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RISK ASSESSMENT OF MODERN LANDFILL STRUCTURES IN FINLAND

Thesis for the degree of Doctor of Science (Technology) to be presented with due permission for public examination and criticism in the Auditorium of the Student Union House at Lappeenranta University of Technology, Lappeenranta, Finland on the 18th of September, 2015, at noon.

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Abstract

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The purpose of this thesis was to investigate environmental permits of landfills with respect to the appropriateness of risk assessments focusing on contaminant migration, structures capable to protect the environment, waste and leachate management and existing environmental impacts of landfills. According to the requirements, a risk assessment is always required to demonstrate compliance with environmental protection requirements if the environmental permit decision deviates from the set requirements. However, there is a reason to doubt that all relevant risk factors are identified in current risk assessment practices in order to protect people end environment.

In this dissertation, risk factors were recognized in 12 randomly selected landfills. Based on this analysis, a structural risk assessment method was created. The method was verified with two case examples.

Several development needs were found in the risk assessments of the environmental permit decisions. The risk analysis equations used in the decisions did not adequately take into account all the determining factors like waste prospects, total risk quantification or human delineated factors. Instead of focusing on crucial factors, the landfill environmental protection capability is simply expressed via technical factors like hydraulic conductivity.

In this thesis, it could be shown, that using adequate risk assessment approaches the most essential environmental impacts can be taken into account by consideration of contaminant transport mechanisms, leachate effects, and artificial landfill structures. The developed structural risk analysing (SRA) method shows, that landfills structures could be designed in a more cost-efficient way taking advantage of recycled or by-products. Additionally, the research results demonstrate that the environmental protection requirements of landfills should be updated to correspond to the capability to protect the environment instead of the current simplified requirements related to advective transport only.

Keywords: landfill, contaminant transport, geological barrier, environmental protection, EC Landfill Directive, Structural Risk Analysing method.

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Oulu, September 2nd, 2015

Hannu Aurinko

Abbreviations

ASTM	The American Society for Testing and Materials
COD_{Mn}	Chemical Oxygen Demand, oxidation with permanganate
DepV	Deutsche Gesellschaft für Geotechnik e.V., German Geotechnical Society
DIN	Deutsches Institut für Normung e.V., German Institute for Standardization
DOC	Dissolved Organic Carbon
EC	European Council
EPA	United States Environmental Protection Agency
EU	European Union
GCL	Geosynthetic Clay Liner
GLO -85	Geotekniset Laboratorio-ohjeet 1985, Geotechnical laboratory instructions 1985
HDPE	High Density Polyethylene
ICT	Intensive Compaction Tester
LPR	Lappeenranta
MSW	Municipal Solid Waste
NH_4^+-N	Ammonium nitrogen
NO_3-N	Nitrate nitrogen
NO_2-N	Nitrite nitrogen
RVF	Renhållningsverksföreningen, Swedish Association of Waste Management
SRA	The structural risk analyzing method
SBP	Sodium Bentonite Polymer
SFS-EN	European Standard implemented in Finland
TASi	Technische Anleitung Siedlungsabfall, German Technical Instructions on Municipal Waste
TOC	Total Organic Carbon
VNp	Valtioneuvoston päätös, Finnish Government decision

Symbols

Γ_B	Biological decay constant
Γ_R	Radioactive decay constant
Γ_S	Volume of fluid removed/unit volume of soil/unit
β_x	Probability coefficient of unidentified risk
Θ	Volumetric water content
γ_{dmax}	Maximum value of dry unit weight
γ_{sat}	Saturated unit weight
λ	The first order decay constant
ρ_d	Dry density
τ	Tortuosity factor
1 D	One Dimensional
2 D	Two Dimensional
3 D	Three Dimensional
1/t	-/time
A	Cross section area
C	Concentration of the solute
C_0	Concentration solute of time (0)
C(t)	Concentration solute of time (t)
d	Thickness of the layer
D	Diffusion coefficient
D_x	Hydrodynamic dispersion coefficient in direction x
e	Void ratio
g	Acceleration due to gravity
h	Thickness of the layer
h_w	Hydraulic head
Δh	The elevations of fluid levels
H_T	Distance from groundwater
i	Hydraulic gradient

k	Hydraulic conductivity
$k_{leachate}$	Hydraulic conductivity determined by leachate
K	Intrinsic permeability
K_d	Distribution coefficient
L	Material layer thickness
m	Mass of contaminant transported into the soil
n	Porosity
N	Total number of measurements
P_x	Ranking value of a risk factor
Q	Flow rate
Q_x	Probability coefficient of identified risk
R	Retardation factor
R_{id}	Identified risk factor
Rud	Unidentified risk factor
R_{total}	The total risk level
S	Coefficient of Sorption
S_m	Quantity of medium sorption
S_r	Degree of saturation
t	Time
v	Darcy velocity
$v_{x,y,z}$	Velocity in x, y and z components
V_g	Velocity of groundwater
V_s	Seepage velocity
w	Water content
w_{opt}	Optimum water content

Glossary

Active phase:	The time period during which waste is deposited at a landfill.
Artificial barrier:	The constructed barrier to contain the landfilled waste and emissions (bottom or surface).
Artificial layer:	The constructed layer of an artificial barrier at a landfill (e.g. HDPE geomembrane).
Base ground:	The layer is consolidated rock or soil on which the landfill is founded.
Design & construction:	The period when the landfill structural design is developed, risk assessment and environmental evaluation is done, as well as construction work is carried out.
Disposal:	The deposition of waste in a landfill.
Geological barrier:	An artificial barrier or natural barrier or their combination to protect the migration of leachate and the migration of biogas to the environment, a mechanical support to the waste and a geological structure to ensure safety in the long term against possible of base ground pollution.

Groundwater balance: The groundwater balance is the balance of a groundwater body in terms of incoming hydraulic flow associated with groundwater inflow into the groundwater body, associated with the outflow and groundwater level.

Human delineated factors: The human manufactured factors effecting on landfill during the life-cycle e.g. built facilities, old landfills, human effects to groundwater.

Hydraulic gradient: The hydraulic gradient is a difference between two or more hydraulic head measurements over the flow path of the material.

Landfill operator: The natural or legal person responsible for a landfill in accordance with the internal legislation and responsible for landfill management (Landfill owner = landfill operator).

Municipal waste: The waste from households and other waste which has a composition similar to waste from households.

Passive phase: The management of a closed landfill, including monitoring, maintenance, aftercare and treatment of emissions, until no more monitoring measures are necessary and landfill does not cause threat to human or environment (Landfill post-closure period = passive phase).

Protection capacity: The environmental protection capacity to protect the humans and the environment from landfill emissions.

Waste management: Waste management includes the following activities:

1. Generation of waste: Storage, collection, transport, treatment and disposal of waste;
2. Waste treatment and environmental considerations: Control, monitoring and regulation of the production, collection, transport, treatment and disposal of waste; and
3. Waste minimization: Prevention of waste production through in-process modification, reuse and recycling.

Contents

1 Introduction

1.1 Background and motivation for the study

Since the 1970s, environmental consequences of waste materials and landfills have become an increasing concern. This has highlighted the importance of the design of landfills and environmental protection structures. In the 1990s, Germany (Deutsche Gesellschaft für Geotechnik e.V., 1996) and the United States Environmental Protection Agency (USEPA, 1995, updated 2008) launched a large-scale project to develop the national requirements for landfill structures. Partly based on these projects, in Europe and the United States, the requirements were determined for landfill structures, waste classification and environmental protection. In Europe, the European Union has set the latest requirements, the European Commission Landfill directive (The European Union Waste Framework Directive EC 98, 2008, The European Union Landfill Directive EC 31, 1999).

Finland had 561 landfills in end of the 1990s and in 2005 only 140 municipal waste landfills. Today 35 modern landfills exist in Finland, and the rest are closed according to EU the directives. None of the modern landfills have been sealed yet.

The EC directives, the waste laws and decrees determine the location of landfills, the terms of environmental protection and the structural dimensions of bottom, sides and surface structures. These requirements set the principles of the landfill management. *"The Landfill Directive describes the general principles for the acceptance of waste in the various classes of landfills upon which the waste classification should be based".* In the directive EC 31 (1999), landfills have been classified into three classes depending on the waste quality: inert, municipal and hazardous waste landfills. In addition, the waste management, which is a part of the landfill management, includes leachate control, collection and treatments and gas control, which are determined in the related EC directive and local regulations (EC 31, 1999; VNp, 1049 1999; VNp 861, 1997).

"European Union member states were required to bring into force the laws, regulations and administrative provisions necessary to comply with the Landfill Directive no later than 16 July 2001. The directive sets requirements for the authorization, design, operation, closure and aftercare of landfills". (European Commission, 2005; European Commission, 1999) The EC Landfill Directive provides the framework for the national legislation, within which the member states must operate. An EU member state may surpass the directives nationally with requirements for the environmental protection of landfills that are stricter, but not less strict, than the directive.

According to the national legislation, the landfill owner has to apply for an environmental permit for landfilling, in which the environmental impacts of landfilling are defined during its whole life-cycle. Landfill structures are licensed by the local authorities, and the environmental permits have to be based on the EC Landfill Directive and national laws. The Landfill Directive and Finnish Government Decision on Landfills determine the framework and recommendations according to which the landfill bottom and surface structures have to be realized. Figure 1 presents a conceptual presentation of a typical landfill structure based on the Finnish Government Decision on Landfills. (EC 31, 1999; VNp 861, 1997)

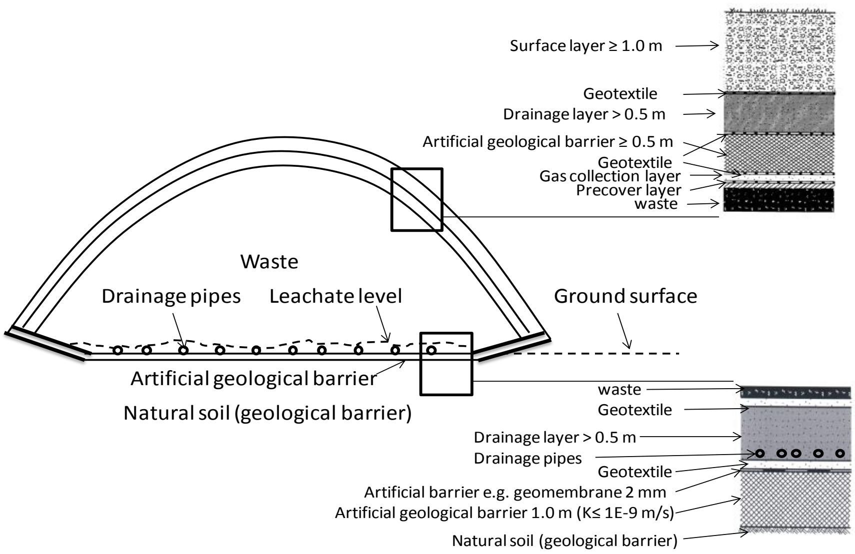

Fig. 1. Conceptual presentation of the landfill structure (VNp 861, 1997).

The objective of the environmental permit procedure is to ensure that the landfill owner has prepared a plan to protect the landfill environment using the Best Available Technique (BAT) and specialists with sufficient expertise. The plan includes structures for the protection of the environment with the help of which the environment, groundwater, surface water and the climate can be protected on a sufficient protection level during the whole life-cycle of the landfill according to the laws and regulations in force.

The Landfill Directive defines structural requirements for their surface and bottom structures, the objective of which is to protect human health and the environment from negative impacts of waste deposition. The directive admits of possible exceptions to the structural requirements if the protection capability of the deviations can be demonstrated to be corresponding with the help of risk assessment. The objective of the risk assessment is to ensure that the planned deviation will not cause an extra risk of environmental pollution due to the landfill for at least 30 years or longer depending on protection demands.

The directive does not require the presence of a geological barrier if risk assessment has been conducted and it can be demonstrated that the risk to soil, groundwater or surface water is acceptable which means that it does not cause any risk to humans. Risk assessments do not need to be conducted in MSW landfills if a natural geological layer fulfilling the hydraulic conductivity values ($k \leq 1 \cdot 10^{-9}$ m/s) can be utilised and if the thickness (1 meter) of the geological barriers is in accordance with the directive. In hazardous landfills, the natural geological barrier demands are for hydraulic conductivity ($k \leq 1 \cdot 10^{-9}$ m/s) and the thickness (5 meter).

"The second alternative is to enhance the geological barrier of the landfill by providing an additional artificial layer to meet an attenuation protection capacity equivalent to those provided by the hydraulic conductivity values and thickness defined in the directive. The interpretation made by the Finnish Authorities was that a 0.5 meter layer was considered as the minimum thickness to guarantee long lasting hydraulic conductivity ($k \leq 1 \cdot 10^{-9}$ m/s) for artificial geological barriers." (EC 31 1999; VNp 861 1997)

Landfills also have to be sealed after the waste deposing period has passed. MSW landfills receive biodegradable waste that formulates landfill gases. Landfill gas has to be collected from the landfill and gas must be treated or used e.g. to produce energy. According to the Government decision, the surface structures of MSW landfills have to include a gas collection layer, a geological barrier at least half a meter deep, and a drainage layer. In addition on the top a surface layer with a depth of one meter for vegetation has to be installed. (EC 31 1999; VNp 861 1997)

The Landfill Directive defines explicit and unambiguous requirements for the structures that can be realized without a need for separate risk assessment. In the case of deviations, the risk assessment is obligatory. However, the directive does not define unambiguous requirements or procedures for it. Based on the literature, the risk factors affecting the landfill bottom structure can be divided into factors related to the landfill operational environment and to the waste content (Guyonnet *et al.* 2009; Cossu *et al.* 2003; Katsumi *et al.* 2001; Giroud *et al.* 2000; Korkka-Niemi & Salonen 1996; Christensen *et al.* 1994; Othman *et al.* 1994; Shackelford & Daniel 1991). Figure 2 presents the division of the most common identifiable factors related to the landfill operational environment and waste content, which affect the environmental protection crucially.

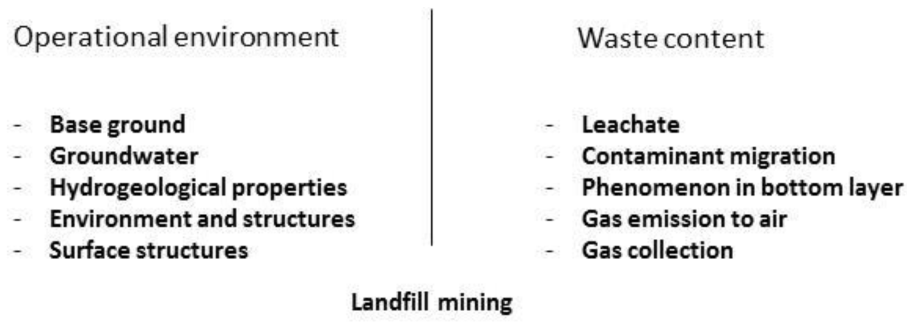

Fig. 2. Effects of the most common identifiable factors related to the landfill operational environment and waste content on the landfill life-cycle information management and environmental protection (Ortner *et al., 2014;* Laner *et al.,* 2012).

In Finland, there is not a single MSW or hazardous landfill with an environmental permit that would have had a natural geological barrier according to the requirements of the Landfill Directive or an artificial geological barrier with a thickness. All realised structures are based on the authorities' interpretation: for MSW landfills a 0.5 m and for hazardous landfills a 1.0 meter or a corresponding additional layer. Based on the interpretation by the Authorities the design of the structures has been based on equivalently calculated transport of harmful substances by clean water through a geological barrier caused by advection in relation to the structure’s layer thickness. In addition to advection, the effects of the ground water and its flowing direction, subsoil and its background concentrations and previous structures on the life-cycle information management have been typically identified in the design phase of the environmental permit process. (Environmental permit registry, 2010)

Based on the literature, it can be concluded that determining factors related to the transport of harmful substances such as the effects of the leachate quality and quantity dominate (e.g. diffusion, advection, dispersion, sorption) (Cossu *et al.,* 2003; Katsumi *et al.,* 2001; Christensen *et al.,* 1997; Shackelford & Daniel, 1991). The dominant factors have unexceptionally been excluded from the environmental permit process, According to environmental permits, only a part of the factors affecting the environmental protection of a landfill are required or identified in the environmental permit process (Environmental permit registry, 2010). However, the natural protection capacity defined in the Landfill Directive has been analysed very briefly, and its effect on the total protection capacity of the environment has not been examined widely enough in the permit process.

In this thesis, deviations in the environmental permits of the existing MSW landfills are examined in relation to the EU Landfill Directive and Finnish Government Decision on landfills, which affect the environmental protection capacity of the landfill geological barrier essentially. In addition, it will be studied whether sufficient data have been defined in the environmental permits, quality control documents and designs of the landfills in the design phase to ensure that the deviations will not influence the environmental protection capacity of the landfill. Furthermore, this thesis highlights innovative final cover structures and takes a stand on the future, e.g. the influence of landfill mining on structural demands, the structural demands regarding incinerated waste and the reuse of structures after landfill mining.

1.2 Objectives of the thesis and research questions

The objective of this thesis is to define the effects of the most essential unidentified and unconsidered factors related to the landfill management and environmental protection capability of the landfills. Typically these factors have been analysed with risk assessment tools, but in Finland, risk assessment tools have not been used during the landfill designing process.

The dissertation has two research questions that represent landfill quality requirements during the life cycle, the design parameters' role and impact on environmental protection capability and focus on requirements for developing life-cycle information management as a basis for sustainable landfilling. In Europe, the landfill structures and quality demands are based on the Landfill Directive EC 1999/31 that gives technical boundary conditions for landfill management. This study also examines and evaluates the effects of crucial factors on the landfill´s sustainability. In this work, the sustainability of landfilling is assessed from a technical perspective using risk analysis.

Research questions:

RQ1 **What are the most significant deficiencies of the present risk analysis practices in Finland?** This research question examines which factors have been identified as the most essential factors that affect the capability of the landfill structures to protect the environment during the landfill life-cycle. Also which essential factors have not been identified during the design phase as risk factors that affect the capability of the landfill environmental protection structures to protect the environment.

RQ2 **How should the risk assessment process in the landfill environmental permits and designs be developed to ensure the landfill sustainability?** This research question examines unidentified factors that affect landfill protection structures behaviour, environmental protection, environmental security and landfill management. In addition, this objective also examines how the technical requirements should be developed in order to optimize the landfill management during the landfill life-cycle.

1.3 Delimitations of the thesis

This study concentrates exclusively on Finnish landfills since the comparison of the Finnish landfills for example with the landfills of the Central Europe would not be relevant because of the climatic factors or differences in the geological conditions. The operation of the landfill surface structures in relation to the EC Landfill Directive recommendations or Finnish Government Decree requirements is also included in this thesis, even though none of the modern landfills have been sealed yet.

The old landfills have not been examined because those have been closed for such long times that enough post-closure results would not be available to describe the change between the active and passive phase. The final covers of existing landfills have no artificial bottom structures, and therefore, the mutual comparison of landfills is not relevant. The observation of the landfill surface structures presumes individualised information on the specific target because for example the surface structure thickness, local rainfall and landfill location have a substantial influence on the observation results.

The transport of the contaminants is examined based on the chemical composition of materials on hazardous waste bottom structure. In addition, the transport of the contaminants is not examined based on the chemical composition of the materials, ion replacement or the absolute composition of the leachate on the surface structure. Hydraulic conductivity tests have been carried out on the leachate of one of the MSW landfills, typifying the Finnish leachate quality.

1.4 Outline of the thesis

This dissertation also aims to develop a risk assessment method that takes local circumstances into account. At first, a literature review of the risk assessment or analysing tools used commonly in the world was carried out during the risk assessment method development process. Environmental permits, design and quality control documents of local MSW landfills have been examined to analyse and calculate the unidentified and unconsidered factors, which could have a dominant effect on environmental protection. The developed risk assessment method will be verified on one MSW landfill surface structure and one hazardous waste landfill bottom structure during the environmental permit process.

In the experimental part, calculations for harmful substance migration through the structures have been conducted for different equations to observe the impact of the calculation method on the transit time. Also the influence of harmful substances on environmental protection structures has been identified. Hydraulic conductivity laboratory tests have been performed with clean water and leachate to determine the need for environmental protection capacity. Hydraulic conductivity tests have been done according to the ASTM D 5084 method by ultra clean water and MSW leachate.

Chapter 1 presents the background and motivation for the research, a discussion of the objectives and scope of the research and a discussion of the research assumptions and process of the study.

Chapter 2 describes the theoretical background of the study. This chapter includes a collection of viewpoints from the literature to enlighten the understanding over the need and challenges of harmful substances' flow through the landfill bottom layer. The chapter presents relevant theories; that is, the essential calculations of contaminant transport that should be considered in the landfill development projects. The purpose of the theoretical part is to provide total perspectives on the landfill design processes. A theory synthesis has been included at the end to highlight the aspects essential for the purpose of this doctoral dissertation.

Chapter 3 discusses the material properties of this study. Also, the chapter presents in detail the methods used for the analysis.

In Chapter 4, the empirical data and their analysis are described. Firstly, the main results are introduced briefly. Then, the data are described and analysed. Finally, an inductive analysis is performed and main findings of the results are presented as a comparison of the obtained results to the literature.

Chapter 5 contains answers to the research questions, theoretical and practical implications, and the evaluation of the research and a summary of the main findings and defines directions for future research.

2 Theory review

Municipal solid waste (MSW) landfills represent the typical waste disposal in many parts of the world, especially in Europe. The proportionally high items of expenditure of treatment and disposal alternatives are the significant reason for the dependency on MSW landfills (Laner *et al.* 2012; Brunner & Fellner, 2007; Hall *et al.* 2007). In the future, landfills role will be changed in Europe, because recycling increases and a part of the waste will be burnt reducing the share of direct landfilling. As a result also the content of the waste fraction disposed changes (Feo & Williams, 2013; Mattiello *et al.,* 2013).

A large number of adverse impacts of landfill management may occur from landfill operations. Damage occurrence can include the infrastructure, for example damage to artificial structures, and consequence of the local environment, such as the contamination of groundwater by leakage, as well as residual soil contamination during landfill life-cycle, after landfill pre-cover or final closure. Also, landfill produces off-gassing of methane, generated by decaying organic wastes, produces methane, which is 34 times more potent than carbon dioxide and can itself be a danger to the inhabitants of an area during first decades (Suopajärvi *et al.,* 2014; IPCC 2013; Solomon *et al.,* 2007; Townsend *et al.,* 2005). Landfill gas could migrate also horizontally, and in certain circumstances gases could move along sewage pipes and cause an explosion (Xie & Chen, 2014). Some damages can occur from harboring of disease-transmitting animals such as rats and flies, particularly from improperly operated landfills (Kumar & Sharma, 2014).

In the future the quality of waste will be changed, because most of the municipal waste will be burnt in the incinerators according to EC laws. The composition of the waste from incinerators differs compared with earlier MSW landfill waste. Incinerator waste could change the composition of leachate compared with typical MSW leachate and leachate migration through the landfill bottom layer (Eichhorst *et al.,* 2013). This sets new challenges for the capability of landfill bottom structures to protect the environment from harmful substances (Feo & Williams, 2013; Mattiello *et al.,* 2013).

The waste in the old landfills could be recovered in the future for e.g. as reuse derived fuel purpose or potential mineral source (Bosmans *et al.,* 2014; Sormunen *et al.,* 2008; Hull *et al.,* 2005). Landfill mining is a term for an approach of excavating landfilled waste in order to utilise the recoverable resources (Bockreis & Knapp, 2011; Otner *et al.,* 2014). However, countries like the USA, Australia, the UK and Finland, are largely depended on landfilling (Brunner & Fellner, 2007).The landfills contain plenty of recoverable or recyclable materials to excavate which is expensive or limitedly utilised. Consequently, the excavation of the old landfills has begun and it has been possible to separate valuable materials from the waste with the help of new techniques (Frändegård *et al.,* 2013).

Typically, landfill management includes six steps before the final completion, and the time frame varies depending on for example the landfill type, disposal period, waste type, waste content, observation demands and influences on the environment (Fig. 3) (Laner *et al.,* 2012). Landfills must be managed and supervised to refrain harmful effects on human health and the environment. Based on these factors, landfill environmental protection structures are typically designed and made as constant structures because landfills can affect the environment for a very long time after disposal. Therefore, actions during the designing process have long-term impacts on the environment and landfill management.

The definition of the MSW landfill life-cycle by Laner et al. (2012) has been developed terminologically to correspond to the Landfill Directive in which the landfill life-cycle is divided into two phases: active and passive phase. In addition, Laner et al. (2012) handle the landfill construction as a unity. In this thesis, the design phase is included to a construction phase.

Landfill management can also be described based on the landfill's technical and structural effectiveness to the environment. In this dissertation, the technical management components are environmental permit, designing and the constructed structures. The structural effectiveness has been divided in the constructing structures, leachate management, waste management, gas and emission management, water management, final coverage, after care monitoring and landscaping (Laner *et al.,* 2012).

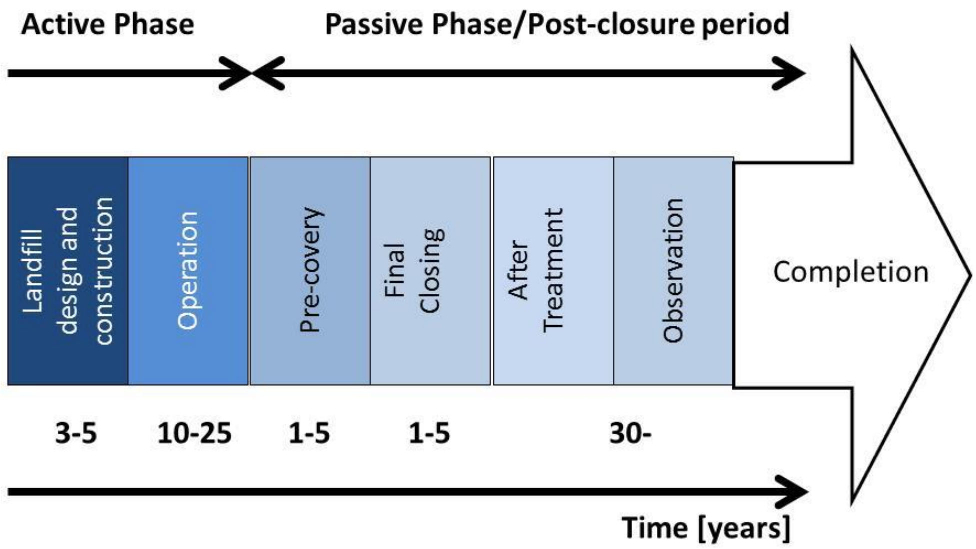

Fig. 3. Landfill management phases throughout the life-cycle; the time-frames are typical for Finland and could vary case by case (Modified after Laner *et al.,* 2012).

According to Laner et al. (2012), the time-frames consist of sectors that the landfill owners have to operate in different stages of the landfill life-cycle. In addition, these parts typically include construction work before final closing. According to literature, landfill management consist of three life cycle stages. The three stages are design and construction, operation (disposal phase) and post-closure period (pre-covery to completion), including the landfill surface, after treatment and observation (Laner *et al.,* 2012; Morris & Barlaz, 2011; Pivato, 2004; Christensen *et al.,* 2001; Christensen *et al.,* 2000; Othman *et al.,* 1994; Champerlain *et al.,* 1990). In figure 4 crucial factors from the landfill management point of view are divided to each stage. These do not include the effect of climate-related, chemical or biological factors on landfills. Also these factors are very important in the future because the MSW waste content will be changed and landfills could be used over and over again based on e.g. landfill mining (Feo & Williams, 2013; Mattiello *et al.,* 2013).

Design and construction has a central role in the landfill life-cycle information management process. In the designing process, the important factors are decided along the landfill life-cycle. The environmental protection requirements are realized according to the Landfill Directive, and the landfill life-cycle is defined based on structural dimensioning.

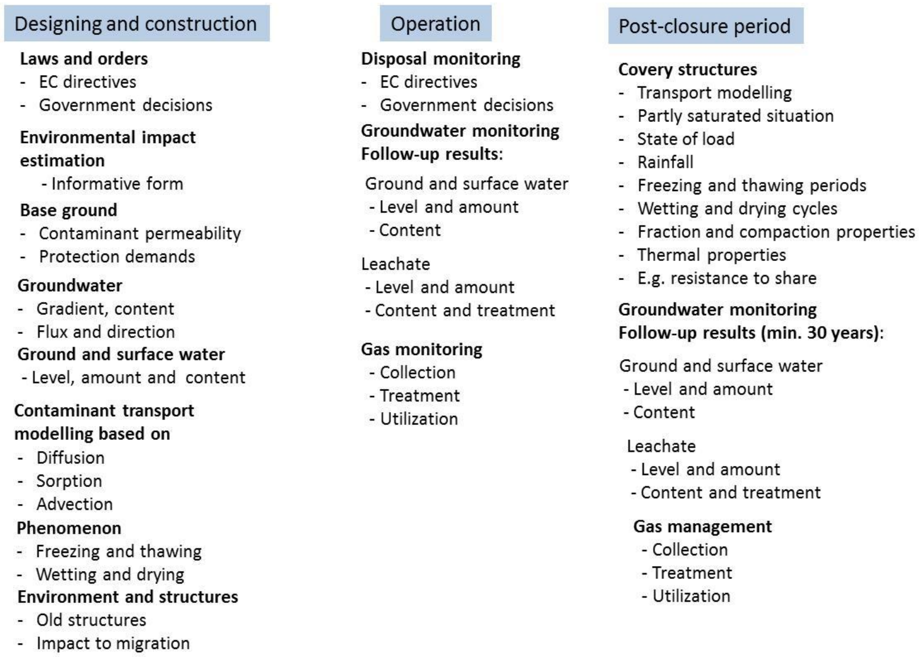

Fig. 4. MSW landfill environmental management divided into three management periods during the life-cycle (Modified after Laner *et al.,* 2012; Morris & Barlaz, 2011; Pivato, 2004; Christensen *et al.,* 2000).

After disposal, the next phase in the life-cycle of landfills is the post-closure period that is much longer compared to the active phase. The aftercare management of landfills in the passive phase includes typically the monitoring of emissions (e.g. leachate and gas) and following-up (e.g. groundwater, surface water, and soil) and maintenance and supervision of the final cover, leachate and gas collection systems. Aftercare process causes costs to landfill owners, and in advance, the post-closure frame is a question mark. This leads to a situation in which cost-effective strategies for the management of landfills are in the interest of both authorities and landfill owners. (Laner *et al.*, 2012; Pivato, 2004; Christensen *et al.,* 2000)

2.1 Review of risk assessment in landfills

Risk assessment is comprehensive process where the landfill operator is typically deciding risk extent and acceptability. The target of the process is to evaluate element of danger and risk factors. Risk analysis is typically included in risk assessment process. Risk analysis identifies enabling factors that could cause danger e.g. technical factors, human actions or environmental circumstances (Zhou *et al.,* 2014; Butt *et al.,* 2008).

A risk could be quantifying as the probability of define dangerous occasion. Environmental risk is common noun for risk, and in case it is realized it could cause environmental damages. Danger is a situation where is a possible to cause e.g. personal injury, property damage, environmental damage or combination of these. (Butt *et al.,* 2008)

According to Neshat et al. (2015) and Zaporozec (2004), the risk can be determined by using the equation (1):

$$Risk = \sum_{i=1}^{R} \text{Probability of event} * \text{Consequence of event} \qquad (1)$$

This event risk happens R times during its life cycle.

Risk assessment and risk analysing is a continually developing branch of science that develops evaluation tools for environmental protection. There are numerous different risk assessment tools for business fields like the construction management or building contract selection. Risk assessment tools and also several computer aided tools for the protection of groundwater from landfill leachate, of landfill leachate liners and drainage systems, of natural hazards like flooding, landslides and gas accumulation has been made for landfills (Butt *et al.,* 2014; Butt *et al.,* 2011; Chowdhury, 2009; Giusti, 2009; Pollard *et al.,* 2006; Aven & Kristensen, 2005).

Monte Carlo Simulation (MCS) approach is one of the most typical applied methods for modelling landfills´ risk level. In the MCS method to reach the distribution of an unfamiliar problematic the tests have been done many times

(Baeurle, 2009). Monte Carlo Simulation has been applied in many computer-aided risk analysing programs (Butt *et al.,* 2011; Aven & Kristensen, 2005). Monte Carlo Simulation could be approach mathematically or as a novel method for landfills risk assessment (Neshat *et al.,* 2015).

Risk assessment processes are typically focused on landfills´ waste products in three phases: solid waste (disposed waste), liquid waste (e.g. leachate) and gas (landfill gas). Landfill may pollute the environment in three ways – atmosphere (air), lithosphere (soil or base ground) and hydrosphere (water or groundwater) (Butt *et al*., 2011). There is also risk analysing processes that is dependent on risk reduction during waste treatment process (Butt *et al.,* 2008).

Risk assessment is important issue for landfills during the design phase. Characteristics may vary widely between case to case, not only in terms of landfill but also management practices and regulations. In many countries, like the USA and the Great Britain, risk assessment is included in environmental regulation even if the EC landfill directive does not call risk assessment into play in all cases (Butt *et al.,* 2014; Coventry *et al.,* 2012; Bonaparte *et al.,* 2002). The identification of risks can help to compare risks in the environmental protection of landfills, and as a result, new landfills are safer than they have been so far.

The landfill structures are affected by a considerable amount of phenomena and background factors that can change over time depending for example on external factors or human delineation. However, the theory does not hold solutions for taking all issues and their mutual complex effects on contaminant migration into consideration. In addition, all factors like the freezing–thawing phenomenon do not affect the landfill structures in all countries, and they have typically been excluded from theoretical studies. The impacts of the artificial environment are significant for example on groundwater levels and flowing, the prognosis of which is reasonably impossible (Heikkinen *et al.* 2002; Mälkki 1999; Korkka-Niemi & Salonen 1996; Lahermo *et al.* 1996). Figure 5 represents, based on the theory, the most essential identifiable factors related to the landfill operational environment and the MSW waste content and their effects on landfill life-cycle information management and environmental protection (Laner *et al*., 2012; Morris & Barlaz, 2011; Pivato, 2004; Christensen *et al*., 2000).

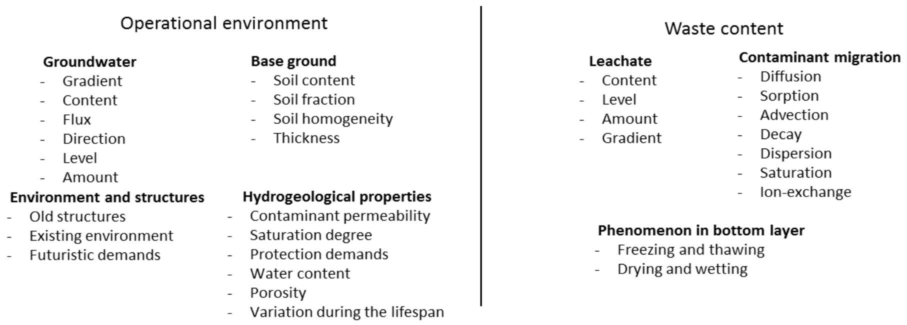

Fig. 5. Identifiable effects related to the landfill operational environment and waste content based on theory (Modified after Laner *et al.,* 2012; Morris & Barlaz, 2011; Pivato, 2004; Christensen *et al.,* 2000).

2.1.1 Key components in landfills' environmental risk assessment

In the landfill life-cycle, the most essential process is the design phase, in which decisions are made about the landfill environmental protection level, landfill related risks are identified, the length of the landfill life-cycle is defined or evaluated and an assessment of the landfill impact on the environment is compiled. In the design phase, the measures that manage risks identified have to be decided and taken into account as a part of the landfill management, and these identified factors have a direct impact on the protective structure requirements and technical realization of the landfill environment. In the design phase, information is produced for the environmental permit, and therefore, the effects of the factors that have not been recognized or identified during it, will be included in the environmental permit.

From the literature, the most important factors have been collected that have an impact on the life-cycle operation of the landfill bottom layer structure. According to several studies, including the studies of Butt et al. (2014 and 2011), Cossu (2007), Mitchell & Soga (2005), Rowe et al. (2004), Katsumi et al. (2001) and Freeze & Cherry (1979), and, the following issues are related to the landfill life-cycle examination from the environmental protection perspective:

- Hydrogeological properties of soil
- Migration mechanisms of contaminants
- Contaminant properties and the examination of effects related to contaminant retention and migration
- Topography
- Meteorology
- Exposure, significance and uncertainty assessment
- Life-cycle of structures
- Risk quantification
- The effects of the environment on protective structure operation, and the effects of the soil on contaminant migration
- Retention as well as the groundwater effects on contaminant migration
- Excavations
- Storage and recovery.

From the landfill life-cycle information management perspective, the leachate releasing or leaching from the waste is in the focal point of the landfill's environmental protection. The leachate content and quantity includes storm water and generation during different activities such as temperature changes inside the landfill or landfill mining. The quantity and quality of leachate affect the structural dimensioning, the landfill internal and external water management and the length of the landfill life-cycle. Leachate management is a crucial factor in landfill management in all the landfill environmental protection phases. The aftercare process includes many different features that have to be focused on separately. (Laner *et al.,* 2012; Barlaz *et al.,* 2002)

Hydrogeological environment has an influence on contaminant transport in soil layers and groundwater due to groundwater movements and flow gradient (Rowe *et al.,* 1995). In Finland, hydrogeological conditions can be divided into a few types based on the groundwater level and its annual variation, soil properties and groundwater flow gradients (Heikkinen *et al.,* 2002; Jokela, 2002; Mälkki, 1999; Korkka-Niemi & Salonen, 1996; Lahermo *et al.,* 1996). In some landfill cases, groundwater is close to the ground surface and simultaneously close to the landfill bottom layer. Alternatively, groundwater is a few meters below the ground surface and varies significantly during the year.

The seasonal variation of groundwater level and short distances between the groundwater and ground surface may in leakage situations result in the migration of leachate from the landfill into groundwater causing soil and groundwater pollution. In Finland, the groundwater surface level may vary, depending on the location, with the range of variation of over 1 m during different seasons (Korkka-Niemi & Salonen, 1996). In the Northern Europe, the special features include, in addition to soil and groundwater features, the annual soil freezing in autumn and thawing in spring, which can affect the long-term durability of the protective structures.

The freezing–thawing cycles are related to the wetting–drying phenomenon since when the soil freezes, it absorbs moisture and expands (Guyonnet *et al.,* 2009; Othman *et al,.* 1994). Based on the literature, the freezing–thawing phenomenon does not affect the well compressed soil layers, but the influence of the phenomenon on Finnish landfills has not been examined thoroughly (Othman *et al.,* 1994; Zimmie, 1992; Champerlain *et al.,* 1990). According to the previous researches, the wetting–drying phenomenon has a remarkable impact on the long-term protection features of expanding mineral structures (Guyonnet *et al.,* 2009; Othman *et al.,* 1994).

However, wetting-drying and freezing-thawing phenomena are typically excluded in substance migration models such as HELP or LandSim (widely used in landfill design) (Wang, 2011; Giroud *et al.,* 2000).

Giroud's model Landfill liner system checklist (2000) and Landfill design.com (2000) checklists is a checklist containing a collection of properties, excluding the effects of soil- and groundwater related and cyclic phenomena. Corresponding models are geotechnical calculation models, material-related dimensioning applications and migration modelling applications.

Based on the literature review, not a single theoretical model exists that would cover the above described factors comprehensively, taking all the cyclic phenomena into account (Leeson *et al.,* 2003; Bonaparte *et al.,* 2002). Landfill management can focus on various perspectives. Models and reviews with larger perspectives have been prepared of the landfill environmental impacts and mainly the realization of surface protective structures. Typical perspectives include the human health and ecological risk and the life-cycle based assessing (Laner *et al.,* 2012; Morris & Barlaz, 2011). These perspectives are observed in more detail in the following sections.

2.1.2 Methods and tools used in landfills risk assessment

In this part of thesis typical methods and tools used in landfills risk assessing processes have been analysed. Methods and tools have been developed or modified in different cases or purposes.

Performance-based managing is focusing on landfill monitoring and performance data. The performance data is in important role when evaluating the landfill conditions, effects on the surrounding environment with guide to appropriate active and passive phase activities (Laner *et al,.* 2012; Barlaz *et al.,* 2002). The evaluation procedures are landfill-specific and provide guidance on landfill managing, protection processes and in long term the reduction of aftercare intensity.

Morris and Barlaz (2011) used a modular approach as a methodology for the evaluation of environmental and human health risk. Modules included data collection in leachate, gas and groundwater contents and content changing after final closing. Evaluation is focused on to reduce aftercare activities and also to protect the environment before completion. Morris and Barlaz (2011) used the performance-based method for defining aftercare requirements at MSW landfills and verified requirements to the evaluation of post-closure care (EPCC) methodology. This methodology provides specific on-time protocols for long-term landfill management. The EPCC method focuses aspects like landfill aftercare monitoring and maintenance. Maintenance includes e.g. leachate and gas management, groundwater protection and final closing.

The methodology establishes landfill site information for further decisions on maintaining, extending, reducing or modifying aftercare activities while sustaining the environmental protection follow-up regulations. Diagrams, which are based on on-time measurements, are produced for each part of aftercare (e.g. leachate, gas, groundwater and structural sustainability). The methodology application requires the waste prospects and demands of the landfill to be considered as advance. The application of the methodology is analysing measurements results and trends like leachate, landfill gas generation and groundwater quality. When the aftercare is realized, the landfill owner should verify the effect of monitoring results. (Laner *et al.,* 2012; Morris & Barlaz 2011)

The EPCC methodology consist three levels of analysis. These all produces different outcome of landfills (Morris & Barlaz, 2011):

(a) Source evaluation; the compliance with target values may be demonstrated at the source (e.g. leachate quality < drinking water standards),
(b) Point of compliance (POC) evaluation; it is demonstrated that the landfill does not pose an unacceptable impact on the POC and
(c) Point of exposure (POE) evaluation; it is demonstrated that the landfill does not pose an unacceptable risk at the POE.

The EPCC method determines a level of care program that is equivalent with requirements of the landfill. The surrounding environment requirements are based on the combination of target values and risk assessment results. The evaluation leads to custodial care program and activities as basis for reducing the aftercare time period (Laner *et al.,* 2012; Morris & Barlaz, 2011; Pivato & Morris, 2005).

Sizirici (2009) developed a set of relevant parameters of leachate criteria: ammonia–nitrogen, chloride, iron, VOCs or landfill gas. Sizirici et al. (2011) and Sizirici & Tansel (2010) presented a procedure of closed landfills. The procedure is based on expert evaluation scale from 1 to 10 of site-specific parameters. The evaluation is included for example climate, operational factors, leachate management and gas management. The parameters is based on a ranking algorithm and assigning weights to different factors. The ranking algorithm includes 11 categories of parameters identifying critical areas which could affect post-closure care (PPC) needs in the future. Each category was further analysed by detailed questions on the site history, location, and specific characteristics. Each question was scored (on a scale of 1-10, 1 being the best and 10 being the worst). The result from the algorithm is used to classify the landfill circumstances as critical, acceptable or good level (Sizirici *et al.,* 2011).

The EPCC method presents the most real time and present situation based on the evaluation procedure, providing operative assessment information to decide on an appropriate level of aftercare, to reduce landfill impact on environment and to get information for risk assessments (Laner *et al.,* 2012; Morris & Barlaz, 2011; Pivato & Morris, 2005). When using the EPCC method, the evaluation requires a high level of expertise because performance-based approaches focus on the landfill life-cycle as a management process instead of authorities requiring financial provisions for a minimum aftercare period.

The LandSim software model has been developed to provide quantitative risk assessments (Environment Agency, 2003). Therefore, the LandSim software could be only a part of total landfill risk assessment. This tool estimates concentrations of leachate and time frame when leachate pollutants reach groundwater or a given point in the base ground. Calculation estimation includes lots of features that could influence to pollutant migration, but model does not observe quantification aspects like groundwater and exposure for people. (Butt *et al.,* 2008)

The model includes two stages; landfill hydraulics and predict the impact on groundwater quality. Hydraulics evaluates whether the drainage system could retain the leachate below the determined maximum level, and anticipate the impact on groundwater quality from the landfill calculated contaminant concentration at the specified receptor over time. Model contaminant transport and transit time calculation is based on LaPlace transform technique to work out the advection-diffusion transport equation that is based on Freeze & Cherry (1979) equation for saturated flow and Van Genuchten (1980) equation for unsaturated flow thru the porous fraction (Butt *et al.,* 2011; Aven & Kristensen, 2005; Environment Agency, 2003).

The LandSim model calculation is based on ready design structures and base ground types that does not exists in Nordic countries. Ground water models could keep as an example, the model assumes base ground typically contain aquifer. In Nordic countries groundwater is typically very close under the ground surface and therefore there exists only very few places where aquifers could have been formulated (Mälkki, 1999; Korkka-Niemi & Salonen, 1996). Hall (2007) has used the LandSim model for the hydraulic modelling calculations. According to this example, the LandSim model results have to be critically evaluated and there could be cases in which this model cannot be applied because all factors are not included to calculate models for example the total amount of percolation for snow. This method is focusing only on the probability of risk and identifies the possible of risks.

The purpose of the landfill liner system checklist is to lead the designer or reviewer to consider the aspects of design for the different components of landfill liner systems including leachate collection, leachate removal and leak detection (Giroud *et al.,* 2000). The checklist is valid for landfill bottom liners all over the world. Industrial Parks have been developed and different models, but the base ground protection demands have been the same. This check list have been modified and updated during this thesis.

The checklist contains ten main points with a significant weight value, according to which it will be defined whether the property has been identified and is a part of the landfill and whether the property is relevant to the specific landfill. Table 2 describes the main properties and their descriptions in the Landfill liner system checklist.

In addition to the main points described in Table 2, the Landfill liner system checklist defines at many points the contract document requirements and installation requirements. Although the method requirements have been compiled for each material, each material is not used in every landfill; for example, the drainage structure is realized either with granular material or geocomposite material. Correspondingly, in the compacted layer, typically neither an artificial compacted structure nor geosynthetic clay liner is used (Giroud *et al.,* 2000).

The checklist made by Giroud et al. (2000) is one example of these types of methods. Landfill design.com webpage is includes similar kind of lists for landfill design. Purpose of these lists is to observe the critical factors of the landfill designing process to identify the consequences.

Table 2. Landfill liner system checklist (Giroud *et al.,* 2000).

Property	Property main description
Protective soil cover/ select waste layer	Will a protective soil cover or select waste layer be used at this site? Does this layer meet the minimum thickness requirement if any? Is the material selected available in the vicinity? Is compaction specified using low ground pressure equipment?
Granular drainage layer, leachate collection and removal system	Has the granular drainage layer been designed to limit the head build-up to less than 300 mm (12 in.) on top of the liner? Is the hydraulic conductivity of the drainage material greater than 1×10^{-4} m/s?
Geocomposite drainage layer, leachate collection and removal system	Has the transmissivity of the geocomposite been evaluated to limit the head within its thickness thus to ensure an unconfined flow)? Have the reduction factors for creep, intrusion, particulate clogging, biological and chemical clogging been considered in the hydraulic assessment of the geocomposite? Have load, gradient, seating period and boundary conditions been specified in the transmissivity requirements of the geocomposite?
Geomembrane	Does the membrane need to be textured? Is the minimum geomembrane thickness met if any
Compacted clay liner (CCL)	Does the clay layer have a saturated hydraulic conductivity of 1×10^{-9} m/s or less? Is the clay layer a minimum of 600 mm (2 ft) in thickness? Has a clay borrow source been identified and tested?
Geosynthetic clay liner (GCL)	Will regulators allow the use of a GCL at this site?
Geonet/ geocomposite drainage layer, Leak Detection Layer	Have the soil retention, filtration, survivability, transmissivity properties of the geotextile been evaluated?
Granular drainage layer	Has the granular drainage layer been designed to limit the head build-up to less than 300 mm (12 in.) on top of the liner? Is the hydraulic conductivity of the drainage material greater than 1×10^{-4} m/s?
Subgrade	Is stabilizing the subgrade using geogrids need to be considered?

According to Cormier et al. (2008), the assessments of human health and ecological risk method evaluate multiplex types and sources of information, analysing wide range of evidence before conclusions. Risk assessors of the US Environmental Protection Agency (USEPA) make use of weight-of-evidence: *"(WOE) approaches to carry out the integration, whether integrating evidence concerning potential carcinogenicity, toxicity and exposure from chemicals at a contaminated site or evaluating processes concerned with habitat loss or modification when managing a natural resource*" (USEPA, 2008). WOE is one of the most commonly used and applied methods for risk assessing. (USEPA, 2008; Cormier *et al.,* 2008)

The WOE "*approach can be defined as a framework for synthesising individual lines of evidence, using methods that are either qualitative (examining distinguishing attributes) or quantitative (measuring aspects in terms of magnitude) to develop conclusions regarding questions concerned with the degree of impairment or risk. In general, qualitative methods include the presentation of individual lines of evidence without an attempt at integration or integration through a standardised evaluation of individual lines of evidence based on qualitative considerations. Quantitative methods include integration of multiple lines of evidence using weighting, ranking or indexing as well as structured decision or statistical models*" (Table 3). (Chapman *et al.,* 2002)

Table 3. Weight of evidence method (Chapman *et al.,* 2002).

Method	Method description
Listing Evidence	Presentation of individual line of evidence without attempt at integration
Best Professional Judgement	Qualitative integration of multiple lines of evidence
Causal Criteria	A criteria-based methodology for determining cause and effect relationships
Logic	Standardised evaluation of individual line of evidence based on qualitative logic models
Scoring	Quantitative integration of multiple lines of evidence using simple weighting or ranking
Indexing	Integration of lines of evidence into a single measure based on empirical models
Quantification	Integrated assessment using formal decision analysis and statistical methods

2.2 Human delineated factors related to landfills

The EC Landfill Directive sets the general requirements for all classes of landfills in Europe. The essential requirements for landfills are location, water control and leachate management, protection of soil and water, gas control, nuisances and hazards, stability and barriers. This dissertation focuses on the location and protection requirements and also partly on water control and leachate management requirements. (EC 31, 1999)

One of the general principles of the Landfill Directive is that:

"*The composition, leachability, long-term behaviour and general properties of a waste to be landfilled must be known as precisely as possible. Waste acceptance at a landfill can be based either on lists of accepted or refused waste, defined by nature and origin, and on waste analysis methods and limit values for the properties of the waste to be accepted. The future waste acceptance procedures described in this Directive shall as far as possible be based on standardised waste analysis methods and limit values for the properties of waste to be accepted*." In addition, during the designing process, criteria for landfill structural acceptance must be based on considerations for the concern to the protection of the surrounding environment (e.g. groundwater and surface water, geological barriers and leachate management), the desired waste stabilisation processes and protection against human health hazards (EC 31, 1999).

The EC Landfill Directive determines requirements for the bottom structure. The directive 1999/31/EC states the following, *"the landfill base and sides shall consist of a mineral layer which satisfies permeability and thickness requirements with a combined effect in terms of protection of soil, groundwater and surface water at least equivalent to the one resulting from the following requirements:*

- Landfill for hazardous waste: $K \leq 1.0 \times 10^{-9}$ m/s; thickness ≥ 5 m,

- Landfill for non-hazardous waste: $K \leq 1.0 \times 10^{-9}$ m/s; thickness ≥1 m,

- Landfill for inert waste: $K \leq 1.0 \times 10^{-7}$ m/s; thickness ≥ 1 m.

- Artificial liner is required hazardous and non-hazardous waste landfills

Where the geological barrier does not naturally meet the above conditions it can be completed artificially and reinforced by other means giving equivalent protection. An artificially established geological barrier should be no less than 0.5 metres thick."

The EC Landfill Directive gives only recommendations for surface sealing. The local authorities decide after consideration the demands of the surface structures. The recommendations are presented in Table 4.

Table 4. MSW EC Landfill Directive recommendations for surface structures (VNp 861, 1997; EC 1999/31).

Landfill category	non-hazardous	Hazardous
Gas drainage layer	Required	not required
Artificial sealing liner	not required	Required
Impermeable mineral layer	Required	Required
Drainage layer > 0.5 m	Required	Required
Top soil cover > 1.0 m	Required	required

The landfill active phase means the time when the waste will be laid into the landfill. The bottom structure consists of an artificial layer, for example geomembrane and the geological barrier. The artificial layer is designed to protect primary the soil and groundwater below the filling of the waste (Cossu *et al.*, 2003; Rowe *et al., 1*995). Drainage structure is constructed over the artificial structure, which leads the leachate into the collection and treatment system. It is generally known that an artificial layer is not fully consistent with the structure of the material properties and the structure implemented from causing damage to individual leaking points in the artificial layer. The geological barrier below the geomembrane structure is ensuring that no leachate could flow into the soil or groundwater under the landfill (Forget *et al.,* 2005; Rowe & Orsini, 2003; EC 31, 1999; VNp 861, 1997; Rowe *et al.,* 1995).

In the landfill post-closure period, surface layers have been installed over the landfill. The surface structure's role as a cap over the waste is to protect the soil and groundwater by preventing the infiltration of surface waters into the waste at unacceptable rates. During the post-closure period in Finland, it is supposed that an artificial layer cannot totally protect the geological barrier on the long term, and the environmental and groundwater protection is based on the geological barrier (EC 31, 1999; VNp 861, 1997).

The Landfill Directive defines the requirements for surface and bottom structures according to environmental protection. The permeability and thickness of the geological barrier are mentioned as the major requirements for protecting landfill environments. The term geological barrier is applied in the directive when assessing landfill bottom layer conditions. A geological barrier has been defined based on landfill geological and hydrogeological conditions (VNp 1049, 1999; EC 31, 1999; VNp 861, 1997).

In addition, the directive requires an adequate risk analysis on soil and groundwater pollution and the EC directive includes the following requirements:

"The location of a landfill must take into consideration requirements relating to:

(a) The distances from the boundary of the site to residential and recreation areas, waterways, water bodies and other agricultural or urban sites;

(b) The existence of groundwater, coastal water or nature protection zones in the area;

(c) The geological and hydrogeological conditions in the area;

(d) The risk of flooding, subsidence, landslides or avalanches on the site;

(e) The protection of the nature or cultural patrimony in the area"

Government Decision 861/1997 mentions that authorities alleviate the requirements if the landfill owner shows that there are no harmful effects to health and the environment in the long term (VNp 861, 1997).

2.2.1 Legislative guidance's impact on environmental protection

The EC Landfill Directive state that the protection capacity of the bottom structure shall satisfies permeability and thickness requirements. The main requirements for the permeability and layer thickness of the soil layer are based on the classification of the waste in the landfill. Neither the EC directive nor the Government Decision VNp (861/1999) provides further specifications as to which factors, such as diffusion, dispersion, or hydraulic gradient over the waste, should be taken into account when defining the protection capacity of a structure or how such factors should be assessed. Existing research, for example by Forget et al. (2005), Rowe & Orsini (2003) and Katsumi et al. (2001), on the topic does not cover, as a whole, those critical factors that most affect the protection capacity of a soil layer in the Nordic countries.

The wording of the Landfill Directive is not explicit and straightforward in aspects relevant to the design of artificial geological barriers. At MSW landfills, this word that refers to construction has been interpreted so that building a mere 0.5 meter thick additional layer would be enough to satisfy the protection requirements set in the directive for layer thickness. The permeability with a combined effect of the interpretation means only the advection, instead of focusing the permeability of the contaminant by all transit mechanisms (Hansen, 2009; Christensen *et al.,* 1994).

According to literature, structures should be designed acknowledging all transport mechanisms (the total amount of mass transport in the cubic element, e.g. diffusion, dispersion, advection, sorption, decay) to protect soil, groundwater and surface water (Rowe *et al.,* 2004). Transport times through landfill layers vary depending on artificial layer transport mechanisms that are related to material hydrogeological properties determining the amount of harmful substances flowing through landfill bottom layers (Varank *et al.,* 2011; Ramke, 2009; Cossu *et al.,* 2003; Shackelford & Daniel, 1991).

The leachate migration transit time through bottom layers is considerably shorter if all the contaminant transport mechanisms have been taken into account instead of the advection only (Sangam, 2005; Katsumi *et al.,* 2001). The contaminant migration to the environment in different time frames depends on the structures, quality and quantity of waste, structural functionality, leachate treating, collection or circulating system operations (Varank *et al.,* 2011; Morris *et al.,* 2009; Olivier *et al.,* 2009; Cossu *et al.,* 2003).

This type of landfill management has resulted in an elemental analysis, which assumes to fulfil the municipal solid waste landfill (MSW) EC directive requirements. The interpretation identifies only the hydraulic conductivity and thickness of the structure and the structural dimensioning are designed according to equivalent calculation. Consequently, based on this type of calculation acknowledging only advection, the bottom layer structure is expected to meet the Landfill Directive requirements. This is a typical national interpretation of the Landfill Directive (e.g. Environmental decisions Dnro PSA-2005-Y-243-121, 2005; Dnro 0800Y0307-111, 2001; Dnro 0295Y0226-124, 1996).

2.2.2 Interpretations of the Landfill Directive

The principle of the EC Landfill Directive is that the landfill has to be located in a place where the soil below the waste meets the directive requirements (EC 1999/31). The soil prevents leachate, which is derived from the waste, from reaching adverse effects on the environment. Leachate will not accumulate over the bottom structure, but it will flow via drainage structure to the cleaning process or water treatment. (EC 1999/31; Christensen *et al.,* 1994)

Figure 5 presents a conceptual structural requirement of MSW landfill´s bottom structures based the on EC 1999/31 Landfill Directive and the interpretations of Germany, Sweden, the Netherlands and Finland. In Germany, German Technical Instructions on Municipal Waste, TASi (TA Siedlungsabfall, 1993) determines that the mineral layer has to be at least a three layered structure, the strength of which should be 0.75 m. German Geotechnical Society, DepV (Deutsche Gesellschaft für Geotechnik e.V.) (2002) reduced the requirement to a double layer with a thickness of 0.5 m. Mineral layer hydraulic conductivity should be designed in a laboratory based on DIN 18130 standard. Drainage layer permeability and thickness can be with a minimum height of 0.5 m (Fig. 6). The Swedish interpretation of the Landfill Directive is almost equal to the German. Departing from the German instruction, in the Swedish interpretation the geological barrier is determined based on the operating time and amount of flux to the environment during operating time.

The landfill bottom structure in the Netherlands was governed by the Environmental Management Act (1993) before the Landfill Directive. In 2001, the Landfill Directive determined a change, which was included in the national law. In the Netherlands, landfills have to build a pipeline system below the mineral layer structure because the ground water (or sea water) can be on a higher level than the bottom structure. Departing from German and Swedish interpretations, the mineral layer permeability has to be less than 20 mm per year ($k < 1 \cdot 10^{-9}$ m/s) when the water pressure is equal to the height of a 0.8 m water column (50 mbar on the surface and 50 mbar below the surface) and the water has an opportunity to flow out below the layer (Fig. 6) (the Landfill Directive 1999/3i, in Germany (*DepV,* 2002), in Sweden (SFS nr 2001: 512), in the Netherlands (*Richtlijnen onderafdichtingsconstructies voor stort- en opslag-plaatsen)* and in Finland (VNp 861/97)).

According to the Netherlands' interpretation of the EC Landfill Directive, high density polyethylene (HDPE) geomembrane has to be at least 2 mm thick. Mineral layer and geosynthetic barrier contaminant permeability can be 5 mm per year at the maximum (Boels *et al.,* 1993). The drainage layer has to consist of sand (and gravel), and a drainpipe system has to be installed in it. Waste filling must be at least 0.7 m on the average above ground water (den Ouden & Backhuijs, 1999; Boels *et al.,* 1993). If the waste filling and the highest ground water height difference is less than 0.5 m, an extra mineral layer with a thickness of at least 0.5 m has to be built. The above described mineral layer structure can be replaced by an alternative structure, which has an *equivalent protection* to the groundwater. Layer thickness can have different values based on the hydraulic conductivity, provided that the equivalent protection capability is achieved. (den Ouden & Backhuijs, 1999; Boels *et al.,* 1993)

The interpretations of the Landfill Directive vary greatly between different countries. This has led to variation in the dimensions and bottom layer proportions of built structures.

However, there is no derogation from the EC landfill directive Article 3.1, so a geological barrier must always be present. In Sweden, there is an example of hazardous waste landfill in Avesta. Hazardous waste landfill has been established directly on the soil based on the directive without external artificial layer (Avesta, 2004). The soil has been determined to be a 3 –5 m layer thick low permeability clay layer, the hydraulic conductivity of which is ranging from 10^{-8} m/s to 10^{-10} m/s. In this case, the authorities have followed the principle of the Landfill Directive that artificial structures do not have to be built if the soil meets the requirements of the geological barrier. Also the authorities are obliged to carry out a risk assessment. (Avesta, 2004)

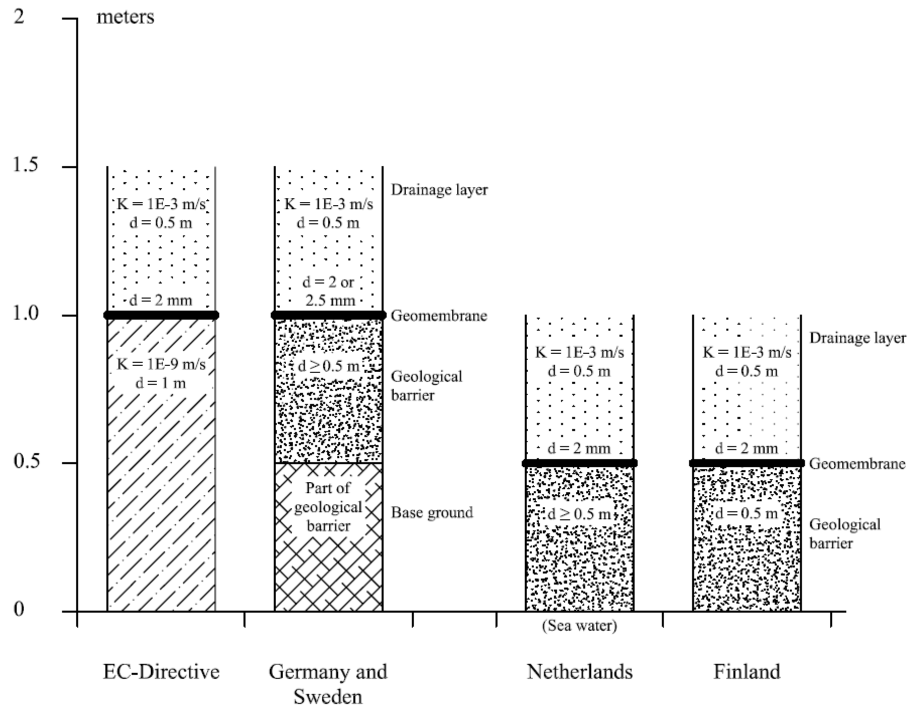

Fig. 6. MSW landfill bottom layer conceptual structural demands based on the Landfill Directive 1999/3i, in Germany (*DepV,* 2002), in Sweden (SFS nr 2001: 512), in the Netherlands (*Richtlijnen onderafdichtingsconstructies voor stort- en opslagplaatsen)* and in Finland (VNp 861/97) (K is the hydraulic conductivity, d the thickness).

The interpretations of the directive differ from each other in Finland and Denmark. An artificially geological barrier consists of a 0.01 meter bentonite liner and membrane where the hydraulic gradient is 6, and there is a 15 cm leachate on the liner. During the passive phase (when leachate pumping has been stopped), based on the Danish interpretation, several meters of leachate is assumed to be on top of the barrier. Also, assuming that a 5 meter leachate layer is over a 1 meter geological barrier, the hydraulic gradient is 6, and for a 5 meter barrier, the hydraulic gradient is 2. For a 0.01 meter bentonite membrane structure, the hydraulic gradient is 500. This means that a 1 cm bentonite liner must have a hydraulic conductivity that is hundreds of times smaller than the clay barrier to have the same flow through in the passive phase.

According to Hansen (2009), the Danish model of calculation indicates that the diffusion of pollutants through a 0.01 m bentonite liner is the dominant flux-component. Assuming the hydraulic conductivity is 5×10^{-11} m/s for a 0.01 m bentonite liner and 0.6 m of leachate on the liner, the flux based on advection was calculated to be 96 mm/year while the diffusive flux (for chloride) was equivalent to a water flux of 770 mm/year (Hansen, 2009).

Examples of the Swedish and Danish interpretations of the EC Landfill Directive show the extent to which the interpretations of these two Nordic countries differ from those in Finland and the bottom structures typically constructed in Finland. In Sweden, according to this example, landfills are purposefully located on soil where the existing base ground can be employed as a part of the geological barrier. In the Danish example, the migration of contaminants due to diffusion, which significantly increases contaminant migration through bottom structures, has also been taken into account in the implementation of the structure.

2.3 Environmental factors´ influence on landfills

In the Nordic countries, environmental circumstances like groundwater balance, wetting-drying and freezing-thawing have an effect on landfills. The influence of these critical circumstances on landfills has not been typically dealt with in risk assessments or environmental permits. Risk assessment tools, e.g. HELP and LandSim, do not take the circumstances of Nordic countries into account (Butt *et al.,* 2014; Wang, 2011; Giroud *et al.,* 2000). In this part of the thesis environmental circumstances influence on the risk assessment is opened.

2.3.1 Groundwater balance and base ground

In Europe, the geological environment varies in different areas and this has to be noticed as an impact on the landfill bottom layer's design. Groundwater levels, seepage gradients and types depend on the hydrogeological condition (Heikkinen *et al.,* 2002; Smith, 1984). This leads to a situation in which a general all-inclusive instruction cannot be determined to cover all the European regions. Hydrogeological environment has to be defined nationally according to the national specific circumstances following the Landfill Directive demands (EC 31, 1999).

There are several factors that affect the landfill hydrogeological environment, which could be categorised as local, regional and temporal factors (Heikkinen *et al.,* 2002; Korkka-Niemi & Salonen, 1996; Mälkki, 1999, Lahermo *et al.,* 1996). These factors and their effectiveness are listed in Table 5.

In Finland, the artificially geological barriers are constructed usually on natural soils or a 0.3–0.5 m layer gravel is used over the natural soil (Environmental decision registry, 2010). The geological barrier may be required retarding the movement of the contaminant along the fractures or through pore holes if the natural soil is fractured or the porosity is high, like with clay or silt based materials (Varank *et al.,* 2011; Cossu *et al.,* 2003). According to this assumption, two of conceptual outward flowing situations are presented in Figure 7a and Figure 7b.

The hydrogeological conditions in Finland comprise typically a natural soil that is till-based material and can be in fully saturated unit weight condition because of till-based material's low permeability, and the groundwater is very close to the ground surface [H_T] and there are changes during the season depending for example on weather conditions (Mälkki, 1999; Korkka-Niemi & Salonen, 1996; Lahermo *et al.,* 1996). In a structure presented in Figure 7a, H_T is over 4.0 m, the hydraulic conductivity k_t is 1×10^{-6} m/s and the seepage velocity $V_s > 0$ m/s. Harmful substances flow through the landfill bottom structure during a long time by the dominant effect of hydrodynamic dispersion. Groundwater becomes contaminated through the region when harmful substance flow occurs in groundwater (Katsumi *et al.,* 2001). This kind of structure has a very long impact on the environment, and the impacts persevere decades after the landfill has been closed (Cossu *et al.,* 2003).

In a structure presented in Figure 7b, $H_T < 1.0$ m (H_T is groundwater distance from the bottom structure base), k_t is 1×10^{-6} m/s (k_t is hydraulic conductivity) and $V_s > 0$ m/s (V_s is seepage velocity). Harmful substances flow through the landfill bottom structure during a short time, and the dominant transport mechanism is the hydrodynamic dispersion (Varank *et al.,* 2011; Rowe *et al.,* 1995). Groundwater becomes contaminated throughout the region when the flow occurs in groundwater (Varank *et al.,* 2011; Cossu *et al.,* 2003; Heikkinen *et al.,* 2002; Katsumi *et al.,* 2001). In addition, this kind of structure has a very long impact on the environment, and the effects persevere until decades after the landfill has been closed or even during the active phase (Cossu *et al.,* 2003).

The permeable layer (natural soil) may operate like a hydraulic control layer as in Figure 6a if the natural soil has relatively low hydraulic conductivity (e.g. 1×10^{-9} m/s or below). According to this concept, the natural soil is saturated and maintained at a soil pressure below the landfill. This creates an outward hydraulic gradient and an advective-diffusive flow (Varank *et al.,* 2011; Cossu *et al.,* 2003; Katsumi *et al.,* 2001). In areas where groundwater is near the landfill bottom structure, as shown in Figure 7b, the advective flow might become a dominant transport mechanism if the hydraulic conductivity is relatively high (e.g. 1×10^{-7} m/s or above). This situation is possible in environments where the load capacity has been increased by using a bearing layer that is made with the gravel of relatively high hydraulic conductivity (El-Zein & Rowe, 2008; Kamon *et al.,* 2005).

The purpose of the landfill base ground is to function as a load-bearing support layer under the environment protection structures. During the passive phase of a landfill, the base ground can function as a protective structure, depending on the suitability of its hydrogeological characteristics (Heikkinen *et al.,* 2002; Mälkki, 1999; Korkka-Niemi & Salonen, 1996; Lahermo *et al.,* 1996). During the passive phase, external water pressure is no longer directed to the waste fill, and if the drying mechanism of the base ground maintains its functions until the end of the landfill's lifespan, its geological barrier should not be subject to the load caused by hydraulic gradient.

In this case, a contaminant migrates via diffusion to the lower layers of the structure where the base ground acts as a contaminant retentive layer. At present, however, the retention and adsorption capacities of the base ground are measured only for particle size distribution and, in some cases, hydraulic conductivity (Environmental decisions 2007, 2005, 2005 and 2001). Recognizing the protective characteristics of the base ground would make it easier to define the dimensions of a requisite geological barrier because the long-term protective effects of the base ground could be taken into account as an additional protective factor. Therefore, it would be reasonable to investigate the environment protective effects of the base ground and take them into account when determining structural dimensions.

Table 5. Hydrogeological condition effect on contaminant transport in different categories in Finland (Heikkinen *et al.,* 2002; Korkka-Niemi & Salonen, 1996).

Factor	Reason	Consequence
Local	Groundwater distance from the structure	Transit time is depending on the distance
	Base ground material (till, sand, silt etc.)	Different contaminant transport time
Regional	Groundwater regional level	Groundwater level ranges in different areas in Finland
	Groundwater flow gradient	Harmful substances pass with groundwater movements
Temporal	Groundwater level variation during the year	Groundwater level could move up few meters during the year

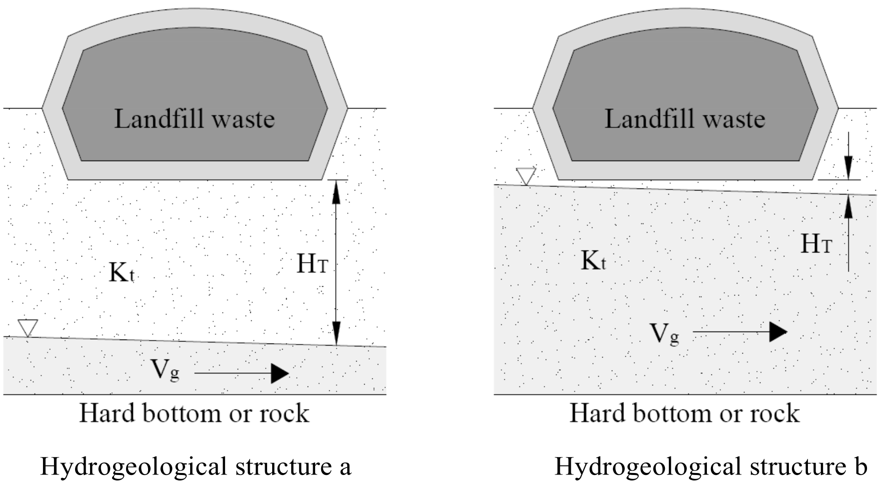

Fig. 7. A typical conceptual hydrogeological conditions in Finland. In Fig. 7a, the distance between groundwater and the bottom of the landfill liners is over 4 meters, and groundwater could reach the bottom of the landfill during the range of its movement. In Fig. 7b, the distance between groundwater and the bottom of the landfill liners is less than 1 meter, and groundwater could reach the bottom structures during the range of its movements. (Mälkki, 1999; Korkka-Niemi & Salonen, 1996; Lahermo *et al.,* 1996)

Hydrogeological environment's influence on the landfill design and environmental protection has been one of the main principles in the Landfill Directive because the artificial layer does not have to exist if the natural soil below the landfill protects the environment as the Landfill Directive demands (EC 31, 1999).

2.3.2 Wetting and drying phenomenon

During the landfill lifespan, its structures may be subjected to the effects of a wide range of different fluids and contaminants (Christensen *et al.,* 1994). The bentonite qualities used in mineral structures typically contain sodium bentonite. Leachate contains typically fluids which have a greater ion-exchange capacity than sodium (ion value 1). Due to ion energy charge e.g. calcium (ion value 2) is more active than sodium (Guyonnet *et al.,* 2003). Ion-exchange is known to affect particularly those structures that have high bentonite content in relation to the total amount of material (in relation to geosynthetic clay liner (GCL), for instance), in which case ion exchange often leads to increased hydraulic conductivity (Benson *et al.,* 2007; Guyonnet *et al.,* 2005; Petrov *et al.,* 1997). In ion exchange, two sodium ions with a value of one, for example, are replaced by a single sodium ion with a value of two, which leads to a single ion opening in the lattice space, which affects a change in the migration of matter.

Fluid load is a crucial factor in the functioning of bentonite clay. Soil bentonite mixes usually contain sodium bentonite, which is an ion with a plus one (+1) charge that can form a bond with a fluid, such as clean water, after which pore volume is filled due to the expansion of bentonite (He & Song, 2011; Guyonnet, 2009; Benson *et al.,* 2007). If clean water is replaced with a calcium chloride solution, the calcium, being an ion with a value of plus two (2+), will detach sodium from its bond with water and, as a result of ion exchange, the bond structure of the bentonite changes, leading to an increase in hydraulic conductivity (Bouazza *et al.,* 2008; Katsumi *et al.,* 2008). In his study, Bouazza (2008) used a calcium chloride solution with a solution of 0.0125 M, which caused a tenfold increase in hydraulic conductivity in a geosynthetic clay liner compared to the effects of de-ionized water when tested in permeameter.

Structure designs do not take into account, for instance, the effects of leachate on hydraulic conductivity. The design of landfill bottom layers is based on the advection of a sample soil saturated with clean water. The thickness of the bottom structure is calculated equivalently based on hydraulic conductivity relative to layer thickness determined in laboratory conditions and is guided by Governmental decisions (Environmental decisions 2007, 2005 a, 2005 b and 2001).

Based on Guyonnet et al. (2009), the impact of NaCl and $CaCl_2$ liquids on swelling based materials and a geosynthetic clay liner have been tested (GCL) over a 385-day period. The influence of the change of fluid from NaCl to $CaCl_2$ was seen to be significant. Hydraulic conductivities have increased from 1.5×10^{-11} m/s on average to 7.2×10^{-10} m/s at the end of the test. The effect of the divalent cation-rich fluid has been an increase in hydraulic conductivity by 48 hours or more (Guyonnet *et al.,* 2009). Based on Benson et al. (2007), MSW leachate typically includes di-, tri- and tetravalent cations that cause cation exchange, which influences the hydraulic conductivity of bentonite mixtures.

When used in a structure, the effectiveness of bentonite is mainly determined by tensions resulting from the structures above it. The saturation level of the structure and changes in the level as well as thermal and fluid load changes in the structure in which the bentonite is located (He & Song, 2011; Guyonnet, 2009; Benson *et al.,* 2007; Guyonnet et al., 2005; Petrov *et al.,* 1997). Since a state of tension (caused by load) compresses matter, tension reduces porosity. According to Eid (2011), increasing load affects saturation levels, and thus, the structure's load-bearing capacity. Variable saturation levels may cause tension changes in bentonite structures, which may result in local fracturing, that is, due to maximum shear. Fractures reduce the load-bearing capacity of bentonite structures, after which the partially saturated bentonite cannot withstand shear as well as the fully saturated structure (Eid, 2011). This factor is particularly significant in partially saturated conditions. Thus, the bentonite structure is liable to fracture locally, and alteration or change in saturation levels can cause a stability problem in an acclivity even though the structure would, according to measurements, appear to work normally when fully saturated.

Changes in saturation levels also affect the hydraulic conductivity characteristics of bentonite structures. The wetting–drying cycle affects the expansion of bentonite clay, and even after a few cycles, their degree of hydration is reduced, resulting in an increase in hydraulic conductivity (Bouazza *et al.,* 2008; Katsumi *et al.,* 2008). This is a factor that affects the suitability of structures for different uses, and according to Bouazza et al. (2008), hydraulic conductivity may become 10 times higher when compared to the situation before the wetting–drying cycle.

Malusis et al. (2011) has researched the behaviour of bentonite soil during wetting–drying cycles for two different bentonite content values (2.7% and 5.6% of dry volume-weight) in identical conditions with identical saturation and load (effective stress being 24 kPa). In the results, the water retention capacity of soil bentonite begun to decrease after three (3) cycles and correlated with the decrease in saturation with relation to the number of cycles (Malusis *et al.,* 2011). On the basis of this, it can be concluded that bentonite clay behaves in like manner regardless of the content value or external conditions. In typical usages of bentonite clay, that is, in landfill bottom and sealing structures, the saturation level of bentonite varies seasonally, due to which the structure's protective properties may decrease after as little as one year of cyclic behaviour.

Rowe et al. (2011) have examined the impact of daily thermal variation on the moisture balance of bentonite mats. In isothermal conditions, daily variation may cause a moisture reduction of 15% in a bentonite mat (Rowe *et al.,* 2011). When fully saturated, the moisture content in a bentonite mat is between 113% and 127%, depending on the mat type used. Changes in the moisture content affect the way in which a structure constricts, whereas constricting in turn affects the water retention capacity of a material and the hydraulic conductivity of a bentonite mat (Rowe *et al.,* 2011). From the perspective of structural design, this new information is significant because the variation in conditions between seasons in Finland is very pronounced.

In many cases, these factors take effect simultaneously, and for this reason, it is worthwhile to examine their combined impact. In landfill protective structures (including the bottom of a landfill at the early phase of its lifespan), for instance, partially saturated bentonite structures are exposed to wetting–drying cycles as well as to changes caused by ion-exchange and thermal variation. On the basis of the existing research, there are no studies regarding the scale of the combined effect of the said factors. The combined effects take place almost without exception due to the conditions in Finland. Furthermore, these structures may deteriorate quite rapidly due to cyclic changes in loads, variation in moisture balance and ion-exchange. (Henken-Mellies & Schweizer, 2011; He & Song, 2011; Guyonnet, 2009; Bouazza *et al.,* 2008; Katsumi *et al.,* 2008; Benson *et al.,* 2007; Meer & Benson, 2007)

Existing studies indicate that it is prudent to take into account the conditions that affect the functioning of a protective structure throughout its lifespan as realistically as possible (Bouazza *et al.,* 2008; Katsumi *et al.,* 2008). On the basis of recent studies, it can be said that, in general, the use of bentonite as a compacted element in the structure of a landfill is not a satisfactory solution when the life expectancy of the structure is taken into account; this is because the structure is subject to cyclic changes in both saturation and thermal variation. In addition, due to structural changes, the analysis of hydraulic conductivity in a fully saturated state does not reveal the actual situation during the early stage in the lifespan of a landfill.

According to Henken-Mellies and Schweizer (2011) and Meer and Benson (2007), research into bentonite structures should focus on the stability of saturation rather than on hydraulic conductivity because variation in saturation has a significant impact on the long-term functioning of environment-protecting structures and the protective effect of structures. The hydraulic conductivity of bentonite mats or bentonite soil mix increases 5- to 40-fold even when there is a one-meter-thick soil layer on top of the structure (Meer & Benson, 2007). On the basis of existing studies, it should be possible to protect bentonite-based environmental protection structures against drying by means of a highly compacted soil layer or a geomembrane (Henken-Mellies & Schweizer, 2011; Meer & Benson, 2007).

2.3.3 Freezing and thawing phenomenon

Seppälä (1999) presented the frost depths in Finland as a function of frost sum for various soil materials. Based on Seppälä´s data, frost depths average value is 2.25 m in sand and 1.8 m in silt. Also Venäläinen (2001) studied the maximum frost depths of road maintenance depots in Finland from 1974 to 1990. The measurements were made in Pudasjärvi, located at approximately the same latitude as Oulu in Finland, Luleå in Sweden and Mo I Rana in Norway. The result showed that the minimum frost penetration is less than 1.2 m and the maximum is more than 2.8 m depending on between-year variations. According to Venäläinen (2001) and Seppälä (1999), it is not possible to give unambiguous recommendations when the landfill structures are protected from freezing-thawing phenomena. Typically, the frozen layer will reach the earth's surface during the winter to the depth of two meters in most of Finland and Northern

Sweden. The winter is typically shorter and the frozen layer does not reach as deep near the Baltic Sea in the south and south-west compared to Northern Finland (Heikkinen *et al.,* 2002).

This phenomenon appears typically on surface structures because landfill surface structures are thinner than 2.0 meter e.g. leachate reservoirs. According to Hansson and Lundin (2006), hydraulic conductivity in partly saturated structures could be 100% lower compared with fully saturated structures.

Freezing and thawing could take place when the new bottom structure has been built, but there is no waste above the substructure before the winter period. Also leachate reservoirs are under the open sky at least over the high water level, and the leachate level could vary depending on the leachate content during the reservoir life-cycle. Freezing-thawing cycles could be numerous during the year.

In most part of Finland and north part of Sweden this problem could be appear during normal winter, when freezing and thawing begins during autumn when the nights are below 0 °C and the temperature rises during the day time. Also in spring time during the normal winter the phenomenon is the same. (Mälkki, 1999; Korkka-Niemi & Salonen, 1996; Lahermo *et al.,* 1996)

This phenomenon could appear during the landfill mining as well. If the landfill will be opened regularly, the landfill owner has to be aware that the demanding protection layer over the landfill bottom layer is retained. This is a phenomenon that none of the risk assessment method or programs does take a part, but at least in Finland cold climate have to be observed while doing risk assessment in landfills.

The impact of the freezing–thawing phenomenon on the conductivity of till based, silt, loam and sand structures has been studied with regard to both partly saturated and fully saturated structures. This phenomenon can be called as annual changes. As the temperature decreases below 0 °C, ice and ice crystals begin to form inside larger pores in soil (Hansson & Lundin, 2006). As water freezes in the pore holes, its volume increases by approximately 9% as a result of ice lens growth. According to Andersland & Anderson (1978) *"an ice lens attracts water from unfrozen layers, thus forming parallel ice lenses of various sizes and shapes, which are defined by the amount of unfrozen water available."*

Benson and Othman (1993) conducted a series of tests in which water was added to a soil layer simultaneously with the formation of a layer of ice, and thawing water was observed to exit the soil layer during thawing. In another test, the soil layer drew the water available in the unfrozen layer in a process of ice-lens formation. Neither freezing type has played a significant role in the results of long-term conductivity measurements performed on compacted clay and soil layers (Othman *et al.,* 1994; Zimmie, 1992; Champerlain *et al.,* 1990). Due to the fact that an unsaturated compacted structure cannot yield enough water for the formation of ice-lenses and because the changes in the volume of a partially saturated structure are considerably smaller when compared to a fully saturated structure, the freezing–thawing phenomenon has no significant long-term effect on the conductivity of a partly saturated soil layer. Porosity, saturation and compactness all affect conductivity, and the effects are further emphasized in landfill bottom structures. Typically, since vacant pore volume allows the structure to change and return to shape, changes in compacted structures are significantly smaller compared to ordinary soil because the amount of saturated pores in compacter layers is smaller (Champerlain, 1992).

According to Othman (1994), "*when soils are permeated with water after freezing and thawing, flow preferentially occurs through the crack network and secondary porosity. Thus, cracks that develop during freeze–thaw cycling increase the hydraulic conductivity of the soil."* The literature shows that natural and compacted clay soils that have hydraulic conductivities in the range from 10^{-9} m/s to 10^{-11} m/s before freeze–thaw typically have hydraulic conductivities in the order of 10^{-8} m/s after freeze–thaw (Othman, 1994). Othman and Benson (1992) found that *"faster freezing rates increased the hydraulic conductivity, but the effect of the rate of freezing on hydraulic conductivity after freeze–thaw was less than one order on magnitude"*.

According to Hewitt and Daniel (1997), "*tests on natural and compacted clays show that increases in the hydraulic conductivity of greater than one order of magnitude typically occur during the first cycle of freeze–thaw. Subsequent cycles result in smaller increases in the hydraulic conductivity, with minimal changes occurring after 3–10 cycles*". Based on the available literature, 3–5 freeze–thaw cycles are sufficient to determine the effects of freeze–thaw (Hewitt & Daniel 1997; Othman *et al.,* 1994; Othman & Benson, 1992; Wong & Haug, 1991; Chamberlain *et al.,* 1990; Zimmie & La Plante, 1990). According to Hewitt and Daniel (1997), "*changes in hydraulic conductivity are similar when samples are frozen one-dimensionally and three-dimensionally"*. Hydraulic conductivity increases less at high effective stress and effective stress influences the predisposition of clays to freeze–thaw damage influences more (Hewitt & Daniel 1997; Omidi *et al.,* 1996; Trast and Benson, 1995; Othman & Benson, 1991).

Freeze–thaw phenomenon can cause increases in hydraulic conductivity in the landfill bottom layer if the landfill is just constructed and there is not yet waste layer on the bottom structures (Othman *et al.,* 1994; Othman & Benson, 1992). The problem could be indicated by measuring hydraulic conductivity. The increased hydraulic conductivity occurs during the first years after construction and has a relationship with thickness of waste that covers and insulates the bottom layer. If the landfill bottom layer has been expanding, there is a possibility to move the existing waste onto a new bottom layer and build an isolation layer above the existing bottom layer. Given the typical conditions in established landfills, a two meters amount of waste usually precludes the possibility of building the isolation layer in most part of Finland (Mälkki, 1999; Korkka-Niemi & Salonen, 1996; Lahermo *et al.,* 1996).

2.4 Technical factors impact on hazard migration

Flow through porous media can occur if there is a potential difference between the interface of masses caused by fluid head, temperature, voltage or chemical potential (Mitchell, 1993). Conduction phenomena can be presented as four flow types for a cross section area A (Mitchell, 1993):

- Water flow, based on the Darcy´s law
- Heat flow, based on the Fourier´s law
- Electrical flow, based on the Ohm´s law
- Chemical flow, based on the Fick´s law

Water flow through the soil plays an important role in problems like seepage, consolidation and stability. Heat flow is relevant for example to frost actions, insulation and thermal pollution. Electrical flow is important to the transport of water and ground stabilisation for example by electro-osmosis, insulation and corrosion. Chemical flow or transport through the soil or ground is related to problems like groundwater pollution, waste disposal and storage, remediation of contaminated sites, leaching phenomena and soil stabilisation. The four types of flow above can be combined to form several types of coupled flows which are important under a variety of circumstances. (Mitchell, 1993)

In the landfill bottom layer, the most important flow phenomena are the water flow caused by the hydraulic gradient and the chemical flow caused by the concentration gradient (Benson *et al.,* 2007; Katsumi *et al., 2001;* Rowe *et al.,* 1995). Electrical and heat flows can exist too, but they do not dominate over the conduction phenomena (Mitchell, 1993). Water and chemical flows are caused by leachate that is stored in the waste and over the landfill bottom layer (Ramke, 2009; Rowe *et al.,* 1995).

2.4.1 Mathematical formulation of contaminant migration

The theoretical equation derived here is a statement of the law of conservation of mass, energy and momentum. Landfill bottom layer contaminant transport mechanisms are advection, dispersion, diffusion, sorption and decays (Sharma & Lewis, 1994). The mechanisms for contaminant transport are discussed in the following subsections. The primary transport mechanisms through the saturated barriers, for example clay or soil based mixtures, are advection and diffusion (Rowe *et al.,* 1995).

Solute transport analyses for geological barrier materials, such as compacted clay liners and soil based on material, are typically performed using solutions to the advective–dispersive equation (Shackelford & Daniel, 1991). In these analyses, the advective term is based on the Darcy law for solution flux in response to a hydraulic gradient, and the solute of diffusion is described by the Fick’s first law for diffusive flux in response to a concentration gradient.

Freeze and Cherry (1979) and Ogata (1970) have described the analytical solution of the derivation of the advection–dispersion equation for solute transport in a saturated porous medium. The analytical solution of transport equation is based on the derivation of Ogata and Banks (1961). The saturated porous medium is assumed homogenous and isotropic, the flow is steady-state and the Darcy´s law applies and the Reynolds number is between 1 and 10 (Bear & Palmer, 1972). In this derivation, the porous medium is assumed to be homogenous and isotropic, the flow is steady state, and the Darcy´s law equation will be valid when the hydraulic gradient is low and the flow is laminar.

Based on Freeze and Cherry (1979), *“the law of conservation of mass for a steady-state flow through a saturated porous medium requires that the rate of fluid mass flow into any elemental control volume is equal to the rate of fluid mass flow out of any elemental control volume”*. To establish the mathematical statement of the conservations of mass, the solute flux into and out of a small elemental volume in the porous medium will be considered in Equation (3). In coordinates the specific discharge v has components (v_x, v_y, v_z), the average linear velocity $\bar{v}$ = v/n has components (v_x, v_y, v_z) and n is the porosity. The rate of advective transport is equal to $\bar{v}$. The concentration of the solute C is defined as the mass of solute per unit volume of solution. The mass of solute per unit volume of saturated porous media is therefore nC. For a homogenous non-compressible medium, the porosity n is a constant, and $\partial(nC)/\partial x = n\, \partial C/\partial x$. The mass of solute transported in the x direction by the two mechanisms of solute transport can be represented as follows:

Mass transport per unit area by advection = $v_x nCdA$ and (2)

mass transport per unit area by dispersion = $nD_x \frac{\partial C}{\partial x} dA,$ (3)

Where C is the concentration of the solute,

n is the porosity,

D_x is the hydrodynamic dispersion coefficient,

x is the thickness of the layer and

dA is the elemental cross-sectional area of the cubic element.

The Darcy´s law states that there is a direct proportionality between the Darcy velocity v or flow rate Q and the hydraulic gradient i for different materials having a different hydraulic conductivity k. Equation (4) present the Darcy´s law

$$Q \sim A\frac{\Delta h}{L} \Rightarrow Q = Ak\frac{\Delta h}{L} \Rightarrow \frac{Q}{A} = ki = v, \tag{4}$$

Where Q is the flow rate,

A is the cross section area is normal in the direction of flow,

Δh represents the elevations of fluid levels,

L is the material layer thickness and

k is the coefficient of permeability or the hydraulic conductivity.

Based on Mitchell (1993), the linearity between the flow rate and the hydraulic gradient is the most significant in the lower range of gradients. In the field, the hydraulic gradient is seldom greater than one (Olivier *et al.,* 2009).

However, in a landfill, over the bottom structure, leachate could cause higher gradients than in the field (Olivier *et al.,* 2009). Based on Ramke (2009) and Olivier et al. (2009), high gradients could cause flow channels or deform the structure of soil.

In the diffusion process, there is an ionic or molecular constituent conduct in the direction of their concentration gradients. The diffusion does not require movement due to hydraulic gradient. This process stops when concentration gradients become negligible.

From the Fick´s first law and the equation of continuity, the rate at which solute could diffuse in porous materials can be given by the following expression (Freeze & Cherry, 1979). Diffusion could be equating the rate at which solute will diffuse in time and is known as the Fick´s second law presented in Equation (5):

$$\frac{\partial C}{\partial t} = D\frac{\partial^2 C}{\partial z^2}, \tag{5}$$

Where C is the concentration content,
t is time,
D is the diffusion coefficient and
z is the thickness of the layer.

Diffusion is not related to an advective transport direction, and diffusion can occur for example in the opposite direction compared to advective transport (Binns *et al.,* 2008; Rowe *et al.,* 1995).

It is possible for the contaminant to migrate from a landfill through the groundwater flow if the flow is directed to the landfill. In this study, groundwater flow is not directed to the landfill. The velocity v_s is positive if the flow is out of the landfill and negative if it is into the landfill (Rowe *et al.,* 1995; Sharma & Lewis, 1994).

F_x is the total solute mass entering an element and $F_x = \partial F_x/\partial_x$ is the mass loss, the difference between the total amount of solute entering the mass. This is shown in Equation (6):

$$F_x = \bar{v}_x nC - nD_x \frac{\partial C}{\partial x}. \tag{6}$$

A negative sign before a dispersive term indicates that the solute moves toward the zone of the lower concentration (Rowe *et al.,* 1995; Sharma & Lewis, 1994). Similarly, expressions in the other two directions y and z are written. Because the dissolved substance is assumed to be nonreactive, the difference between the flux into the element and the flux out of the element equals the amount of dissolved substance accumulated in the element (Rowe *et al.,* 1995). The complete conservation of mass expression can be written as Equation (7):

$$\frac{\partial F_x}{\partial x} + \frac{\partial F_y}{\partial y} + \frac{\partial F_z}{\partial z} = -\frac{\partial C}{\partial t}. \tag{7}$$

In a homogenous medium in which the seepage velocity $\bar{v}$ is steady and uniform, dispersion coefficients D_x, D_y and D_z do not vary through space and in one dimension, the result will be in the following Equation (8):

$$\left[D_x \frac{\partial^2 C}{\partial x^2} + D_y \frac{\partial^2 C}{\partial y^2} + D_z \frac{\partial^2 C}{\partial z^2}\right] - \left[\bar{v}_x \frac{\partial C}{\partial x} + \bar{v}_y \frac{\partial C}{\partial y} + \bar{v}_z \frac{\partial C}{\partial z}\right] = \frac{\partial C}{\partial t}. \tag{8}$$

Equation (9) is the one-dimensional form of the advection–dispersion equation for nonreactive solute transport in saturated soil. According to Sharma and Lewis (1994), this equation can be modified if the dissolved contaminant is removed from the solution due to sorption and biodegradation processes. The resulting equation can be presented in the form:

$$D_x \frac{\partial^2 C}{\partial x^2} - v_x \frac{\partial C}{\partial x} = R \frac{\partial C}{\partial t}, \tag{9}$$

$$R = 1 + \frac{\rho_d}{n} K_d, \tag{10}$$

Where K_d is the distribution coefficient,
n the porosity of the transport medium,
ρ_d the dry density and
R the retardation factor.

Dispersion is a mathematical term in the solute transport equation accounting for dilution or mixing according to concentration gradients (Freeze & Cherry, 1979). Mechanical dispersion is commonly negligible relative to molecular diffusion for the low-flow conditions and short distances of transport (≤ 1 m) typically associated with engineered containment barriers (Mitchell, 1993; Shackelford, 1988). Solute spreading is caused by mechanical dispersion that can arise at the pore-scale due to fluids moving faster at pore centres due to less friction (Freeze & Cherry, 1979). Larger pores allowing faster fluid movement and the flow are varying depending on tortuosity around grains. At a larger scale, macro dispersion is controlled by the distribution of hydraulic conductivity (Fatta *et al.,* 2000).

The sorption can be defined as linear or nonlinear, irreversible or reversible, and therefore, the mass of contaminants removed from solution is proportional to the concentration in the solutions (Rowe *et al.,* 1995). Adsorption describes water-soluble substance attachment to soil surfaces based on the electronic forces. Absorption describes the substance pile inside of soil. Absorption and adsorption are normally concerned with the same process: sorption (Freeze & Cherry, 1979).

The relationship between the sorption quantity of medium S_m [mg/kg] and the concentration content C [mg/dm^3] has been used for adsorption isotherm and can be either nonlinear or linear (Freeze & Cherry, 1979). Distribution between refined and soluble content has been described by the distribution coefficient K_d [dm^3/kg]. Adsorption isotherm is a linear function when K_d is constant (Bear & Palmer, 1972).

In the simplest case, shown in Equation (11), the sorption processes can be defined as linear and reversible, and therefore, the mass of the contaminant removed from solution, S is proportional to the concentration in solution, C:

$$S = K_d C, \tag{11}$$

Where S is the mass of solute removed from solution per unit mass of solid and other terms are defined above.

A plot of the variation in the solid-phase concentration S versus the solution-phase concentration under equilibrium is called an isotherm. The case represented by Equation (11) is a linear isotherm and is usually regarded as a reasonable approximation for low concentrations of contaminant. At high concentrations, sorption is non-linear, and more complex relationships between the solid-phase concentration S and the solution concentration have been devised. (Rowe & Booker, 1995)

The half-life describes the time during which the concentration is to be reduced to a half of the original concentration. The substances that undergo first order decay, the rate of reduction of concentration is proportional to the current concentration that is presented in Equation (12):

$$\frac{\partial C}{\partial t} = -\lambda C, \tag{12}$$

Where λ is the first order decay constant [1/t].

The first order decay constant λ has three components due to radioactive decay, biological decay and fluid withdrawal, respectively, as shown in Equation (13):

$$\lambda = \Gamma_R + \Gamma_B + \Gamma_s, \tag{13}$$

Where Γ_R is the radioactive decay constant,
Γ_B is the biological decay constant and
Γ_S is the volume of fluid removed per unit volume of soil per unit time.

This equation has the analytical solution that is show in Equation (14):

$$C(t) = C_0 e^{-\lambda_t}, \tag{14}$$

Where C_0 is the concentration solute of time (0),
C(t) is the concentration solute of time (t) and
e is the void ratio.

Radioactive decay is controlled by an element´s atomic structure and is essentially independent of the environment in which there are substantial available data that can be used to estimate the decay constant Γ_R. Biological decay depends on many factors such as the presence of appropriate bacteria, substrate, temperature, chemical conditions, pH, etc. The rate of decay will be specific to a given environment. (Rowe *et al. 1*995)

2.4.2 Factors affecting contaminant migration

Landfill contaminant transport could be focused on at least in three time categories: short term (0–30 years), medium term (30–100 years) and long term (> 100 years) (Huber-Humer & Lechner, 2009; Cossu *et al.,* 2003; Katsumi *et al.,* 2001). The contaminant migration to the environment in a different time frame is depending on the structures, quality and quantity of waste and structural functionality, for example drainage system performance (Varank *et al., 2011;* Morris *et al.,* 2009; Olivier *et al.,* 2009). In short and medium term, flow could be illustrating infinity flow, but in long term, flow illustrates finite flow (Kamon *et al.,* 2002; Katsumi *et al., 2*001).

Based on the literature, the concentration of potential contaminants generally increases during the operation of the disposal facility, reaches the peak and then declines (Morris *et al.,* 2009; Rowe *et al.,* 2004). The increase in concentration may be related to 1) the physical processes of the leaching of the contaminant from solid waste as water infiltrates through the waste or 2) the chemical and biological processes, which generate the chemical species of interest from the synthesis, or breakdown, of existing chemical species in the waste (Hudson *et al.,* 2009; Morris *et al.,* 2009; Olivier *et al.,* 2009; Rowe *et al.,* 2004). The decrease in concentration with time may be related to 1) the physical process of removal of the contaminant from the landfill (in the form of leachate) or 2) chemical and biochemical processes, which result in precipitation or the synthesis or breakdown of the chemical species of interest into other chemical forms (Hudson *et al.,* 2009; Morris *et al.,* 2009; Olivier *et al.,* 2009; Rowe *et al.,* 2004).

In many cases, the terms dispersion, diffusion and hydrodynamic dispersion are often used to describe the same phenomenon. In this case, dispersion has been used for the general phenomenon of the scatter. Molecular diffusion is then due to concentration gradient and mechanical or hydrodynamic dispersion is due to advection. (Lo, 2003 and Lo, 1996)

Advection transport is the dominant means for chemical flow for soils having a hydraulic conductivity greater than about 1×10^{-9} m/s (Katsumi *et al.,* 2001; Rowe *et al.,* 1995). Chemical transport by diffusion becomes significant relative to advective chemical transport in soils with hydraulic conductivity values less than 1×10^{-9} m/s. According to literature, for most soils, the molecular diffusion is in the range from 2×10^{-10} m^2/s to 2×10^{-9} m^2/s (Rowe *et al.* 1995; Mitchell, 1993). However, affecting a change in the molecular diffusion of soils is quite difficult, assuming that the layer is homogeneous and isotropic. (Mitchell, 1993)

Soil hydraulic conductivity may be affected by factors such as increasing the bentonite content of the mixture or increasing the compaction work. However, to affect a change in the molecular diffusion of soils is quite difficult, assuming that the layer is homogeneous and isotropic. (Mitchell, 1993; Rowe, 1988; Freeze and Cherry, 1979; Ogata, 1970)

Permeability and layer thickness are the most important properties of a landfill's bottom layer artificial geological barrier design, based on the Landfill Directive. Permeability has been defined in the Landfill Directive (EC 31, 1999) as the hydraulic conductivity requirement. Hydraulic conductivity is related to permeability and fluid and solid matrix properties. The relevant fluid properties are the density ρ and the viscosity μ. The relevant solid matrix properties are the fabric or fraction, the shape of the grains, tortuosity, specific surface and porosity (Bear & Palmer, 1972). Hydraulic conductivity can be expressed as (Nutting, 1930) in Equation (15):

$$k = K\rho g / \mu, \tag{15}$$

Where k is the hydraulic conductivity and
K is the intrinsic permeability.

Many models have been developed for the prediction of the hydraulic conductivity of soil materials based on water content and dry density or saturated unit weight, for example by Atuahene (2008), Boardman and Daniel (1996), Benson and Daniel (1994), Acar and Haider (1990), Kozeny-Carmen (1972), Harleman et al. (1963), Krumbein and Monk (1943) and Hazen (1911).

The deviation of the results varies depending on the materials and experimental arrangements. The variation is greater when the materials are compared in field and laboratory tests (Purdy & Suryasamita, 2006; Folkes, 1982). Based on the study of Purdy and Suryasamita (2006), a single hydraulic conductivity test does not provide a sufficiently reliable view of the structure. Parallel determinations can be used to achieve a reliable picture of the structure of hydraulic conductivity.

One of the main reasons for different results between laboratory and field tests is the goal of determining the lowest hydraulic conductivity of material, which invariably leads to far-processed frames of production and, consequently, results of small mutual dispersal. Based on Folkes (1982), laboratory tests give typically lower hydraulic conductive results than field tests.

Hydraulic conductivity is not related to hydraulic gradient, assuming that the Darcy law applies (Mitchell, 1993). Hydraulic gradient plays an important role in regard to contaminant migration through the landfill bottom layer. Flow velocity is directly related to hydraulic gradient, and gradient changes also influence the quantity of flow. In the laboratory, tests use higher hydraulic gradients, commonly over 10, but also up to several hundreds, speeding up the test and increasing the utility suitability of the laboratory test results (Sharma & Lewis, 1994).

A high hydraulic gradient can cause loading intensity on the thin samples, and at the same time the loaded sample consolidates in the cell. In this case, hydraulic gradient can reduce hydraulic conductivity, and the sample consolidation can cause significant differences between laboratory-measured and field-measured hydraulic conductivities (Mitchell, 1993). An important aspect of hydraulic conductivity determination is that the linearity of the laboratory conditions versus field conditions provided no changes in the sample fabric during the testing of the water-saturated samples (Rowe *et al.,* 1995).

The degree of saturation and porosity or the total number of pore classes are parts of an equation in which the hydraulic conductivity of partly saturated soils is dependent on the degree of saturation caused by the negative pore water pressure (Mitchell, 1993). This causes the different degrees of hydraulic conductivity in soils, compared with fully saturated situation.

Based on Rowe et al. (1995), inactive liners will be nearly saturated compressible on loading, which are compacted with water contents higher than the standard Proctor optimum moisture content and not allowed subsequently to dry out. Liners should be behaving like a fully saturated barrier. This leads to a situation in which the primary transport mechanism through a well-designed compacted liner will be molecular diffusion (Rowe *et al.,* 1995). Unsaturated hydraulic conductivity is the mainstay, modulating water and chemical transport in the field (van Genuchten *et al.*, 1980). Significant events such as runoff, drainage, soil reclamation and chemical transport are related to unsaturated water transport, and this requires advanced knowledge of water flux under unsaturated conditions and spatial variation of this flux. Especially contaminant transport under unsaturated conditions is affected by soil hydraulic and chemical properties and process in soil. (Thomasson & Wierenga, 2003)

The advective–diffusive movement of contaminants through partly saturated soil is more complicated than through saturated soils. The partial differential equation governing one-dimensional movement is given in Equation (16) by:

$$\frac{\partial}{\partial t}\left(n\Theta C\right)=\frac{\partial}{\partial x}\left(n\Theta D\frac{\partial C}{\partial x}\right)-\frac{\partial}{\partial C}\left(nv_x C\right)-n\Theta\lambda C, \tag{16}$$

Where n is the effective porosity of the soil (Θ volumetric water content equal to the porosity for a saturated soil)

All other terms are as previously defined. The movement of contaminants through the partly saturated soils is a very complex phenomenon. The simplest case is when there is negligible advective transport through the partly saturated soil. This situation can only arise when the net infiltration is negligible for example below the geomembrane. Under these circumstances the migration of the contaminant in a solution will be very slow because the migration will be pure by diffusion and it has been very slow (Rowe *et al.,* 2004).

The partly saturated soil will be usually hydraulically active, and advective transport must be taken into account. The advective transport depends on the hydraulic conductivity of the soil. This tends to increase with the volumetric water content of the soil up to a maximum value for a saturated soil (van Genuchten *et al.,* 1991; van Genuchten, 1980). The hydraulic conductivity of partly saturated soil will be more sensitive to point-to-point variations in grains size distribution than saturated soils, and this makes the determination of representative hydraulic conductivities substantially more difficult (Rowe *et al.,* 2004).

A literature review of the effect of chemicals on hydraulic conductivity related that chemicals or leachates may, for example, influence clays through their effects on the composition of the clay, and hydraulic conductivity can increase or decrease depending on the chemical and fabric (King *et al.,* 1993; Bowders & Daniel, 1987; Mitchell & Madsen, 1987).

2.4.3 Artificial structures' impact on contaminant migration

The bottom layer receives the effective stress via the waste, and thus, long-term impacts on advection increase are in most cases impossible (Bouazza *et al.,* 2008; Katsumi *et al.,* 2008). Therefore, the effect on contaminant transport is not so significant because advection is not the dominant transport mechanism in the bottom layer. Cracks, migration and porosity changes, in most cases, are short-term effects because effective stress impacts the bottom layer and the structure returns to its original form.

The essential guiding principle in estimating the serviceable life of landfill bottom structures is the protection of the landfill environment from the effects of a waste fill. Making an estimate like this requires information and assumptions about the amount and type of waste that will be brought into the landfill in the future, changes in environmental legislation and the methods of waste management and landfill environment requirements. Furthermore, a landfill bottom structure is typically a permanent structure that cannot be modified or repaired during its lifespan.

Estimates on the life expectancy of a landfill bottom structure must be made with regard to the different structures and materials that it consists of, taking into account contaminants, migration mechanisms, environmental conditions and the stability and durability of underlying structures. Due to interaction between and variation in the materials used in protective structures constructed out of natural materials, it is very difficult to form estimates about mineral protective structures. The effects of migration, retention and conduction of materials as well as the phenomena that affect structures have been discussed earlier in the present study. In equivalency comparisons between materials used in structures, it must be acknowledged that, in different structural solutions, different factors interact, and their combined effect may have a detrimental impact on the functioning of a structure. (Bouazza *et al.,* 2008; Katsumi *et al.,* 2008)

By means of a risk analysis of environmental protection structures, it is possible to examine a situation during the design phase in which a geological barrier functions as an environmental protection structure. In addition, it is possible to predict the lifespan of a protective structure or artificial layer above the geological barrier as well as the end of its serviceable life and resulting contaminant penetration into the geological barrier (Fig. 8). The leak area size and leak amount variations can be used when identifying a situation in which contaminants have penetrated protective structures due to various migration mechanisms and reached the structures below, such as groundwater. Regarding protective structures, contaminant retention capacity should be examined in addition to defining the migration of contaminants, taking into account, among other things, the effects of diffusion. The functioning of a structure throughout its entire life expectancy should be ascertained by making an estimate of the serviceable life of the structure that includes the effects of contaminant migration (Fig. 8). (Bouazza *et al.,* 2008; Katsumi *et al.,* 2008)

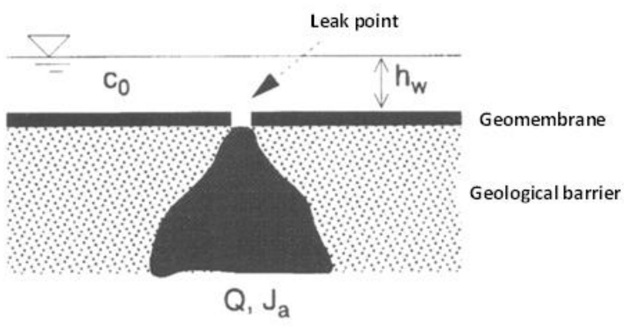

Fig. 8. Katsumi leakage model (Modified after Katsumi *et al.,* 2001).

The migration of contaminants is mainly affected by the number and size of calculated leakage areas, leachate gradient and concentration, as well as the hydrogeological characteristics of the geological barrier and structure thickness (Xie *et al.,* 2010, Foose *et al.,* 2002; Kalbe *et al.,* 2002). The number of factors is very high; additionally, they interact and depend on each other. Furthermore, the mechanisms affecting migration do not all the time function in the same way. On the basis of this, it can be said that accomplishing an absolutely accurate modelling of the phenomena is extremely difficult as the number of factors is high and needs to be limited when making a model (Xie *et al.,* 2010).

According to Foose et al. (1999), Rowe (1998) and Park et al. (1996), the molecular diffusion is a more significant factor in contaminant migration when compared to leakage because organic substances have access to and can affect the entire area of the geomembrane in a bottom structure and, in the long term, contaminant leakage through the geomembrane is significant. The significance of the layers beneath the geomembrane is emphasised because, in some leakage situations in which the leakage area is large in size or the adhesion area between the geomembrane and the structure beneath it is inadequate, flow can become strong enough to increase the effect of advection due to hydraulic gradient. At the present time, design processes and risk estimates do not adequately acknowledge the effects of molecular diffusion during the serviceable life of a bottom structure.

According to Forget et al. (2005), artificial structures, e.g. geomembranes, could cause mechanical strain or stress during installation, transport or production. The amount of leaks or holes depends on the material thickness and quality control procedures during the installation process. Typically, leaks are very small and locating them with the eye is not possible without using e.g. an electrical conductivity detector. According to Grellier et al. (2006), temperature changes could influence leachate conductivity. During landfill mining, HDPE liner could cause a leakage peak under the landfill because of leachate temperature and liner temperature changes. It could increase the amount of holes in the liner, extend the size of holes, liner degradation during the life-span and accelerate the leachate movements (Grellier *et al.,* 2006).

2.5 Typically used risk assessment tools in landfills

Butt et al. (2014) have introduced the baseline study modules and its position in relation to overall risk assessment structure. It is a review of critical factors related to environmental risk assessment. It contains the state of the art of the previous studies and illuminates the reliance between the overall risk assessments. It is a model of landfills risk assessment parts, but without the baseline study modification could not be used in practice. The baseline study modules give extensive knowledge, but it has to be relinked between the aspects and factors.

One of the famous risk assessment methods is the US EPA risk assessment forum. The US EPA has used e.g. the Monte Carlo method as a tool in risk assessments. This method has been used as a starting point for the development of other risk assessment methods like LandSim.

2.5.1 Computer-aided landfill risk assessment tools

This literature review focuses on landfill risk assessment tools that approach landfills as a combination of soil, groundwater, protection structures, leachate, gas emissions and local risk assessment elements. The relevant computer-aided approaches that are recognised to be closely related to those are (the list modified after Butt *et al.* 2014):

- FRAMES-3MRA (Environment Agency, 2002)
- LandSim (Environment Agency, 2003, 2001, 1996)
- HELP – Hydro-geological Evaluation of Landfill Performance (Scientific) Software (water balance calculations)
- GasSim (Attenborough *et al.,* 2002; Golder Associates, 2003)
- GasSimLite (Environment Agency, 2002)
- LandGEM (Landfill Gas Emissions Model, Enrivonment Agency 2005)
- RIP – Repository Integration Programme (Landcare Research, 2003; Golder associates, 1998)

FRAMES-3MRA is a software model developed by the US EPA for assessing hazardous waste risks and risk management. This model includes 17 modules which can be used to simulate the effects of risks on the environment. (Babendreier & Castleton, 2005)

The LandSim model focuses on leachate production and collection, chemistry, harmful substances migration and leakage through artificial barrier to geological barrier and base ground. LandSim model allows landfill operators and authorities to consider and observe the environmental performance of different artificial barrier/liners and leachate collection systems, and to take account of the large variety of geological and hydrogeological conditions. The model analyses leachate migration through the unsaturated zone to the ground by assessing the impact on the aquifer. LandSim also includes finished design models which can be used to model the harmful substance migration. The model uses Monte Carlo probabilistic performance assessment and could be used for new or existing landfills. The LandSim software model has been developed to provide probabilistic quantitative risk assessments of the performance of specific landfill sites in relation to groundwater protection. (Environment Agency, 2003)

The HELP model (Hydro-geological Evaluation of Landfill Performance) is a programme to design, evaluate and optimise landfill hydrology and groundwater recharge. The HELP model is used and recognised all over the world as the accepted standard for modelling landfill hydrology, and has become an integral component for projects involving landfill operating and closure permits. (Schroeder *et al.,* 1994) LandSim is an advanced method of HELP.

GasSim, GasSim lite and LandGEM simulate the emission of landfill gas. "*The models use information on waste composition and quantity, landfill engineering, and landfill gas management techniques to enable assessment of the best combination of control measures for a particular design and rate of filling*". The model could be used as a part of a total risk assessment and cases where landfill gas emission could cause disadvantages to the environment. (Attenborough *et al.,* 2002; Golder Associates, 2003)

2.5.2 Risk assessment tools for environmental systems

Some computer-aided software programmes have an integrated probabilistic simulator for environmental systems and assessing risks to human health and the environment. These methods, such as the RIP – Repository Integration Programme, GoldSim, ConSim, the Contaminant Land Exposure Assessment (CLEA), Spatial Analysis and Decision Assistance (SADA), have not been developed for landfill risk assessment (Butt *et al.,* 2011; Leavesley & Nicholson, 2005; Whittaker *et al.,* 2001; Riggenbach *et al*., 1991).

All computer-aided software typically contains only some part or aspects of landfill risk assessment, instead of overall examination (Butt *et al.,* 2011). These models or methods could be used as design tools for landfill structures. Methods or models could also be combined to obtain a more relevant risk assessment of specific cases (Chowdhury, 2009; Giusti, 2009; Pollard *et al.,* 2006).

2.6 Summary of landfill risk assessment

This section includes essential risk factors affecting landfill structures in Finland. Table 6 represents four essential factors which have to be focused and the factors relevance to the landfills´ structures. The factors have been ranked depending on the relevance to landfill bottom, surface and reservoir structures. The approach cannot be applied without modification.

Table 6. The essential factors focused depending on the relevance to landfills structures.

Structure type	Contaminant migration	Groundwater balance	Wetting-drying	Freezing-thawing
Landfill bottom	***	***	*	*
Landfill surface	*	*	**	**
Reservoir	***	***	***	***

Note. *** is ranked more important than *.

Contaminant migration affects the surrounding environment during the landfills life cycle. Landfill bottom layers´ and leachate reservoirs´ function are to prevent the leachate influence and protect the environment. Surface structures typically prevent only rainwater migration to waste and therefore the relevance is not so essential.

Groundwater balance is in the essential role in landfills risk assessment. Several national Environment Agency has produced guidance documentation to assist the waste management and authorities' interpretation of groundwater protection (Bonaparte *et al.* 2002). Typically, guidance provides the requirements for groundwater risk assessment and the setting of groundwater control and trigger levels. These documents contain a technical guidance to landfill operators, designers and authorities. The guidance describes a complex approach to landfill risk assessment according to e.g. England hydrogeological requirements.

Wetting-drying and freezing-thawing phenomenon has essential relevance for deforming materials like bentonite soil materials or clay. Reservoirs´ top part is unprotected during the structures whole life cycle. Climate changes and annual changes could cause huge amount of cyclic loads to the reservoirs and transform the fabric of the artificial materials. Typically, surface and bottom structures are protected by soil layers after construction and e.g. cold climate areas annual changes could cause cyclic load to the structures.

3 Materials and methods

The EC Landfill Directive and the Finnish Government decision determine the grounds for the implementation of the environmental permit procedure, requirements for the environmental protection structures and the grounds based on which deviations from the requirements are possible. The most important possibilities for deviations are provided by the risk assessment in the designing phase in which the effects of the deviations have to be compared with the environmental protection capability of structures that are in accordance with the requirements (VNp 861/1997; EC 31/1999; VNp 1049/1999). Figure 9 presents a simplified flowchart of the environmental permit process depending on whether the permit application meets the landfill structure requirements as such or whether risk assessment is needed for the approval of the landfill structures.

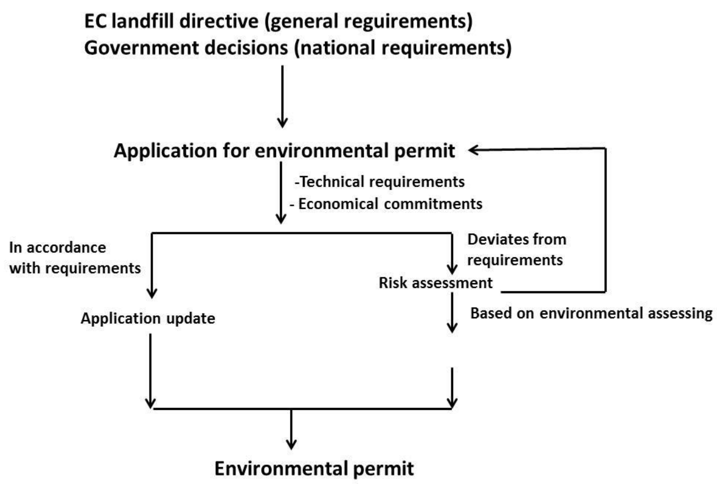

Fig. 9. Flowchart of the environmental permits process in Finland.

3.1 Comparing scientific literature against Landfills' environmental decisions made

The environmental permit defines, in addition to the environmental protection structures, obligations for the landfill owner, for example technological monitoring obligations and economical obligations. The environmental permit decisions of 12 MSW landfills that differ from the Finnish Government decision 86/1997 were chosen for the source material for this study. The information of landfills is based on existing data from the building plans and environmental decisions (all the bottom layer structures are described in Figure 13 and are based on structure 2, 3 or 4). In these environmental permit decisions, factors affecting the structures, contaminant retention capacity and life-cycle are studied, along with their effects. The selected landfills environmental decisions has been made between the years 1996 and 2004 without identifying the decisions, and the information was wanted to be kept anonymous. These environmental decisions were focused on the basis of technical requirements, the bottom structure thickness and hydraulic conductivity. In addition, the bottom structures' hydraulic conductivities are calculated according to an advection equivalent calculation without any other transport mechanisms, which are the main reasons why these landfills have been chosen. The environmental permit decisions have been collected from the Internet where they are freely available. Source material data has been compiled and classified into three main categories:

- **i) Human delineated factors related to risk assessment**
- **ii) Environmental factors effects to risk assessment**
- **iii) Technical factors impacts to risk assessment**

In this thesis, the risk factors identified during the MSW landfill environmental permit processes are compared to the environmental risks that are, based on the literature, the most significant (Varank *et al.,* 2011; Ramke, 2009; Guyonnet *et al.,* 2009; Cossu *et al.,* 2003; Giroud *et al.,* 2000; Korkka-Niemi & Salonen, 1996; Rowe *et al.,* 1995; Shackelford & Daniel, 1991). The reference data will be used to identify the effects of the most important factors and the relation to the current conventions.

Data processing was realized utilizing inductive reasoning, in which premises lead to true conclusions. The reasoning proceeds by conclusions from a general statement into a specific conclusion (Fig. 10) (Eskola & Suoranta, 1996).

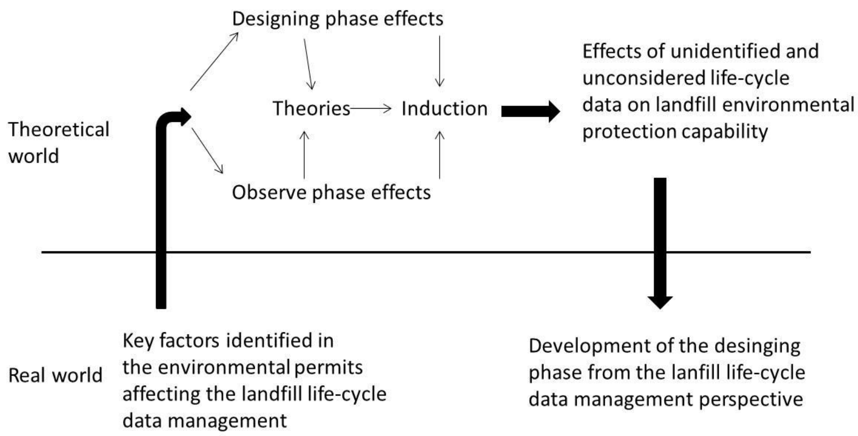

Fig. 10. Research data processing using the inductive method in this thesis.

Structural risk assessment method (SRA method) has been developed and verified with two environmental cases in this thesis. One of the structures is surface structure and the other one hazardous waste bottom structure. The surface structure has had the environmental permit during 2013 and the hazardous waste structure 2014.

The modern MSW landfill is located in Ostrobothnia and verified structure is surface structure. The landfill infrastructure and bottom structures have been made 2001-2003. The bottom structure includes HDPE membrane and half meter thick gravel and polymer sand-bentonite structure over the base ground. The waste was typical MSW waste, waste embankment height was 22 meters, the slope angle was 14 degrees, inside the waste was gas collection system and over the bottom layer leachate collection system leading leachate to reservoir. Landfill is located in the forest natural environment and the closest neighbour is over 500 meters. In the landfill area does not exits any built environment influencing landfill hydrogeology or causing human influence.

The hazardous waste landfill will be located near the Gulf of Bothnia. The existing landfill bottom structure includes HDPE membrane and 60 cent meter thick compacted peat structure over the base ground. The waste was steel mill hazardous waste, waste embankment height was 44 meters, the slope angle was 14 degrees, the waste was does not formulate any landfill gas and over the bottom layer leachate collection system leading leachate to the reservoir. Landfill is located in the industrial area and the closest neighbour is over one-kilo meters.

The premise of the risk assessment process was to find a solution for cost-effective structure that is fulfilling the environmental protection demands and could be able to use steel mill by-product as much as possible. A part of the mills by-products have been CE-marked.

3.2 Literature review for developing SRA method

SRA method has been tested before it was finalized in this dissertation. SRA method has been developed based on the theory review, observing local circumstances and special features of the climate and soil. This method is partly based on the same elements as the study (Butt et al. (2014). The SRA method is also partly based on the Landfill liner system checklist, evaluation of post-closure care (EPCC) methodology and weight-of-evidence (WOE) approaches (Laner *et al.,* 2012; Morris & Barlaz, 2011; Sizirici *et al.,* 2011; Pivato & Morris, 2005; Giroud *et al.,* 2000).

The Monte Carlo simulation (MCS) was applied as a novel method to analyse risk assessing sections uncertainty and to assess the environment pollution occurrence (Neshat *et al.,* 2015). The Monte Carlo simulation is used for sensitivity analysis and quantified probabilistic analysis (Baeurle, 2009). MCS is useful for simulating curtain phenomena significant uncertainty, like hazards migration thru the landfill bottom layer (Caflisch, 1998).

The developed method differs from other management methodologies by focusing also on the design management requirements like freezing–thawing and wetting–drying phenomena, leachate content, hydrogeology, meteorology and soil effect as a part of the risk management. Despite local weather and soil variations compared to Central Europe or the United States, most of the methods do not take these issues into consideration.

3.3 Methods for calculating contaminant migration times

This study focuses on landfills with fully saturated geological barriers and concentrates only on considering leachate flows through geological barriers. In addition, the hydraulic gradient is assumed constant over geological barriers. Based on literature, in the worst case, there may be a couple of meters leachate stored in the waste over the bottom layer (Hudson *et al.,* 2009; Morris *et al.,* 2009; Olivier *et al.,* 2009; Ramke, 2009). The transit time through the bottom layer has been calculated according to horizontal equivalent hydraulic conductivity caused by advection equation and one dimensional form of advection-dispersion equation.

The effect of landfill environmental protection structures on the environmental protection capability is examined by comparing the contaminant migration times in bottom structures using two different calculation models. It will be possible to estimate the migration time comparison whether the currently used calculation model in risk assessment produces similar results with the total migration equation.

The environmental protection capability of the landfill bottom structure is examined by comparing the contaminant migration times computationally using the advection and Ogata-Bank's equations one dimensionally for four different bottom structure types. The Ogata-Bank's equation was selected because it has been commonly used for proving contaminant migration computationally. (Sangam & Rowe, 2005; Katsumi *et al.,* 2001)

In the advection calculation, it will be calculated how long it will take for the contaminant to migrate through the landfill bottom structures with different thickness values and different hydraulic conductivity values. Based on Ogata and Banks (1961), an analytical solution of contaminant transport was made for Equation (17). This equation can be written in the form:

$$\frac{C}{C_0} = \frac{1}{2}\left[erfc\left(\frac{h - vt}{2(Dt)^{1/2}} \right) + \exp\frac{vh}{D} erfc\left(\frac{h + vt}{2(Dt)^{1/2}} \right) \right], \tag{17}$$

Where C is the concentration at time t,
C_0 is the concentration at time=0,
h is the thickness of the layer,
v is the Darcy velocity(m/s),
t is time and
D is the diffusion coefficient.

Based on the comparison results, it can be concluded whether the calculation practice influences the contaminant migration time. Also it can be demonstrated if the bottom structure will achieve a sufficient environmental protection impact from the perspective of landfill management during the whole landfill life-cycle by the migration based solely on advection. In the analysis of the results, it is taken into account that, based on the literature, advection is not the dominant transport mechanism if the hydraulic conductivity value is below 1×10^{-9} m/s. (Katsumi *et al.,* 2001; Rowe, 1995)

In the data processing, explanations or validations are searched for in the theory to support the interpretations of the findings from the data. Based on the source data, a hypothesis was formed according to the contaminant migration transit times thru the landfill bottom structures should be similar independent of the calculation method. Pulse or continuous flow situation can be calculated analytically only on limited conditions. The boundary conditions can demonstrate the worst case on the landfill bottom, and on these bases, the harmful contaminants' influence on the environment could be less than the results prove. Hydrogeological conditions, the type of flow and contaminant characteristics are related to the calculation (Rowe *et al.,* 1995). Typical boundary conditions that have been adapted as source data in this work will be discussed in the following.

The surfaces of geological barriers are assumed to be impermeable, and the contaminant flux is considered as one dimensional (1 D) and the flow across barriers to be zero. The partial water vapour pressure in pores is equal to the atmospheric pressure that is fully saturated. In addition, the steady state situation is reached, there is no change in the effective stress and the barrier will undergo neither deformation nor volume change. Furthermore, mass balances in landfills are assumed to be stable. Also, the temperature is assumed to be constant and not causing any influence on the results.

In addition, when the contaminants distribution is assumed to be stable and that there will be no environmental changes such as changes in advective velocity or any other conditions within the landfill. Groundwater conditions and flow gradients are assumed to be constant. Also, this study assumes that the contaminants will not react with each other; that is, the contaminants are assumed to be conservative solute and dispersions not impacting contaminant migration. Thermal or electrical flows are assumed not to influence the conditions or particles inside or outside landfills.

Based on the experiments and the literature flows through geological barriers are assumed to be one dimensional (Mitchell, 1993; Ogata, 1970). Below the geological barrier, the flow may be 1–3 D depending on the hydrogeological conditions, quantity or quality of contaminants and base ground materials (Rowe e*t al.,* 2004). Analytical calculations are based on constant leachate flows from landfills though the bottom structures. Only 1 D flow was considered, because horizontal flow can be expected to be negligible in relatively thin layers (Hall *et al.,* 2007). Hydraulic water pressure and chemical gradients are assumed to be caused by advection and hydrodynamic dispersion (Rowe *et al.,* 2004).

In Figure 11, the first structure describes the EC directive type of structure (Sarkkila *et al.,* 2006; EC 31, 1991). The other three structures are based on a 0.5 meter thick artificial geological barrier; the double thin layer structure and the triple thin layer structures that are typical bottom layer structures in Finland (Sarkkila *et al.,* 2006; Finnish Environment Institute, 2002).

The analysis assumes that the drainage structure was fully saturated with leachate. The landfill was assumed to be in the passive phase, or that the artificial geomembrane structure is damaged or does not operate properly. Fluid properties are assumed to be equal in all structures. Criteria of the structures are listed in Table 7.

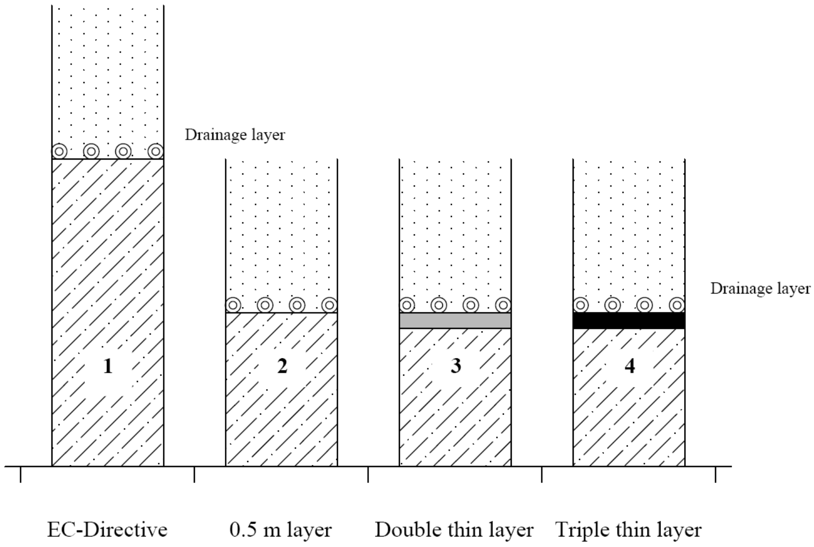

Fig. 11. Four different conceptual bottom layer structures and dimensions used in this study. 1) EC directive and 2) 0.5-meter layer has single homogenous material as a mineral layer, 3) double layer has very thin and low hydraulic conductivity layer and below higher hydraulic conductivity layer, 4) triple layer has two very thin and quite low hydraulic conductivity layers and below higher hydraulic conductivity layer. (Environmental permit registry, 2010)

Table 7. The criteria of four different conceptual liners, dimensions, hydraulic and diffusion coefficients values based on Fig. 11. Liner 1 is based on the EC directive and the other three liners are typical bottom layer structures in Finland.

Criteria/type of liner	Liner 1	Liner 2	Liner 3	Liner 4
Hydraulic conductivity k_1 [m/s]	1×10^{-9}	0.67×10^{-10}	4.9×10^{-10}	4.6×10^{-11} (air void 2.5%)
Thickness h_1 [m]	1.0	0.5	0.09	0.05
Hydraulic conductivity k_2 [m/s]	-		1×10^{-8}	1.5×10^{-9} (air void 3.1%)
Thickness h_2 [m]	-		0.21	0.06
Hydraulic conductivity k_3 [m/s]	-		-	$6,67\times10^{-10}$
Thickness h_3 [m]	-		-	0.25
Diffusion coefficient D_1 [m^2/s]	2×10^{-10}*	2×10^{-10}*	2×10^{-10}*	6×10^{-12}**
Diffusion coefficient D_2 [m^2/s]	-		2×10^{-10}*	6×10^{-12}**
Diffusion coefficient D_3 [m^2/s]	-		-	2×10^{-10}*

Note. * is based on Katsumi (2001), dependent on the contaminant, and ** is based on Deponieverordnung (2002). Hydraulic conductivity values are based on hydraulic conductivity tests made with leachate.

3.4 Materials and fluids used in hydraulic conductivity tests

The EC Landfill and the Finnish Government decision define a value requirement for hydraulic conductivity (EC 31/1999; VNp 861/1997). In hydraulic conductivity tests, ultra clean water is typically used as a fluid. Unlike clear water, leachate causes a contaminant burden on the geological barrier of the landfill bottom structure.

This dissertation focuses on the hydraulic conductivity of selected mineral liners and identifies the main factors affecting the hydraulic conductivity phenomena. The laboratory hydraulic conductivity determinations have been made, and the results are collated for various tested different mineral liners, an industry-made geosynthetic clay liner and various sodium-bentonites with different mineralogical characteristics. The empirical tests for this dissertation have been conducted using both clear water and MSW landfill leachate. The studied materials were typically used in bottom layers as artificial geological barriers in Finnish landfills (Finnish Environment Institute, 2002).

In research data processing, inductive reasoning has been applied, in which tests with the same material should end up into same end results, independent of the test materials or reference sample. Table 8 presents the studied material and the amount of tests using clean water and leachate.

Table 8. Tested materials and number of hydraulic conductivity tests with clean water and leachate (N=18).

Material	Clean water [n]	Leachate [n]
Sand mixed with polymer bentonite (spb)	2	2
Geosynthetic Clay liner (GCL)	5	2
Natural Clay (Jämsä) (Clay 1)	1	1
Natural Clay (LPR) (Clay 2)	3	2

Note. Abbreviation LPR refers to Lappeenranta.

Table 9 and Table 10 illustrate the main characteristics of the materials. Granulation was determined by the material manufacturer. The grain size distribution curves of sand and till mixed with polymer bentonite materials are presented in Figure 12.

Table 9. Main characteristics of swelling clay material.

Materials	Water content w_{opt} [%]	Maximum dry unit weight γ_{dmax} (kN/m^3]	Grain size < 0.063 mm [%]	Porosity
Sand mixed with polymer bentonite 1	8.4	16.52	5	0.42

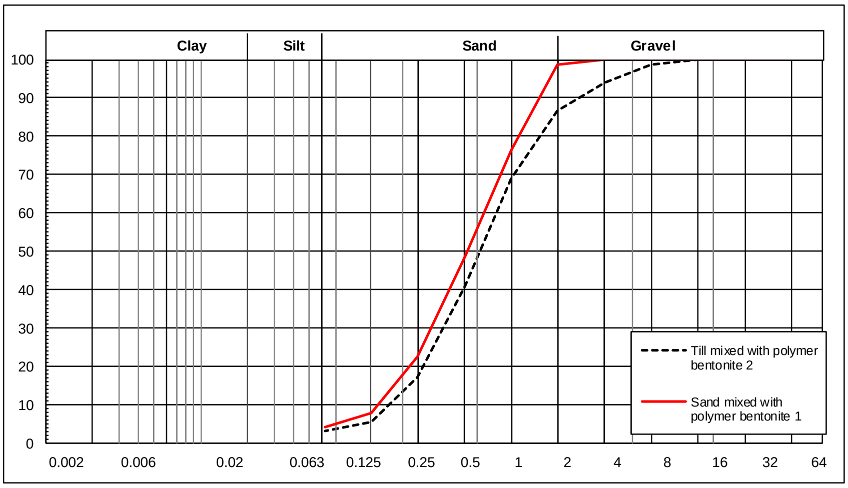

Fig. 12. Grain size distribution curves of sand and till mixed with polymer bentonite material.

Table 10. Main characteristics of natural clay materials.

Materials	Water content w_{opt} [%]	Maximum dry unit weight γ_{dmax} [kN/m^3]	Grain size < 2 µm [%]	Porosity
Clay 1	27 $^{*)}$	15.15	35	0.58
Clay 2	32 $^{*)}$	15.55	29	0.56

Note. *) natural water content

Figure 13 and Figure 14 illustrate the grain size distribution curves of materials Clay 1 and Clay 2.

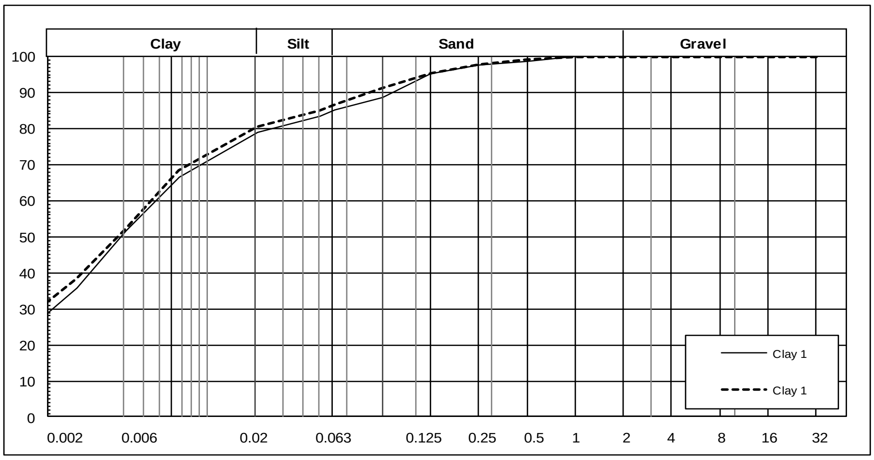

Fig. 13. Grain size distribution curves of Clay 1.

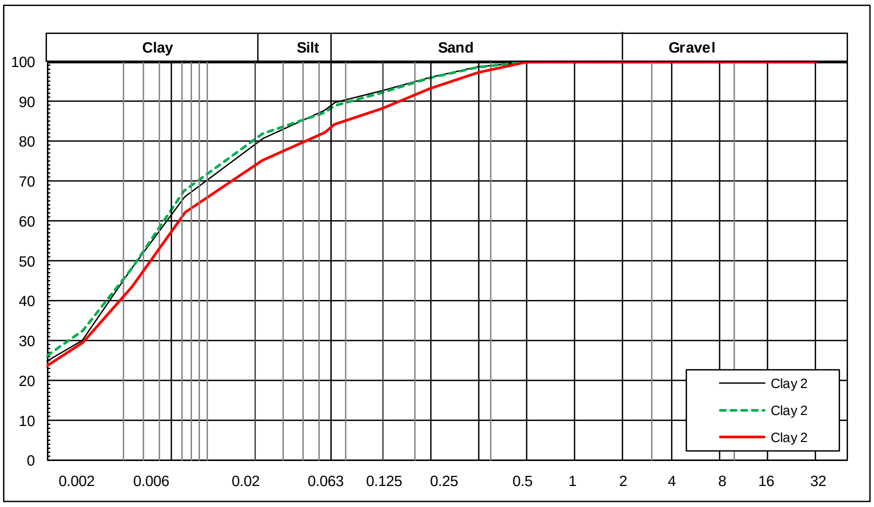

Fig. 14. Grain size distribution curves of Clay 2.

The tested liquids were chosen to correspond to the actual situation in the MSW landfill bottom. After the leachate tests, all the equipment had to be disinfected properly, and usually some part of the equipment was no longer usable due to the influence of leachate. No pH analysis was made for the ultra-pure water. Table 11 presents the MSW leachate and clean water contents. Ultra-clean water was manufactured for laboratory test purposes.

Table 11. MSW leachate and clean water contents.

Measured parameter	unit	MSW leachate***	(ultra) clean water
pH	-	6.5 **	
Electrical conductivity	(mS/m)	903 **	0.00005
Redox-potential	(mV)		
COD_{Mn}	(mg/l)	817 **	
Total P	(mg/l)	7 **	
Total N	(mg/l)	605**	
NO_2 and NO_3	(µg/l)	85 **	
NH_4–N	(mg/l)	490 **	
Chloride	(mg/l)	612 **	
Arsenic	(µg/l)	7 *	
Cadmium	(µg/l)	< 1 *	
Chromium	(µg/l)	322 *	
Soluble Chromium	(mg/l)		
Zinc	(mg/l)	1.46 *	
Lead	(µg/l)	< 5 *	
Mercury	(µg/l)	< 0.1 *	
Soluble Molybdenum	(mg/l)		
Nickel content	(mg/l)		
TOC content	(mg/l)		<0.00001
DOC content	(mg/l)		

Note. * Average of three analysis determinations, ** Average of six analysis determinations. *** The MSW leachate represents typical landfill leachate in Finland and the tests are made by the same leachate.

3.5 Hydraulic conductivity testing provisions

In this dissertation, the hydraulic conductivity of all the samples, with the exception of GCL, was measured with flexible wall and a backpressure permeameter according to ASTM D 5084 (2000) standard. The test properties, that is, pressures, varied depending on material and sample dimension, for example height. The back pressure was between 60 kPa and 128 kPa during the whole measurement and was increased up to 250 kPa. The front and cell pressures were changed as the experiment continued, front pressure ranging between 81 kPa and 360 kPa and cell pressure between 131 kPa and 375 kPa. At the end of the experiment, the effective stress was 30.0 kPa or 50.0 kPa. The hydraulic gradient ranged between 1 cm H_2O/cm and 30 cm H_2O/cm. During the saturation of the sample, the gradient was 1–7, and during the measurement, 10–30, 50 or 100. The saturation section of the experiment lasted 7–165 days and the measurement phase 13–307 days. The temperature was at a constant 22 °C in the laboratory during the test experiments.

The hydraulic conductivity of the swelling clay product (GCL) samples was measured with a solid wall permeameter according to ASTM D 5887 (1999) standard. Test samples were placed in the measuring cell inside a cylinder. During the test, the sample was under water pressure so that the solid wall squeezed the sample tightly, preventing potential edge flow. The samples were saturated with clean water, and after the saturation the samples were conducted with clean water or leachate.

The hydraulic conductivity was calculated based on the Darcy's law. Hydraulic conductivity was analysed from the samples by measuring the water volume that passed through the sample and experiment time. A total pressure of 550 kPa and a back pressure of 515 kPa were used during the measurements. The front pressure was increased to 530 kPa. The pressure differential was 15 kPa throughout the measurement process.

The swelling clay product hydraulic conductivity was determined for both clean water and leachate (ASTM D 5887, 1999). The test sample was produced by cutting a 0.3 m×0.3 m size out of the test sample. A 100 mm diameter sample was cut from the GCL piece, and water was led to the sample edges during the trimming. The averaged diameter of two and four height measurements was determined for the samples on a pusher scale, and the mass was measured with a laboratory scale. Figure 15 illustrates a typical flexible wall permeameter sample.

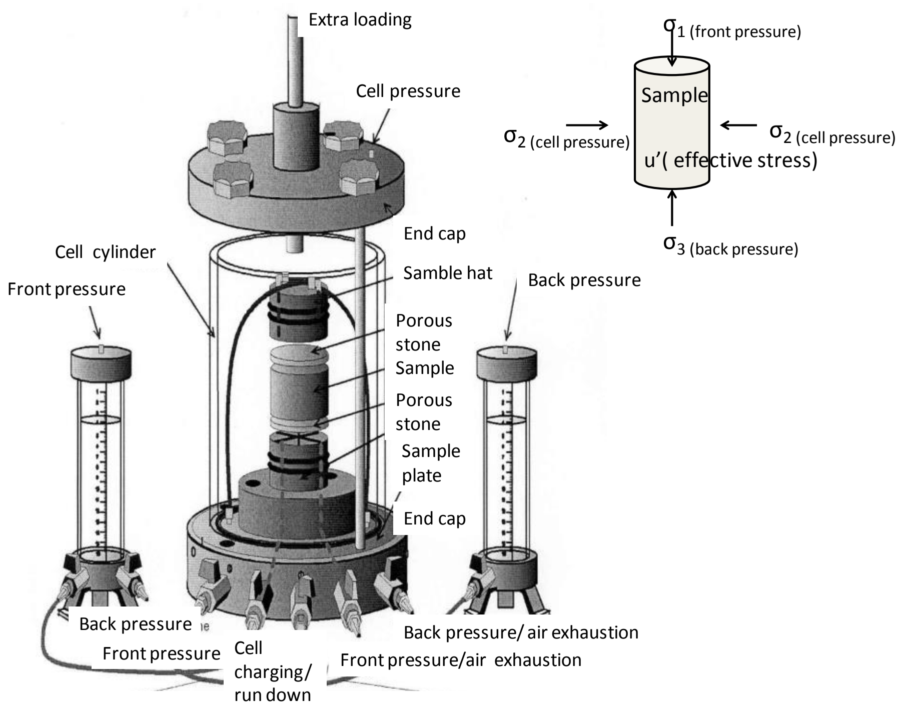

Fig. 15. The sample has been in a flexible wall permeameter under water pressure to squeeze rubber ring tightly against the sample, preventing potential edge flow (Modified after Virtanen, 2001).

The standard Proctor compaction test was made on the swelling clay material (GLO, 1985) to determine the moisture content (W is 9.8% sand mixed with polymer bentonite). The samples were made using the ICT equipment (Intensive compaction tester) at 92% of the greatest density of the maximum dry weight volume. Working pressure was 4.0 bar. The required density was reached after 15 rounds of work with the ICT equipment (Nordtest Build 427, 1994).

During the test, the sample was measured for test duration [d], saturated unit weight [γ_{sat}] (GLO, 1985), dry density [ρ_d] (GLO, 1985) and water content [w] (GLO, 1985) (which were determined before and after the hydraulic conductivity test), hydraulic conductivity [k] and hydraulic gradient [i]. The samples were measured for the degree of saturation [S_r] (ASTM C 566, 2004) before and after the test.

The natural clay samples were saturated using clean water or leachate, and after the saturation the samples were conducted with clean water or leachate. During the tests on the natural clay material (Clay), the following were measured: test duration [d], saturated unit weight [γ_{sat}] (GLO, 1985), dry density [ρ_d] (GLO, 1985) and water content [w] (GLO, 1985) (which were determined before and after the hydraulic conductivity test), hydraulic conductivity [k] and [$k_{leachate}$] and hydraulic gradient [i]. For the samples the degree of saturation was defined [S_r] (ASTM C 566, 2004) before and after the test. Clay samples were summarized in the standard Proctor compaction test in 3–5 layers, and compaction of the intermediate layers were scratched (GLO, 1985).

The diameter and height average over four measurements was measured for the samples on a pusher scale, and the weight was determined with a laboratory scale. The water content was measured from the compacted mass using oven drying (GLO, 1985).

The samples were saturated using clean water or leachate, and after the saturation the samples were conducted with clean water or leachate. Hydraulic conductivity was monitored for 357 days.

4 Results and discussion

This dissertation analyses the inadequacy of the environmental permits and risk assessments of 12 arbitrarily chosen modern landfills. The risk assessments the landfills were compared with a theory review, observing local circumstances and special features of the climate and ground. Also a risk assessment tool, the Structural Risk Assessment (SRA) method has been developed. The aim of the SRA method is to identify and minimise environmental pollution, assess the protection capability of landfill structures and the consequences of the chosen solution. In this dissertation, two environmental permit risk assessments performed with the SRA method have been verified with risk assessment methods typically applied in Finland.

4.1 Deficiencies in environmental permit risk assessment

In Finland, landfill risk assessment is currently based on the evaluation of environmental effects included in the environmental permit process. However, in the environmental effect evaluation should be included and defined issues like topography, hydrogeology, meteorology or tools computer-aided models. General instructions for risk assessment are included in the Landfill Directive and Government Decision, and these instructions have formed the conventions for their implementation within the field.

The EU Landfill Directive (1999) "*defines that in the case of deviations from the minimum structures defined in the directive, the applicant for the environmental permit has to be able to demonstrate based on risk analysing that the deviating structures will not cause problems for the environment and that their protection capability is corresponding with the minimum requirements of the directive*". Landfills could cause problems and pollute the environment in three main areas: the air, the lithosphere and the hydrosphere (Fig. 16). However, the

Directive leaves the responsibility for analysing the landfill risk assessment to the applicant without taking a stand on how to demonstrate it.

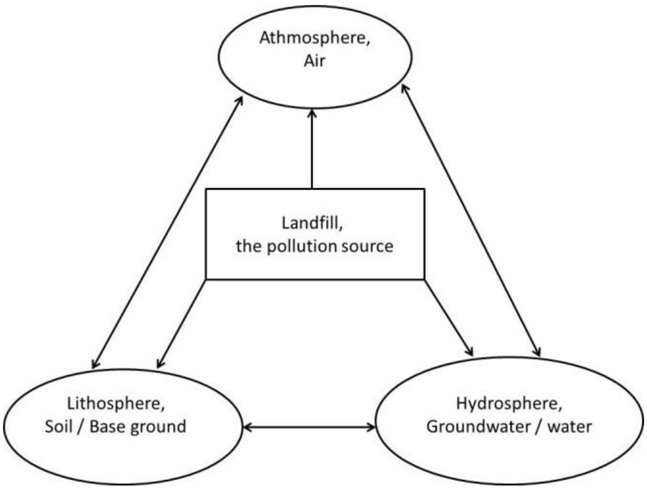

Fig. 16. Main areas where landfills could cause problems and pollute the environment. (Modified after Butt *et al.,* 2014 and Moriarty *et al.,* 1993).

The data collected from the environmental permits has been compiled into tables 11, 12, and 13 in which the central risk factors are compared with the environmental permit decisions' risk assessment. The landfills, which are all MSW landfills, are labelled from A to L and all the landfills are active. The targets of the comparison are the differences between the selected and the identified risk factors that have been defined in the theory (Butt *et al.* 2014) (Fig. 17). In addition, it will be examined how the risk assessment procedure is defined in the EC landfill directive as mentioned in General Requirements for all Classes of Landfills:

"The location of a landfill must take into consideration requirements relating to: (a) the distances from the boundary of the site to residential and recreation areas, waterways, water bodies and other agricultural or urban sites; (b) the existence of groundwater, coastal water or nature protection zones in the area; (c) the geological and hydrogeological conditions in the area; (d) the risk of flooding, subsidence, landslides or avalanches on the site; (e) the protection of the nature or cultural patrimony in the area."

Human delineated factors	**Environmental factors**	**Technical factors**
➢ **Site management**	➢ **Topography and Geology**	➢ **Risk quantification**
➢ **Human influence**	- Soil content	➢ **Migration assessment**
➢ **Natural environment**	- Soil fraction	- Diffusion
➢ **Built environment**	➢ **Hydrogeology**	- Sorption
➢ **Life time expanse**	- Gradient, content	- Advection
➢ **After care**	- Flux	➢ **Hazards identification**
	- Flux direction	- Contaminant permeability
	➢ **Meteorology**	- Protection demands
	- Rainfall amount	➢ **Uncertainly assessment**
	- Annual changes	- Freezing - thawing
		- Wetting - drying

Fig. 17. Impact of the identified factors on the landfill bottom structure risk assessment and life-cycle information management (Modified after Butt *et al.* 2014).

4.1.1 Human delineated factors related to risk assessment

The effects have been divided into six areas: the effect of humans, the natural and built environment, the structure's life-cycle, the requirements of the monitoring method from the environmental protection perspective, and aftercare demands. Table 12 includes the long-term effects of the existing landfill.

Table 12. Comparison of the factors of landfill environmental permits and the minimum requirements of structures in relation to existing environment, life-time expectancy, monitoring method and aftercare phase.

Examined item	A	B	C	D	E	F	G	H	I	J	K	L
Human influence	-	-	-	-	-	-	-	-	-	-	-	-
Natural environment	-	-	-	-	-	-	-	-	-	-	-	-
Built environment and structures	–	–	–	–	–	–	–	–	–	–	–	–
Life time expectancy	+	+	+	+	+	+	+	+	+	+	+	+
Monitoring method	M	M	M	M	M	M	M	M	M	M	M	M
Aftercare demands (years) a	30	30	30	30	30	30	30	30	30	30	30	30
and methods e.g. landfill mining	-	-	-	-	-	-	-	-	-	-	-	-

Note: + determined, − not determined; monitoring method: M manual.

Typically, the risk assessment for the environmental permits does not take into consideration the impact of the landfills on the contaminant load of the new building areas, such as residential areas. Additionally, the evaluation of an old landfills' impact on humans and the built environment is absent from the current risk assessment procedure. Similarly, the effects on new areas near the landfill, life-time expectancy and the monitoring method are inadequately determined.

In the life-time expectancy calculation of the risk assessment method, attention should be paid to the structure's capability to protect the environment from the landfill leachate. Also, the life-time expectancy was only 30 years after the sealing structures have been built. Nevertheless, landfills could influence the environment 50-300 years depending on waste prospects and new techniques. During that period, the built environment could expand and become adjacent to the landfill, which could cause e.g. gas emissions or groundwater pollution to the environment.

The monitoring method can produce real-time information on the water balance changes in the landfill area and on their impacts on landfill management. From the environmental protection perspective, the aim is to achieve a sustainable landfill environment optimising the structures' operation cost-effectively during the whole life cycle. Consequently, a monitoring method should become a requirement, yielding measurement results from the soil and groundwater.

Landfill mining as a method for aftercare will be adopted in Finland and has therefore been excluded from the consideration during the design and environmental permit phase. Similarly, all the other waste prospects or possibilities for reuse have been ignored. Typically, landfill aftercare was based on the assumption that the waste in the landfill is a permanent structure after the landfill is sealed.

Risk assessments have focused in great detail on human delineated factors for landfills. Typically, landfills have a negative influence to the environment, but waste embankment could have a positive one. Near the city center of Oulu in Northern Finland, an old landfill has been made a downhill skiing center.

4.1.2 Effect of environmental factors on risk assessment

Environmental factor source data is presented in Table 13. The source data is defined from soil, groundwater and meteorology. The comparison results demonstrate that in all landfills, the basic soil characteristics have been determined from soil samples, and the levels and amounts of groundwater and surface waters have been determined. In addition, the groundwater content has been determined, and in 9 cases of 12, also the groundwater flux direction. However, based on the literature, it is reasonable to argue that both the groundwater gradient and flux direction should be measured to obtain a more realistic idea of the landfill groundwater movements. Also, annual seasonal and rainfall changes have been ignored in all cases. In addition, the sample amounts used in the definitions are typically small, which affects the reliability of the evaluation results since small amounts of samples may result in human errors in the sample taking or analysis.

Table 13. Comparison of the environmental factors of landfill environmental permits and the minimum requirements of structures in relation to soil content, ground and surface water and meteorology.

Examined item	A	B	C	D	E	F	G	H	I	J	K	L
Soil content	+	+	+	+	+	+	+	+	+	+	+	+
Soil fraction	+	+	+	+	+	+	+	+	+	+	+	+
Soil background content	-	-	-	-	-	-	-	-	-	-	-	-
Ground and surface water level and amount	+	+	+	+	+	+	+	+	+	+	+	+
Groundwater content	+	+	+	+	+	+	+	+	+	+	+	+
Groundwater gradient	−	−	+	−	−	+	+	+	+	+	−	+
Groundwater flux	−	−	+	−	−	+	+	+	+	+	−	+
Groundwater flux direction	+	+	+	−	−	+	+	+	+	+	−	+
Meteorology	-	-	-	-	-	-	-	-	-	-	-	-

Note: + determined, − not determined

The results demonstrate that the background content of the soil has not been determined in any of the landfills and that all contaminant transport modelling is based on the hypothesis that advection is the dominant transport mechanism (Table 13). The purpose of the background content definition is to model the initial situation comprehensively enough to define the environmental protection requirements, contaminant concentration limits and monitoring procedures. In addition, the soil characteristics have not been determined in the landfill risk assessment broadly enough to include the soil in the environmental protection modelling as a part of the geological barrier.

4.1.3 Impact of technical factors on risk assessment

Technical factors have been divided into five categories: 1) analysis of the dominant transport mechanism, 2) leachate collection, 3) gas collection, 4) hazard identification and 5) uncertainty assessment (Table 14). The analysis of the dominant transport mechanism covers diffusion, sorption and advection, and the test amount used in their analysis. The hazard assessment studies whether the modelling is carried out with clean water or with a typical MSW landfill leachate content. Uncertainty assessment focuses on critical phenomena such as drying – wetting.

Table 14 presents a comparison of the landfill environmental permits with the requirements of the EU Landfill Directive and Government decision and analyses the reasons for the deviations from the minimum requirements. The analysis includes a comparison of the deviations from the minimum requirements based on the methods in which equivalent calculations and calculations including total transport mechanisms have been applied.

The general requirements for all classes of landfills in the EC Landfill Directive state the following:

"Where the geological barrier does not naturally meet the above conditions it can be completed artificially and reinforced by other means giving equivalent protection. An artificially established geological barrier should be no less than 0,5 metres thick."

The results show that typically in Finland the geological barrier is only a 0.3 m–0.5 m thick artificial extra layer, and the natural geological barrier, soil, has not been recognised, not even as a part of the geological barrier (Table 14). The inclusion of soil would have required a more extensive examination of its characteristics to act as a natural geological barrier.

The hydraulic conductivity of the structures met the minimum requirements in the environmental permits of all landfills. In equivalent calculations, the hydraulic conductivity had to be lowered to the level 6×10^{-10} m/s–7×10^{-10} m/s, so that the 0.5 m thick geological barrier would computationally meet the EU directive requirement.

In the geological barrier properties, deviations from the EU directive and Government decree are possible if it can be reliably demonstrated that the structure provides protection corresponding to that defined in the directive and Government decree. Laws and decrees do not take a stand on how the corresponding protection should be demonstrated and which factors should be taken into account, but the authority can make the decision case-specifically, considering the local conditions.

Table 14. Comparison of the technical factors of landfill environmental permits and the minimum requirements of risk quantification.

Examined item	A	B	C	D	E	F	G	H	I	J	K	L
Migration assessment												
-Transport mech. advection	+	+	+	+	+	+	+	+	+	+	+	+
-Transport mech. sorption	–	–	–	–	–	–	–	–	–	–	–	–
-Transport mech. diffusion	–	–	–	–	–	–	–	–	–	–	–	–
- Artificial layer thickness	*	*	***	**	**	**	**	***	**	**	**	**
Leachate collection	+	+	+	+	+	+	+	+	+	+	+	+
Gas collection	W	W	W	W	S	S	W	S	W	W	S	S
Hazards identification												
- Contaminant permeability	–	–	–	–	–	–	–	–	–	–	–	–
- Protection demands	–	–	–	–	–	–	–	–	–	–	–	–
Uncertainly assessment												
- Drying–wetting phenomenon	–	–	–	–	–	–	–	–	–	–	–	–
- Freezing–thawing phenomenon	–	–	–	–	–	–	–	–	–	–	–	–

Note: + determined, − not determined, * < 0.5 m, ** ≥ 0.5 m, *** ≥ 1.0 m, W gas collection is inside the waste filling, S gas collection layer is under the sealing structure.

In all analysed landfills, the thickness of the geological barrier is below the 1.0 m thickness defined in the EU directive, and in two landfills, even the additional layer is thinner than the 0.5 m minimum thickness of the artificial geological barrier. According to the EU directive and Government decree, this is absolutely allowable if it can be demonstrated that the structure is capable of protecting the environment correspondingly to the structures defined in the directive and Government decision (Table 14). According to Malusis et al. (2003), Katsumi et al. (2001) and Christensen et al. (1994), the total amount of mass transport should have been used as the dominant transport mechanism because the hydraulic conductivity k has been defined to be below 1×10^{-9} m/s for the material. Similarly, structural dimensioning calculated equivalently taking only advection into account is not relevant in these cases.

All the selected landfills have the required gas collection systems for waste filling or a gas collection layer on top of the waste before sealing. Similarly, leachate collection pipes or systems have been set up to collect leachate into the reservoir or directly into the treatment plant. Every modern MSW landfill in Finland has been designed the same protection structures to the waste filling areas, leachate reservoir or recycled waste treatment areas. Also, all the protection structures have been designed by equivalent calculations focusing on advection, and the tests have been conducted with clean water instead of leachate.

In all landfills, the equivalently calculated relation between the advection-based hydraulic conductivity and the layer thickness has been computationally determined to be the reason for the deviation of the geological barrier. In structures deviating from the requirements, the correspondence with the set requirements can be the most reliably demonstrated when the deviating structure is based, in addition to the total transport equation, also on the inclusion of the soil as a part of the geological barrier. The calculation of the structure's environmental protection capability according to the total transport equation is a more reliable calculation method than the equivalently calculated examination of advection because the factors affecting the contaminant transport in a dominant manner are taken into consideration more thoroughly.

The boundary condition for the investigation of the factors is that the protection structure is presumed to be a layer through which a minimal amount of fluids can pass. Also, the direction of flow is obviously vertical, and the horizontal flow in the protecting structure is presumed to be zero. Nevertheless, from the perspective of environmental protection, hydrodynamic dispersion is often a significant factor when estimating the migration of contaminant mechanisms beside hydraulic conductivity. These factors are generally not considered or included in the quality requirements for mineral protection structures.

In other words, the structure is in a passive phase, which implies that the artificial layer has protected the bottom structure and withheld contaminants until the landfill sealing structures are constructed. After this, the assumption is that the life expectancy of the artificial layer comes to its end. The results of this dissertation shows, the bottom structure of the landfill is not presumed to prevent contaminant migration through it since this is theoretically impossible. Contaminants penetrate or migrate through all protective layers, depending on the characteristics of the contaminant and migration mechanisms.

Inadequate identification of the transport mechanism affects the structural dimensioning and can provide an excessively optimistic idea of the capability of the geological barrier to protect the environment in the long run.

In addition, in Finland, the analysis of the drying–wetting and freezing–thawing phenomena has been ignored. In a landfill, protective, partially saturated bentonite structures are exposed to wetting–drying cycles as well as to changes caused by ion-exchange and thermal variation. The presumption is that the drainage layer and the waste filling prevent these phenomena. The structural dimensioning of landfills is rather complex, and the effect of the drying–wetting phenomenon on the dimensioning can be examined using Equation (18).

In all landfills, the risk analysis carried out for the environmental permit is so simplified that it cannot be used as a basis for determining whether the protection properties correspond with the minimum requirements of the EU directive. Based on these results, the assumption in the risk assessment has been that the artificial layer, e.g. the geomembrane, is completely secure although it has not been required in any environmental permit. In addition, in every landfill, it has been presumed that the geological barrier operates mainly in the passive phase of the structure's life-cycle, which based on the literature is not very probable, but the artificial layer will leach contaminants already during the active phase. Therefore, the estimation of the bottom structure life expectancy is absolutely impossible because of the lack of essential dimensioning information and because the structures can during their life-cycle cause unpredictable stress to the environment and groundwater.

4.1.4 Discussion on environmental permits' inadequacy

The risk assessments conducted for the environmental permits of the selected and existing 12 MSW landfills were compared based on the literature to the most important factors affecting environmental protection, which are the EC Landfill Directive and the Government decision. All 12 MSW landfills were permitted after the EC directive was issued. Of all the analysed landfills, two central factors arose that deviated from the requirements: the hydraulic conductivity and layer thickness of the mineral barrier. Their impact has been examined in the following sections from the perspectives of contaminant migration and factors affecting the hydraulic conductivity. The examined structures are based on the landfill bottom structures that have been realised according to the studied environmental permits.

In the present study, landfill management refers to the critical factors in the environmental safety of the features of the design and dimensions of a landfill. This affects the functioning of the bottom structure of the landfill as well as other phenomena that contribute significantly to the long-term functioning of the bottom structure and compliance with the level of protection described in the EC Landfill Directive. One of the most important criteria is the mineral liner's effectiveness in protecting the environment from contaminants when selecting a material for a landfill bottom structure. The required level of protection is often expressed in terms of hydraulic conductivity (environmental decisions between 2003 and 1997).

In this dissertation, four different types of structure options were examined. These four structures are 12 randomly selected corresponding landfill bottom structures. The calculation principals for horizontal equivalent hydraulic conductivity were observed by advection and calculated by Ogata-Bank's equation. The calculations assume that the structure can be represented by horizontal equivalent hydraulic conductivity and that the transport process is caused by advection.

The transit time calculations are based on studied hydraulic conductivity values so that the results can demonstrate the real situation in the landfill bottom. In these calculations, it was assumed that the harmful substance flux through the bottom layer is caused by the hydraulic gradient.

Figure 18 illustrates the horizontal equivalent calculation transit times based on advection. Structure 1 had a layer thickness of 1 m, and the other three structures had thicknesses of 0.3 m or 0.5 m. Based on the equivalent calculation, the layer thicknesses were directly related to the transit time. Structure 3 had the shortest transit time, which is due to it being the thinnest of the structures. The transit times of all the other structures were over 45 years based on advection equivalent calculation.

The same four different types of structure options were examined by emphasising diffusion as the dominant transport mechanism. The contaminant transport mechanisms of the first and second structure are expected to operate in the same way through the whole structure. The transit time calculations are based on the hydraulic conductivity values tested in this dissertation so that the results can demonstrate the real situation in the landfill bottom. In these calculations, it was assumed that the hydraulic and concentration gradient causes harmful substance flux through the bottom layer.

The structures 1 and 2 were equal in terms of material, soil and fluid properties, but they differ in thickness. The thickness of the structure 2 is only half of that of the structure 1.

In the structure 3, the advection is assumed to pause over the upper layer under the assumption that the structure is intact and the layer thickness is the same in the whole structure. The lower layer cannot get enough flux from the upper layer due to advection. The advection effect should be focused on only when the overhead structure is fully saturated and flux slowly leaks to the lower structure. This transit time was much longer compared to the diffusion transit time. The diffusion coefficient was based on literature where the polymerbentonite and silt-based materials were at the same level (Table 15) (Mitchell, 1993). The calculations were based on the assumption that the structure 3 is as one equivalent.

For the structure 4, dense asphalt, the primary transport mechanism was advection because the dense asphalt had a small diffusion coefficient. Based on this, the diffusion was the secondary transport mechanism (Sarkkila *et al.*, 2006). Below the dense asphalt layers, the silt-based layer was calculated to function like the structure 1. (Fig. 18)

The results of this study indicate that all the structure examinations emphasised that the landfill was in the passive phase. The geomembrane or artificial layer was unable to arrest the adverse material or was damaged. The access of harmful substances to the artificial geological barrier was caused by the hydraulic and concentration gradient. Sorption was assumed to be uniform in all barriers (R=1) and calculations were conducted with two concentration contents: C/C_0 = 0.1 and C/C_0 = 0.5.

The calculation results of transit times varied greatly in different structures (Table 15). The results show that the layer thickness is clearly related to the transit time. The layer 2 has thinned to half compare to the structure 1, and the transit time decreased from 11 years to 4.3 years when the concentration content C/C_0 was 0.1.

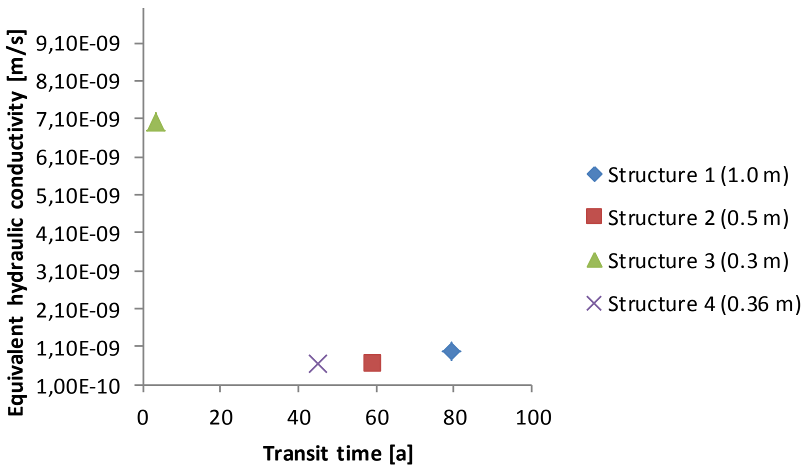

Fig. 18. Transit times of four different structures have been calculated based on the equivalent calculation caused by advection. Hydraulic conductivity values were the examined hydraulic conductivity values in this study.

The structure 3 transit time was less than one year (R is 1) (Table 15). Diffusion was a more dominating influence on the transit time compared to the low hydraulic conductivity of the thin upper structure. According to this dissertation and Katsumi et al. (2001), a low advection coefficient could not protect the lower layers.

In the structure 4, dense asphalt structure, the air void volume was influenced by hydraulic conductivity and total transit time. If the air void volume were 2.5% or less, it would reduce the hydraulic conductivity value 10–15 times. The theory supports this study in that the effects of the low diffusion coefficient of dense asphalt on harmful substance transport are based on hydrodynamic dispersion that will become the primary transport mechanism.

Table 15. Contaminant transit times of four different bottom structures in two concentrations based on hydraulic and concentration gradients. R is 1, D is 2×10^{-10} m^2/s, v and h depend on the liner type, H_f is 1.0 m.

Type of structure	1	2	3	4
T $c/co = 0.1$ [a]	11	4.3	< 1*	4.1
T $c/co = 0.5$ [a]	68	51	7.8	51

Note. * Transit time is less than a year.

In this study, 18 hydraulic conductivity tests were conducted on a clay liner (GCL), sand mixed with polymer bentonites and clay. The tests were carried out with both clean water and leachate. For the geosynthetic clay liner (GCL) and sand mixed with polymer bentonite, 11 hydraulic conductivity tests were carried out with clean water and seven with leachate.

The geosynthetic clay liner hydraulic conductivity was designed for level 1×10^{-11} m/s. The hydraulic conductivity varied between 1.7×10^{-11} and 9.5×10^{-12}m/s. Hydraulic conductivity was 1.7–1.9 times higher when leachate was used as a test liquid compared with clean water (Fig. 19).

The hydraulic conductivity of the sand mixed with polymer bentonite (SPB) ranged between 5.7×10^{-11} m/s and 5.9×10^{-11} m/s with clean water and with leachate ranged between 4.6×10^{-10} m/s and 5.1×10^{-10} m/s (Fig. 20). This indicates that the results of the samples vary very little. These samples were also created using the same material path as the clean water-tested samples.

Comparing the hydraulic conductivity and saturated unit weight results of the SPB, leachate increased the hydraulic conductivity eight to nine times more than clean water, but it did not influence the saturated unit weight (Fig. 20). The results show that the effect of leachate increased the hydraulic conductivity of the SPB.

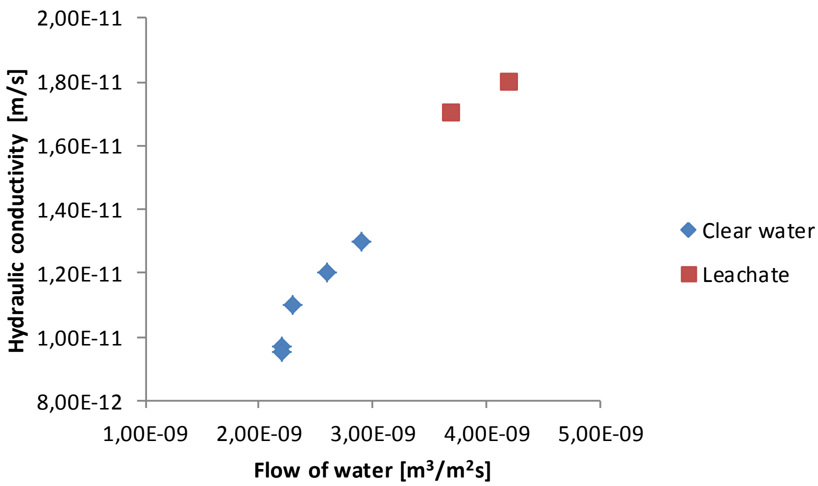

Fig. 19. Geosynthetic clay liner hydraulic conductivity results tested with clean water and leachate.

For natural clay, four hydraulic conductivity tests were conducted with clean water and three with leachate. The hydraulic conductivity of natural clay materials ranged between 1.3×10^{-10} and 1.9×10^{-10} m/s for clean water tests, and between 1.1×10^{-10} and 1.3×10^{-10} m/s for leachate (Fig. 21). Based on the results, the interpretation can be made that the effect of leachate did not increase the hydraulic conductivity of the samples during the 357-day test. Also Figure 21 supports this interpretation, but the amount of samples is not statistically relevant for the analysis.

The results of the study show that hydraulic conductivity increased for all tested materials when leachate was led to the permeameter as a substitute for clean water. The hydraulic conductivity in sand and till mixed with polymer bentonite materials was approximately 11 times higher for leachate compared with clean water. Guyonnet's et al. (2009) research supports the results of this dissertation, that is, when the test liquid was used as leachate or a leachate substitute for soil mixed with polymer bentonite material. The literature also supports the results of this study that the effect of leachate increases hydraulic conductivity when bentonite inside GCL is prehydratated via soil moisture. This is a typical saturation phenomenon in landfill bottom structures, providing that the geomembrane does not leak and was not damaged during installation.

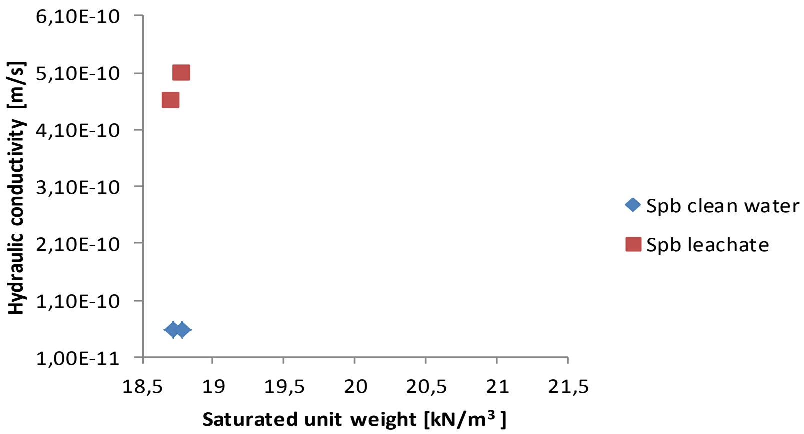

Fig. 20. Sand mixed with polymer bentonite hydraulic conductivity (m/s) results testing with clean water and leachate.

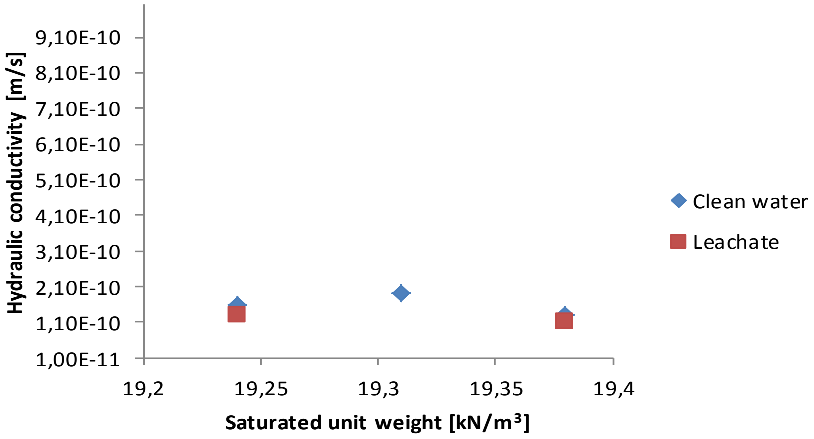

Fig. 21. Hydraulic conductivity of natural clay materials tested with clean water and leachate.

According to Ashmawy et al. (2002) polymer treatment is believed to render clay or soil non-reactive to many organic and inorganic chemicals. The literature supports this dissertation, indicating that polymer treatment is more beneficial in general if the clay is first saturated by water before coming in direct contact with leachate (Guyonnet *et al.,* 2009). Bentonites have a high swelling potential, resulting in polymer treatment being more advantageous, especially when low hydraulic conductivity is required.

According to Guyonnet et al. (2009), and in agreement with this study, the hydraulic conductivity of sand mixed with polymer bentonite increases when using leachate as the test liquid in hydraulic conductivity tests. The number of samples was not sufficient for statistical analysis, but the results for both tests showed similar results. On the other hand, the duration of these tests was 306 days, which increases the reliability of the results.

The literature is in line with the main finding of this study, that leachate increases the hydraulic conductivity of GCL (Lee *et al.,* 2005; Norotte *et al.,* 2004; Egloffstein, 2001; Jo *et al.,* 2001). According to the literature, the swelling capacity and hydraulic conductivity are related to the size of hydrated monovalent cations.

Based on the calculations in this dissertation, the examined structures may leak contaminants in low concentrations already after a couple of years during their passive phase, depending on the hydraulic water pressure, geological barrier permeability, sorption, decay, and groundwater surface levels and movements. The equation results of Acer and Haider (1990) support this study in maintaining that in the long term (50–300 years), landfills are a risk factor for the environment and the groundwater depending on the properties, thickness and hydrogeological conditions of the landfill's bottom layer structures. Harmful substances inside a landfill can flux to the lower layers during the first couple of decades, should it not be adequately protected. This result must be acknowledged when designing landfill structures, for example MSW landfills, in order to protect the surrounding environment even after the landfill has been closed.

This dissertation highlights how harmful substance transport is largely related to material thickness. The Dutch Waste Management Association also supports this view. Also experts Gronow (2008), Hansen (2008), Hjelmar (2008) and van der Sloot (2008), from three member states that *"were involved in drafting the Landfill Directive have confirmed that their intention was to regulate hydraulic conductivity instead of permeability, and hydraulic conductivity applies to the entire geological barrier rather than a specific artificial layer material property."*

The interpretation of the Dutch Waste Management Association on the Landfill Directive states the following: *"Article 3.4 tells us that member states do not have to abide by the values given in articles 3.2 and 3.3, if they carry out a risk assessment of a proposal for engineering and the assessment demonstrates there is no unacceptable risk to soil, groundwater or surface water. However, there is no derogation from article 3.1, so a geological barrier must always be present. If you do not intend to carry out a risk assessment then either you can profit from a natural geological (mineral) layer fulfilling the K values and thickness provided in the Directive, or you need to enhance the geological barrier by providing an additional artificial one of at least 0.5 meter thick to provide an attenuation capacity equivalent to those provided by the K values and thickness provided in the Directive. At the time 0.5 meter was considered a minimum thickness to guarantee the long lasting $K < 10^{-9}$ m/s for engineered artificial geological barriers."*

In order to meet the EC requirements, all transport mechanisms are to be taken into account in the risk assessments if the artificial geological barrier thickness is considered less than 0.5 meters and the hydraulic conductivity for any part of an artificial bottom layer is less than 1×10^{-9} m/s. Decreasing hydraulic conductivity does not result in reduced soil hydrodynamic dispersion. According to Rowe et al. (1995), diffusion is the dominant transport mechanism, and consequently decreasing hydraulic conductivity ($k \leq 1\times10^{-9}$ m/s) does not justify thinner layers. The results of the calculations show that leachate migration modelling by advection equation gives excessively optimistic results compared with Ogata-Bank's equation calculations.

Misinterpretations of the Landfill Directive are partially caused by the directive itself being somewhat unclear. The current Landfill Directive could be improved by specifying the maximum leachate infiltration. This type of addition would make the directive less ambiguous. The directive intends to regulate leachate infiltration by providing limit values for hydraulic conductivity, but fails to specify the maximum infiltration.

The Dutch Waste Management Association states that *"the intention to clearly describe and regulate a limited infiltration of leachate into soil and groundwater has not been fully achieved. Annex 1 paragraph 3 requires that if a geological barrier of less than 0.5 meter thickness is built, this can only be acceptable after a risk assessment that not only addresses the limitation of infiltration but also the attenuation capacity of the engineered construction. For some constructions diffusion is an issue to address as part of the risk assessment. Based on the different national regulations that are in place and the constructions that member states allow, it can be questioned whether this has been sufficiently understood."*

It could also be highlighted that current Finnish risk assessment practices do not take into account all of the crucial factors affecting environmental protection. For example, the waste filling bottom structure has been typically designed similarly to leachate reservoirs. However, the metrological conditions vary in leachate reservoir during the year and could cause e.g. freezing-thawing because part of the reservoir surface is open without any insulation layer.

Also, the hydraulic gradient could vary between the leachate reservoir and the waste filling structures. The protection structure in a reservoir could have a leachate head of several meters. In contrast, in the waste filling area, the leachate head is designed to be less than a half meter. These critical matters have not been taken into account in any of the 12 landfill designs and environmental permits.

In addition, from recycling and composting fields the landfilled waste will be moved after processing for further use, and the compost will no longer protect the field. The temperature inside the compost could rise during the composting process and fall after moving the processed compost away from the composting field. This could cause temperature and moisture content changes inside the bottom protection structures. Such landfill mining affects to protection structures have not been identified in any risk analyses in Finland.

4.2 Development of a landfill risk assessment method

Based on the inadequate results of landfill risk assessment, it can be stated necessary to introduce further developed landfill risk assessment methods. The Structural Risk Assessment (SRA) method has been developed in this dissertation. This method helps to define the risk factors, which have been identified in the designing phase and how their impact has been taken into account. The SRA method is a tool for a sustainable landfill designing process. The SRA method gives substantially more information for landfill management, risk management and risk identification compared to the risk assessments conducted in the environmental permit processes.

The SRA method has been verified in the environmental permit processes of two landfills. The first verified case was an innovative surface structure and the second one a hazardous waste bottom structure. These verified structures used local products, by-products or recycled materials as much as possible to supplant natural materials and offer a substitute to expensive materials, such as sand bentonite. Also, these materials have to fulfil environmental regulations; otherwise authorities may not issue an environmental permit.

4.2.1 Introduction of the Structural Risk Assessment method

The SRA method is useful to all landfill stakeholders: owners, designers, authorities and residents in the vicinity of the landfill. However, its most essential task involves design. The SRA method a tool for landfills risk assessment, which can be divided into three main categories (Fig. 22):

1) Human factors
2) Environmental factors
3) Technical factors.

In the first phase, the identification of the current operational environment begins by evaluating source information, i.e. by comparing monitoring information, which can be called reference data, and test results. The second phase concentrates on the impact of various mechanisms, local conditions and phenomena in the design phase in relation to the required environmental protection level. In addition, during the design process, the initial conditions in the current operational environment and their impact on the environmental protection requirements will be taken into account. In the third phase, the achieved total result of the design phase will be compared to the minimum environmental protection requirements that are based on laws and decrees. In the last phase, the planned landfill life-cycle length will be examined in relation to the monitoring system with the help of which the changes or damages in the structures can be predicted, along with their environmental impacts. Also the risk quantification safety factors and the ranking between the risk assessment factors have to be defined. The risk quantification safety factors are based on all heretofore made risk assessments, findings and environmental influence evaluations. Sections 4.2.2 and 4.2.3 present two examples of risk assessments conducted with the SRA method. The ranking between the risk assessments factors has been made by the landfill operator. The operator is responsible for the landfill and also the risk level.

Each category item has been further divided into sub-categories. The risk quantification can be identified more easily in the design phase with the help of the classification. In the risk analysis, the selected factors are examined based on the hypothesis that unidentified risk management does not exist but only identified risks could be accepted. The method emphasises the observation of every single factor, but examines analytically whether the factor is identified as a risk or not. Based on the reference data, it can be analysed whether the important factors have been identified in the design phase. Also, the identified factors impact the life-cycle information from the management perspective. In addition, the classification would have indicated how much and what type of essential additional information would have to be produced with the help of the evaluation procedure.

The SRA method begins with the identification of the source data out of each entity of affecting factors, based on which it can be confirmed in the designing phase that there is enough source data available for decision-making in the design process. The minimum amount of source data depends on the target: is the landfill new or old, will the risk assessment being full or partial? Typically, the source data would be more reliable if statistics could be calculated or the frequency of the data could be evaluated.

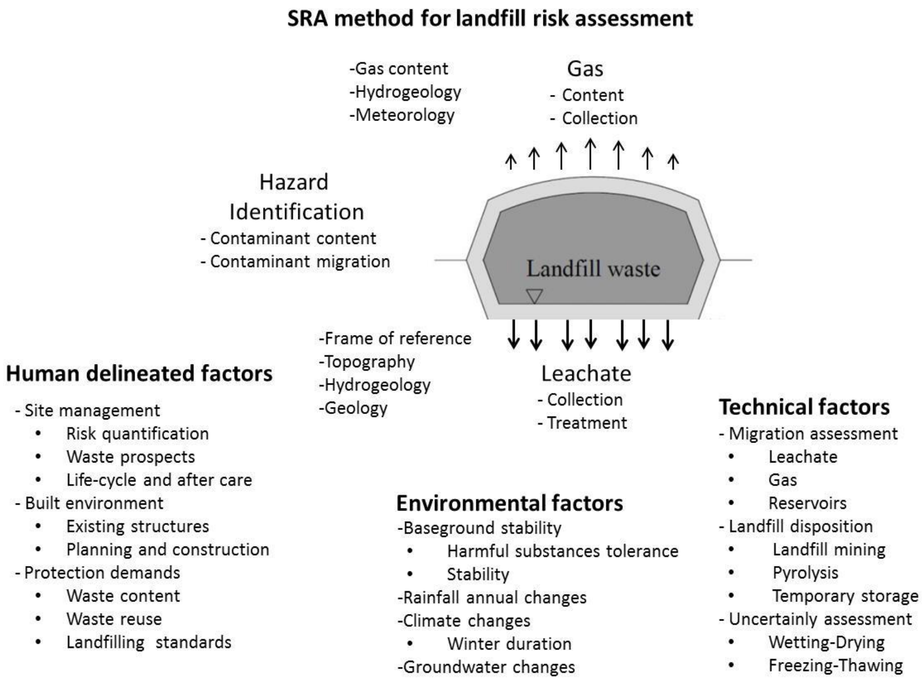

Fig. 22. Risk assessment according to the SRA method.

Unidentified risk factors will not necessarily cause risks during the whole landfill life-cycle, but the risk assessment is challenging if a risk has not been identified and its impacts have not been acknowledged in the design phase. The method does not guarantee risk-free environmental protection at the chosen risk level, because it is mainly a tool for the design phase. Errors during the construction phase may result in deviations in the selected life-cycle information management risk level and cause problems to the environment. For this reason, monitoring information should be utilised in the life-cycle information management after the design and construction phases.

Risk factors have to be ranked (P_x) after identifying them from the source data. P_x is the ranking value of an identified risk factor. Also, the landfill management or the designer has to determine the acceptable total risk level (R_{total}). The total risk level includes identified risk factors (R_{id}) and unidentified risk factors R_{ud}. The risk factor ranking depends on the structure protection demands, regulations, protection environment and human factors. This will lead to the situation that every risk assessment process has to be determined case by case. In addition, all the dominant risk factors are not included in regulations and environmental permits. Landfill management must identify the risk level (R_{ud}) of these unidentified risk factors.

The total risk level calculation is shown in Equation 18:

$$R_{total} = \sum_{x=1}^{n} R_{id} + \sum_{x=1}^{n} R_{ud} \tag{18}$$

The identified risk level (Ri_d) can be determined by using Equation 19:

$$R_{id} = \sum_{x=1}^{n} P_x * Q_x \tag{19}$$

Q_x is the probability coefficient of the identified risk.

The unidentified risk level (R_{ud}) can be determined by using Equation 20:

$$R_{ud} = \sum_{x=1}^{n} P_x * \beta_x \tag{20}$$

β is the probability coefficient of the unidentified risk.

The probability coefficient Q_x of identified risk factors is always higher than demanded in regulations or environmental permits. The probability coefficient of unidentified risk factors could vary because increasing the reliability of the other risk factors could compensate for it. For example, the drainage layer risk probability is due to the fact that the soil is not being able to protect environment by increasing contaminant migration. Furthermore, the total risk level has to fulfil the demanded risk level. Figure 23 presents an example of a landfill risk assessment with the SRA method.

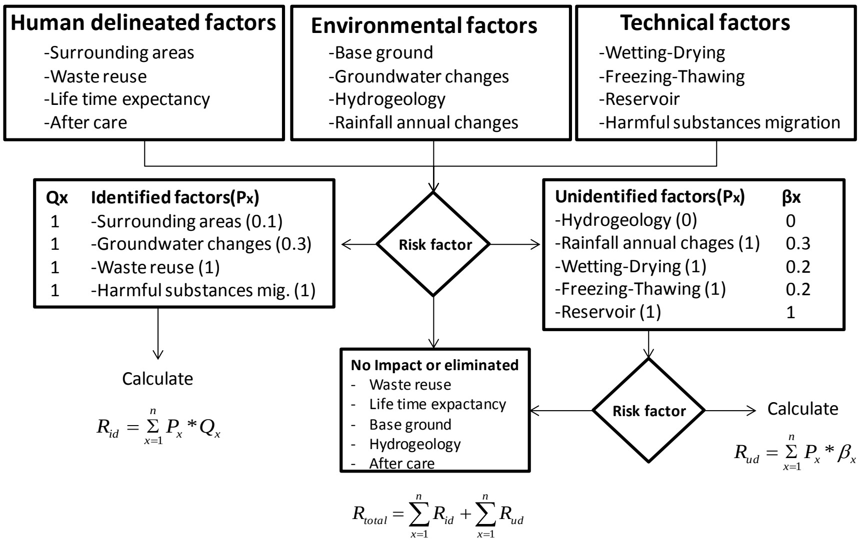

Fig. 23. An example of landfill risk assessment with the SRA method.

Figure 24 presents an example flowchart of a landfill bottom structure dimensioning using the SRA method. Structural dimensioning was begun by examining whether the selected structure is in accordance with the requirements. If the structure is in accordance with the requirements, the SRA method does not cause additional data evaluation. In the risk analysis, the structure will be analysed to determine the dominant transport factor or the factors based on which the structural dimensions are defined, and if necessary, their calculation principles will be defined in a laboratory. Laboratory determinations are always based on material-specific requirements and the burdening contaminant, and thus, the risk analysis corresponds with the realistic situation. Dimensions can be altered when needed with the aim that the structure will meet the landfill bottom structure requirements. The dominant factors should also be ranked e.g. depending on the natural environmental demands if the structure is close to sensitive areas.

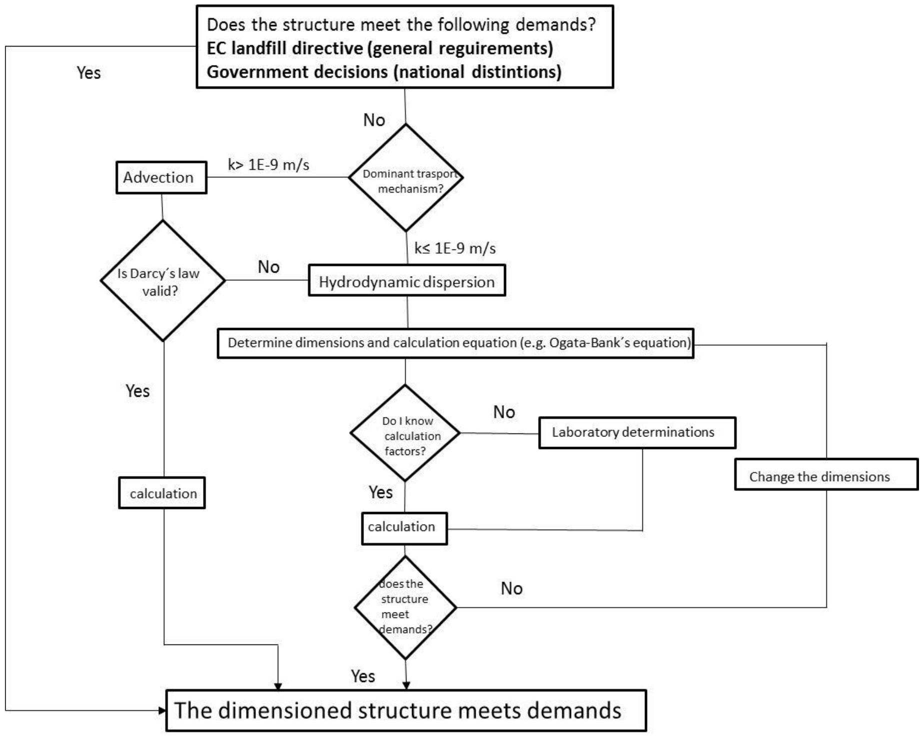

Fig. 24. Flowchart showing the landfill bottom structure risk analysis with the SRA method.

The utilisation of the SRA method as a tool for landfill life-cycle management improves the current risk management practices essentially in the landfill environmental permit phase. In the SRA method, significantly more data must be produced for decision-making processes to support the current, rather simplified risk assessment. Thus, from the environmental protection perspective, more important data will be available on the landfill environmental impacts to support the landfill management.

The construction of geoenvironmental technology has lacked requirements for the functionality or estimations of structure life expectancy in relation to materials or structures. The technical requirements for structures are based on requirements for functionality and not on the interaction of causal relationships.

This is further emphasised in structures in which the effects of technological innovation and the application thereof have been very limited.

4.2.2 Landfill surface structure risk assessment with the SRA method

The objective of the risk assessment process was to find a solution for a lightweight structure that fulfils the environmental protection demands and is cost-effective. The incinerator has been activated recently and the surface structure should be possible to open for landfill mining purposes. The life expectancy exceeded 50 years because of the incinerator, landfill mining and the separation of waste. Landfill gases generate undesired odours in the vicinity of the landfill, and this could be avoided after the surface structures have been installed. Near the landfill site is a moraine area that has been tested during the landfill site construction, and the material could be used in the sealing structures. According to measurements by the Finnish Meteorological institute, the average drainage capacity is 548 mm/year. The temperature is below 0 degrees Celsius for 158 days a year. Figure 25 represents the structural dimensioning of the surface structure.

The study was initiated with laboratory tests, calculations of gas emissions and water migration through porous media in a partly saturated situation. The risk quantification for uncertain assessment calculations had to be at least 1.5. The exposure assessment demands were set to control the water migration and electrical conductivity measurement results outside the landfill site in real time. The significance of electrical conductivity measurements is to control and secure that surface structures do not affect the surrounding environment. Electrical conductivity was chosen because the measurements easily indicate if harmful substances increase in the landfill areas, and comprehensive advance material is available as a reference (Fig. 26) (Grellier *et al.,* 2006).

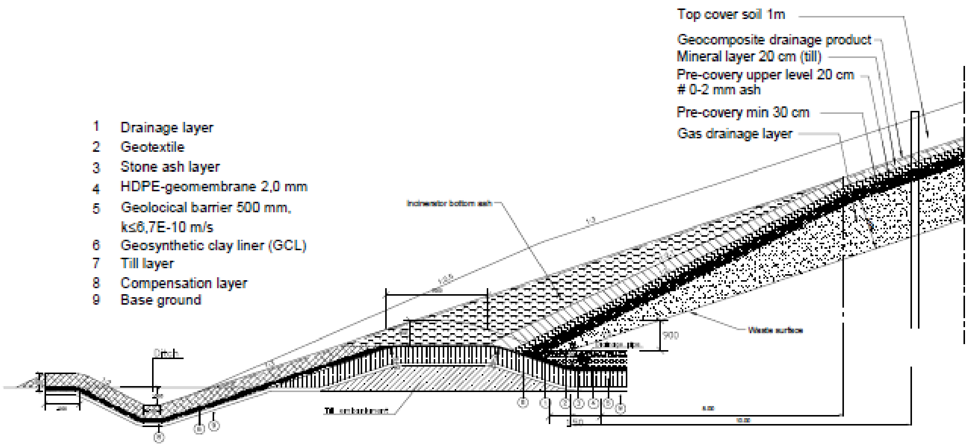

Fig. 25. The structural dimensioning of the surface structure.

The hydrogeological properties, drying and shrinkage factors, insulation properties, frost resistance and mechanical properties of till were tested in laboratory conditions. The calculations of mineral layer hydraulic properties were simulated according to the Van Genuchten equation for partially saturated soils, the drainage system capacity according to the unit gradient method and the bearing capacity with the general shear failure method. A drainage core was included in the gas collection layer.

Table 16 illustrates the laboratory tests. Tables 17 and 18 include mineral layer calculation results of dimensioning and safety factors. The dimensioning is calculated including and excluding safety factors. According to calculations, the environmental protection capacity of a mineral layer with a thickness of 0.09 meters could meet Government recommendation if fully saturated. Laboratory tests substantiate that the mineral layer could not reach full saturation in any meteorology condition. Consequently, partial saturation is a relevant calculation principal. Tammirinne et al. (2004) have made a model for surface structure drainage, but this model assumes that the temperature above 0 degrees Celsius throughout the year. The assumption is meteorologically irrelevant, but the model has been modified to be more realistic and relate to weather conditions in the landfill area.

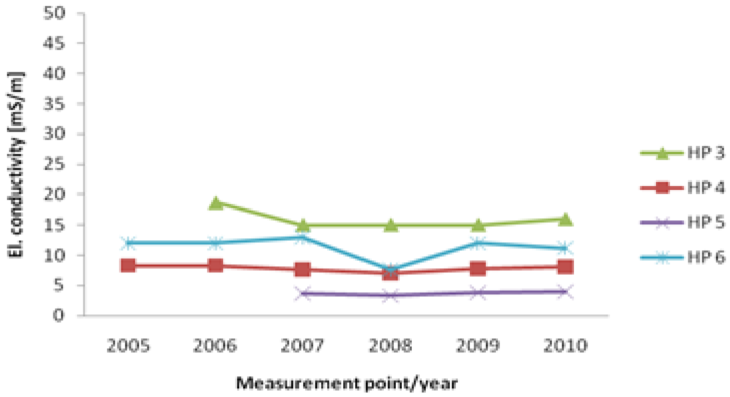

Fig. 26. Electrical conductivity results between 2005 and 2010 in four measurement points (HP means measurement point).

Table 16. Till's laboratory tests results.

Property, [unit]	test 1	test 2	test 3
Hydraulic conductivity [m/s]	2.2 E-8	1.7 E-8	6.6E-9
Effective stress [kPa]	26	40	20
Water content [w %]	5.2*	5.0*	9.5*
Dry unit weight [kN/m^3]	20.11	20.49	21.26
Volume shrinkage [%]	0	0	0
Compression strength [kPa]	73	75	72
Shearing strength [kPa]	36.5	34	37.2

Note. * Till optimum water content is 2.6%.

The layer thickness has been increased form 0.09 meters to 0.2 meters because of construction and dimensioning problems. A very thin-layer construction over the angle slope would be challenging to excavate. Under Finnish regulations, the maximum fraction size of soil material, such as till, has to be at least five times smaller than the constructed structure thickness. The maximum fraction size of till is 32 mm. Based on these factors, the 32 mm fraction size is a good reason for the thickness of the layer. According to frost resistant tests, water sources could not determine the freezing-thawing phenomenon for the mineral layer structure. The calculated total risk factor value was 2.43. The total risk factor value is the sum of the identified risk factors, and the sum of the ranking of the risk factors of this structure is (1). Table 19 represents the calculation details of the total risk factor.

Table 17. Risk acceptance calculation results for a surface structure mineral layer.

Property,[unit]	Including safety factor	Maximum safety factor	Risk factor
Hydraulic conductivity[m/s], (partly saturated conductivity)	$4.6 \cdot 10^{-11}$	$1.3 \cdot 10^{-11}$	3.5
Effective stress [kPa]	26	13.7	1.9
Saturation degree [%]	40	30…50	---
Mineral layer thickness [m]	0.2	0.09	2.2

Table 18. Risk acceptance calculation results of the surface structure drainage layer.

Property,[unit]	Including risk factor	Excluding risk factor	Risk factor
Flow rate [l/m^2s]	$2.38\ 10^{-1}$	$7.26 \cdot 10^{-2}$	3.3
Effective stress [kPa]	20	13.3	1.5
Gradient [-]	0.25	0.14	1.8

Table 19. The calculation details of the total risk factor.

Substance	Identified risk factor	Ranking factor	Risk factor
Hydraulic conductivity	3.5	0.2	0.70
Effective stress	1.9	0.2	0.38
Mineral layer thickness	2.2	0.1	0.22
Flow rate	3.3	0.18	0.59
Effective stress	1.5	0.12	0.18
Gradient	1.8	0.2	0.36
Total Risk Factor [Σ]			**2.43***

Note: * Total risk factor of the SRA method.

The freezing-thawing phenomena have been focused in the risk assessment process. The till based material has been tested by the freezing-thawing, the maximum deformation and the maximum shrinkage tests. Any of these test results did not indicate that freezing-thawing cycles could cause changes on the till material during the life cycle. The phenomena were not determining the factor of used till based material and have been ignored of final risk assessment.

The cost-efficiency of the structure compared with Government recommendations is 14% more affordability per hectare. The cost-efficiency will increase if the structure is opened in the future, because the structures are thinner and easier to excavate.

The collected data was based on at least triplicate tests or test series. Also during the experiments the changes emulate the circumstances as far as possible to be able to evaluate the reliability of the risk assessment. The test result dispersion has been low, and none of the test results have been rejected e.g. based on the Dixon test. This testifies to the high reliability and quality of the test results. In consequence, risk assessments on a landfill surface structure could be based on a reliable source of information and are an important issue for landfill management as well as environmental regulators.

In the SRA analysis of mineral structures, the functional role of the protective structures is slightly altered compared to the current concept. At the moment, the focus of the structural dimensioning is on hydraulic conductivity and its computational demonstration according to equivalent calculations. In the SRA method, the factors related to contaminant migration through the unsaturated layer are identified and recognised more comprehensively. Additionally, the mutual dependence and impacts of the identified mechanisms related to migration are analysed using the total transport equation. Therefore, the structure life expectancy can be calculated more reliably.

4.2.3 Risk assessment of the bottom structure of a hazardous landfill with the SRA method

The objective of the risk assessment process was to find a solution for a sustainable structure that fulfils the environmental protection demands and the life-cycle expectancy exceeded 50 years. The structural dimensioning was partly based on meteorological background information. Also, the result of the absorption tests was essential.

The average drainage capacity, founded on the precipitation statistics of the Finnish Meteorological Institute, is 513 mm/year. By 2012, the maximum precipitation during the preceding 40 years was 900.5 mm/year. The air temperature was below 0 degrees Celsius for 108 days and the soil temperature 217 days. Figure 27 represents the structural dimensioning of the innovative bottom structure.

The background research included laboratory tests, a site investigation by boring drill holes into the solid base rock, taking samples from the drill holes, and absorption tests of water sludge made from existing landfill leachate. The objective of the absorption experiments was to examine the absorption capacity of water sludge and leachate, the impact of leachate on electrical conductivity, pH, and the chromium, molybdenum and nickel liquid-solid content. The chosen parameters are based on the heretofore made research results and the estimation of the most critical components that could cause impact to the groundwater and environment. The risk quantification for uncertain assessment calculation had to be a minimum of 1.5 or at the same level as the frame of reference values. The exposure assessment demands were to control the electrical conductivity measurement results outside the landfill site in real time.

One of the primary principles was to make sure that the HDPE geomembrane does not include any holes after installation. There are several methods to confirm the integrity of the membrane. In this case, the electrical tension difference method has been used. The integrity of the membrane was crucial for structural dimensioning.

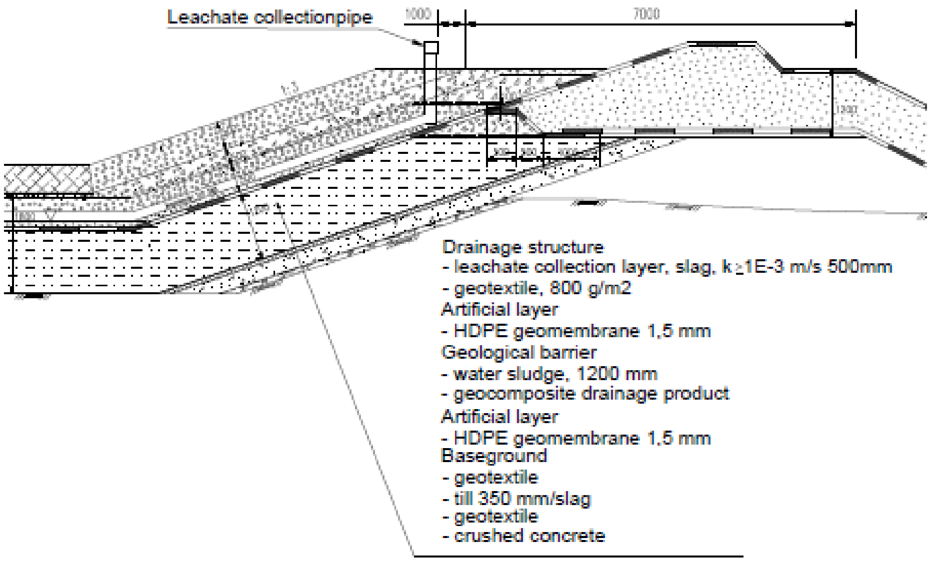

Fig. 27. The structural dimensioning of the innovative bottom structure.

Mineral liner dimensioning was based on the analysis by Rowe et al. (2004) of contaminant transit through the porous media. The laboratory experiments of leachate absorption with by-products were used as the source information of dimensioning. Figure 28 represents the chromium solubility test results.

The results of the absorption test indicate that water sludge did not have a harmful effect on the leachate, and similarly, no particles dissolved from the water sludge to the leachate. The HDPE geomembrane will be installed above and under the water sludge layer, also called the geological barrier. A double layer geomembrane could secure the migration of harmful substances of water sludge to the soil. The drainage layer and the leachate will be collected separately when the HDPE geomembrane is installed. This enables the control of the leachate source and content in both layers. Leachate is conducted to the reservoir and further to water treatment.

The structural dimensions of the geological barrier are shown in Table 20. The barrier is 1.2 meters thick, and the calculated risk level is 1.78. Hydraulic conductivity determined by 100 kPa effective stress, which is 1/6 of the final situation when the landfilling is finished. The porosity could be much lower in the last few years of landfilling because the effective stress increases and causes loading. The hydraulic conductivity will decrease impact on porosity and also affect the transport equation.

The drainage system capacity has been dimensioned according to the maximum rain capacity of the year. The drainage system dimensioning is only estimation, because the first part of the landfill had not reached its maximum height. In addition, the water was assumed to affect the drainage system during the landfill's active phase. Also, the amount of leachate collected from the landfill served as the basis for calculations for the drainage capacity dimensioning. The risk level was determined to be 1.3.

The uncertainty of the risk assessment has been estimated by determining the content of the drilled ground samples. The sample test results have been compared with the Government decree limit values and natural values (VNa 214/2007). Table 21 shows the reference values for soil.

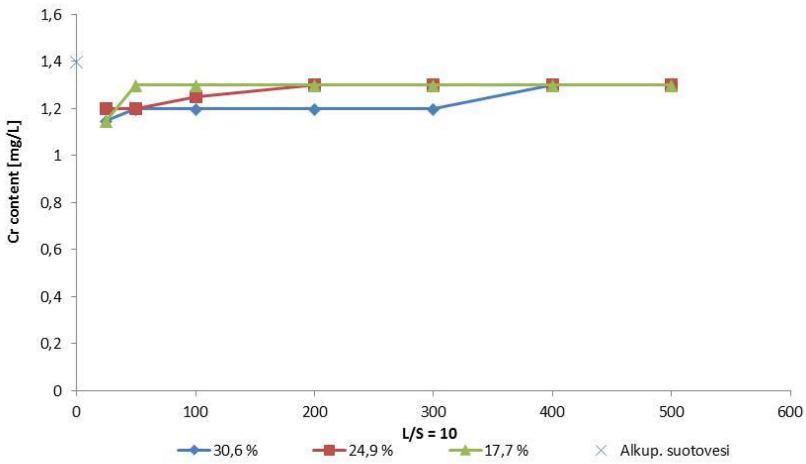

Fig. 28. The chromium solubility test results with the leachate (alkup. suotovesi means leachate in English).

The soil test results are clearly within the reference values. According to the results, the existing soil has not been polluted by the effects of the mill. The landfill structures encounter problems, if the values rise after landfilling.

Table 20. Geological barrier's structural dimensions, calculation coefficients, porosity and contaminant transport times for three layer thicknesses.

C/C_0 =0.1 [Years]	Layer thickness [m]	Hydraulic conductivity [m/s] *	Hydraulic gradient [-]	Diffusion coefficient [m^2/s] **	R [-]	Porosity (-)
63	1.0	$4.5 \cdot 10^{-10}$	1.5	$2.0 \cdot 10^{-10}$	1	0.4
89	1.2	$4.5 \cdot 10^{-10}$	1.42	$2.0 \cdot 10^{-10}$	1	0.4
118	1.4	$4.5 \cdot 10^{-10}$	1.33	$2.0 \cdot 10^{-10}$	1	0.4

Note: * The hydraulic conductivity value was determined by 100 kPa effective stress that is 1/6 of the final situation when the landfilling is finished. ** The diffusion coefficient is based on calculations by Katsumi et al. (2001).

Table 21. The examples test results compared with Government decree limit values and natural values.

Substance	Unit	Frame of reference	Limit value	Natural content	Lower reference value	Upper reference value
Chromium (Cr)	mg/kg	30*(25-48)	100	31(6-170)	200	300
Molybdenum Mo)	mg/kg	<1*	-	<1	<1	<1
Nickel (Ni)	mg/kg	10*(8-13)	50	17(3-100)	100	150
pH	-	7*(6.2-7.8)	-	-	-	-

Note: * Outside of the bracket is the mean value of 12 measurements and inside of the bracket are the lowest and the highest determined values.

The total risk value is the sum of the identified and unidentified risk factors and the sum of the ranking of the risk factors of this structure is (1). The total risk factor (R_{total}) was 1.79 (Table 22). The cost-efficiency of the structure compared with the Government decision is 43% more affordable per hectare. The total cost-efficiency is influenced most by the possibility to use by-products as a part of the structure. Table 22 represents the calculation details of the total risk factor.

Table 22. The calculation details of the total risk factor.

Substance	Identified risk factor	Ranking factor	Risk factor
Hydraulic conductivity	4.5	0.08	0.36
Mineral layer thickness	1.2	0.15	0.18
Drainage capacity	1.3	0.3	0.39
Chromium	2.1	0.22	0.46
Molybdenum	1	0.2	0.20
Nickel	3.8	0.05	0.19
Total Risk Factor [Σ]			**1,78***

Note: * Total risk factor of the SRA method.

Typically in environmental protection structures, attention is paid mainly to the hydraulic conductivity of the mineral layer. The landfill structure analysis should take into account the leachate management, and in leakage situations the leachate content, human-related factors and waste prospects in the risk assessment. Essential factors for the securing of the landfill barrier structures' operation are the functionality of the drainage layer and the intactness and life-expectancy of the artificial barrier. If it is possible to conduct the leachate from the bottom structure through the drainage layer into treatment during the whole life-cycle, the hydraulic gradient to the bottom structure is not significant. This reduces the leachate stress on the mineral layer including all transport mechanisms. From this example, the effect of the hydraulic gradient is eliminated because the artificial layer will be controlled to avoid damage after installation. Therefore, during the active phase the artificial and drainage structure will eliminate the effect of the hydraulic gradient.

Factors related to material transport and retention and contaminant migration have been discussed above. Based on them, it can be concluded that an unambiguous correspondence between the materials and structures is not easy to determine. The definition of reference materials or structures is also difficult because some materials may be better than others in some respect. Instead, materials and structures can be compared in relation to functional requirements. Dominant factors may in some applications be the amount of penetrating water and in some applications the penetrating contaminants or their concentrations or the time needed for their accomplishment. In structural analysis, the identification of the dominant factors and the setting of boundary conditions are essential and significant. According to the results of this thesis, the boundary condition setting has to be based on landfill-specific requirements, which may vary significantly depending on e.g. the location of the landfill, hydrogeology, meteorology and topography.

The limit conditions stated in the EC Landfill Directive should be developed further by means of structure life expectancy calculations, which can be implemented using various methods, such as the conversion factor method, statistical measuring and limit state of fatigue measurements, and the risk assessment safety factor. Although all of the methods mentioned above can be employed when determining the serviceable life of geosynthetic structures, the SRA method is the most suitable for materials like these, that is, industrially by-produced materials for which the manufacturer can provide a serviceable life estimate for comparison. The SRA method is much lighter risk assessment tool comparing e.g. with U.K. Environmental Agency Hydrogeological Risk Assessments for Landfill. Statistical measurements support the SRA method for measuring structure dimensions, since the measuring equations are presented as a fatigue parameter and a response parameter so that the parameters are time-dependent. One example of this is the contaminant migration equation in which a concentration differential forms as a function of time due to the effects of fatigue caused by different loads.

4.2.4 Discussion on risk assessment with the SRA method

Landfill management risk assessments were carried out with the SRA method based on a relatively wide range of information, but is still much lighter method comparing with e.g. U.K. EPA method. The risk quantification results include real-time measurements on landfill surface and sides, and also a total safety factor. Waste will play a major role in the future, because humans are going to need more and more materials for reuse from landfills. Landfill mining or pyrolysis could be common techniques already in the near future, and landfills as a material source should be excluded from the risk assessment process.

In both examples, the SRA method proved profitable to the landfill owners. An inadequate risk assessment provides false information of the possible risks. Inconsistency in risk assessments could mislead authorities and landfill owners in decision-making. It is essential to take into account all possible risk factors and also statistical descriptions like safety factors, the most likely values and natural values. Uncertainty assessment and significance assessment help to avoid inconsistency in risk assessment in the future.

Risk assessment can be useful in the creation of effective and economic structures. Cost-efficient solutions could equally protect the environment without expensive materials. Also local materials and by-products can be used in effectually indicating their properties and focusing sufficient safety factors.

In structure modelling, presumptions have to be formulated before the model building, which may result in differences between the real situation and the presumed situation. Migration through structures using various transport mechanisms is as a mathematical model a known and controllable entity, but the factors in the laboratory environment do not simulate the environment in all cases. Because of these factors the SRA method, and thus, the functional properties of structures could be evaluated more extensively. The design process should be extended to risk analysis. In addition, the mutual dependence of the functional factors should be identified more extensively, and thus, the prognosis for the contamination caused by and the structural life expectancy of the landfill would be possible computationally using various calculation methods. Contaminant migration modelling by equivalent calculations is quite imperfect in current risk analysis because the analysis focuses only on advection.

Based on these cases, the SRA method was proved to be an applicable method for the partial or full risk assessment of landfills. Both of these risks assessments have accepted by the local authorities and the structures were constructed in 2013 and 2014. Also authorities were in favour of the development of risk assessments of local landfills and waste areas, taking advantage of by-products, for instance.

5 Conclusions

5.1 Answering the research questions of the thesis

The overall aim of this research was to provide further information on how the landfill risk assessment process has to be focused according to human-related, environmental and technical factors while evaluating the possible risk assessment to the environment (EC 31, 1999). The objective of this thesis was to find answers to the research questions:

i) What are the most significant deficiencies of the present risk analysis practices in Finland?

ii) How should the risk assessment process in the environmental permits and designs of landfills be developed to ensure landfill sustainability?

i)

The technical requirements of the present landfill protection structures cannot achieve sustainable protection by focusing on the landfill life-cycle risk assessment information. The most essential problems in the design period are the unidentified and unrecognised risk factors and their effects on the landfill's environmental protection capability. The most significant factors are the inadequacies in the contaminant transport modelling and in the identification of the factors affecting it. Another essential factor is the identification of the factors affecting the environmental protection structures' capability and their life-cycle. In addition, the impacts of the existing structures or infrastructure on landfills have not been identified.

The comparison of environmental permits with the SRA method factors demonstrated significant deficiencies in the identification of risks in all three main categories: human-related, environmental and technical factors. The most important unidentified factors were the identification of the dominating transport mechanism for harmful substances and its impact in relation to the environmental protection capability of thinner structures. The life-cycle information management problems will be discussed case by case in more detail in the following.

The main transport mechanisms influencing the contaminant transit time through the artificial mineral bottom layer in landfills include *advection* and *hydrodynamic dispersion*. This is evident both in the literature and in the results of this dissertation (Kamon, 2002; Kamon, 2005; Katsumi, 2001; Benson, 1999; Acer & Haider, 1990; Baer & Palmer, 1972).

The transport time of contaminant materials due to advection diminishes substantially when the hydraulic conductivity is less than 1×10^{-9} m/s, and **hydrodynamic dispersion becomes the dominant transport mechanism** (Rowe *et al.,* 1988). The structural thickness significantly influences the impact of hydrodynamic dispersion in contaminant transport through the landfill bottom layer (Smith *et al.,* 2004; Bell *et al.,* 2002; Lo *et al.,* 1999; Barone *et al.,* 1992).

In addition to the advection and layer thickness, one must acknowledge the impact of diffusion. Merely increasing the layer thickness is not sufficient to eliminate diffusion as the concentration gradient initiates diffusion. Consequently, the diffusion coefficient for the structure should be small (Katsumi *et al.,* 2001).

According to the calculations conducted in this dissertation, analysing contaminant transit purely based on advection leads to overly optimistic results. **According to Katsumi et al. (2001), hydrodynamic dispersion has a central role in evaluating the environmental impacts of landfills. The EC directive 31/1999 and the environmental permits granted by Finnish authorities do not, however, acknowledge the impact of hydrodynamic dispersion when guiding the design of landfill bottom layers (Environmental decision registry, 2010). Consequently, the true environmental impacts need to be assessed case by case.**

Based on the literature and the results of this study, it is practically impossible to build a structure that would be capable of fully protecting the environment from leakages in the long term. The bottom structures should be designed to protect groundwater and landfill surroundings at least for 50–100 years, like Huber-Humer (2009), Cossu (2003) and Katsumi (2001) have suggested, supporting the results of this dissertation. Risk assessment typically has to be delimited to a certain time-frame like the examples in section 4.2. The 50-year time-frame is so long that it is impossible to evaluate the changes, e.g. waste prospects.

The passive phase of MSW landfills is typically 30–50 years, or one human generation, after building the surface structure (Cossu *et al.,* 1997). In bioreactor type MSW landfills, the passive phase is 50–100 years or longer (Rinkinen *et al.,* 2009). During the passive phase, excessive water separates from waste, penetrates through the drainage structure and is fed into water treatment. As a result, the hydraulic gradient is formed above the bottom structure as the drainage structure cannot be maintained. The concentration of contaminants simultaneously decreases as the leachate is removed from the waste and the surface structure prevents rain and melting waters from entering the waste if the structure is maintained. Smith (2004), Kugler (2002), Giroud & Bonaparte (1989) and Rowe (1987) also support this view of the importance of the surface structure.

This dissertation shows that the hydraulic conductivity of natural clay materials tested for the effect of leachate decreases by 20% compared to clean water, but the result is based on limited amount of results. The literature supports the results of this dissertation that in some cases the effect of leachate may decrease the hydraulic conductivity of natural clays (Binns *et al.,* 2008; Lakshmikantha & Sivapullaiah, 2006; Ozcoban *et al.,* 2006; Jo *et al.,* 2005, Sharma & Lewis, 1994). Hydraulic conductivity increased during the experiments when MSW leachate was used as a fluid in the case of all natural clay materials. However, when clean water was used as a fluid, the role of material properties was highlighted. The hydraulic conductivity in a GCL product and sand mixed with polymer bentonite material increased similarly in leachate tests.

ii)

The Finnish interpretation of the directive differs from those of other EU countries. The constructed landfill structures in Finland only consider artificial geological barriers while the soil beneath is ignored. The purpose of the directive is that landfills will be located primarily in geological environments where the soil can naturally protect harmful substances from flowing into the groundwater and the surrounding environment.

This dissertation confirms conception early presented by Gronow (2008), Hansen (2008), Hjelmar (2008) and van der Sloot (2008) that the directive’s intention is to regulate the maximum infiltration of leachate into the soil and groundwater underneath the landfill. Unfortunately, it seems that numeric goals are missing in the directive. The directive only defines hydraulic conductivity and layer thickness, enabling different interpretations. The directive should be developed to include complete guiding principles for the risk assessment scenario. The inconsistency in risk assessment leads to overly optimistic solutions and does not observe crucial requirements such as the local climate and weather.

The results of this dissertation confirm the above-mentioned criticism aimed at the Finnish interpretation and also the EC landfill directive. To confirm the conclusions drawn by the researcher, the EC working group managed by the Dutch Waste Management Association, which was responsible for landfill construction, was contacted in writing. The working group confirmed the criticism against the Finnish interpretation.

Landfills have a wide-ranging impact on the environment, and leachate is an essential part of that. **A landfill can pollute the environment directly or indirectly and transmit contaminants into atmosphere, hydrosphere and lithosphere.** Therefore, more comprehensive and complete risk assessment methods will be needed in the future. Also the role of landfills has and will change due to new techniques and needs, e.g. landfill mining and pyrolysis.

These factors form the most essential unidentified risk factors in the landfill design phase. With the help of the SRA method developed from the risk assessment results in this dissertation, the crucial factors can be identified in the early stages of landfill design through more extensive and thorough analysis. A detailed examination is required of the effect of dominant factors on contaminant transport, the effects of soil, the effects of built environment and the phenomena affecting the operation of the structures in the different phases of the landfill life-cycle.

The 12 landfill bottom structures studied in this thesis may meet the EU directive requirements, but an analysis of their correspondence with the SRA method is not possible due to the lack of information on the environmental permits and designs. Verified landfill risk assessments include a wide range of background studies focusing on collective risk quantification, human-related, environmental and technical factors, and also waste prospects. Based on the results of the thesis, risk assessment should be developed according to the SRA method to analyse and identify the technology and human-related risks as a part of the risk quantification to support landfill life-cycle management and sustainability in the long term.

5.2 Theoretical implications

The artificial landfill bottom structure is always heterogeneous regardless of the materials and working methods used. The literature has discussed this issue mostly through analysing damages in the artificial layer, for example in the geomembrane (Katsumi *et al.,* 2001). This study, in contrast to previous literature, analyses the landfill geological barrier as a heterogeneous structure.

The landfill bottom structure and geological barrier form an entity which should include, in addition to the artificial layer, the existing circumstances within the area, soil layers below the landfill and groundwater circumstances as part of the landfill life-cycle information management. In this thesis, the SRA method has been developed using inductive reasoning, and the results demonstrate, supported by the theoretical background, that the method enables the identification of significant risk factors that affect the landfill's environmental risks and protection. The earlier simplified environmental permit decisions should be supported with the analysis of a substantially larger amount of risk factors that are significantly more important compared to the factors analysed so far. One of the most essential factors is transport modelling and its correspondence with the realistic circumstances in the nature. An unidentified risk factor may be a threat to the environment.

According to the literature, calculations are typically made assuming that the flows are constant (Rowe *et al.,* 1995; Sharma *et al.,* 1994; Ogata & Banks, 1961). The results of this study contradict the mainstream literature by highlighting the fact that leachate flows are not constant, but change as a function of time. Hence, this study is in line with the studies by Huber-Humer *et al.* (2009) and Katsumi *et al.* (2001), pointing out that all contaminant transport mechanisms must be taken into account. In addition, this study supports the results by Katsumi *et al.* (2001) who argue that after 50 years the diffusion's significance has typically decreased and become negligible. The concentration differential between soil and waste should be balanced. The soil will be polluted in the same concentration as the waste inside the landfill if the landfill has been sealed for 50 years. The sealing structures prevent external water flow to the waste and the waste content stands stable. Otherwise, this default information is irrelevant.

This study provides new information for the scientific community by analysing landfill structures with a thickness of 0.5 m, half the minimum thickness of 1 m set by the EC directive. A thinner bottom structure may lead to contaminant leakages to the environment sooner than specified in the EC requirements. In addition, the results of this study show that merely analysing contaminant transfer through advection only is not adequate. These results support the theory of layer thickness having a correlation with transit time. The United Kingdom Environmental Protection Agency (EPA) has developed by Golder Associates computer-aided calculation models for tools to evaluate the impact of structural dimensions on risk assessment. LandSim and HELP computer solutions are examples of EPA's development work to improve full or parts of risk assessment.

Laboratory scale experiments are typically conducted with clean water, but true landfill geological barriers are in contact with leachate. The results of this study show that leachate or leachate stimulants may cause an increase in the hydraulic conductivity and must be taken into account in the landfill bottom layer design. According to Kamon et al. (2005 and 2002) and Rowe et al. (1995 and 1991), it is important to ensure that hydraulic conductivity test results illustrate the reliability of sample hydraulic conductivity. In addition, typically the laboratory results could be too optimistic compared to field measurements. On these bases, the assessment data should be evaluated carefully. The results of this dissertation are in line with these studies, indicating that the saturation time is material-specific.

The literature has typically analysed individual contaminants or stimulants and their influence on hydraulic conductivity (Rowe & Iryo, 2005; Katsumi *et al.,* 2001). This study creates new knowledge by analysing the influence of true MSW leachate, pointing out the inadequacy of currently utilised tests methods and bentonite based materials. This study supports the results of Xue & Zhang (2014), Bradshaw & Benson (2014) and Ashmawy et al. (2002).

The requirements for contaminant migration in the EC Landfill Directive should be amended to bring into account contaminant penetration amounts, and during the planning phase of a landfill, a model should be developed to account for how the contaminant penetration amounts correlate through different structure types with the environmental burden of waste compacting. At the moment, the limitations to a landfill's impact on the environment are unspecified already during the active phase, and thus, the responsibility of defining and localising requirements on the migration of contaminants is left to local authorities. However, based on international literature, it can be said that the modelling of the migration of contaminants is a well-known field which simply has not been adequately applied in the EC

Landfill Directive and national environmental decisions (Katsumi *et al.,* 2001; Sharma *et al.,* 1994; Ogata & Banks, 1961).

5.3 Practical implications

In this thesis, the SRA method has been developed for future use as a risk assessment tool in the landfill design phase. It has been verified in this thesis that the method helps in recognising the possible environmental damages caused by risky situations remarkably better compared to the current procedure if the risks can be identified extensively in the design phase. This thesis has introduced two examples of risk assessments conducted with the SRA method. The SRA method enabled the use of a thinner surface structure compared with typical solutions and also by-products that have never been used before. In addition, both risk assessment solutions were extremely cost-effective and qualified the set boundaries to safety factors. Risk identification using the SRA method leads to the better risk management and thus eliminates problems.

The SRA method is applicable as a practical landfill designing tool in which the effects of unidentified factors can be anticipated as a part of the landfill life-cycle information management. As an example of this the meaning of hydraulic conductivity can be mentioned, which is currently in the focus of design and which is expected to be a dominant factor for structural design. Additionally, the SRA method demonstrates that the phenomena of wetting–drying and freezing–thawing have typically been unidentified during the environmental permit process and that their impacts in the short or long term are unknown. Typically, targets can be mentioned in almost every landfill in which these phenomena have an effect during the whole landfill life-cycle. Leachate ponds are a typical example since they are susceptible to water level fluctuation during different seasons depending on precipitation and cumulative leachate amounts. In addition, the edges of the ponds have not been protected at all, and for example the impact of winter on the contaminant retention capability of the leachate ponds has not been identified. Correspondingly, the durability of the mineral protection layers in the leachate ponds, which are made of bentonite-based materials, towards the drying–wetting phenomenon is not known. The leachate pond bottom structures are typically under the load of several meters of leachate, and thus, in seepage situations, contaminant flux can be regarded as continuous, which can result in significant environmental stress to the landfill environment. With the SRA method, the impacts of for example the drying–wetting and freezing–thawing phenomena can be highlighted and identified in the designing phase.

This study analyses the characteristics of hydraulic conductivity in landfill bottom layers. In Finland, this is typically done through laboratory experiments instead of theoretically analysing different transfer mechanisms of harmful substances. Simple advection analyses are not adequate enough to describe the long-term environmental impacts of landfills that may extend over hundreds of years. Opposed to this mainstream approach, this study is based on calculating both advection and dominant transfer mechanisms, for example diffusion. This is further utilised to assess the Finnish interpretations of the EC Landfill Directive.

The implications of this study include the need to consider landfill structures as a whole instead of analysing them merely based on the transfer of single harmful substances or sorption factors.

The Finnish environmental permit procedure emphasises less important factors for environmental risks, such as potential traffic incidents causing oil leakages or explosions, during the landfill active phase (Environmental decision registry, 2010). With the existing landfill structures, thinner than required by the EC directive, landfill risk assessments should be based on the long-term impact of harmful substances. Consequently, environmental risk assessments have been inadequate in Finland.

This study highlights that landfills ought to be primarily located in such geological areas where the soil layers beneath are capable of protecting the surrounding environment in the long term. Unfortunately, only in rare cases, the hydro-geological properties, such as groundwater levels, the natural environment like a sea, river or lake, and human influence, are defined in the environmental permits or in landfill design. The long-term environmental impact of landfills ought to be clarified to avoid harmful substance flux to the environment before any permanent damages. Modern landfills should, according to the EC directive, be located where the landfill cannot pollute hydro-geological properties, but old landfills might cause problems.

The results of this study emphasise the importance of utilising risk assessment like the SRA method for designing landfill structures. The design parameters should be based on true technical and environmental factors, human influence, waste prospects and risk quantifications for the final result to acknowledge the existing hydro-geological environment.

In the future, demands concerning landfills will change because of reuse, recycling and using landfills as a material source. In consequence, landfills will be used as temporary storage in the future. Therefore, tools such as the SRA method will be needed to evaluate the life-cycle of landfill structures from different angles. Also, there will be pressure to extend the temporary storage period beyond the present three years. The collection of materials such as water sludge, slag, ash, sediment and contaminated masses should be possible over a longer period depending on how much recyclable mass is produced per year. According to the EC directive, the material owner has to pay the tax for the material after three years of temporary storage. After taxes, the materials are no longer cost-efficient for reuse purposes. An amendment to the EC directive to allow longer temporary storage without taxes could open new possibilities for materials like sludge, slag, ash, sediment and contaminated masses for reuse or recycling.

The existing EC directive clearly defines the design requirements for landfill structures. However, the interpretations on the directive vary in different parts of Europe. Even in Finland, there are differences in the instructions given by different local authorities. More detailed instructions, either at the EC or national levels, are required to complement the existing directive. In Finland, there is a clear need to re-analyse the long-term environmental impacts of the existing landfills. This type of re-assessments would provide necessary information for landfills' risk potential by focusing e.g. on landfill mining or the structure life-cycle.

The most significant problem is the lack of knowledge on landfills' risks and their impact on the environment. This thesis has highlighted some practical implications of the risk analysis deficiencies impacting landfill risk management.

Structural dimensioning is based on simplified equivalent calculation. This thesis and literature have shown that leachate increases hydraulic conductivity (Xue & Zhang, 2014; Bradshaw & Benson, 2014; Ashmawy *et al.,* 2002). In practise, any of the risk assessments, design or environmental permits of the examined 12 landfills did not take into account the possibility that e.g. the protection capacity of reservoir sides could change because of increasing hydraulic conductivity.

According to Forget et al. (2005), holes were generated in the geomembranes holes after installation. Katsumi et al. (2001) and Park et al. (1996) have introduced the waste stream calculation equations for pollution. E.g. reservoirs include leachate during the whole life-cycle of the landfills. The reservoir's environmental protection structures have been typically designed equally compared with the waste filling bottom structures or recycling field. Drainage pipes and a drainage structure have typically been installed over the bottom structure to avoid a hydraulic gradient. However, a reservoir could hold a couple of meters leachate over a similar bottom structure as under the waste filling. Therefore, reservoirs could become risk factors because the structures are designed according to waste filling, not the reservoir's demands.

Many of the modern landfills have been established alongside old landfills that do not have any artificial protection layer under the waste embankment. For example, one reservoir was located in the middle of the old and modern bottom structures. Even if the old landfill has been sealed a few years ago, the leachate could migrate from the waste to the soil. The migration could be identified from the control water samples by increased concentration. It could not be able identify the pollution target if the reservoir leaks or old landfill leaks, because the location of the reservoir.

According to the EC directive and Government decision, reservoirs should be able to protect the environment at least 30 years after landfill sealing and duration could be much longer depending on aftercare. In many landfills, the total life-cycle during the active and passive phase could be over 50 years. Landfill bottom structures have been designed to protect the environment at least 30 year after sealing, but the structures were not protected under the waste from changes in the weather. According to literature, none of the geomembranes have been tested for 50 years under the local weather conditions in Finland. If the geomembrane is damaged, what will the protection capacity of the artificial protection structure under the reservoir be like after 30 years, taking into account different phenomena and meteorological changes? By focusing on the combined effect of wetting-drying, freezing-thawing and leachate migration through the holes, none of the landfill managers, designers or authorities have knowledge of the combined effects on reservoirs or the recycling fields life-cycle to protect the environment.

In the future, landfills owners should focus more aggressively on the critical aspects of environmental protection and avoid possible pollution. This thesis has raised several essential points which have been typically downplayed or ignored. Leachate plays the main role in a landfill's impact on the environment, but the impact of leachate on the features and tolerance of materials has been explained straightforwardly. Also, human influence on a landfill is very important issue. Landfills are closing in on human residential areas, and the impacts could be two-way. More attention should be paid to site management and residential areas in risk assessments. It could be concluded that in practice the focus should be on the future and the better evaluation of future challenges, like waste prospects.

5.4 Evaluating the research

The calculations will be not valid if some of the properties or the equation is changed. The theoretical calculation takes into account only the bottom layers over the soil. The soil has an influence on the results, but it has to be examined before inclusion in the calculation. Consequently, the soil has been ignored, and the calculations have been carried out with an equation which describes the seeping of harmful substances through the landfill bottom. The vertical dimension is assumed to be the dominant direction, and calculations have been made one-dimensionally according to Jaiswal et al. (2011).

The landfill waters that were used in this study consisted of water from leachate basins, including solids; and the waters were visually estimated to be of a similar colour. While in the canister, the water became aged and precipitated as some of the heavier particles sank to the bottom, and a lighter-coloured residue formed on the surface of the leachate. The solid matter in the water may have affected the results as it may have remained in some of the samples or formed compounds or sorbed within the sample.

Artificial water was manufactured in a laboratory. Potential causes for unreliable results are the accuracy of the measurement scales, the thoroughness of the rinsing of the containers used in the measurements and the filling of the measurement bottle up to the mark on the bottle precisely. Regarding the artificial water, the unreliability of the acquired results was affected by the initial concentrations and chemical character of the water rather than the manufacturing process.

All samples were stored in the same place at a constant temperature and insulated from any contact with air as well as possible for the duration of storage. Interaction between the clay matter and air may cause changes in the dampness of the sample, and thus affect the outcome of the test.

Errors in measurement results are either parasitic or systematic and result from both integral errors in measurement devices and errors in measurements. Parasitic errors are caused by errors in the readings of a measurement device. These errors, however, do not have a significant impact on the reliability of the results in this study since the measurement devices were operated by a single person throughout the examination and the operator was both trained to perform the task and highly experienced in such tasks. Systematic errors in the results are usually caused by a device that is not functioning properly or by an improperly calibrated or broken device.

The repeatability of the hydraulic conductivity tests depends to a great extent on the stability of the characteristics of the testing material even when the same batch of samples is used. The results of the tests in which the results of triplicate measurements show similar tendencies and values are reliable. On the other hand, in tests in which parallel measurements have not been taken, the reliability of the results is supported by the fact that nothing uncommon was found in the samples during their unpacking and that they appeared similar to the eye when compared with each other.

5.5 Further studies

During this research, a number of further study themes have emerged. The themes below are proposed as a continuation of this study.

i) *National Landfill Directive interpretation's influence on the environment.* In Finland, the design process for landfill structures takes into account only the artificial geological barrier while the ground is ignored. This kind of interpretation differs from the interpretations of some other European countries. It would be important to evaluate how the national Landfill Directive interpretation influences the environment in Finland and for example in Germany, Sweden and Denmark.

ii) *Environmental risk assessment of existing landfills*
From the perspective of direct and indirect risks to the atmosphere, lithosphere and hydrosphere, landfills are very different in terms of their location. None of the surface or bottom layer structures is fully capable of protecting the environment in the long term. A risk assessment model, or a method like the SRA method, should be used for analysing the existing landfills for potential risks to the environment. Also develop to identify risk factors and the ranking factors.

iii) *Contaminant transport phenomena's influence on the environment and groundwater in different geological circumstances*. In this study, calculations have been made for various mineral liner materials' contaminant transit times. The national interpretation of the Landfill Directive must be updated taking into account all of the transport mechanisms and geological circumstances influencing the environment. All transport mechanisms do not affect soil materials in the same manner. Consequently, further research could focus on materials which have typically been used in landfill bottom layers and the soil materials below the landfill.

iv) *Landfill life-cycle modelling.* Harmful substances may seep from the landfill to lower layers over decades should there be no adequate protection. For example, in MSW landfills, energy production keeps the landfill active for a couple of decades after the waste has been sealed. According to previous studies, it is theoretically possible to develop a landfill life-cycle model that helps to understand landfills' influence on the environment.

v) *Landfill in the future.* In the future, landfills could be temporary storage areas. This could affect the landfill risk assessment because the waste material content will be changed, the storage time-frames vary depending on the waste and new techniques might demand special properties of the protection structures. Landfilling must adapt to updated European and local regulations.

The topics for future studies above could be implemented as combined laboratory and field experiments supported by modelling. Many of these research topics require large-scale studies and several years of field measurements to quantify and qualify the overall and general patterns of process variations. This study provides a good starting point and background information for these topics, which need further research in the future.

6 References

Acar, Y.B. & Haider, L. 1990. Transport of low-concentration contaminants in saturated earthen barriers. *Journal of Geotechnical Engineering,* 116 (7), pp. 1031–1052.

Advanded GeoTech Systems. 2000. Landfill Design.com. URI: http://www.landfilldesign.com/main.html.

Andersland, O.B. & Anderson, D.M. 1978. *Geotechnical Engineering for Cold Regions*. New York: McGraw-Hill Book Co, pp. 278.

Anon. 1997. Deutsche Gesellschaft für Geotechnik, GDA-Empfehlungen Geotechnik der Deponien und Altlasten, pp. 69.

Ashmawy, A.K., El-Hajji, D., Sotelo, N. & Muhammad, N. 2002. Hydraulic performance of untreated and polymer-treated bentonite in inorganic landfill leachates, *Clays and Clay Minerals,* 50 (5). pp. 546–552.

ASTM Standard C566. 1997. Standard test method for total evaporable moisture content of aggregate by drying. American Society for Testing and Materials, West Conshohocken, Pennsylvania, USA, pp. 6.

ASTM Standard D5084. 2000. Standard test method for measurement of hydraulic conductivity of saturated porous materials using flexible wall permeameter. American Society for Testing and Materials, West Conshohocken, Pennsylvania, USA, pp. 46.

ASTM Standard D5887. 1999. Standard test method for measurement of index flux through saturated geosynthetic clay liner specimens using a flexible wall permeameter. American Society for Testing and Materials, West Conshohocken, Pennsylvania, USA, pp. 14.

Attenborough, G.M., Hall, D.H., Gregory, R.G. & McGoochan, L., 2002. Development of a landfill gas risk assessment model: GasSim. In: Proceedings for Solid Waste Association of North America, 25 Annual Landfill Gas Symposium, Monterey, CA, USA, pp. 25–28

Atuahene, F. 2008. Prediction of the hydraulic conductivity of clay liners for efficient construction and monitoring of clay liners in landfills. *Journal of Solid Waste Technology and Management,* 34 (3), pp. 149–159.

Auriault, J. & Lewandowska, J. 1996. Diffusion/adsorption/advection macrotransport in soils, *European Journal of Mechanics*, A/Solids, 15 (4), pp. 681–704.

Aven, T. & Kristensen, V. 2005. Perspectives on risk – review and discussion of the basis for establishing a unified and holistic approach. *Reliability Engineering and System Safety*, 90, pp. 1–14.

Avesta IPPC (2004). Deldom om Outokumpu Stainless Aktiebolag tillstånd enligt miljöbalken till befintlig och utökad verksamhet vid bolagets anläggningar vid Norra och Södra verken I Avesta (Environmental permit decision of Avesta). pp. 50.(In Swedish.)

Babendreier, J.E. & Castleton, K.J. 2005. Investigating uncertainty and sensitivity in integrated, multimedia environmental models: Tools for FRAMES-3MRA. *Environmental Modelling and Software*, Volume 20, Issue 8, pp. 1043-1055.

Baeurle, S. 2009. Multiscale modeling of polymer materials using field-theoretic methodologies: A survey about recent developments. *Journal of Mathematics Chemistry,* 46 (2), pp. 363-426.

Barlaz, M.A., Rooker, A.P., Kjeldsen, P., Gabr, M.A. & Borden, R.C. 2002. Critical evaluation of factors required to terminate the postclosure monitoring period at solid waste landfills. *Environmental Science and Technology,* 36 (16), pp. 3457–3464.

Barone, F.S., Rowe, R.K. & Quigley, R.M. 1992. A laboratory estimation of diffusion and adsorption coefficients for several volatile organics in a natural clayey soil. *Journal of Contaminant Hydrology,* 10 (3), pp. 225–250.

Bear, J.H.F. & Palmer, R.S.J. 1972. *Hydraulics of groundwater.* New York: McGraw-Hill.

Bell, L.S.J., Binning, P.J., Kuczera, G. & Kau, P.M.H. 2002. Rigorous uncertainty assessment in contaminant transport inverse modelling: A case study of fluoride diffusion through clay liners. *Journal of Contaminant Hydrology,* 57 (1–2), pp. 1–20.

Benson, C.H. & Daniel, D.E. 1994. Minimum thickness of compacted soil liners: II. Analysis and case histories. *Journal of Geotechnical Engineering—ASCE*, 120(1), pp. 153–172.

Benson, C.H., Daniel, D.E. & Boutwell, G.P. 1999. Field performance of compacted clay liners, *Journal of Geotechnical and Geoenvironmental Engineering,* 125(5), pp. 390–403.

Benson, C.H. & Othman, M.A. 1993. Hydraulic conductivity of compacted clay frozen and thawed in situ. *Journal of Geotechnical Engineering,* 119 (2), pp. 276–294.

Benson, C.H., Thorstad, P.A., Jo, H. & Rock, S.A. 2007. Hydraulic performance of geosynthetic clay liners in a landfill final cover. *Journal of Geotechnical and Geoenvironmental Engineering,* 133 (7), pp. 814–827.

Benson, C.H., Zhai, H. & Wang, X. 1994. Estimating hydraulic conductivity of compacted clay liners. *Journal of Geotechnical Engineering—ASCE*, 120 (2), pp. 366–387.

Berger, K., Gröngröft, A. & Harms, C. 2009. Water balance and effectiveness of a landfill cover with layers of dredged materials: The Hamburg-Francop case. 12 th International Waste Management and Landfill Symposium, 5–9 Oct. 2009, S. Margherita di Pula-Cagliari, Sardinia, Italy, pp. 675–679.

Binns, A., Robinson, P.N.J. & Eccles, C.S. 2008. The properties of Lias Clay for landfill liners. Proceedings of the Institution of Civil Engineers: Geotechnical Engineering, 161 (4), pp. 197–207.

Boardman, B.T. & Daniel, D.E. 1996. Hydraulic conductivity of desiccated geosynthetic clay liners. *Journal of Geotechnical and Geoenvironmental Engineering*, 122 (3), pp. 204–208.

Boels, D., Koenis, E.P.W. & Loovers, E.M. 1993. Suitability of tertiary clays and water glass for lining landfills. The Netherlands, Wageningen: DLO Winand Staring Centre. Rapport 291.

Bonaparte, R., Daniel, D.E. & Koerner, R.M. 2002. Assessment and Recommendations for Improving the Performance of Waste Containment Systems. EPA/600/R-02/099, pp. 1039.

Bosmans, A., De Dobbelaere, C. & Helsen, L. 2014. Pyrolysis characteristics of excavated waste material processed into refuse derived fuel. *Fuel*, Volume 122, 15 April 2014, pp. 198-205.

Bowders, J.J. & Daniel, D.E. 1987. Hydraulic conductivity of compacted clay to dilute organic chemicals. *Journal of Geotechnical Engineering,* 113, pp. 1432–1448.

Bouazza, A., Vangpaisal, T. & Jefferis, S. 2008. Effect of wet-dry cycles and cation exchange on gas permeability of geosynthetic clay liners. *Journal of Geotechnical and Geoenvironmental Engineering,* 132 (8), pp. 1011–1018.

Bockreis, A. & Knapp, J. 2011. Landfill Mining - landfill sites serving as raw materials sources. *Osterreichische Wasser- und Abfallwirtschaft*, 63 (2-3), pp. 70-75.

Bradshaw, S.L. & Benson, C.H. 2014. Effect of municipal solid waste leachate on hydraulic conductivity and exchange complex of geosynthetic clay liners. *Journal of Geotechnical and Geoenvironmental Engineering*, Volume 140, Issue 4, Article number 04013038.

Brown, K.W. & Thomas, J.C. 1998. A comparison of the convective and diffusive flux of organic contaminants through landfill liner systems. *Waste Management and Research*, 16 (3), pp. 296–301.

Brunner, P.H. & Fellner, J. 2007. Setting priorities for waste management strategies in developing countries. *Waste Management and Research*, 25 (3), pp. 234–240.

Butt, T.E., Gouda, H.M., Baloch, M.I., Paul, P., Javadi, A.A & Alam, A. 2014. Literature review of baseline study for risk analysis – The landfill leachate case. *Environment International,* Volume 63, pp. 149-162.

Butt, T.E., Ingles, A.J.D. & Baloch, M.I. 2011. A conceptual model outline for integrated exposure assessment. *Environmental Program Sustainable Energy Journal*, Volume 30 (4), pp. 696–708.

Butt, T.E., Lockley, E. & Oduyemi, K.O.K. 2007. Risk assessment of landfill disposal sites – State of the art. *Waste Management,* 28 (2008), pp. 952-964.

Caflisch, R.E. 1998. Monte Carlo and quasi-Monte Carlo methods. Acta Numerica 7. Cambridge University Press. pp. 1–49.

Chamberlain, E.J. 1992. Freeze-thaw effects on the permeability of Shakopee and Resemont II soils. U.S. Army cold Regions Research and Engineering Laboratory, Hanover, N.H. 1992.

Chamberlain, E.J., Iskander, I. & Hunsiker, S.E. 1990. Effect of freeze-thaw on the permeability and macrostructure of soils. Proc., Int. symp. on Frozen soil Impacts on Agric., Range, and Forest Lands, pp .145–155.

Chapman, P.M., McDonald, B.G., & Lawrence, G.S. 2002. Weight-of-evidence issues and frameworks for sediment quality (and other) assessments. *Human Ecological Risk Assessment*, 8 (7), pp. 1489–1515.

Christensen, T.H., Bjerg, P.H. & Kjeldsen, P. 2000. Natural attenuation: a feasible approach to remediation of groundwater pollution at landfills? *Ground Water Monitoring and Remediation*, 20 (1), pp. 69–77.

Christensen, T.H., Kjeldsen, P., Bjerg, P.L., Jensen, D.L., Christensen, J.B., Baun, A., Albrechtsen, H.-J. & Heron, G. 2001. Biogeochemistry of landfill leachate plumes. *Applied Geochemistry,* 16, pp. 659–718.

Christensen, T.H., Cossu, R. & Stegmann, R. 1994. Principles of landfill barrier systems. In: Christensen, T.H., Cossu, R. & Stegmann, R. (Eds.). *Landfilling of Waste: Barriers.* London: E & FN Son, pp. 631-640.

Chowdhury, M. 2009. Searching quality data for municipal solid waste planning. *Waste Management*, Volume 29 (8), pp. 2240–2247.

Cormier, S.M., Paul, J.F., Spehar, R.L., Shaw-Allen, P., Berry, W.J. & Suter II, G.W. 2008. Using field data and weight of evidence to develop water quality criteria. *Integrated Environmental Assessment and Management,* 4, pp. 490–504.

Cossu, R. 2007. Landfill sustainability—concepts and models. In: Proceedings Sardinia 2007, Eleventh International Waste Management and Landfill Symposium, 1–5 Oct. 2007, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Cossu, R., Frongia, G., Muntoni, A., Nobile, A. & Raga, R. 1997. Use of pumping tests for the assessment of leachate flow regime, waste hydraulic parameters and well efficiency. Proc. Sardinia 13-17 Oct. 1997, 6th International Landfill Symposium, Cagliari, Italy.

Cossu, R., Raga, R. & Rossetti, D. 2003. The PAF model: an integrated approach for landfill sustainability. *Waste Management,* 23, pp. 37–44.

Cossu, R., Raga, R. & Rossetti, D. 2003. Full scale application of in situ aerobic stabilization of old landfills. Proceedings Sardinia 6-10 Oct. 2003, Ninth International Waste Management and Landfill Symposium, Cagliari, Italy.

Coventry, Z.A., Tize, R. & Karunanithi, A. 2012. Environmental life-cycle assessment of waste-to-energy technologies. Air and Waste Management Association—International Conference on Thermal Treatment Technologies and Hazardous Waste Combustors17-19 May, pp. 159–174

Daniel, D., Bowders, J. & Gilbert, R. 1997. Laboratory hydraulic conductivity testing of GCLs in flexible-wall permeameters. Testing and Acceptance Criteria for Geosynthetic Clay Liners, ASTM STP 1308, pp. 208–226.

Deutsche Gesellschaft für Geotechnik e.V. 1996. Empfehlungen für die Ausführung von Asphaltarbeiten im Wasserbau-EAAW 86/96, Essen. (German Guidelines for the execution of asphalting in hydraulic engineering.) (In German.)

EC Landfill Directive 1999/31/EC. 1999. Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste.

Eichhorst, J., Morgan, S. & Hoskins, J. 2013. Saturated hydraulic conductivity of bioretention media with municipal waste incinerator bottom ash. World Environmental and Water Resources Congress 2013: Showcasing the Future - Proceedings of the 2013 Congress 2013, pp. 3000-3008.

Eid, H.T. 2011. Shear strength of geosynthetic composite systems for design of landfill liner and cover slopes. *Geotextiles and Geomembranes*, 29 (20101), pp. 335–344.

Egloffstein, T.A. 2001. Natural bentonites—Influence of the ion exchange and partial desiccation on permeability and self-healing capacity of bentonites used in GCLs. *Geotextiles and Geomembranes*, 19 (7), pp. 427–444.

El-Zein, A. & Rowe, R.K. 2008. Impact on groundwater of concurrent leakage and diffusion of dichloromethane through geomembranes in landfill liners. *Geosynthetics International,* 15 (1), pp. 55–71.

Environmental decision registry. 2010. URI: http://www.ymparisto.fi/default.asp?contentid=243974&lan=fi&clan=fi

Environmental decision. 2007. Dnro UUS-2004-Y-823-111 No YS 998. 14.11.2013

Environmental decision. 2005. Dnro PSA-2004-Y-253-111. 14.11.2013

Environmental decision. 2005. Dnro PSA-2005-Y-243-121. 14.11.2013

Environmental decision. 2001. Dnro 0800Y0307-111. 14.11.2013

Environmental decision. 1996. Dnro 0295Y0226-124.14.11.2013

Environment Agency, 2002. GasSimLite User Manual. Golder Associates.

Environment Agency. 2005. Landfill Gas Emissions Model (LandGEM), Morrisville.

Environment Agency. 2003. LandSim 2.5 - Groundwater risk assessment tool for landfill design, Bristol.

Environment Agency. 2001. LandSim, Release 2 (V.2.02) – Landfill Performance Simulation by Monte Carlo Method. R&D Publication 120.

Environment Agency. 1996. LandSim Performance Simulation by Monte Carlo Method (a LandSim Software Manual). Golder Associates Ltd.

Eskola, J. & Suoranta, J. 1996. Johdatus laadulliseen tutkimukseen (Introduction into qualitative research). University of Lapland. Publications of the Faculty of Education. C:13. (In Finnish.)

EU Waste Framework Directive 2008/98/EC.

EU Landfill Directive 1999/31EC.

Fatta, D., Naoum, C., Karlis, P. & Loizidou, M. 2000. Numerical simulation of flow and contaminant migration at a municipal landfill. *Journal of Environmental Hydrology*, 8, pp. 41–42.

Feo, G..D., & Williams, I.D. 2013. Siting landfills and incinerators in areas of historic unpopularity: Surveying the views of the next generation. *Waste Management,* volume 33, Issue 12, pp. 2798-2810

Finnish Environment Institute. 2002. Kaatopaikan tiivistysrakenteet SYKE 36, pp. 145. Helsinki: SYKE—Finnish Environment Institute. (In Finnish.)

Folkes, D.J. 1982. Fifth Canadian Geotechnical Colloquium: Control of contaminant migration by the use of liners. *Canadian Geotechnical Journal V,* 19 (3), pp. 320–344.

Foose, G.J., Benson, C.H. & Edil, T.B. 2002. Comparison of solute transport in three composite liners. *Journal of Geotechnical and Geoenvironmental Engineering*, 128 (5), pp. 391–403.

Forget, B., Rollin, A.L. & Jacquelin, T. 2005. Lessons learned from 10 years of leak detection surveys on geomembranes. In: Proceedings Sardinia 2005, tenth International Waste Management and Landfill Symposium, 3–7 Oct. 2005, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Freeze, A.R. & Cherry, J.A. 1979. *Groundwater.* London: Prentice Hall.

Frändegård, P., Krook, J., Svensson, N. & Eklund, M. 2013. A novel approach for environmental evaluation of landfill mining. *Journal of Cleaner Production*, 55, pp. 24-34.

Giroud, J.P. & Bonaparte, R. 1989. Leakage through liners constructed with geomembranes—Part II. Composite liners. *Geotextiles and Geomembranes,* 8 (2), pp. 71–111.

Giroud, J.P., Zornberg, J.G. & Zhao, A. 2000. Hydraulic design of geosynthetic and granular liquid collector layers. *Geosynthetics International, Special issue on Liquid Collection Layers,* 7, pp. 4–6.

Giusti, L. 2009. A review of waste management practices and their impact on human health. *Waste Management*, Volume 29 (8), pp. 2227–2239.

GLO-85. 1985. Geotekniset laboratorio-ohjeet 1. Luokitusohjeet (Geotechnical laboratory instructions 1. Categorization instructions). Finnish Geotechnical Society. Helsinki. Rakentajain Kustannus Oy. (In Finnish.).

Golder Associates (UK) Ltd. 1998. Repository Integration Programme (RIP) (Integrated Probabilistic Simulator for Environmental Systems). http://www.golder.com/rip/intro.htm#What.

GoldSim Technology Group LLC. 2003. GoldSim (Probabilistic User Environment). Version 8.01, User's Guide, GoldSim Technology Group LLC, October.

Grellier, S., Robain, H., Bellier, G. & Skhiri, N. 2006. Influence of temperatue on the electrical conductivity of leachate from municipal solid waste. *Journal of Hazardous Marterials*, Volume 137, Issue 1. pp. 612-617.

Guyonnet, D., Cazaux, D., Vigier-Gailhanou, H. & Chevrier, B. 2009. Effect of cation exchange on hydraulic conductivity in a sand-bentonite-polymer mixture. 12 th International Waste Management and Landfill Symposium, 5–9 Oct. 2009, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Guyonnet, D., Gaucher, E., Gaboriau, H., Pons, C., Clinard, C., Norotte, V. & Didier, G. 2005. Geosynthetic clay liner interaction with leachate: Correlation between permeability, microstructure, and surface chemistry. *Journal of Geotechnical and Geoenvironmental Engineering*, 131 (6), pp. 740–749.

Guyonnet, D., Gourry, J., Bertrand, L. & Amraoui, N. 2003. Heterogeneity detection in an experimental clay liner. *Canadian Geotechnical Journal*, 40 (1), pp. 149–160.

Hall, D. H., McDougall, J., Drury, D., Blakey, N., Braithwaite, P. & Rosevear, A. 2007. 11th International Waste Management and Landfill Symposium S. Margherita di Pula, Cagliari, Italy; 1 - 5 October 2007.

Hansson, K. & Lundin, L.C. 2006. Equifinality and sensitivity in freezing and thawing simulations of laboratory and in situ data. *Cold Regions Science and Technology*, 44, pp. 20-37.

Hansen, J.A. 2009. Editorial. *Waste Management and Research*, 27 (9), pp. 837–838.

Harland, C.E. 1994. *Ion Exchange: Theory and Practice.* 2. Ed. Bath, UK: Royal Society of Chemistry Paperbacks.

He, J. & Song, G. 2011. Hydraulic conductivity of clay liners under wet-dry cycling: State of the art. 2011 International Conference on Electric Technology and Civil Engineering, ICETCE 22–24 Apr. 2011—Proceedings, pp. 4174–4177.

Heikkinen, P.M., Korkka-Niemi, K., Lahti, M. & Salonen, V.-P. 2002. Groundwater and surface water contamination in the area of the Hitura nickel mine, Western Finland. *Environmental Geology*, 42 (4), pp. 313–329.

Henken-Mellies, W.-U. & Schweizer, A., 2011. Long-term performance of landfill covers—Results of lysimeter test fields in Bavaria (Germany). *Waste Management and Research*, 29 (1), pp. 59–68.

Hewitt, R. & Daniel, D. 1997. Hydraulic Conductivity of Geosynthetic Clay Liners after Freeze-Thaw. J. Geotech. Geoenviron. Eng., 123(4), pp. 305–313.

Hudson, A.P., Beaven, R.P., Knox, K. & Powrie, W. 2009. Investigation of the clogging of landfill tyre drainage layers. 12th International Waste Management and Landfill Symposium, 5–9 Oct. 2009, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Huber-Humer, M., & Lechner, P. 2009. Future visions: sustainable landfilling of a sustainable society without landfills? 12th International Waste Management and Landfill Symposium, 5–9 Oct. 2009, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Hull, R.M., Krogmann, U. & Strom, P.F., 2005. Composition and characteristics of excavated materials from a New Jersey landfill. *Journal of Environmental Engineering,* 131, pp. 478-490.

IPCC, 2013: *Climate Change 2013*: *The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 1535 pp.

Jaiswal, D.K., Kumar, A. & Yadaw, R.R. 2011. Analytical solution to the One-Dimensional Advection-Diffusion Equation with Temporally dependent Coefficients. *Journal of Water Resource and Protection* (3), pp. 76–84.

Jo, H.Y., Benson, C.H., Shackelford, C.D., Lee, J. & Edil, T.B. 2005. Long-term hydraulic conductivity of a geosynthetic clay liner permeated with inorganic salt solutions. *Journal of Geotechnical and Geoenvironmental Engineering* 131 (4), pp. 405–417.

Jo, H.Y., Katsumi, T., Benson, C.H. & Edil, T.B. 2001. Hydraulic conductivity and swelling of nonprehydrated GCLs permeated with single-species salt solutions. *Journal of Geotechnical and Geoenvironmental Engineering* 127 (7), pp. 557–567.

Jokela, J. 2002. Landfill operation and waste management procedures in the reduction of methane and leachate pollutant emissions from municipal solid waste landfills. Doctoral Thesis. University of Jyväskylä.

Kalbe, U., Müller, W.W., Berger, W. & Eckardt, J. 2002. Transport of organic contaminants within composite liner systems. *Applied Clay Science* 21 (1–2), pp. 67–76.

Kamon, M., Katsumi, T., Inui, T. & Hamada, S. 2005. Environmental acceptability of the installation of piles through the bottom clay barrier at coastal landfill sites. *Environmental Geotechnics,* pp. 3633–3647.

Kamon, M., Katsumi, T., Inui, T. & Hamada, S. 2005. Experimental study on the interface transmissivity between clay layer and steel pile installed at waste disposal site. *Zairyo—Journal of the Society of Materials Science, Japan* 54 (11), pp. 1100–1104.

Kamon, M., Katsumi, T., Inui, T. & Morimoto, T. 2002. Suitability assessment of bentonite-soil mixtures as the landfill bottom liner material. *Zairyo—Journal of the Society of Materials Science, Japan,* 51 (1), pp. 36–41.

Katsumi, T., Benson, C.H., Foose, G.J. & Kamon, M. 2001. Performance-based design of landfill liners. *Engineering Geology*, 60 (1–4), pp. 139–148.

Katsumi, T., Ishimori, H., Ogawa, A., Maruyamam, S. & Fukagawaiv, R. 2008. Effects of water content distribution on hydraulic conductivity of prehydrated GCLS against calcium chloride solutions. *Soils and Foundations*, 48 (3), pp. 407–417.

King, K.S., Quigley, R.M., Fernandez, F., Reades, D.W. & Bacopoulos, A. 1993. Hydraulic conductivity and diffusion monitoring of Keele Valley Landfill liner, Maple, Ontario. *Canadian Geotechnical Journal*, 30 (1), pp. 124–134.

Korkka-Niemi, K. & Salonen, V.P. 1996. Maanalaiset vedet–pohjavesigeologian perusteet, Turun yliopiston täydennyskoulutuskeskus julkaisuja A:50 (Undeground waters—Intruduction into groundwater technology,

Publications of the Centre for continuing education of University of Turku, Finland, A:50). (In Finnish.)
Kugler, H., Ottner, F., Froeschl, H., Adamcova, R. & Schwaighofer, B. 2002. Retention of inorganic pollutants in clayey base sealings of municipal landfills. *Applied Clay Science,* 21 (1–2), pp. 45–58.
Kumar, A. & Sharma, M.P. 2014. Estimation of GHG emission and energy recovery potential from MSW landfill sites. *Sustainable Energy Technologies and Assessments* Volume 5, March 2014, pp. 50-61.
Lahermo, P., Väänänen, P., Tarvainen, T. & Salminen, R. 1996. Suomen geokemian atlas, Osa 3: Ympäristögeokemia – purovedet ja sedimentit. (Geochemical atlas of Finland. Part 3: Environmental geochemistry—Stream waters and sediments.) Geological Survey of Finland, Espoo. (In Finnish.)
Lakshmikantha, H. & Sivapullaiah, P.V. 2006. Relative performance of lime stabilized amended clay liners in different pore fluids. G*eotechnical and Geological Engineering*, 24 (5), pp. 1425–1448.
Laner, D., Crest, M., Scharff, H., Morris, J.W.F. & Barlaz, M.A. 2012. A review of approaches for the long-term management of municipal solid waste landfills. *Journal of Waste Management*, 32, pp. 498–512.
Landcare Research (Manaaki Whenua Landcare Research – a New Zealand Crown Research Institute). 2003. Risk assessment model reviews. http://www.contamsites.landcareresearch.co.nz/risk_assessment_models_reviews.htm.
Leavesley, G.H. & Nicholson, T.J. 2005. Interagancy steeing committee on multimedia environmental models (ISCMEM).http://www.isc-mem.org/WorkGroup_02.thm.
Lee, J., Shackelford, C.D., Benson, C.H., Jo, H. & Edil, T.B. 2005. Correlating index properties and hydraulic conductivity of geosynthetic clay liners. *Journal of Geotechnical and Geoenvironmental Engineering*, 131 (11), pp. 1319–1329.
Leeson, J., Edwards, A., Smith, J.W.N. & Potter, H.A.B. 2003. Hydrogeological Risk Assessments for Landfills and the Derivation of Groundwater Control and Trigger Levels. Environment Agency, pp. 104.
Li, G., Yang, W. & Dan, T. 2008. Hydraulic conductivity of GCLs in MSW landfills. *Journal of Harbin Institute of Technology (New Series),* 15 (4), pp. 455–458.

Lo, I. 2003. Innovative waste containment barriers for subsurface pollution control. Department of Civil Engineering, Hong Kong University of Science and Tech. Clear Water Bay, Kowloon, Hong Kong. Publisher American Society of Civil Engineers, *Practice Periodical of Hazardous, Toxic and Radioactive Waste Management*, 7 (1), pp. 37–45.

Lo, I. 1996. Role of organic attenuation in saturated clay barrier system. Hong Kong University of Science and Technology, Kowloon, Hong Kong. The 1995 3rd IAWQ Specialized Conference on Appropriate Waste Management Technologies for Developing Countries, Nagpur, India 1995. Publisher Pergamon Press Inc, Tarrytown, NY, USA. *Water Science and Technology*, 33 (8), pp. 145–151.

Lo, I.M., Law, W.K. & Shen, H.M. 1999. Risk assessment using stochastic modeling of pollutant transport in landfill clay liners. *Water Science and Technology*, 39 (10–11), pp. 337–341.

Malusis, M.A., Yeom, S. & Evans, J.C. 2011. Hydraulic conductivity of model soil-bentonite backfills subjected to wet-dry cycling. *Canadian Geotechnical Journal*, 48 (8), pp. 1198–1211.

Mattiello, A., Chiodini, P., Bianco, E., Forgione, N., Flammia, I., Gallo, C., Pizzuti, R. & Panico, S. 2013. Health effects associated with the disposal of solid waste in landfills and incinerators in populations living in surrounding areas: A systematic review. *International Journal of Public Health*, Volume 58, Issue 5, October 2013, pp. 725-735.

Meer, S.R. & Benson, C.H. 2007. Hydraulic conductivity of geosynthetic clay liners exhumed from landfill final covers. *Journal of Geotechnical and Geoenvironmental Engineering*, 133 (5). pp. 550−563.

Mitchell, J.K. 1993. Fundamentals of soil behaviour. Second edition, Canada, Wiley.

Mitchell, J.K. & Madsen, F.T. 1987. Chemical Effects on Clay Hydraulic Conductivity. Geotechnical Practice for Waste Disposal ´87. *Geotechnical Special Publication*, 13, pp. 87–116.

Mitchell, J.K. & Songa, K. 2005. Fundamentals of soil behaviour. Third edition, Canada, Wiley.

Morris, J.W.F. & Barlaz, M.A. 2011. A performance-based system for the long-term management of municipal waste landfills. *Waste Management,* 31 (4), pp. 649–662.

Morris, J.W.F., Crest, M., Åkerman, A., Bucka, A., Presse, D., Sayward-Higham, S. & Barlaz, M.A. 2009. Application of the EPCC methodology to evaluate functional stability at closed MSW landfills in Europe. Proceedings Sardinia 2009, 12th International Waste Management and Landfill Symposium, 1–5 Oct. 2009, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Mälkki, E. 1999. Pohjavesi ja pohjaveden ympäristö (Groundwater and groundwater environment). Helsinki: Tammi Oy. (In Finnish.)

Neshat, A., Biswajeet, P. & Saman, J. 2015. Risk assessment of groundwater pollution using Monte Carlo approach in an agricultural region: An example from Kerman Plain, Iran. *Computers, Enviroment and Urban Systems,* 50 (2015), pp. 66-73.

Norotte, V., Didier, G., Guyonnet, D. & Gaucher, E. 2004. Evolution of GCL hydraulic performance during contact with landfill leachate. Advances in Geosynthetic Clay Liner Technology: 2nd Symposium, ASTM STP 1456, R.E. Mackey and K. von Maubeuge, Eds., ASTM International, West Conshohocken, PA, May 2004. pp: 41–52.

Ogata, A. 1970. Theory of dispersion in granular medium. US Geological Survey Professional Paper 411-I.

Ogata, A. & Banks, R.B. 1961. A solution of the differential equation of longitudinal dispersion in porous media; Fluid movement in Earth materials.

Olivier, F., Oxarango, L., Mugnier, V., Tinet, A.J. & Marcoux, M.A. 2009. Estimating the drawdown of leachate in a saturated landfill: 3D modelling based on field pumping tests, in: Proceedings Sardinia 2009, 12th International Waste Management and Landfill Symposium, 1–5 Oct. 2009, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Den Ouden, J. & Backhuijs, F. 1999. Dutch guidelines for bottom isolation and site specific alternatives. Proceedings Sardinia 99, Seventh International Waste Management and Landfill Symposium 14–18 Oct. 1999, S. Margherita di Pula, Cagliari, Italy.

Omidi, G.H., Thomas, J.C. & Brown, K.W. 1996. Effect of desiccation cracking on the hydraulic conductivity of a compacted clay liner. Texas A&M University, Soil and crop science department, College Station Texas 77843-2474, USA. *Water, Air & Soil Pollution*, 89 (1–2), pp. 91–103.

Othman, M.A. & Benson, C.H. 1993. Effect of freeze-thaw on the hydraulic conductivity and morphology of compacted clay. *Canadian Geotechnical Journal*, 30 (2), pp. 236–246.

Othman, M.A. & Benson, C.H. 1992. Effect of freeze-thaw on the hydraulic conductivity of three compacted clays from Wisconsin. Transp. Res. Rec. 1369, Transp. Res. Board, Washington, D.C., pp. 118–125.

Othman, M.A. & Benson, C.H. 1991. Influence of freeze-thaw on the hydraulic conductivity of a compacted clay. Proc., 14th Annual Madison Waste conference, University of Wisconsin, Madison, Wis., pp. 296–312.

Othman, M.A., Benson, C.H., Chamberlain, E.J. & Zimmie, T.F. 1994. Laboratory testing to evaluate changes in hydraulic conductivity of compacted clays caused by freeze-thaw: state-of-the-art. Hydraulic conductivity and waste containment transport in soils, ASTM STP 1142, D. E. Daniel and S. J. Trautwein, eds., American society for Testing and Materials, Philadelphia, Pa., pp. 227–254.

Ortner, M.E., Knapp, J.& Bockreis, A. 2014. Landfill mining: Objectives and assessment challenges. *Proceedings of Institution of Civil Engineers: Waste and Resource Management*, Volume 167, Issue 2, pp. 51-61.

Ozcoban, M.S., Erarslan, I. & Tufekci, N. 2006. Influence of leachates on permeability and removal rate of compacted clay. *Fresenius Environmental Bulletin*, 15 (9 B), pp. 1106–1110.

Ozcoban, M.S., Tufekci, N., Tutus, S., Sahin, U. & Celik, S.O. 2006. Leachate removal rate and the effect of leachate on the hydraulic conductivity of natural (undisturbed) clay. *Journal of Scientific and Industrial Research,* 65 (3), pp. 264–269.

Park, J.K., Sakti, J.P. & Hoopes, J.A. 1996. Transport of aqueous organic compounds in thermoplastic geomembranes. II: Mass flux estimates and practical implications. *Journal of Environmental Engineering, ASCE,* 122 (9), pp. 807–813.

Petrov, R.J., Rowe, R.K. & Quigley, R.M. 1997. Comparison of laboratory-measured GCL hydraulic conductivity based on three permeameter types. *Geotechnical Testing Journal,* 20 (1), pp. 49–62.

Pivato, A. & Morris, J.W.F. 2005. Performance-based methodology for assessing the aftercare end-point for landfills. In: Proc. International Seminar and Workshop on Sustainable Landfilling, 13–15 June, 2005, Abbey of Praglia (Padua), Italy.

Pivato, A. 2004. Environmental sustainability of landfills: performance-based methodology for assessing the aftercare end-point. PhD Thesis. IMAGE Department. University of Padova, Italy.

Pollard, S.J.T., Smith, R, Longhurst, P.J., Eduljee, G.H. & Hall, D. 2006. Recent developments in the application of risk analysis to waste technologies. *Environment International,* Volume 32, pp.1010–20.

Purdy, S. & Suryasasmita, V. 2006. Comparison of hydraulic conductivity test methods for landfill Clay liners. In: Proceedings Sardinia 2007, Eleventh International Waste Management and Landfill Symposium, 1–5 Oct. 2007, S. Margherita di Pula-Cagliari, Sardinia, Italy.

Ramke, H.G. 2009 Leachate collection systems in bottom liner systems. 12th International Landfill Symposium 5–9 Oct. 2009, in Sardinia, Italy.

Riggenbach, J.D., Streeter, R.A. & Hess, S.C. 1991. Risk assessment methods for establishing clean closure levels. *Water Science and Technology*, Volume 24,Issue 12, pp. 57-76.

Rinkinen, J., Sikiö, J., Forsman, J., Uusihakala, M. & Pammo, R. 2009. Maintainance facilities at the Ämmässuo bioreactor landfill. 12th International Landfill Symposium 5–9 Oct. 2009, in Sardinia, Italy.

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

Rowe, R.K. 1998. Geosynthetics and the minimization of contaminant migration through barrier systems beneath solid waste. Proceedings of the Sixth International Conference on Geosynthetics 1, pp. 27–102.

Rowe, R.K. 1987. Pollutant transport through barriers. *Geotechnical Practice for Waste Disposal, ASCE, GSP*, 26, pp. 159–181.

Rowe, R.K. & Booker, J.R. 2005. POLLUTE v7—Pollutant migration through a nonhomogeneous soil, distributed by GAEA Environmental Engineering, Ltd., Ontario, Canada.

Rowe, R.K. & Booker, J.R. 1995. A finite layer technique for modelling complex landfill history. *Canadian Geotechnical Journal,* 32 (4), pp. 660–676.

Rowe, R.K. & Brooker, J.R. 1991. Pollutant migration through liner underlain by fractured soil. *Journal of Geotechnical Engineering*, 117 (12), pp. 1902–1919.

Rowe, R.K. & Iryo, T. 2005. Effect of geosynthetics on the hydraulic performance of leak-detection systems. *Canadian Geotechnical Journal*, 42 (5), pp. 1359–1376.

Rowe, R.K., Lake, C.B. & Petrov, R.J. 2000. Apparatus and Procedures for Assessing Inorganic Diffusion Coefficients for Geosynthetic Clay Liners. *Geotechnical Testing Journal,* 23 (2), pp. 206–214.

Rowe, R.K. & Orsini, C. 2003. Effect of GCL and subgrade type on internal erosion in GCLs under high gradients. *Geotextiles and Geomembranes,* 21 (1), pp. 1–24.

Rowe, R.K., Rayhani, M.T., Take, W.A., Siemens, G. & Brachamn, R.W.I. 2011. GCL hydratation under simulated daily thermal cycles. *Geosynthetics International*, 18 (4), pp. 196–205.

Rowe, R.K., Quigley, R.M. & Booker, J.R. 2004. Clayey Barrier Systems for Waste Disposal Facilities. Second edition, Oxon United Kingdom, Spon Press.

Rowe, R.K., Quigley, R.M. & Booker, J.R. 1995. Clayey Barrier Systems for Waste Disposal Facilities. First edition, Oxon United Kingdom, Spon Press.

Rowe, R.K. & San, E.K.W. 1994. Effect of leachate mounding and sand lenses on contaminant migration from a landfill cell. *Computers and Geotechnics,* 16 (3), pp. 173–204.

RVF, Sverige AB. 2006. Funktion för geologisk barriär och bottentätning. 01 (Funktion of the geological barrier and covering). (In Swedish.)

Sangam, H.P. & Rowe, R.K. 2005. Effect of surface fluorination on diffusion through a high density polyethylene geomembrane. *Journal of Geotechnical and Geoenvironmental Engineering*, 131 (6), pp. 694–704.

Sarkkila, J., Kuusiniemi, R., Forsten, L. & Manni-Rantanen, L. 2006. Asfalttiset ympäristönsuojausrakenteet. Suomen Ympäristökeskus; ympäristöopas (Asphalt structures in environmental protection. Finnish Environment Institute: Environmental guidebook). (In Finnish.)

Schroeder, P.R., Dozier, T.S., Zappi, P.A., McEnroe, B.M., Sjostrom, J.W. & Peton, R.L. 1994. The Hydrologic Evaluation of Landfill Performance (HELP) Model:Engineering Documentation for Version 3, EPA/600/R-94/168b, US. Environmental Protection Agency, Risk Reduction Engineering Laboratory, Cincinnati, OH.

Scientific Software Group. 1998. Environmental Software and Publications (Catalogue), Washington, DC 20020 3041. http://www.scisoftware.com/products/visual_help_details/visual_help_details.html.

Seppälä, M. 1999. Geomorphological aspects of road construction in a cold environment in Finland. *Geomorphlogy*, 31, pp. 65-91.

Shackelford, C.D. & Glade, M.J. 1997. Analytical mass leaching model for contaminated soil and soil stabilized waste. *Ground Water,* 35 (2), pp. 233–242.

Shackelford, C.D. & Daniel, D.E. 1991. Diffusion in saturated soil. II. Results for compacted clay. *Journal of Geotechnical Engineering,* 117 (3), pp. 485–506.

Shackelford, C.D. 1988. Diffusion as a transport process in fine-grained barrier materials. 25, pp. 24–27.

Sharma, H.D. & Lewis, S.P. 1994. Regulatory framework and waste characterization and soil-waste interaction. Waste Containment Systems, Waste Stabilization, and Landfills: Design and Evaluation. First Edition, Canada, Wiley.

Sizirici, B. 2009. Risk based post closure care analysis for Florida landfills. College of Engineering and Computing. PhD. Florida International University, Miami.

Sizirici, B. & Tansel, B. 2010. Projection of landfill stabilization period by time series analysis of leachate quality and transformation trends of VOCs. *Waste Management,* 30 (1), pp: 82–91.

Sizirici, B., Tansel, B. & Kumar, V. 2011. Knowledge based ranking algorithm for comparative assessment of post-closure care needs of closed landfills. *Waste Management,* 31 (6), pp: 1232–1238.

Smith, D., Pivonka, P., Jungnickel, C. & Fityus, S. 2004. Theoretical analysis of anion exclusion and diffusive transport through platy-clay soils. *Transport in Porous Media*, 57 (3), pp: 251–277.

Smith, R.M. 1984. Images of the world. An atlas of satellite imagery and maps, with interpretive supplement.

Solomon, S.D., Qin, M., Manning, Z., Chen, M., Marquis, K.B., Averyt, M. & Miller, H.L. (eds.) 2007. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007. Cambridge, United Kingdom and New York, NY, USA, pp. 1007

Sormunen, K., Ettala, M. & Rintala, J. 2008. Detailed internal characterisation of two Finnish landfills by waste sampling. *Waste Management* 28, pp. 151-163.

Suopajärvi, H., Pongracz, E. & Fabritius, T. 2014. Bioreducer use in Finnish blast fucnace ironmaking – Analysis of CO_2 emission reduction potential and mitigation cost. *Appl. Energy 124,* pp. 82-93.

TA, Abfall. 1991. Zweite allgemeine Verwaltungsvorschrift zum Abfallgesetz. Teil 1: Technische Anleitung zur Lagerung, chemisch/physischen, biologischen Behandlung, Verbrennung und Ablagerung von besonders überwachungsbedürftigen Abfällen (Second general administrative provision to the Waste avoidance and waste management act, Part 1: Technical instructions on the storage, chemical, physical and biological treatment, incineration and deposition of waste requiring particular supervision). 12.3.1991. (In German.)

TA, Siedlungsabfall. 1993. Dritte Allgemeine Verwaltungsvorschift zum Abfallgesetz. Technische Anleitung zur Vermeidung, Verwertung, Behandlung und sonstigen Entsorgung von Siedlungsabfällen (Third general administrative provision to the Waste avoidance and waste management act: Technical instructions on recycling, treatment and disposal of municipal waste). 14.5.1993. (In German.)

Tammirinne, M., Juvankoski, M., Laaksonen, R. & Rathymayer, H. 2004. Kaatopaikan tiivistysrakenteiden ja –materiaalien tuotehyväksyntä. Menettelyopas vapaaehtoiselle tuotehyväksynnälle. VTT Rakennus- ja yhdyskuntatekniikka, Espoo. pp. 24 (in Finnish).

Thomasson, M.J. & Wierenga, P.J. 2003. Spatial variability of the effective retardation factor in an unsaturated field soil. *Journal of Hydrology,* 372, pp. 213–225.

Townsend, T.G., Wise, W.R. & Jain, P. 2005. One-dimensional gas flow model for horizontal gas collection systems at municipal solid waste landfills. *Journal of Environmental Engineering,* 131 (12), pp. 1716-1723.

US EPA (United States Environmental Protection Agency) 2008. Application of watershed ecological risk assessment methods to watershed management. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment. No. EPA/600/R-06/037F.

US EPA (United States Environmental Protection Agency). 2004. Test Methods Subtopics.

US EPA (United States Environmental Protection Agency). 1997. Guiding Principles for Monte Carlo Analysis.

van Genuchten, M.T. 1980. Closed-form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Science Society of America Journal*, 44 (5), pp. 892–898.

van Genuchten, M.T., Leij, F.J. & Yates, S.R. 1991. The RETC Code for Quantifying the Hyrdaulic Function of Unsaturated Soil EPA/600/2-91-065. U.S. Protection Agency, Washington DC.

Varank, G. 2010. Investigation of contaminant transport through solid waste landfills liners and rehabilitation of landfill liners, Ph.D. Thesis, Institute of Science, Department of Environmental Engineering, Yildiz Technical University, Istanbul, Turkey. (In Turkish.)

Varank, G., Demir, A., Top, S., Sekman, E., Akkaya, E., Yetilmezsoy, K. & Bilgili, M. 2011. Migration behavior of landfill leachate contaminants through alternative composite liners. *Science of the Total Environment,* 409 (2011), pp. 3183–3196.

Venäläinen, A., Tuomenvirta, H., Lahtinen, R. & Heikinheimo, M. 2001. The influence of climate warming on soil frost on snow-free surface in Finland. *Climate Change*, 50, pp. 111-128.

VNa 214/2007. 2007. Valtioneuvoston asetus maaperän pilaantuneisuuden ja puhdistustarpeen arvioinnista. (Finnish Government degree of the base ground putridness and the estimation of cleaning demands. 1.3.2007. (In Finnish.)

VNa 202/2006, 2006. Valtioneuvoston asetus kaatopaikoista annetun valtioneuvoston päätöksen muuttamisesta. Ympäristöministeriö. (Finnish Government decree on altering the government decision on landfills. Ministry of the Environment.) 23.3.2006. (In Finnish.)

VNa 552/2001, 2001 Valtioneuvoston asetus kaatopaikoista annetun valtioneuvoston päätöksen muuttamisesta annetun valtioneuvoston päätöksen voimaantulopäätöksen muuttamisesta. Ympäristöminiteriö. (Finnish Government decree on altering the implementing decision of the altering of the government decision on landfills. Ministry of the Environment.)13.6.2001. (In Finnish.)

VNp 1049/99, 1999. Valtioneuvoston päätös kaatopaikoista annetun valtioneuvoton päätöksen muuttamisesta. Ympäristöministeriö. (Finnish Government decision on altering the government decision on landfills. Ministry of the Environment.) 18.11.1999, pp. 36. (In Finnish.)

VNp 861/97, 1997. Valtioneuvoston päätös kaatopaikoista. Ympäristöministeriö. (Finnish Government decision on landfills. Ministry of the Environment.) 4.9.1997. (In Finnish.)

VROM (Netherlands Ministry of Housing, Physical Planning and the Environment), 1993. Richtlijn onderafdichtingsconstructies voor stort- en opslaglaatsen. (Guidelines for sealed structures in landfills and final storages.) Soil Protection publication Series 1993/2. (In Dutch.)

VROM (Netherlands Ministry of Housing, Physical Planning and the Environment), 1991. Richtlijnen voor dichte eindafwerking op afval- en reststofbergingen. (Guidelines for the construction of impervious final covers for waste-product storing sites and landfills). *Soil protection publication series* 1991/2. (In Dutch.)

Wang, H. 2011. Design for composite liners of landfill under alternating cycles of wetting-drying and freezing-thawing. *Advanced Materials Research*, volume 163-167, pp. 4464-4469.

Weber, C.T. & Zornberg, J.G. 2007. Leakage through geosynthetic dam lining systems. In: Proceedings Sardinia 2007, Eleventh International Waste Management and Landfill Symposium, 1–5 Oct. 2007, S. Margherita di Pula-Cagliari, Sardinia, Italy, pp. 835-844.

Whittaker, J.J., Buss, S.R., Herbert, A.W. & Fermor, M. 2001. Benchmarking and guidance on the comparison of selected groundwater risk assessment models. *National Groundwater*, 7 Contaminated Land Center (NGACLC) Report NC/00/14, Environment Agency, pp. 36.

Wong, L.C. & Haug, M.D. 1991. Cyclical closed-system freeze-thaw permeability testing of soil liner and cover materials. *Canadian Geotechnical Journal*, 208 (6), pp. 784–793.

Xie, J. & Chen, J. 2014. Numerical simulation of landfill gas migration around a vertical extraction well in Xiangshan landfill. *Energy, Sources,* Part A: Recovery, Utilization and Environmental Effects, volume 36, Issue 16, pp. 1764-1773.

Xie, H.J., Chen, Y.M. & Lou, Z.H. 2010. An analytical solution to contaminant transport through composite liners with geomembrane defects. *Science China, Technological Sciences*, 53 (5), pp. 1424–1433.

Xue, O. & Zhang, O. 2014. Effects of leachate concentration on the integrity of solidified clay liners. *Waste Management and Research*, Volume 32, Issue 3, pp. 198-206.

Ym 426 Jätelaki V:1.1.1994 A:3.12.1993, SK:1072/1993, 1994. Ympäristöministeriö. (Finnish Waste Act. Ministry of the Environment.). (In Finnish.)

Ym 427 Jäteasetus V:1.1.1994 A:22.12.1993, SK:1390/1993, 1994. Ympäristöministeriö. (Finnish Waste Decree. Ministry of the Environment.). (In Finnish.)

Zaporozec, A. 2004. Groundwater contamination inventory: A methodological guide. International Hydrological Program (IHP) VI Groundwater Series No. 2. UNESCO, pp. 160.

Zhou, J., Liu, Y., Zhang, X. & Wang, D. 2014. Uncertain risk aversion. *Journal of Intelligent Manufacturing,* December 2014, pp. 10.

Zimmie, T.F. 1992. Freeze-thaw effects on the permeability of compacted clay liners and covers. *Geotechnical News*, 6/1992 pp. 28–30.

Zimmie, T.F. & LaPlante, C.M. 1990. The effects of freeze/thaw cycles on the permeability of fine-grained soil. Proc. 22nd Mid-Atlantic Industrial Waste Conference, Drexel University, Philadelphia, Pa., pp. 580–59.

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