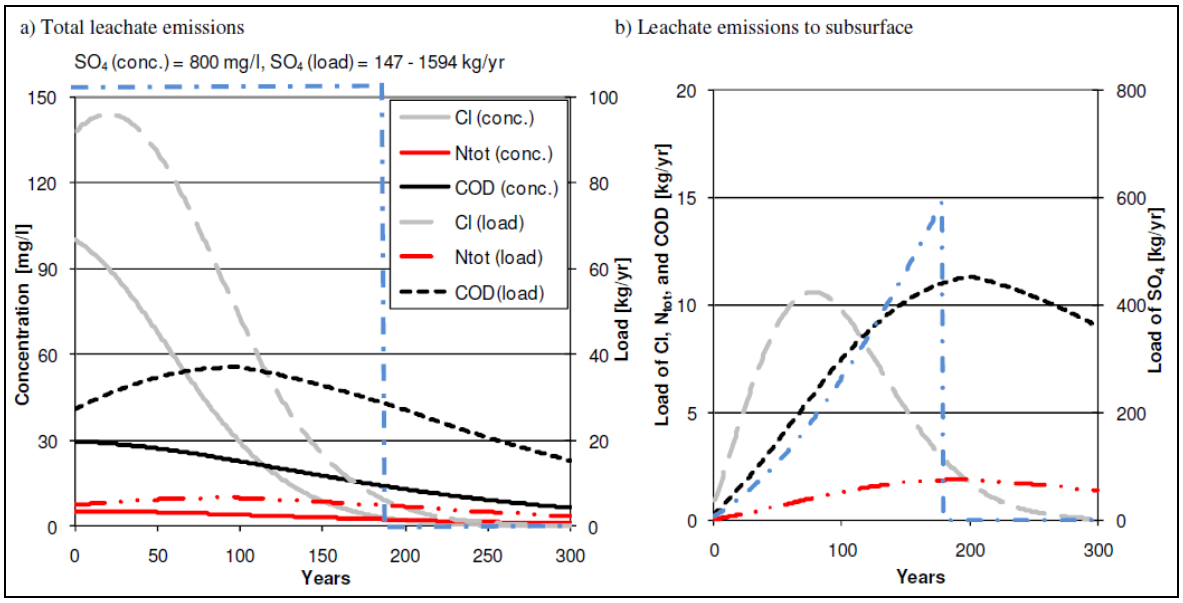


Figure 6-23: Leachate emissions for Scenario A (gradual decrease of barrier performance) at Landfill C (left: total leachate emissions, right: leachate emissions to subsurface)

The substance loads released into the subsurface below the landfill increase for all parameters from the beginning to reach their maxima after 80 years (10.6 kg Cl /a), after 180 years (590 kg SO_4/yr), and after 200 years (1.9 kg N/yr and 11.3 kg COD/yr), respectively. Afterwards, the loads decrease (instantaneously in case of SO_4) to 0 kg SO_4/yr, to almost 0 kg Cl/yr, to 1.4 kg N/yr, and to 9 kg COD/yr, respectively (see Figure 6-23, right).

With respect to Scenario A* the total leachate emission levels of Scenario A (see Figure 6-23, left) are the emission levels to the subsurface, as the Scenario A* assumes the inefficiency of base lining and leachate collection after the termination of aftercare.

Scenario B

Within Scenario B the failure of the containment system at Landfill C is investigated in view of its effect on leachate emission levels. The worst performance level of the top cover allows 16% of average annual precipitation to infiltrate into the waste body. All the generated leachate drains to the subsurface below the landfill.

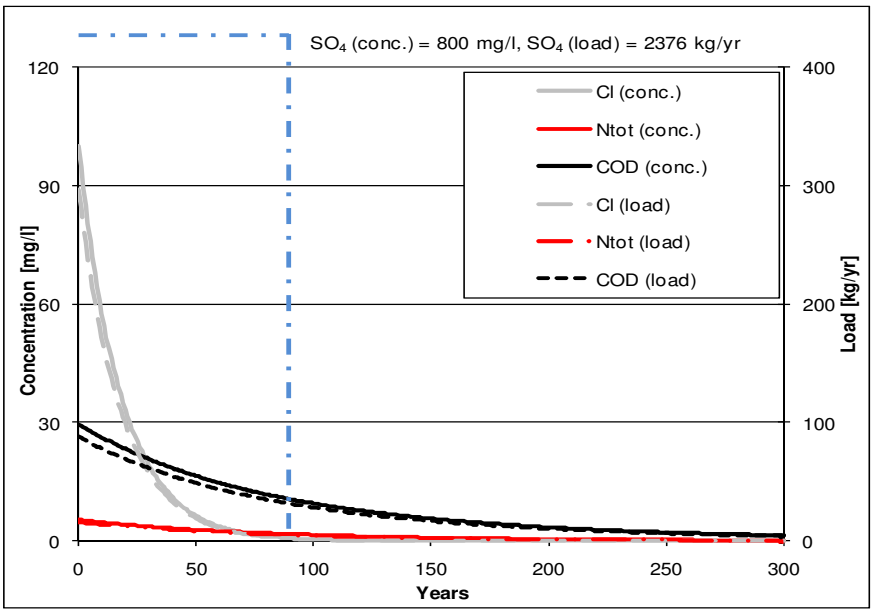


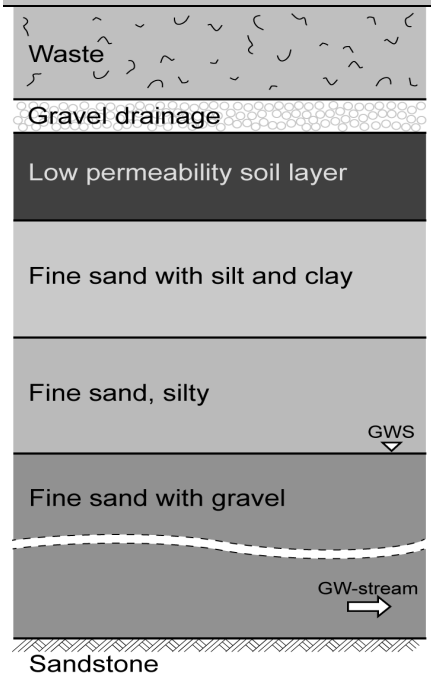
Figure 6-24: Leachate emissions for Scenario B (complete barrier failure) at Landfill C

The emission levels estimated for Scenario B are shown in Figure 6-24. Due to the higher water throughput compared to the other scenarios, the concentrations of Cl, N_{tot}, and COD in the leachate decrease faster and the concentration of SO_4 drops from 800 mg/l to 0 mg/l after 90 years. The substance loads are initially high with maxima of 300 kg Cl/yr, 16 kg N/yr, 88 kg COD/yr, and 2380 kg SO_4/yr, respectively (see Figure 6-24). After 100 years, however, the respective loads fall to 1 kg of Cl/yr, 4.5 kg of N/yr, 28 kg of COD/yr, and 0 kg of SO_4/yr. Finally, at the end of the modeling period the COD load is 4 kg/yr and the other substance loads are below 0.1 kg/yr.

6.3.3 Pollutant migration

The effect of a leachate release from Landfill C on pollutant concentrations in the subsurface is evaluated for the vadose zone, the groundwater, and the river water. The soil layers between the landfill base and the groundwater are described in Table 6-27. The landfill bottom liner (low permeability earthen barrier) and the loamy foundation layer below the liner are described as one layer of 1.5 meters thickness, because they were constructed together and they exhibit similar soil properties. The distance from the bottom of the engineered barrier (low permeability soil layer) to the groundwater surface is four meters, made up of sandy soil with varying amounts of silt and clay (see Table 6-27). The content of clay is decreasing with increasing depth and the aquifer is mainly composed of sand and gravel. The thickness of the groundwater is around 10 meters with a sandstone aquitard at the bottom.

Table 6-27: Layout of the subsurface below Landfill C and material properties of the soil layers

Layout of the subsurface below the Landfill C	**Thickness, grain size distribution and soil type** (according to AG Boden 2005)	**Hydraulic conductivity, field capacity, pore volume, and dry bulk density**
	**Low permeability soil layer:** Thickness: 1.5 m; 30 – 40 % sand, 40 – 50 % silt, 17 – 25 % clay (soil type: Ls2)	$k_f < 10^{-9} – 10^{-8}$ m/s field capacity: 34 % total pore volume: 43 % dry bulk density: 1.8 g/cm³
	Silty and clayey fine sand: Thickness: 2.0 m; 40 – 60 % sand, 30 – 50 % silt, 10 – 20 % clay (soil type: Slu)	$k_f \approx 10^{-8}$ m/s field capacity: 33 % total pore volume: 43 % dry bulk density: 1.5 g/cm³
	Silty fine sand: Thickness: 2.0 m; 50 – 75 % sand, 25 – 40 % silt, 0 – 10 % clay (soil type: Su3)	$k_f \approx 10^{-8} – 10^{-7}$ m/s field capacity: 29 % total pore volume: 43 % dry bulk density: 1.5 g/cm³
	Fine sand with gravel (aquifer): Thickness ≈ 10 m	Average groundwater velocity ≈ 500 m/yr
	Bedrock: Sandstone	

The points of compliance to evaluate pollutant migration in the subsurface are the vadose zone just above the groundwater surface (PoC1), the mixing zone of leachate and groundwater (PoC2), the groundwater 100 m downstream of the landfill (PoC3), and the river Danube (PoC4). A scheme of the landfill-environment system is shown in Figure 6-25. The local groundwater quality is strongly affected by agricultural activities in the region. Nitrate concentrations in the groundwater are around 50 mg/l, ammonia concentrations are below 0.1 mg/l, and concentrations of dissolved oxygen range between 8 and 12 mg/l. Chloride concentrations vary seasonally from 10 to 15 mg/l (peaks in winter due to road de-icing salts) and sulfate concentrations are typically around 30 mg/l in the groundwater[24]. In addition, there are several other landfill sites in the vicinity of Landfill C which could potentially affect the groundwater. Although the subsequent transport modeling does not include any other pollutant sources or consider the upstream pollutant levels already present in the groundwater, it should be noted that cumulative effects of several pollutant sources or background pollutant levels might be significanct for determining acceptable pollution levels of the environmental media.

[24] As an example, the calculation of total concentration levels is illustrated for PoC2:
Total concentration at PoC2 (c_{tot} in mg/l) = background concentration (c_{back} in mg/l) * leachate dilution factor (DF)[a)] + concentration because of leachate release at PoC2 (c_{PoC2} in mg/l) → $c_{tot} = c_{back} * DF + c_{PoC2}$
[a)]DF = affected groundwater flow [m³/yr] / (leachate flow [m³/yr] + affected groundwater flow [m³/yr])

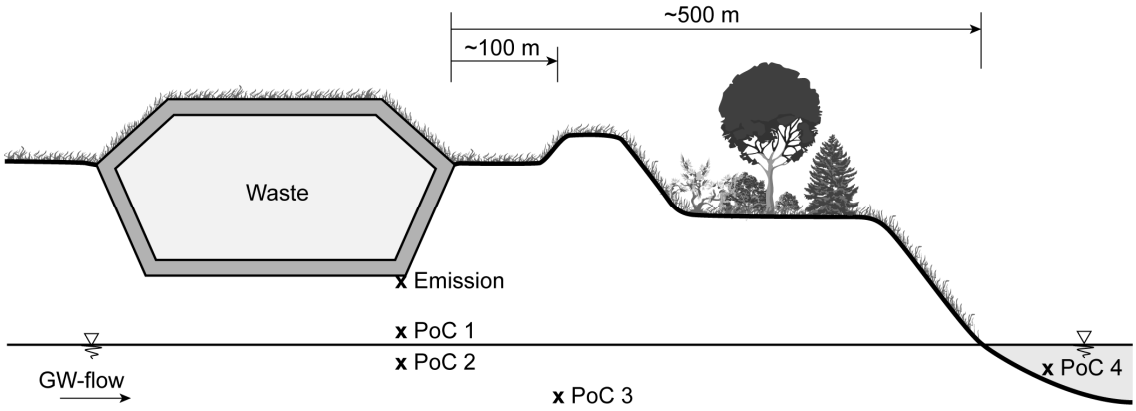


Figure 6-25: Scheme of the landfill-environment system and relevant points of compliance (PoC) at Landfill C (cross-section along the ground water flow direction)

Table 6-28: Parameters used for pollutant migration modeling in the subsurface of Landfill C

Vadose zone	Diffusion coefficient in water [m²/yr]	k_d-value [l/kg] (low perm. soil layer)	k_d-value [l/kg] (stones, gravel & sand)	Comments
Cl	0.0315	0	0	No degradation
NH_4-N	0.0378	0.5	0	No degradation
SO_4	0.0189	0	0	No degradation
Groundwater	Groundwater velocity [m/yr]	Mixing depth [m]	Contaminated width [m]	Comments
General parameters	500	0.25	100	No degradation *

* The dissolved oxygen content in the groundwater is used as an indicator to estimate the potential for aerobic degradation of organic substances and the nitrification of ammonia.

The migration of Cl, SO_4, and N_{tot} (modeled as NH_4-N) in the vadose zone is estimated based on the AF model (Schneider and Stöfen 2004) for scenarios with constant leachate generation rates. For scenarios with varying leachate generation rates and for COD in general, pollutant migration in the vadose zone is evaluated based on a mass balance approach using the maximum loads released into the subsurface. The concentration in the leachate at the time of maximum load is equal to the concentration at PoC1[25]. Main parameters for pollutant migration modeling in the subsurface are given in Table 6-28. The release of contaminants is assumed throughout the landfill width perpendicular to the groundwater flow direction. The assumed mixing depth is 0.25 meters and the average groundwater velocity is 500 m/yr. In the vadose zone, sorption is only considered in the low permeability soil layer at the landfill base for ammonia-nitrogen. As potential degradation is not considered in the modeling, a reduction of pollutant concentration occurs mainly due to hydrodynamic dispersion (including diffusion) and dilution processes.

[25] In case of COD concentrations for scenarios with constant leachate generation rate, the maximum loads are averaged over the travel time in the vadose zone (the travel time of Cl from the AF model calculations is used for COD).

Table 6-29: Data for the AF model calculations to estimate concentrations at PoC1 and PoC2at Landfill C

Scenario	Parameter	$c_{0,max}$ [mg/l]	Release rate [mm/yr]	AF [-]	Travel time t_m [yrs]	c_{PoC1} [mg/l]	GW flow [m³/yr]	AF_{GW} [-]	c_{PoC2} [mg/l]
Status quo	Cl	100	0.3	166	5710	0.6	12500	1360	0.0
Status quo	NH_4-N	5.3	0.3	58	8350	0.1	12500	1360	0.0
Status quo	SO_4	800	0.3	30	5710	26.7	12500	1360	0.0
Scenario B	Cl	100	99.0	1.8	18	55.6	12500	5.2	10.7
Scenario B	NH_4-N	5.3	99.0	1.0	26	5.3	12500	5.2	1.0
Scenario B	SO_4	800	99.0	1.0	18	800	12500	5.2	154
Status quo*	Cl	100	30.7	2.0	57	50.0	12500	14.6	3.4
Status quo*	NH_4-N	5.3	30.7	1.4	84	3.8	12500	14.6	0.3
Status quo*	SO_4	800	30.7	1.0	57	800	12500	14.6	55

Table 6-30: Data for the calculations based on maximum emission loads into the subsurface to estimate concentrations at PoC1 and PoC2 at Landfill C

Scenario	Parameter	$C_{0,max}$ [mg/l]	Release rate [mm/yr]	Load [kg/yr]	Travel time t_m [yrs]	c_{PoC1} [mg/l]	AF [-]	AF_{GW} [-]	C_{PoC2} [mg/l]
Status quo	COD	30	0.3	0.3	5710	29.6	1.0	1360	0.0
Scenario A	Cl	100	7.6	10.6	230	46.3	2.2	55.6	0.8
Scenario A	NH_4-N	5.3	29.0	1.9	90	2.2	2.4	15.4	0.1
Scenario A	COD	29.6	29.0	11.3	60	13.0	2.3	15.4	0.9
Scenario A	SO_4	800	24.4	591	71	800	1.0	18.1	44.7
Scenario B	COD	29.6	99.0	79.0	18	26.6	1.1	5.2	5.1
Status quo*	COD	29.6	30.7	27.2	57	29.6	1.0	14.6	2.0
Scenario A*	Cl	100	36.4	86.0	48	78.8	1.3	12.4	6.3
Scenario A*	NH_4-N	5.3	52.0	6.6	50	4.2	1.3	9.0	0.5
Scenario A*	COD	29.6	53.2	37.0	33	23.2	1.3	8.8	2.6
Scenario A*	SO_4	800	66.3	1594	28	800	1.0	7.3	110

[a] Average travel time of COD in the vadose zone is assumed the same as for Cl.

The calculation results and data for the evaluation of pollutant concentrations at PoC1 (vadose zone above groundwater surface) and PoC2 (mixing zone in the groundwater) based on the AF model are shown in Table 6-29. The calculations based on a mass balance approach using the maximum substance loads released to the subsurface to estimate pollutant concentrations at PoC1 and PoC2 for COD and scenarios with varying leachate generation rates are shown in Table 6-30.

After the mixing of leachate and groundwater (PoC2), pollutants travel at the same velocity as the groundwater, as no retardation and no degradation of pollutants are assumed in the groundwater. However, between PoC2 and PoC3 the pollutant concentrations decrease due to the spreading of the plume in the groundwater, which amounts for 1 percent of the travel distance. Hence, the thickness of the plume increases by 1 meter between PoC2 and PoC3. No further dispersion of the plume is assumed afterwards and the concentration in the contaminated groundwater remains constant until the groundwater mixes with the river water at PoC4. Because of the very high discharge volume of the river Danube, the concentrations in the river resulting from a release of leachate into the subsurface are very low. The corresponding attenuation factors for the investigated pollutants between the different points of compliance are shown in Figure 6-26. The

scenario- and pollutant-specific attenuation factors in Figure 6-26 are the basis for the reverse calculation of tolerable pollutant levels in the landfill leachate and are used to derive completion criteria with respect to leachate quality.

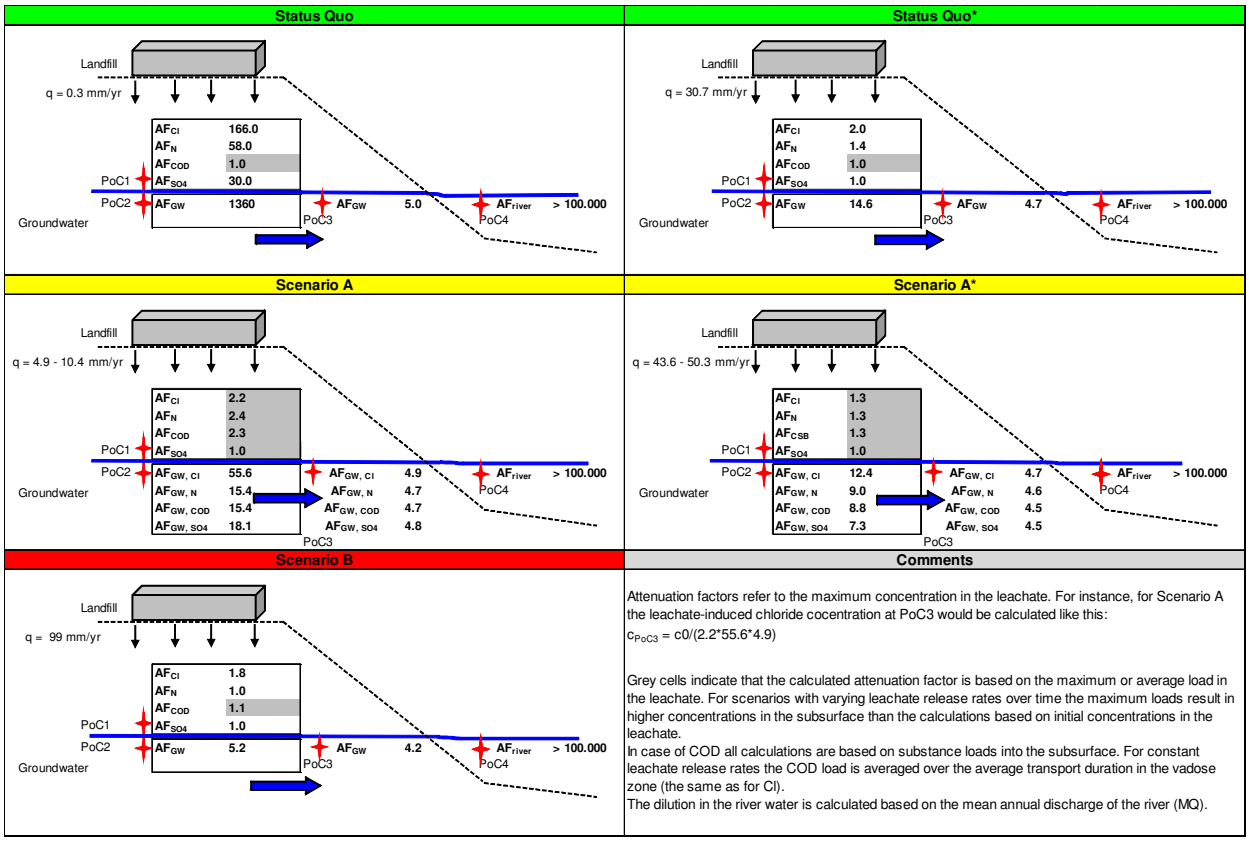


Figure 6-26: Attenuation factors to estimate the concentrations at relevant PoCs due to a release of leachate from Landfill C for the different scenarios

6.3.4 Completion criteria and aftercare duration

The leachate generated at Landfill C has to be extracted at the landfill base with pumps. As the functionality of this pumping system may not be guaranteed after the termination of aftercare, it is assumed that the generated leachate will percolate through the base lining system of the landfill and pollutants will be released to the subsurface. Hence, at Landfill C the status quo*, Scenario A*, and Scenario B are regarded as most relevant to evaluate the effect of potential future leachate emissions on the local groundwater (see Table 6-31). As the groundwater at the site is already polluted and further degradation should be prevented (authorities aim at improving groundwater quality for the affected groundwater body), the PoC2 (leachate mixing zone in the groundwater) is defined as the critical point of compliance, where drinking water quality standards have to be met. Thus, the limit values are applied at PoC2 and the reverse calculation of the maximum leachate concentration (based on the attenuation factors shown in Figure 6-26) results in the tolerable pollutant concentrations in the landfill leachate at the end of aftercare shown in Table 6-31. The tolerable concentrations at the source are generally above the current levels in the leachate, only in case of complete barrier failure (Scenario B) the current concentrations of nitrogen in the leachate

would be too high[26]. As a sudden failure of technical barriers is not expected, Scenario A* (a gradual decrease of top cover performance and the complete inefficiency of the base lining system) is regarded as the most relevant scenario to evaluate the aftercare completion. Consequently, the leachate of Landfill C can be considered tolerable in the local environment and the operator could be released from aftercare obligations. However, due to the very short monitoring period and the poor quality of leachate data at Landfill C, further aftercare (especially representative monitoring) will be necessary to provide additional data as a basis for validating the models underlying the emission scenarios. Emphasis should be primarily put on observations to confirm the parameters of the emission model and the assumptions on the performance level of the final top cover during the next decade (e.g. desiccation cracks).

Table 6-31: Tolerable leachate concentrations (C_0) at the source in order not to exceed the limit values specified at PoC2 for relevant emission scenarios of Landfill C

Parameter	PoC2	Status quo*	Scenario A*	Scenario B
	C_{limit} [mg/l]	C_0 [mg/l]	C_0 [mg/l]	C_0 [mg/l]
Cl [a)]	100	2919	**1580**	938
NH_4-N [a)]	0.5	10	**6**	3
COD [a)]	5	73	**56**	29
SO_4 [a)]	250	3649	**1822**	1302

a) Limit values for leachate induced concentrations at PoC equal Austrian drinking water quality standards (MoE 2001). In case of ammonia-nitrogen limit values for ammonia are applied at PoC3. No nitrification of ammonia is assumed.

In conclusion, post-closure care at Landfill C needs to be prolonged to gather appropriate monitoring data to back up specific model parameters and assumptions. If the model results are verified by future monitoring, landfill completion could be achieved within the next decade at the site. If monitoring data do not support the model estimates, models need to be adjusted to the collected data and the evaluation of landfill environmental compatibility needs to be repeated based on the adapted models and scenarios.

6.4 Discussion of caste study results

In general, the evaluation of aftercare and environmental risks at the case study landfills is associated with substantial uncertainties. Apart from the fact that future conditions are unknown, the quality of available data is often quite limited (e.g. landfill gas data for Landfill A, data to estimate subsurface transport processes at all sites, and leachate generation rates for Landfill C) and the monitoring periods to establish appropriate models are rather short (e.g. post-closure data for Landfill C is available for one year only). Therefore, it should be emphasized that the purpose of the evaluation is primarily to illustrate the applicability of the developed methodology. In order to use the evaluation within a regulatory procedure for evaluating aftercare completion, further investigations and monitoring data would be necessary at the sites. While the range of parameters measured at short intervals (i.e. less than two years) could be reduced, some parameters (e.g.

[26] Note the assumption that all the nitrogen is released as ammonia-nitrogen and no nitrification takes place in the subsurface.

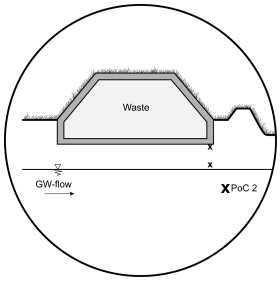
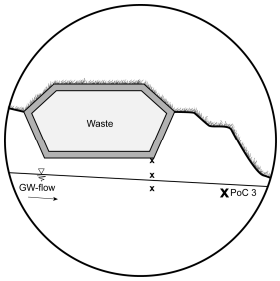
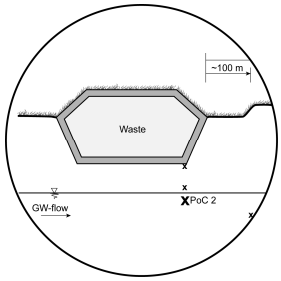
leachate generation rates, electric conductivity) should be measured frequently, under different hydraulic conditions, and over an extended period of time. Apart from the continuous gathering of information about the landfill and associated emissions, the investigation of specific properties (e.g. mobilizable substance fractions) may be necessary to establish more reliable estimates on future emission levels. In general, the evaluation of environmental compatibility of a closed landfill is probably associated with additional investigations of the landfill-environment system (e.g. waste properties or subsurface conditions) and is only possible if appropriate monitoring data time series exist.

A summary of basic data and the results of the case study evaluations are shown in Table 6-32. The evaluations differ with respect to various aspects, such as the conditions at the site, the deposited waste material, the relevant points of compliance, or the most probable emission scenario after aftercare completion, which was used to derive completion criteria.

Different points of compliance are defined at the different sites to show the relevance of the location of the definitive point of compliance, on the one hand, and to reflect the level of protection assigned to a specific environmental compartment, on the other hand. However, within a regulatory procedure the pollutant levels already present in the affected environmental media, potential additional sources, and the intended use at and around the site would probably be factors significant for the choice of the critical PoC and the associated environmental quality standards. During the case study evaluations drinking water quality standards (MoE 2001) were used to specify an acceptable level of landfill-borne pollution at the PoCs in the subsurface. Nevertheless, other quality standards could be chosen in view of the vulnerability of the affected environment and the intended level of protection (e.g. marginal effect levels (LAWA 2004) or limit values elative to background levels and fluctuations (cf. Bräcker et al. 2004)). Nonetheless, the methodology illustrated here can be used with alternate input assumptions and constraints.

The emission estimates, which were used to evaluate the release of leachate at the points of compliance, relate to Scenario A* (gradual decrease of top cover performance and complete inefficiency of the base lining system) for Landfill A and Landfill C and to Scenario A (gradual decrease of top cover and base lining system performance) for Landfill B, see Table 6-32. While the extraction of collected leachate at the base of Landfill A and Landfill C depends on functional pumping systems, which cannot be guaranteed in the absence of aftercare, gravity drainage of collected leachate at Landfill B can be expected also after the end of regulated aftercare. This implies that at Landfill B the appropriate management of the collected leachate needs to be assured also in the absence of aftercare. It should be noted that conditions assumed within the relevant emission scenarios represent specific criteria, which need to be fulfilled. For instance, geotechnical stability of the case study landfills is assumed within all scenarios, which implies that this is an inherent criterion for aftercare completion. Also the persistence of water flow patterns in the landfill and constant release mechanisms are assumed for all scenarios, for instance requiring protection from human intrusion. In general, to complete aftercare it has to be demonstrated that the scenario models are appropriate for the site and that the landfill conditions of the emission scenario underlying the evaluation can be expected at the site (i.e. after-use concepts and custodial care program).

Table 6-32: Overview on the evaluation of the case study landfills and the estimated remaining aftercare periods

	Landfill A **Closed MSW landfill**	**Landfill B** **Closed MSW landfill**	**Landfill C** **Closed C&D landfill**
General landfill data	Precipitation: 956 mm/yr Pot. evaporation: 633 mm/ yr Operation period: 1972-2005 Waste volume: 1700000 m³ Average deposition height: 15 m	Precipitation: 1960 mm/ yr Pot. evaporation: 550 mm/ yr Operation period: 1988-2003 Waste volume: 135000 m³ Average deposition height: 11 m	Precipitation: 613 mm/ yr Pot. evaporation: 630 mm/ yr Operation period: 1992-2008 Waste volume: 400000 m³ Average deposition height: 13 m
Emission parameters	Leachate: Cl, COD, NH_4-N, Cr_{tot}; Landfill gas: CH_4	Leachate: Cl, COD, NH_4-N; Landfill gas: CH_4	Leachate: Cl, COD, N_{tot}, SO_4
Most probable emission scenario after termination of aftercare	Gradual decrease of top cover performance and complete inefficiency of base lining system (Scenario A*)	Gradual decrease of top cover and base lining system efficiency (partial collection and management of leachate required – Scenario A)	Gradual decrease of top cover performance and complete inefficiency of base lining system (Scenario A*)
Leachate generation $\begin{pmatrix} Initial\,level \\ after\ 100\ years \\ after\ 300\ years \end{pmatrix}$	$\begin{pmatrix} 30\ mm/yr \\ 100\ mm/yr \\ 150\ mm/yr \end{pmatrix}$	$\begin{pmatrix} 20\ mm/yr \\ 100\ mm/yr \\ 320\ mm/yr \end{pmatrix}$	$\begin{pmatrix} 30\ mm/yr \\ 50\ mm/yr \\ 80\ mm/yr \end{pmatrix}$
Leachate release rate to subsurface $\begin{pmatrix} Initial\ level \\ after\ 100\ years \\ after\ 300\ years \end{pmatrix}$	$\begin{pmatrix} 30\ mm/yr \\ 100\ mm/yr \\ 150\ mm/yr \end{pmatrix}$	$\begin{pmatrix} 0.2\ mm/yr \\ 5.0\ mm/yr \\ 95\ mm/yr \end{pmatrix}$	$\begin{pmatrix} 30\ mm/yr \\ 50\ mm/yr \\ 80\ mm/yr \end{pmatrix}$
Definitive point of compliance (PoC) in the subsurface	Groundwater (GW) plume 	GW 100 m downstream landfill 	Leachate mixing zone in the GW 
Acceptable level at PoC	Drinking water quality	Drinking water quality	Drinking water quality
Critical parameter (leachate)	**Ammonia-nitrogen** (Nitrification considered)[1]	**Ammonia-nitrogen** (No nitrification)	**-**
Aftercare duration[2]	**Minimum: 80 years** **Maximum: 450 years**	**Minimum: 11 years** **Maximum: 110 years**	**Most probable scenario is environmentally tolerable**
Additional criteria	a) Undisturbed landfill body (e.g. geotechnical stability); b) Performance of top cover decreases gradually;	a) Collection and management of leachate is guaranteed for; b) Undisturbed landfill body (e.g. geotechnical stability); c) Performance of top cover decreases gradually;	a) Undisturbed landfill body (e.g. geotechnical stability); b) Performance of top cover decreases gradually; c) Aftercare necessary to collect representative monitoring data;

[1] It is assumed that the ammonia is nitrified before or after the exfiltration into the adjacent surface water systems (Limit values for nitrate-nitrogen are used to evaluate groundwater contamination).

[2] Minimum aftercare duration: Water infiltration rate equals the local groundwater recharge rate. Maximum aftercare duration: Water infiltration rate under the assumption of constant optimal top cover performance.

The estimated aftercare durations for the three case study landfills are shown in Table 6-32. Depending on the assumed aftercare strategy at the site, the time periods to reach the completion criteria (in all cases ammonia-nitrogen in the leachate constitutes the critical parameter) are different. For Landfill A the aftercare duration is 80 years at the minimum (intensive water infiltration during aftercare) and more than 450 years at the maximum (continuous optimal function of the top cover system). For Landfill B the equivalent durations are 11 years at the minimum to 110 years at the maximum provided that (partial) collection and treatment of the leachate can be assured after the termination of aftercare. In case of Landfill C, it would be not necessary to maintain aftercare based on the criteria addressing tolerable emission levels. Nevertheless, further aftercare is necessary to confirm the models and assumptions within the evaluation due to the poor quantity (only one year post-closure monitoring) and quality of monitoring data. However, if future monitoring supports the established models and scenarios, it may be possible to terminate aftercare within the next decade without unacceptable environmental risks at the Landfill C.

The case studies illustrate that the developed aftercare completion criteria can be used to estimate the duration of aftercare for different long-term management concepts. The comparison of strategies in view of the desired state of the landfill in the absence of aftercare represents a basis to link the duration and intensity of long-term management to the environmental risks associated with the closed landfill. Therefore, the case study evaluations can be the foundation to develop cost-effective long-term management concepts for the investigated landfills.

7 Summary and conclusions

7.1 Summary

Landfills are key elements in modern waste management, as they represent sinks for substances which cannot be recovered due to economic or ecologic reasons. In general, landfills need to be managed after closure to assure the long-term protection of humans and the environment. This period of "aftercare" or "post-closure care" typically comprises the monitoring and control of landfill emissions, the maintenance and control of landfill facilities, as well as site surveillance. It can be brought to an end when the authorities consider the landfill to no longer pose a threat to humans and the environment. While simple qualitative criteria are often used as a basis to evaluate aftercare, rigorous methodologies and data to evaluate aftercare and aftercare completion are difficult to build and hardly available. Different approaches have been suggested to evaluate landfill aftercare and its completion (e.g. Stegmann et al. 2006; Environment Agency 2010a; Morris and Barlaz 2011; Scharff et al. 2011), but a consistent methodology to derive aftercare completion criteria in view of long-term environmental risks associated with a closed landfill has not yet been established. However, such a methodology is needed to support the authorities' decision on appropriate management strategies and completion criteria for aftercare. Therefore, in this thesis the long-term landfill management was investigated in view of the associated environmental risks. The goal was to develop a methodology for defining an environmentally compatible closed landfill at the end of aftercare. The methodological framework to evaluate landfill environmental compatibility included assessments of landfill emission behavior, estimates on containment system performance, and evaluations of pollutant migration in the environment.

7.1.1 Landfill emission behavior

Landfill behavior and associated emissions were analyzed for different types of landfills containing municipal solid waste (MSW), municipal solid waste incineration (MSWI) ash, and construction and demolition (C&D) waste, respectively. Based on literature data and landfill data gathered at sites in Austria and Switzerland, emission characteristics of different landfills and their evolution over time, remaining emission potentials, and possible approaches to predict future landfill emissions were evaluated.

MSW landfills

The emission behavior of closed MSW landfills is dominated by the degradation of organic matter (landfill gas and organic leachate pollution) and the leaching of soluble salts and inorganic macrocomponents (cf. Farquhar and Rovers 1973; Bookter and Ham 1982; El-Fadel et al. 1997; Kjeldsen et al. 2002). While landfill gas emissions typically drop to very low levels within a few decades after landfill closure (e.g. Krümpelbeck 2000), in the leachate several substances are present at high concentrations even several decades after the end of waste deposition. The concentrations of ammonium and organic pollutants (i.e. COD, TOC, BOD_5) in the leachate of Austrian MSW landfills were shown to be, on average, one order of magnitude above the corresponding limits for direct discharge into surface waters even 20 years after landfill closure (see Table 3-4). In addition,

average concentrations of chloride, despite an observed decreasing trend, are still high (>500 mg/l) in the leachate. Apart from the leachate parameters mentioned above, Fe, AOX, and Cr_{tot} are of significance at some sites. Heavy metal concentrations are typically low in the leachate of closed MSW landfills. Although emissions from MSW landfills decrease after closure (concentrations and loads), a large fraction of mobilizable substances remains in the landfill for an extended period of time, especially if waste leaching and waste degradation is minimized by an impermeable cover on top of the landfill. Thus, models to evaluate long-term emission characteristics at closed MSW landfills need to consider the level of interaction between the deposited waste and the environment, the mobilizable potential of substances remaining in the landfill, the water flow regime in the waste, and potentially changing conditions at the landfill. On the short term (decades) emission trends and models based on observed emission characteristics can be used to predict emissions. On the long term (centuries to millennia) fundamental relationships could be used to establish emission models. However, due to long modeling periods and limited process understanding such models are associated with large uncertainties. Emission models always need to be adapted to the specific situation of a particular landfill, as emission characteristics vary substantially from one landfill to another.

MSWI bottom ash landfills

Numerous reactions, which are partly associated with environmentally relevant emissions, take place in landfills containing bottom ash from the incineration of municipal solid waste[27]. High temperatures may occur in the waste body due to exothermic reactions potentially affecting the landfill containment system. Landfill gas consists primarily of water vapor, hydrogen (from the reaction of metallic aluminum and water to aluminum oxide and hydrogen gas), and possibly methane as a product of the anaerobic degradation of residual organic matter contained in the bottom ash. The analysis of leachate at Swiss MSWI ash landfills showed that the concentrations of soluble salts (i.e. chloride) in the leachate are in the range of several thousand mg/l during the first decades of landfill existence. In addition organic leachate pollution (e.g. TOC) and ammonium-nitrogen concentrations might be significant in view of the corresponding discharge standards, whereas heavy metal concentrations in the leachate typically comply with drinking water quality standards. However, the release of heavy metals represents a long-term threat with respect to leachate emissions, as increased mobilization might occur as a consequence of decreasing pH in the landfill over a period of many thousands of years (e.g. Förstner and Grathwohl 2007). The long-term emission behavior of MSWI ash landfills can be addressed by geochemical models, though the results may be of limited significance for aftercare decisions due to the long timeframes (cf. Hellweg 2000). The immediate attention in terms of leachate quality is therefore focused on soluble salts, organic pollution, and ammonium-nitrogen. Trend extrapolations based on real-scale monitoring data or results from laboratory experiments simulating current landfill conditions can be used to evaluate the evolution of these parameters in the leachate over time periods from decades to centuries. In all cases, estimates of landfill emissions need to be based on the specific conditions at

[27] It should be noted, that the quality of residues from thermal processes depends also on process parameters, which are continuously optimized. Therefore, the composition and properties of future MSWI bottom ash might well be different from currently deposited material.

the site (and potential changes of these), with the significance of trend extrapolations strongly depending on the availability of appropriate aftercare monitoring data.

C&D waste landfills

The composition of construction and demolition (C&D) waste is of major importance for the emission characteristics observed at C&D waste landfills. At older sites, where waste containing higher amounts of organic matter has been deposited, organic leachate pollution as well as the concentrations of nitrogen and soluble salts in the leachate are substantially higher than at sites where stringent waste acceptance criteria have been in place all the time. Consequently, landfill emission characteristics differ from one site to another. However, parameters such as the concentrations of soluble salts, organic leachate compounds (COD, DOC, AOX), and ammonium-nitrogen (or total nitrogen) may be significant at many C&D waste landfills. As the analyzed monitoring data stem from landfills with ongoing waste deposition, data on emission characteristics during the post-closure period of C&D waste landfills will prove valuable to evaluate future emissions. Typically, models on the evolution of leachate characteristics at C&D waste sites are based on laboratory experiments (e.g. shaking leaching tests, column leaching tests, lysimeter studies) and direct observations at the closed site used for model calibration and validation. Although comparisons of results from leaching experiments to larger systems (lysimeters) allow one to infer that experimental data are valuable to evaluate emissions from C&D waste landfills (e.g. Susset and Grathwohl 2011), appropriate monitoring data from closed landfills will be necessary to transfer laboratory-scale results to real-scale landfill situations.

7.1.2 Technical and natural barriers

A landfill is not in thermodynamic equilibrium with its surrounding environment due to its different composition. This induces two-way substance and energy flows, balancing the differences between these systems. Substance flows to air, soil, surface waters, and groundwater may be of environmental concern. The impact of substances mobilized from the deposited waste is dependent on the performance of technical barriers at the landfill to limit and control the interaction of the deposited waste with the surrounding environment, on the one hand, and the ability of the natural environment to mitigate potential negative effects of released substances on environmental media, on the other hand. The two systems represent barriers for pollutants and may prevent adverse effects on human health and the environment.

Technical barriers

Technical barriers typically consist of several components with low-permeability mineral layers (e.g. clay liners), geomembranes (e.g. HDPE liners), geosynthetic clay layers, drainage layers, and recultivation layers being among the most important ones. 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 (cf. Inyang 2004). The elements of a technical barrier system have different service lives. Estimates on the long-term performance of different barrier components (under the assumption that no maintenance and repair is taking place) range from several decades

for leachate drainage systems to geologic time periods (thousands of years) for low-permeability mineral liners (e.g. 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 uncertainty. Therefore, a scenario-based approach was implemented to evaluate future barrier performance. A best and a worst case scenario investigate the effect of constant barrier performance (best case) and complete barrier failure (worst case). A scenario of gradual barrier deterioration is based on an evaluation of different factors to address the conditions at the landfill site and the resistance built into the system. The weighting of the factors for different service periods was based on evaluations of experts on landfill geotechnics who had been invited to list important factors with respect to the performance of technical barriers at landfills and subsequently evaluate the importance of these factors for the performance of the system for different time periods (e.g. 0-50 yrs, 50-100 yrs, above 100 yrs). These evaluations were related to the best and the worst barrier performance levels and the scenario of gradually decreasing barrier performance could be estimated for the specific system at a particular site.

Natural barriers

The potential of natural systems to attenuate negative effects of landfill-borne pollutants is based on naturally occurring processes reducing the mass, toxicity, extent, mobility or concentration of pollutants in the environment. As the release of landfill leachate is of potential concern with respect to groundwater contamination for long periods of time, the focus of evaluating natural attenuation potentials in the environment is placed on the release of leachate from the landfill and the corresponding subsurface migration pathways. Although natural attenuation potentials have been in the focus of contaminated site management and remediation for some time (e.g. Christensen et al. 2000; National Research Council 2000), knowledge about the migration of typical leachate constituents such as ammonium-nitrogen or with respect to aggregate parameters on organic leachate pollution is still rather limited (cf. Luckner 2010). Apart from the mixture of a multitude of substances in the leachate, also the lack of site-specific data on the characteristics of the subsurface system and the relevant boundary conditions render the evaluation of natural attenuation potentials a challenging task. Therefore, although there is evidence of such processes limiting groundwater pollutant plumes of unlined landfills to several hundreds of meters (cf. Table 4-13), conservative models are preferred to assess the migration of landfill-borne pollutants and their impact on specific environmental media. In general, convective transport and dispersion/diffusion processes in the vadose and groundwater zone are well investigated and understood, whereas large knowledge gaps exist with respect to processes of substance immobilization and transfer (cf. National Research Council 2000; Luckner et al. 2008). In essence, the complexity of the modeling approach should reflect the quality and availability of data on the investigated system. While simplifications and assumptions in simple models can cause the ignorance of significant processes in the real system, the advantages of such models are the limited amount of spatially distributed input data required to establish the model, the easier handling and application of the model, the robustness of the model with respect to reverse calculations of tolerable emission levels, and the possibility of using the simple model as a basis to adapt a higher tier model in a consecutive evaluation. Therefore, a comparatively simple analytical model (one-dimensional solution of the

advection-dispersion equation) is suggested to evaluate substance transport processes in the vadose zone within the evaluation methodology. Substance transport in the groundwater is described based on a mass balance approach taking into account primarily dilution of the leachate plume in the groundwater. However, provided that appropriate data are available to establish and calibrate more complex migration models (e.g. numerical models), the evaluation of attenuation potentials along the subsurface migration pathway may be also based on such models.

7.1.3 Methodology to derive aftercare completion criteria

The method to derive aftercare completion criteria forms the basis for evaluating environmental risks associated with the closed landfill. Within the evaluation, models about the emission behavior of the deposited waste, the performance of technical barriers, and the potential attenuation of pollutants in the environment are combined to derive aftercare completion criteria (cf. Figure 7-1).

The basis of the evaluation is the collection of data and information about the conditions at a site (e.g. climate, geology, topographic location), the containment system (=technical barriers), and the deposited waste and associated emissions (see Figure 7-1, 1.). In general, data need to be critically evaluated. For monitoring data this means, for instance, checking for representativeness of the sampling location, measurement protocols, collection efficiencies (with respect to landfill gas) or leachate discharge rates during sampling (with respect to landfill leachate). Monitoring at closed landfills could be optimized by gathering information in view of the situation at the site and the significance of the data for modeling purposes (e.g. Zweifel et al. 1999). For instance, measurements including a wide range of parameters could be done once every few years, whereas routine monitoring could be restricted to a few parameters (e.g. electric conductivity, leachate discharge rate, ammonium, and TOC) measured at a higher frequency/resolution to gain system understanding for model development. Provided that conditions at the site are constant (e.g. after top cover installation) the observation of a few parameters can also allow for statements about the behavior of other parameters for which correlations have been established (e.g. Kylefors 2003; Laner et al. 2011b). In all cases, input data and models are uncertain, at least due to natural variability, and should be treated accordingly.

The relevant emission parameters are identified based on the observations at the site and emission models are established to predict future emission characteristics (Figure 7-1, 2.). The substance concentrations in the leachate of a closed landfill were described by a model which depends on the mobilizable amount of substances in the waste, the heterogeneity of water flow in the landfill, the observed concentrations of the substance, and the rate of water infiltration. The effects of different or changing conditions at the landfill were illustrated via emission scenarios. For instance, scenario analysis typically included scenarios investigating the effect of different future performance levels of technical barriers. The layout and choice of the investigated scenarios depended on the range of potentially occurring conditions at the site with a significant effect on landfill emission behavior as well as on the scope of the evaluation (e.g. management periods in the focus of the evaluation). For instance, emissions associated with landfill flooding might be a relevant emission scenario at some sites, while insignificant at others (cf. Laner et al. 2009). The estimates on future emission levels derived from the individual scenarios formed the input for the pollutant migration modeling in the third step (Figure 7-1, 3.). Pollutant migration models were used to evaluate the concentrations of

selected pollutants in the affected environmental media. Scenario- and pollutant-specific attenuation factors were calculated along the migration pathways at defined points of compliance (PoC). Based on the attenuation factors, the quality criteria to be met at the designated point of compliance, and the choice of the critical long-term emission scenario, aftercare completion criteria were determined for the landfill in the fourth step (Figure 7-1, 4.). As the derived emission-related criteria were based on a specific long-term emission scenario, additional completion criteria could relate to the scenario-conditions underlying the evaluation. Such additional criteria may comprise geotechnical aspects (e.g. geotechnical stability of the landfill body), the functionality of landfill elements (e.g. containment system), surveillance and maintenance activities at the site, or restrictions of use. Finally, the models used as a basis of the emission scenario need to be confirmed by future monitoring and site observations (Figure 7-1, 5.). If model estimates and scenario assumptions are supported by the observations, the evaluation results and the corresponding completion criteria are validated. In case emissions and/or conditions at the site develop differently, the evaluation needs to be adapted or repeated to account for the observed landfill behavior. Thus, at many sites the application of the evaluation methodology and the determination of completion criteria resemble iterative processes.

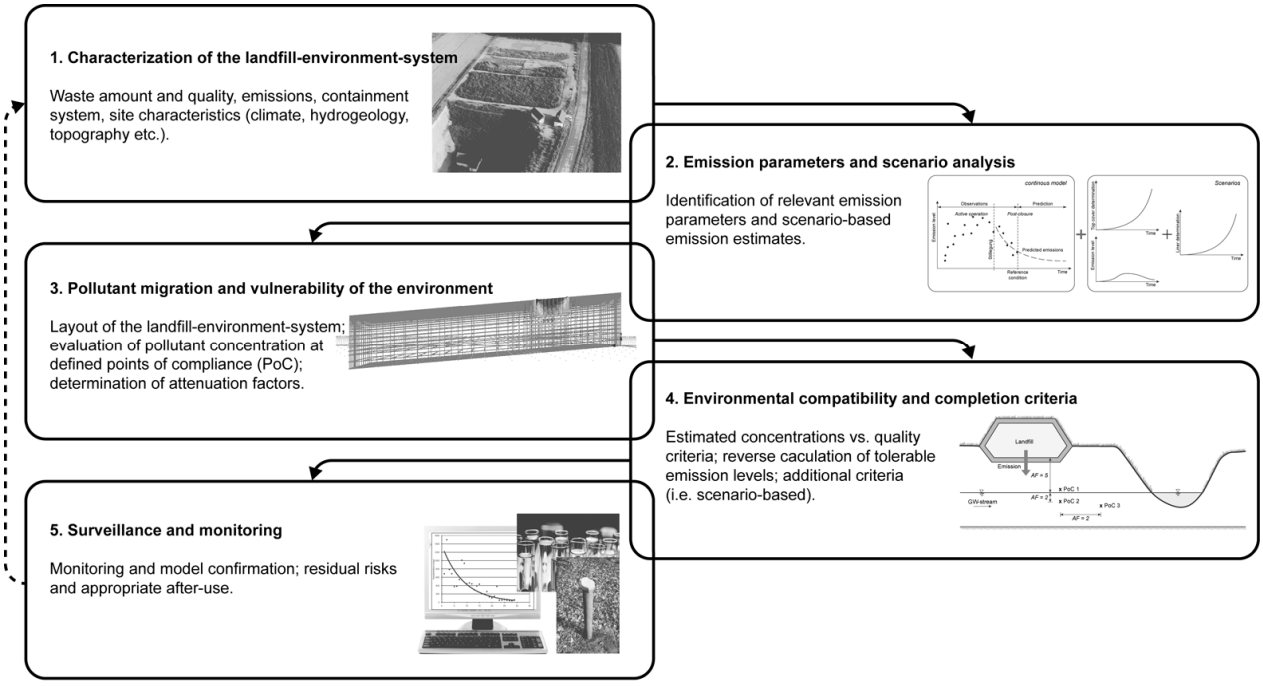


Figure 7-1: Schematic illustration of the procedure to derive aftercare completion criteria at a closed landfill site

7.1.4 Application of the methodology

The developed methodology was applied at three closed landfill sites to derive aftercare completion criteria. The case study landfills were chosen based on three factors: the availability of data, different waste types and varying environmental settings, and a positive attitude towards the evaluation by the landfill owner. At two landfill sites, municipal solid waste (Landfill A and Landfill B) was deposited and one landfill contains primarily construction and demolition waste (Landfill C). The volume of Landfill A is 1.7 million m³ with the deposition period from 1972 to 2005. The different

compartments of the landfill are equipped with different technical barrier systems ranging from mineral liner systems to composite liner systems at the landfill base. The climate at the site is temperate with an average annual precipitation of 956 mm/yr. Landfill B contains approximately 135000 m³ of MSW which was deposited at the site between 1988 and 2003. The landfill is equipped with a temporary cover (i.e. soil layer) at the top and a composite liner system at the bottom. The climate at the site is humid with an average annual precipitation of 1960 mm/yr. Landfill C was operated from 1992 to 2008 and contains 400000 m³ of construction and demolition waste. Mineral liner systems are installed at the top and bottom of the landfill. The climate at Landfill C is arid with 613 mm/yr of annual precipitation and 630 mm/r of annual potential evaporation.

At Landfill A and Landfill C the emission scenario assuming the gradual decrease of top cover performance and a total failure of the base lining system after the end of aftercare was used to derive aftercare completion criteria (= Scenario A*), because of the lack of gravity leachate drainage at these sites. At Landfill B leachate can gravity drain to a storage tank outside the landfill. Therefore, the scenario of gradual barrier performance decrease at the top and the bottom of the landfill (= Scenario A) was chosen as a basis to derive completion criteria for Landfill B (cf. Table 6-32)[28]. At Landfill A and Landfill C, the mixing zone of leachate and groundwater was the relevant point of compliance where drinking water quality standards had to be met. At Landfill B these quality standards had to be met 100 m downstream of the landfill in the groundwater. At all the case study landfills, ammonium-nitrogen was the most problematic leachate parameter with respect to the quality standards at the point of compliance (cf. Table 6-32). The time until the calculated completion criteria could be met at a landfill was estimated for different aftercare strategies. One strategy was to allow for high rainwater infiltration rates (i.e. minimum aftercare durations due to intensified leaching) and another strategy was to minimize rainwater infiltration effectively (i.e. maximum aftercare duration due to very slow decrease of mobilizable substance potentials). Consequently, for Landfill A the minimum aftercare duration was estimated to be another 80 years at the minimum (intensive water infiltration during aftercare) and more than 450 years at the maximum (continuous optimal function of the top cover system) until the completion criteria would be reached. For Landfill B, the equivalent durations were 11 years at the minimum to 110 years at the maximum, provided that (partial) collection and treatment of the leachate could be assured in the absence of aftercare. In the case of Landfill C, it would not be necessary to maintain aftercare based on the determined tolerable leachate emissions, but due to the poor quantity (only one year post-closure monitoring) and quality of monitoring data, further aftercare is necessary to confirm the underlying models and scenario assumptions. If future monitoring supports the evaluation, aftercare could be ended at Landfill C within the next decade without unacceptable environmental risks. A general finding was that high water infiltration rates flush out contaminants faster and thereby shorten necessary aftercare periods. However, at all case study sites additional completion criteria have to be fulfilled based on the emission scenarios underlying the evaluation

[28] Apart from Scenario A and A*, three more scenarios on barrier performance were investigated for the case study landfills: Status quo: constant barrier performance at the top and bottom of the landfill; status quo*: constant barrier performance at the top and failure at the bottom; Scenario B: total barrier failure at the top and at the bottom of the landfill.

(e.g. undisturbed content or gradual decrease of top cover performance in the absence of aftercare). Finally, it should be highlighted that the case study evaluations were meant to illustrate the applicability and flexibility of the developed evaluation method. An application of the methodology as a decision support tool for aftercare completion by the authorities may well be different with respect to the underlying emission scenarios, points of compliance, or quality standards.

7.2 Conclusions and research outlook

The evaluation of landfill environmental compatibility needs to address different aspects significant to the environmental risk associated with a landfill. This includes 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. The evaluation of long-term landfill management and aftercare completion is based on the environmental risks associated with a closed landfill at the end of aftercare. Thus, the evaluation of aftercare resembles a site-specific endeavor about acceptable risks (cf. Scharff et al. 2011). As data available to characterize the system are limited, system understanding is incomplete, and future conditions are unknown, there is substantial uncertainty inherent in the evaluation. Consequently, the developed methodology builds on scenario-based assessments to determine the environmentally tolerable state of the landfill at the end of aftercare. Aftercare completion criteria are derived based on the combination of different models and include tolerable emission levels at the site as well as criteria associated with the long-term emission scenario underlying the evaluation.

The methodology was applied to three closed landfills to illustrate the determination of aftercare completion criteria. For the two closed MSW landfills the evaluation showed, that aftercare periods of hundreds of years have to be expected until leachate emissions become environmentally tolerable, if the landfills are equipped with (constantly) impermeable top covers after closure. If higher water infiltration is maintained after landfill closure to intensify contaminant leaching, management periods could be shortened. However, for the relatively large Landfill A aftercare would still have to be maintained at least for a period of 80 years from now. For Landfill B, a site with humid climate and high groundwater recharge rates, the minimum duration of aftercare was estimated to be a few decades, provided a cover allowing for high water infiltration rates would be maintained at the site. The evaluation of Landfill C (closed construction and demolition waste landfill) showed that aftercare may be terminated within the next decade(s), provided that the underlying models and scenario assumptions can be verified by future monitoring. The low emission potential of the waste, the arid climate at the site, and the high dilution potential in the groundwater below the landfill are the main factors enabling the completion of aftercare without inacceptable environmental risks at this site.

The presented methodology allows for defining the desired state of a landfill at the end of aftercare, which is a crucial element to optimize aftercare strategies in view of environmental risks associated with a closed landfill. The costs for aftercare can be determined for different long-term management concepts ranging from the encapsulation of the waste to additional efforts (e.g. enhanced emission reduction) to accelerate the transition of the landfill into an environmentally tolerable state in the absence of aftercare. Therefore, the evaluation methodology may present the basis to

optimize the use of available financial resources to assure cost-effective protection of humans and the environment at closed landfills. Overall, a consistent and transparent evaluation of aftercare is a prerequisite for estimating the full cost of landfilling including long-term care obligations at the landfill sites.

The evaluation framework presented in this work is intended to be used by authorities to decide on appropriate aftercare strategies and aftercare completion at a site. However, experience with implementing the developed approach is yet to be gathered. Well documented case studies that include regulatory review and acceptance are needed to evaluate the utility of the evaluation methodology as a decision support tool on landfill aftercare and its completion. In addition to the assessment of the method's practicability, the increasing availability of data from case study landfills will provide further insights about the emission behavior of closed landfills, the performance of barriers at the field scale, and the environmental conditions encountered at closed landfills. The experience and data gathered during the application of the methodology at different sites and by different stakeholders should be subject to critical discussion in order to arrive at a regulatory procedure for assessing the environmental significance of closed landfills and the associated aftercare requirements.

The importance of site-specific conditions for the evaluation of landfill aftercare completion is highlighted by the results of the case study applications. The climatic conditions, the geologic and hydrogeologic setting, and the vicinity of vulnerable uses are important factors with respect to the determination of environmentally tolerable emission levels. Particularly in view of mid- to long-term environmental compatibility, the significance of site specific-conditions should be emphasized during site selection for new landfills. An insensitive environmental setting (e.g. large thickness of clayey soil layer above the groundwater, high dilution potential in the groundwater stream, absence of sensitive (current and future) uses, etc.) facilitates the maintenance of a low level of environmental risk at a closed landfill and is potentially associated with a reduced intensity of long-term management relative to landfills at sites with less favorable conditions (e.g. gravel layer of low thickness between landfill base and groundwater). However, for many existing, closed landfills the accrued funding might be insufficient to fulfill the required aftercare obligations (e.g. in Austria funding has been accrued for 40 years of aftercare at MSW landfills). Thus, concerted efforts of landfill owners and authorities are needed to develop integrated concepts for the long-term management of these sites as well as for landfills which are currently building up their financial aftercare provisions.

While the evaluation of environmental risks associated with a closed landfill is the basis for cost-effective protection of humans and the environment, the significant uncertainties present in the evaluation results counteract their use for decision-support. Long-term estimates are particularly confronted with large knowledge gaps with respect to dominant release processes and process rates, the potential degradation of materials (waste and containment), and the variation of external forces at the site (e.g. climate change, extreme events, human intrusion, etc.). Further research is needed to increase the understanding of the landfill emission behavior as well as the interaction of emissions with technical and natural barriers. For instance, the effect of water flow heterogeneity and its potential change on landfill emissions is of major importance to understand the evolution of the mobilizable substance potential accessible to mobile water. Investigations focusing on different

hydraulic regimes in the waste body and the associated fractions of preferential flow are essential to understand the influence of different water infiltration rates on substance concentrations in the leachate. Research about the lasting effect of enhanced emission reduction measures (e.g. aeration and flushing) is necessary to evaluate the effect of such measures on mobilizable substance potentials and future emission rates. Due to the long periods of time over which residual emission levels may persist at landfills, the investigation of low-intensity technologies to manage these emissions (e.g. methane oxidation in bio-active covers or extensive leachate treatment) is important to optimize landfill management. Research efforts to define mobilizable substance potentials in the waste in view of different release mechanisms, time frames, and environmental conditions would be valuable for long-term emission estimates, but need to cope with substantial uncertainty due to the extended timeframes involved in such evaluations. In addition, research in the field of geotechnical landfill engineering (e.g. performance levels of technical barrier systems over time) or geohydraulics (e.g. natural attenuation of typical leachate constituents) can increase the understanding of the landfill-environment system. However, due to the long time frames and the complex interaction of system components, uncertainty is inherent in an evaluation of risks associated with landfilling. Apart from increasing the reliability of the evaluation of environmental compatibility, a joint effort involving all stakeholders in waste management will be necessary to optimize long-term landfill management.

8 Literature, Figures, and Tables

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8.2 List of Figures

8.3 List of Tables

Appendix

Appendix 1

Basic landfill data (2 pages)

Appendix 2

Expert evaluations of long-term barrier performance (5 pages)

Appendix 1: Basic landfill data

a) Municipal solid waste (MSW) landfills

Table A-1: General characteristics of the sites included in the data set on MSW landfills

#	Landfill age* [yrs]	Years after closure* [yrs]	Precipitation [mm/yr]	Leachate quality (Param./Sampling times)	Gas data
MSW1	28	7	1485	21 / 85	yes
MSW2	18	-	1150	21 / 69	yes
MSW3	36	9	1150	21 / 85	yes
MSW4	36	32	956	52 / 53	yes**
MSW5	32	29	956	52 / 52	yes**
MSW6	28	23	956	52 / 49	yes**
MSW7	23	19	956	52 / 40	yes**
MSW8	18	8	956	52 / 29	yes**
MSW9	19	8	956	52 / 31	yes**
MSW10	11	3	956	52 / 16	yes**
MSW11	14	6	956	52 / 22	yes**
MSW12	22	20	633	21 / 72	no
MSW13	22	20	633	21 / 72	no
MSW14	22	20	633	21 / 72	no
MSW15	25	13	1300	18 / 43	no
MSW16	20	-	923	24 / 37	yes
MSW17	24	9	926	10 / 43	no
MSW18	17	0	1220	11 / 194	yes
MSW19	26	20	658	25 / 22	no
MSW20	28	17	525	75 / 10	no
MSW21	16	8	525	75 / 10	no
MSW22	16	4	520	75 / 10	no
MSW23	23	4	562	75 / 16	no
MSW24	32	4	490	75 / 10	no
MSW25	31	12	500	75 / 16	no
MSW26	32	19	547	75 / 16	no
MSW27	26	19	810	75 / 10	no
MSW28	26	19	810	75 / 10	no
MSW29	22	11	840	75 / 10	no
MSW30	31	4	770	75 / 20	no
MSW31	23	6	710	75 / 10	no
MSW32	25	9	660	75 / 14	no
Mean	**24**	**13**	**832**	**50 / 39**	-

* Data on the landfill age and PCC duration refer to the year 2008.

** For the landfill compartments MSW4 to MSW11 gas data is available only in aggregate (for the whole landfill site).

b) Municipal solid waste incineration (MSWI) bottom ash landfills

Table A-2: General characteristics of the sites included in the data set on MSWI bottom ash landfills

#	Landfill age* [yrs]	Average waste age [yrs]	Closure	Precipitation [mm/yr]	Leachate quality (Param./Sampling times)
BA1	10	3.8	no	1100	43 / 32
BA2	22	14.5	no	1117	37 / 52
BA3	18	9.5	no	1035	28 / 98
BA4	12	7.7	no	1035	28 / 96
Mean	**15.5**	**8.9**	**-**	**1070**	**34/70**

* Data on the landfill age and PCC duration refer to the year 2008.

c) Construction and demolition (C&D) waste landfills

Table A-3: General characteristics of the sites included in the data set on C&D waste landfills

#	Landfill age* [yrs]	Average waste age [yrs]	Closure	Precipitation [mm/yr]	Leachate quality (Param./Sampling times)
C&D1	14	7.9	no	1155	37/34
C&D2	7	2.5	no	900	22/6
C&D3	4	2.1	no	1179	73/17
C&D4	?	?	no	620	4/9
C&D5	Leachate quality monitoring data from 8 C&D waste landfills (no additional information)				8/18

Appendix 2: Expert evaluations of technical barrier systems

A survey was conducted in two steps during the year 2010 and involved 23 experts on landfill geotechnics from Austria and Germany. In the first step, 23 experts were invited to create a list of relevant factors for barrier performance based on a suggestion by the author. 13 of the 23 invited experts accepted the invitation and participated in the list of relevant factors. In the second step, the 13 experts were asked to evaluate the importance of each factor for the standard top cover and base lining systems shown in Figure 4-2 and Figure 4-4 at a landfill site with typical Austrian climate (e.g. precipitation around 1000 mm/yr). For three different time periods (several decades, around a century, and several centuries) each factor had to be evaluated with scores ranging from 1 to 5 (1 being insignificant and 5 being highly significant for the performance of the barrier system). The histograms of the expert evaluations are shown for each factor and period below for the two bottom barrier and the two top cover systems (cf. Table A-5 to Table A-8). More information about the expert survey, the original survey forms, and more extensive analysis of the expert evaluations can be obtained from the report published by Laner et al. (2011)[1].

Table A-4: Affiliations of the 13 experts involved in the evaluation of factors determining the performance of technical barrier systems

Affiliation	Authorities*	Consulting and engineering companies	Research and academic institutions
Number of experts	4	4	5

*Authorities: Environmental Protection Agency, Ministries, and Provincial governments

Note:

8 experts have their primary background in geotechnical engineering, 5 experts in waste management.

Note:

The percentages do not add up to 100% for some histograms in the Tables A-5 to A-8 due to missing score assignments for the respective factor and period by the experts.

[1] Laner, D., J. Fellner, Brunner, P.H. (2011): Standortbezogene Kriterien zur Beurteilung der Umweltverträglichkeit von Deponieemissionen unter dem Aspekt der Nachsorgedauer (Projekt SKUDENA). Final report of the research project SKUDENA. Institute for Water Quality, Resources and Waste Management, Vienna University of Technology, Vienna. 452 pages + appendices.

Table A-5: Histograms of expert scores to evaluate the importance of a factor for the performance of a top cover system with a composite liner (cf. Figure 4-2) during different service periods

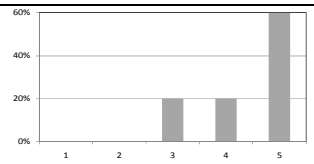
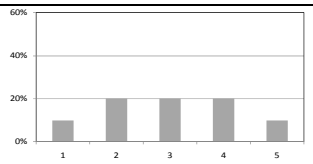
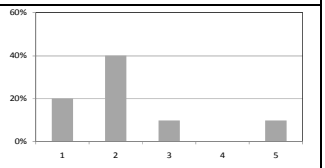
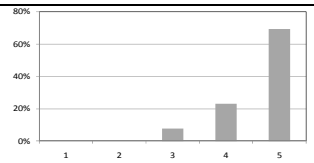
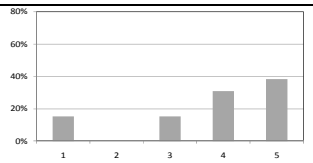
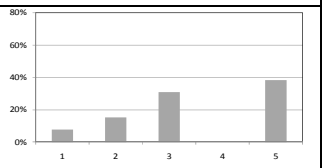
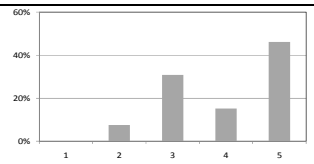
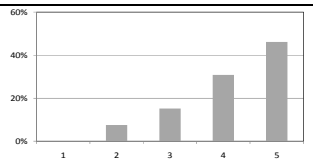
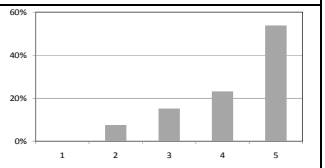
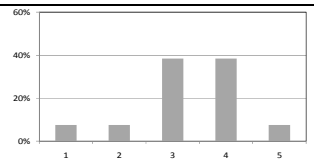
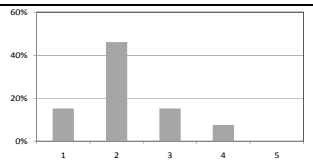
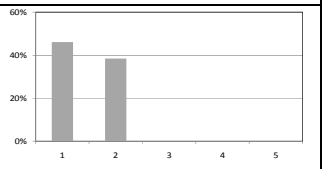
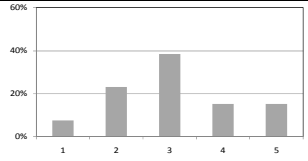
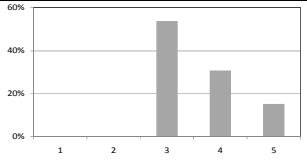
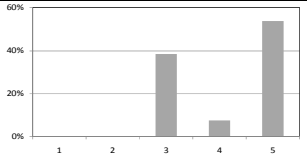
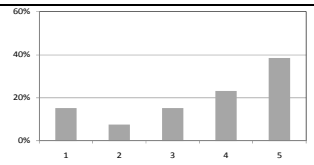
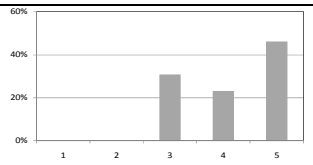
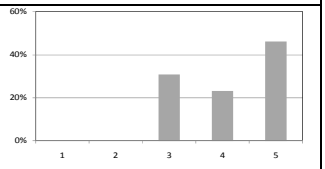
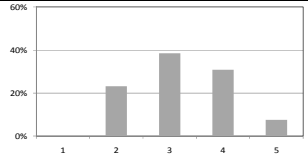
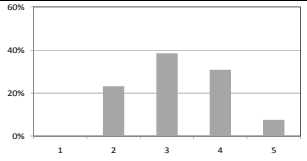
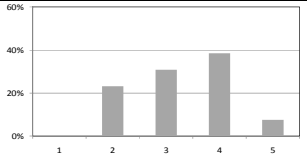
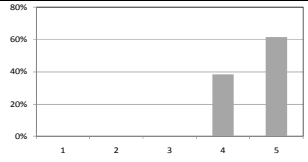
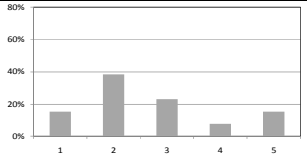
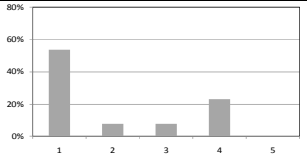
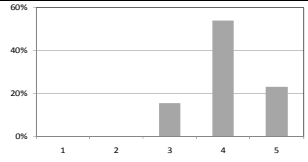
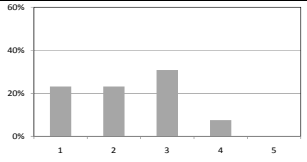
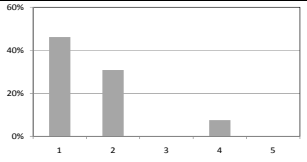
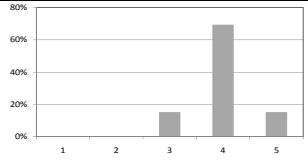
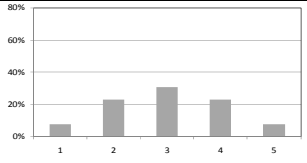
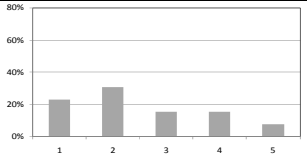
Criterion	**$1_{insignificant} - 5_{very\ high\ significance}$**		
	0-50 yrs	**50- >100 yrs**	**>> 100 yrs**
Observed barrier performance			
Construction & design quality assurance program			
Recultivation layer (vegetative cover thickness)			
Heat production in the waste			
Climate (e.g. temperature, precipitation)			
Vegetation and projected after-use			
Relief			
Expected settlements			
Lining control systems (direct monitoring)			
Drainage system			

Table A-6: Histograms of expert scores to evaluate the importance of a factor for the performance of a top cover system with a mineral liner (cf. Figure 4-2) during different service periods

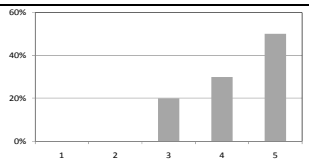
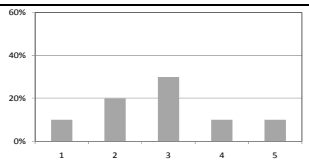
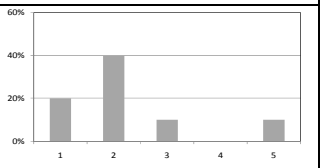
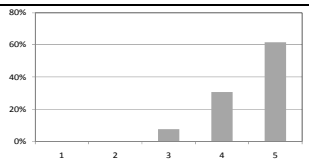
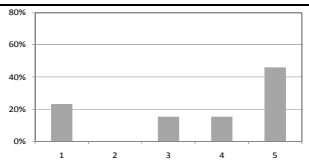
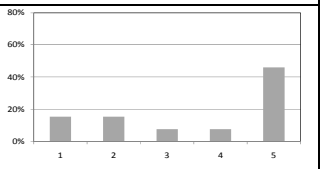
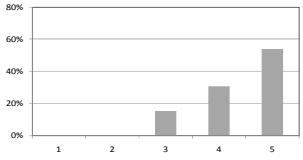
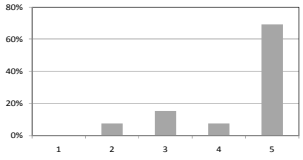
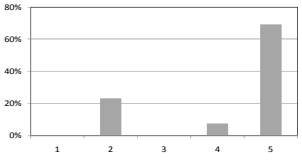
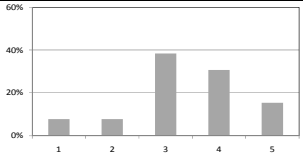
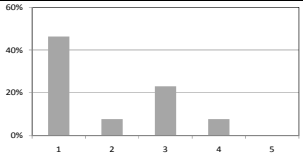
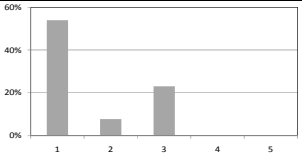
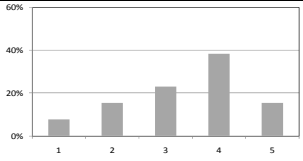
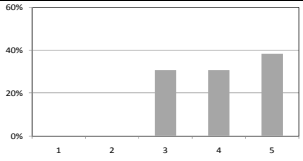
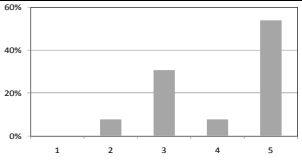
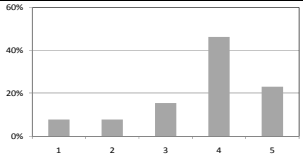
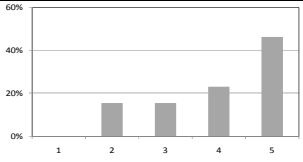
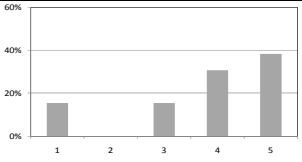
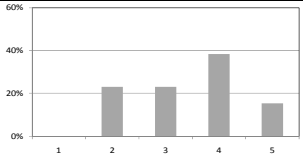
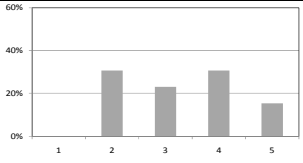
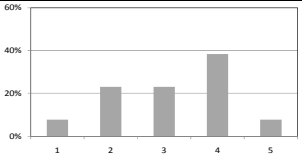
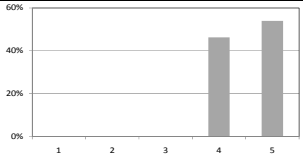
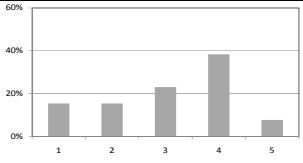
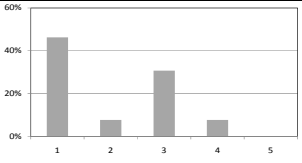
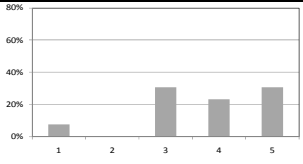
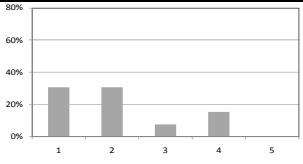
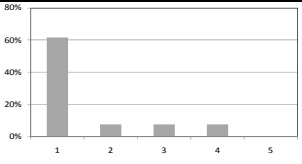
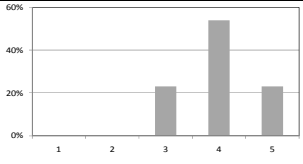
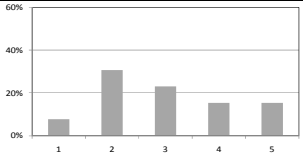
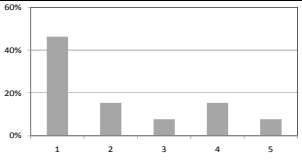
Criterion	$1_{\text{insignificant}} - 5_{\text{very high significance}}$		
	0-50 yrs	**50- >100 yrs**	**>> 100 yrs**
Observed barrier performance			
Construction & design quality assurance program			
Recultivation layer (vegetative cover thickness)			
Heat production in the waste			
Climate (e.g. temperature, precipitation)			
Vegetation and projected after-use			
Relief			
Expected settlements			
Lining control systems (direct monitoring)			
Drainage system			

Table A-7: Histograms of expert scores to evaluate the importance of a factor for the performance of a bottom barrier with a composite liner (cf. Figure 4-4) during different service periods

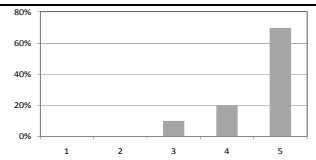
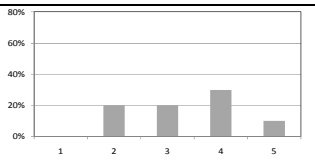
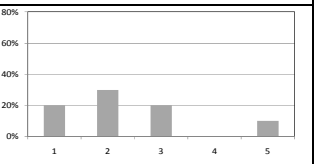
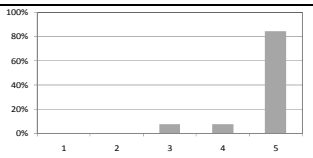
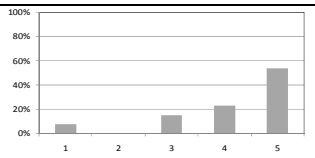
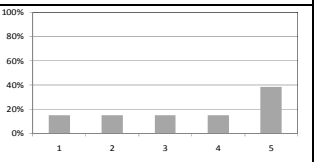
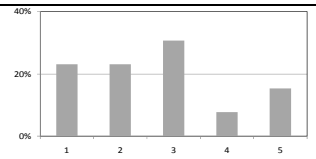
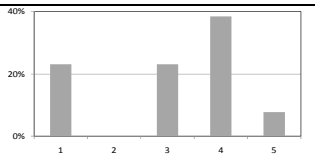
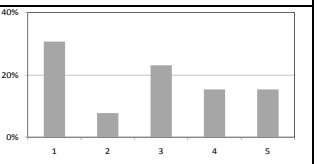
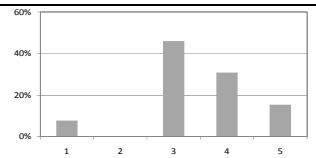
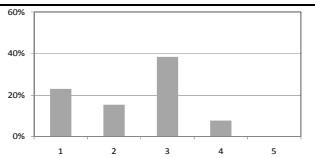
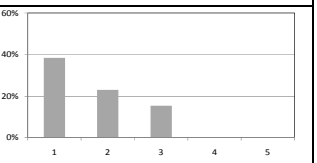
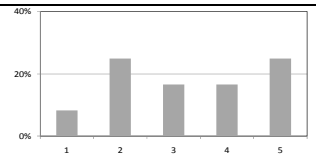
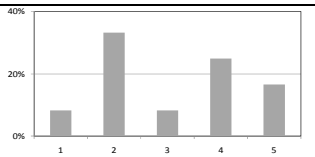
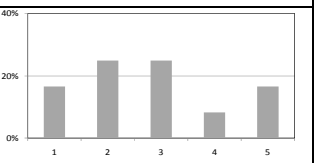
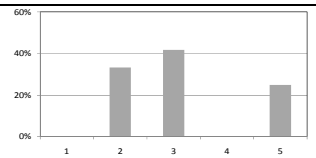
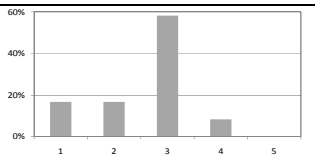
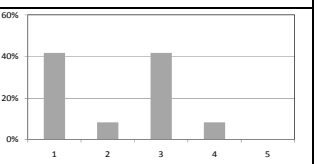
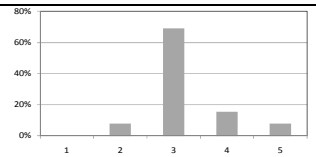
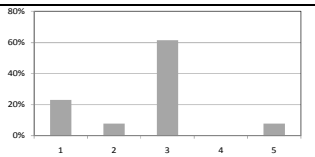
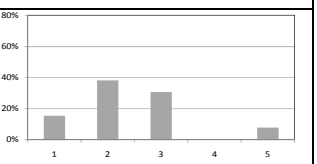
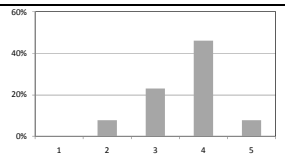
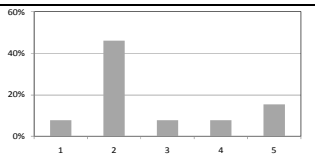
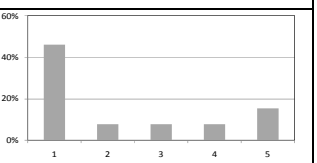
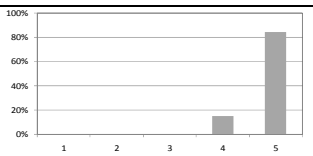
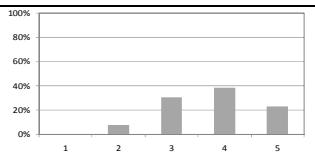
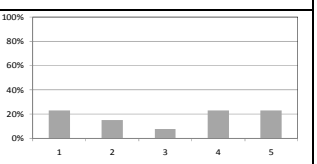
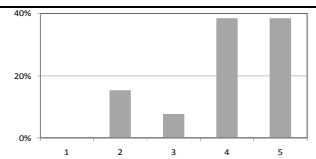
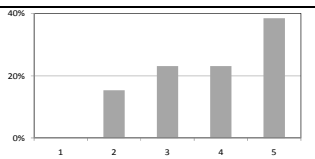
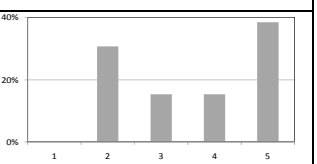
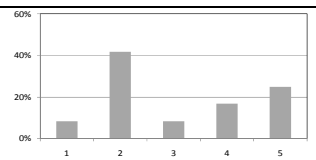
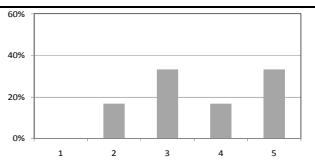
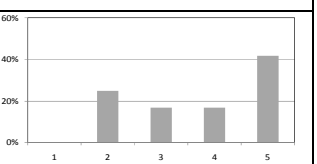
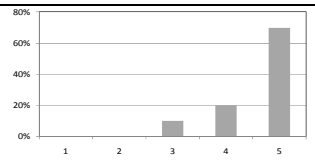
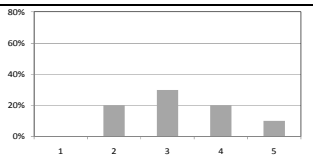
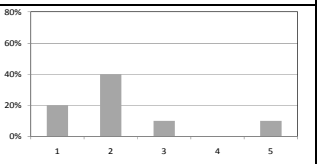
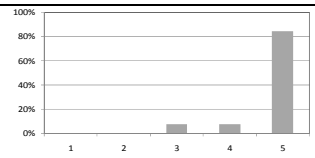
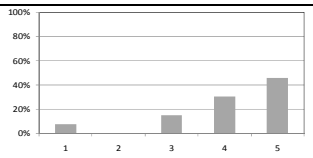
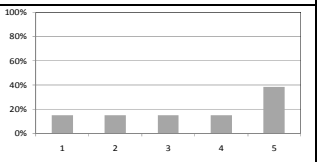
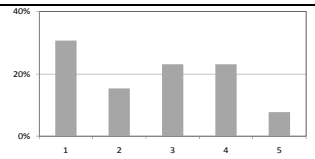
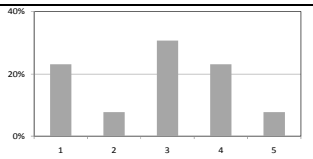
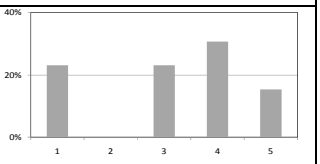
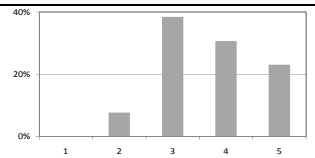
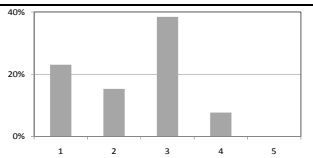
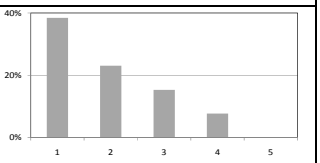
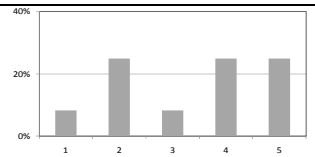
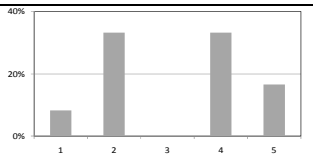
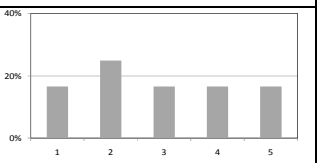
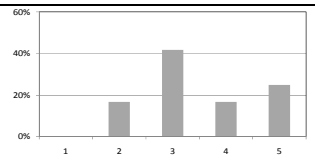
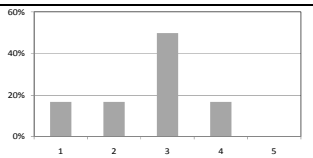
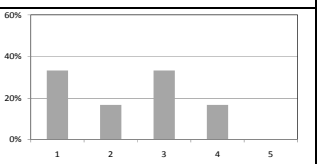
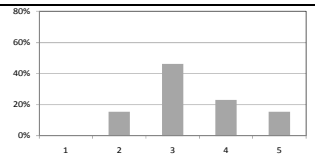
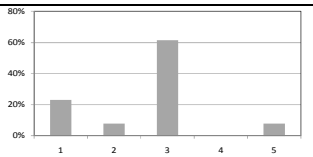
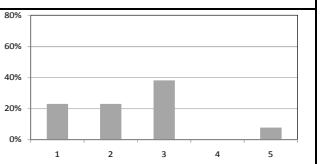
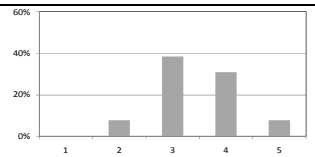
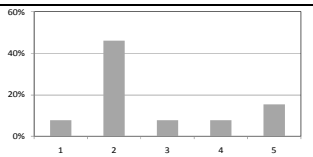
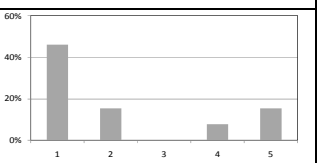
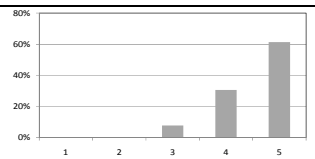
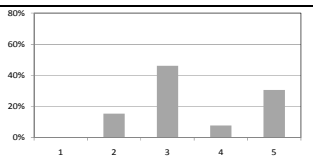
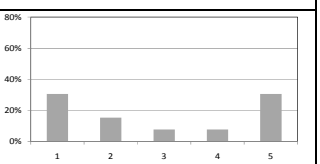
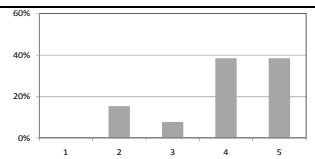
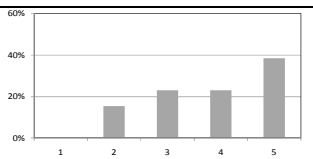
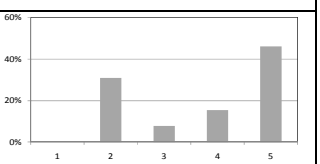
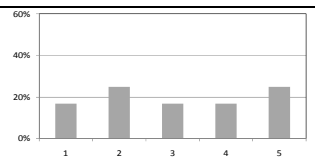
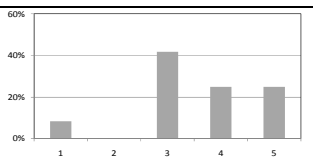
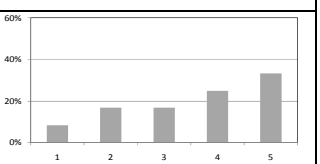
Criterion	**$1_{insignificant} - 5_{very\ high\ significance}$**		
	0-50 yrs	**50- >100 yrs**	**>> 100 yrs**
Observed barrier performance			
Construction & design quality assurance program			
Climate (e.g. temperature, precipitation)			
Heat production in the waste			
Distance to groundwater surface			
Overburden pressure			
Leachate quality			
Monitoring (e.g. groundwater monitoring)			
Leachate drainage system			
Stability of foundation			
Landfill geometry (heap vs. cavity)			

Table A-8: Histograms of expert scores to evaluate the importance of a factor for the performance of a bottom barrier with a mineral liner (cf. Figure 4-4) during different service periods

Criterion	**$1_{insignificant} - 5_{very\ high\ significance}$**		
	0-50 yrs	**50- >100 yrs**	**>> 100 yrs**
Observed barrier performance			
Construction & design quality assurance program			
Climate (e.g. temperature, precipitation)			
Heat production in the waste			
Distance to groundwater surface			
Overburden pressure			
Leachate quality			
Monitoring (e.g. groundwater monitoring)			
Leachate drainage system			
Stability of foundation			
Landfill geometry (heap vs. cavity)			

Curriculum Vitae of David Laner

Vital statistics

Date of birth:	25.04.1979
Place of birth:	Schärding
Nationality:	Austrian

Education

X. 2006 – VI. 2011	Dotoral studies at the Vienna University of Technology, Faculty of Civil Engineering.
II. 2011 – III. 2011	Visiting researcher at the North Carolina State University, Department of Civil, Construction, and Environmental Engineering.
X. 2006 – IX. 2008	Participation in the Post Graduate School of Industrial Ecology, Norwegian University of Science and Technology (Marie Curie Program).
X. 1999 – XI. 2005	Master studies of "Land and Water Management and Engineering" at the University of Natural Resources and Applied Life Sciences, Vienna.
X.1997 – IX. 1998	Master studies of "Land and Water Management and Engineering" at the University of Natural Resources and Applied Life Sciences, Vienna.
VI. 1997	A-Levels at the grammar school (Bundesrealgymnasium) in Schärding, Upper Austria.

Professional experience

IV. 2006 -	Research associate, Vienna University of Technology, Institute for Water Quality, Resources and Waste Management
XII. 2005 - III. 2006	Research associate, Groundwater Research Center Dresden (Scholarship of the foundation: "Stiftung zur Förderung der wissenschaftlichen Schule Zunker-Busch-Luckner")
2003 - 2005	Employment as a student worker at the consulting engineers „Büro Dr. Lengyel" for 3-4 months each year
1994 - 2003	Various jobs for a few months per year during summer vacations (e.g. City of Vienna, Department for the sewer system (MA 30): internship; Gesta Fertighäuser: construction site worker).
X. 1998 – IX.1999	Civil service at the Lebenshilfe Oberösterreich (Tagesheimstätte Münzkirchen)