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Impact of the technical cover systems and landfill reclamation works on selected environmental components

Wpływ technicznych sposobów przykrycia i rekultywacji składowiska na wybrane komponenty środowiska

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Statement of the supervisor of the doctoral thesis

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List of selected key abbreviations in the thesis:

Symbol	Description	Units
γ	unit weight of the cover soil	kN/m^3
ϕ	friction angle of the cover soil	°
c	cohesion of the cover soil	kPa
CCL	compacted clay liner	m
FS	factor of safety	-
GM	geomembrane	mm
h	thickness of the cover soil layer	m
HDPE	high-density polyethylene	-
HELP	Hydrologic Evaluation of Landfill Performance	-
HMs	heavy metals	mg/l, mg/kg d.m.
IG	germination inhibition	%
IR	root inhibition	%
k	hydraulic conductivity	m/s
LFG	landfill gas	%, m^3
LOI	loss on ignition	%
LPI	leachate pollution index	-
MSW	municipal solid waste	kg, m, kg/m^3
WM	waste management	-
w_L	liquid limit	%
w_p	plastic limit	%
I_p	plasticity index	%
I_L	liquidity index	%

Streszczenie

Wpływ technicznych sposobów przykrycia i rekultywacji składowiska na wybrane komponenty środowiska

Praca doktorska koncentrowała się na kompleksowej ocenie technik rekultywacji składowisk odpadów komunalnych poprzez porównanie dwóch najczęściej stosowanych systemów izolacji powierzchniowych: mineralnego i syntetycznego, zalecanych odpowiednio w Polsce i Republice Czeskiej. Badania przeprowadzono na dwóch częściowo-zrekultywowanych składowiskach odpadów komunalnych w Zakroczymiu (Polska) oraz w Zdounkach (Republika Czeska), z zastosowaniem holistycznej, ośmiomodułowej metodologii integrującej inżynierię lądową, geotechnikę środowiskową, hydrogeologię, nauki chemiczne i biologiczne. Do oceny skuteczności systemu przykrycia składowiska porównano zabiegi rekultywacji technicznej i biologicznej, wykorzystując wieloletni monitoring wód podziemnych, odcieków i gazu składowiskowego. Wykonano modelowanie prognozujące ilość odcieków, emisję gazu składowiskowego i stateczność skarp. Analizę uzupełniły testy biomonitoringowe z nasionami *Sinapis alba* L. oraz badania respiracji gruntu, które wykazały wpływ odcieków na aktywność biologiczną i rozwój roślin. Uzyskane wyniki posłużyły do opracowania praktycznych rekomendacji dotyczących doboru i zastosowania systemów uszczelniających przy zamykaniu składowisk odpadów uwzględniających uwarunkowania techniczne, środowiskowe, ekonomiczne oraz społeczne, dostarczając projektantom narzędzi wspierających podejmowanie decyzji w doborze systemu przykrycia składowiska. Zaproponowana koncepcja wypełnia lukę w ogólnodostępnych wytycznych, oferując interdyscyplinarne podejście do bezpiecznego i zrównoważonego zagospodarowania zamykanych składowisk. Realizacja celów pracy wykazała, że połączenie badań terenowych, laboratoryjnych oraz modelowania umożliwia uzyskanie kompleksowego obrazu procesów zachodzących w składowiskach odpadów.

Słowa kluczowe: składowisko, odcieki, gaz składowiskowy, zanieczyszczenie, badania monitoringowe, badania modelowe, rekultywacja

Summary

Impact of the technical cover systems and landfill reclamation works on selected environmental components

The dissertation focused on a comprehensive evaluation of the municipal solid waste landfill reclamation methods by comparing the two most commonly used covers systems: mineral and synthetic recommended in Poland and the Czech Republic, respectively. The study was conducted on two partially reclaimed landfills in Zakroczym (Poland) and in Zdounky (Czech Republic) using a holistic, eight-module methodology integrating civil engineering, environmental geotechnics, hydrogeology, chemical and biological studies. To evaluate the effectiveness of the landfill cover system, technical and biological reclamation treatments were compared using long-term monitoring of groundwater, leachate and landfill gas. Modeling was conducted to predict the amount of leachate, landfill gas emissions and slope stability. The analysis was supplemented by biomonitoring tests with *Sinapis alba* L. seeds and soil respiration tests, which showed the effect of leachate on biological activity and plant growth. The results were used to develop practical recommendations for the selection and application of cover systems for landfill closure, considering technical, environmental, economic and social aspects, providing engineers decision-support tools for selecting a landfill cover system. The proposed concept fills a gap in publicly available guidelines, offering an interdisciplinary approach to the safe and sustainable management of landfill closures. The realization of the thesis objectives showed that the combination of field, laboratory and modeling studies makes it possible to obtain a comprehensive insight of landfill processes.

Keywords: landfill, leachate, landfill gas, pollution, monitoring studies, modelling, reclamation

1. Introduction

This dissertation is the result of scientific research conducted during three long-term internships abroad and intensive research and practical activities at the Institute of Civil Engineering at SGGW.

As part of the prestigious Fulbright Junior Research Award scholarship, Aleksandra Jakimiuk, M.Sc. completed a research internship at Stanford University in the USA (01.10.2024–31.01.2025 conducting a research project entitled "*Research on the impact of various technical methods of covering and reclaiming landfill sites on internal processes within them and the overall environmental safety*". Ph.D. candidate also developed her research skills during an Erasmus+ internship at Okayama University in Japan (01.09.2022–01.11.2022), where she worked on the project "*Research on sustainable waste management and food loss modelling*". Additionally, as part of the Własny Fundusz Stypendialny SGGW scholarship (01.04.2022–30.06.2022), she completed an internship at Mendel University in Brno in Czech Republic, working on the project *"Influence of reclaimed landfills on environmental components*" during which she completed biomonitoring studies. Aleksandra Jakimiuk, M.Sc. is a co-author of 23 scientific papers (*h*-index=8, according to the Scopus database), including 12 of them, the Ph.D. candidate research results were partially published in the following papers:

- **Jakimiuk, A.**, Koda, E., Goli, V.S.N.S., Podlasek, A., Winkler, J., Singh, Y., Vaverková, E., Varma, M., Zarębska-Michaluk, D., Singh D.N., Vaverková, M.D. (2025). COVID-19 pandemic-induced medical waste in the Anthropocene: Generation, management, and environmental impact. *The Anthropocene Review, 0*(0).

- **Jakimiuk, A.**, Podlasek, A., Vaverková, M.D., Koda, E. (2025). Impact of the Reclamation and Capping System of MSW Landfills on the Environment. *Sustainable Infrastructures, Proceedings of EGRWSE-23,* Vol. 3 Springer Nature.

- Podlasek, A., Koda, E., Kwas, A., Vaverková, M.D., **Jakimiuk, A**. (2025). Anthropogenic and Natural Impact on Surface Water Quality: The Consequences and Challenges at the Nexus of Waste Management Facilities, Industrial Zones, and Protected Areas. *Water Resources Management 39*, 1697–1718.

- Podlasek, A., Vaverková, M.D., **Jakimiuk, A.**, Koda, E. (2024). A comprehensive investigation of geoenvironmental pollution and health effects from municipal solid waste landfills. *Environmental Geochemistry and Health*, *46*, 97.

- Podlasek, A., Vaverková, M.D., **Jakimiuk, A.**, Koda, E. (2024). Potentially toxic elements (PTEs) and ecological risk at waste disposal sites: An analysis of sanitary landfills. *PLoS One*, *19*(5), e0303272.

- Vaverková, M.D., Paleologos, E., Goli, V.S., Koda, E., Mohammad, A., Podlasek, A., Winkler, J., **Jakimiuk, A.**, Cerny, M., Singh, D.N. (2023). Environmental impact of landfills: perspectives on biomonitoring. *Environmental Geotechnics*, 1-11.

- Podlasek, A., Vaverková, M.D., Koda, E., **Jakimiuk, A.**, Barroso, P.M. (2023). Characteristics and pollution potential of leachate from municipal solid waste landfills: Practical examples from Poland and the Czech Republic and a comprehensive evaluation in a global context. *Journal of Environmental Management*, 332, 117328.

- **Jakimiuk, A.**, Matsui, Y., Podlasek, A., Koda, E., Goli, V.S.N., Voberková, S., Singh, D.N., Vaverková, M.D (2023). Closing the loop: A case study on pathways for promoting sustainable waste management on university campuses. *Science of the Total Environment, 892*, 164349.

- **Jakimiuk, A.**, Matsui, Y., Podlasek, A., Vaverková, M.D. (2022). Assessment of landfill protection systems in Japan – a case study. *Acta Scientiarum Polonorum Architectura*, *21*(4), 21-31.

- Podlasek, A., **Jakimiuk, A.**, Vaverková, M.D., Koda, E. (2021). Monitoring and Assessment of Groundwater Quality at Landfill Sites: Selected Case Studies of Poland and the Czech Republic. *Sustainability*, *13*(14), 7769.

- Rose P. E., Guidi N. W., Adamcová D., Podlasek A., **Jakimiuk A.**, Vaverková M.D. 2022. *Sinapis alba* L. and *Triticum aestivum* L. as a biotest model species for evaluating municipal solid waste leachate toxicity. *J. of Environmental Management*, 302, 114012.

- **Jakimiuk, A.** (2022). Review of technical methods of landfill sealing and reclamation in the world. *Acta Scientiarum Polonorum Architectura, 21*(1), 41-50.

Additionally, the Ph.D. candidate actively participated in the preparation of research expert opinions concerning sanitary and geotechnical expert opinion, and environmental impact assessments for the following projects:

- Sanitary expert opinion for the landfill site in Zakroczym, related to the planned construction of a 500 kWp photovoltaic installation on a reclaimed landfill cell. Client: PG INWEST Sp. z o.o., March 2023.

- Geotechnical expert opinion for the landfill site in Zakroczym, related to the planned construction of a 500 kWp photovoltaic installation on a reclaimed landfill cell. Client: PG INWEST Sp. z o.o., May 2023.

- Sanitary expert opinion for the reclaimed landfill site in Łubna, related to the planned investment project "*Construction of the Warsaw Green Energy Center in Łubna*." Client: MPO Warsaw, June 2023.

- Geotechnical expert opinion for the reclaimed landfill site in Łubna, related to the planned investment project "*Construction of the Warsaw Green Energy Center in Łubna*." Client: MPO Warsaw, June 2023.

- Expert opinion on the potential environmental impact of adopted design solutions for the planned Municipal Waste Landfill of the City of Waco, located in McLennan and Limestone Counties, Texas, USA. Client: SAXON Loomis Consulting Group, August 2023.

- Environmental review of the landfill site in Łubna, Kalwaria Municipality. Client: MPO Warsaw, October 2023.

- Report from conducting two field tests of natural filtration coefficient and two field tests of an artificial geological barrier at the construction site of the southern cell –stage II, landfill site in Zakroczym, Byłych Więźniów Twierdzy Zakroczym Street. Client: PG INWEST Sp. z o.o., July 2022.

- Documentation of soil investigation for the project of reinforcing the subsoil of Cell No. 4 embankment and basin at the landfill site in Rusko. Client: ENERIS Sp. z o.o., October 2023.

1.1. The problem of landfilling

Around the world, waste management (WM) seems to be a challenging task with serious implications for the health, conservation of natural resources, stability, and sustainable wealth of a nation (Yaashikaa et al., 2022). Landfilling of untreated municipal solid waste (MSW) is still seen as an acceptable practice in many developed countries (Madon et. al., 2020). According to Sabour et al. (2020) and Yatoo et al. (2024), landfilling is the most common method of waste disposal worldwide, accounting for a total of 70% of WM, including 33% open dumps, 25% unspecified landfills, 8% sanitary landfills and 4% controlled landfills. The method of MSW disposal in landfills is still widely accepted and practiced due to its economic advantages (Vaverková, 2019). Nevertheless, the European Union (EU) is taking steps to minimize landfilling, and in 2022, about 22.8% of the MSW generated in the EU was landfilled, down from 51.1% in 2000. This rapid decline is driven by the EU target of sending no more than 10% of MSW to landfill by 2035 (Statista, 2024). This is due to the negative impacts observed at landfills, which the most common include: the generation of greenhouse gases (GHG) (landfills account for approximately 29% of all GHG, representing more than 15% of the average global share (Mor and Ravinda, 2023), air pollution from volatile organic compounds (VOCs), particulate matter, and allergenic pollen (Sharma and Sinha, 2023), contamination of groundwater and surface water by leachate, as well as soil contamination and threats to human health (Podlasek et al., 2024), production of unpleasant odours (Li et al., 2024) and degradation of the natural landscape (Wen et al., 2023). Methane (CH_4) emissions from landfills account for about 10% of all anthropogenic CH_4 emissions worldwide, and about 50 Tg per year (Wang et al., 2024). CH_4 is currently the largest source of GHG emissions from the solid waste sector worldwide, and emissions are expected to increase owing to increasing waste generation, especially in countries where biodegradable waste is still landfilled (Gebert et al., 2022). CH_4 is more than 28 times more potent than carbon dioxide (CO_2) at trapping heat in the atmosphere (EPA, 2025). Reducing CH_4 emissions is also a priority for mitigating climate change (Maasakers et al., 2022). All the landfill connections and their associated risks and environmental impacts are well illustrated by the co-occurrence network based on 2363 records in Fig. 1.1.

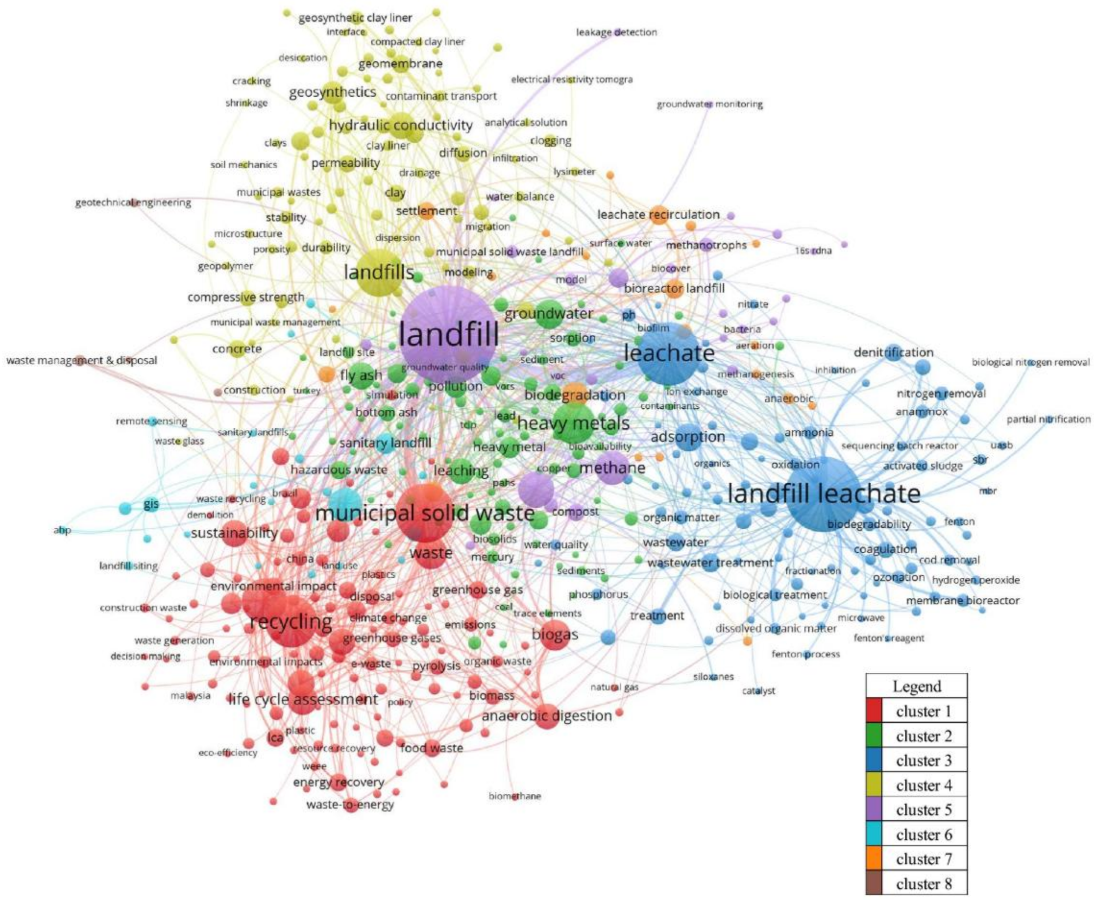


Figure 1.1. Keyword collaborations based on co-occurrence relationships that met the repetition threshold, acc. to Sabour et al. (2020).

Landfills are unconventional engineering structures, and their specificity results from their large surface area (up to several dozen hectares), high capacity (up to several million cubic meters), considerable thickness (up to several dozen meters), long operational period, and potential environmental impact even after the end of their use (Koda, 2011). Location in inadequate geotechnical conditions and lack of properly selected sealing and soil protection are the main reasons for the formation of environmental risks from landfills. In some circumstances, the locations of landfills could be exposed to natural hazards, such as flooding and earthquakes (Przydatek, 2019). Problems related to landfills also occur, when they are appropriately designed, constructed and operated. This can be associated with waste decomposition processes, which take up to several decades, and with the formation of landfill gases (LFG) and leachates, as a result of these processes. Landfill reclamation is a process that contributes to the restoration of the natural and functional values of degraded land and slow down the landfill processes.

The lack of uniform and detailed guidelines on how to perform landfill reclamation, as well as gaps in the scientific literature on which materials are more suitable, create a need for scientific research in this direction. Research on the mechanical properties of various landfill covers has long attracted significant interest. The first studies focusing on mineral and synthetic cover appeared as early as 1997 (Manassero et al., 1997), and since then, further attempts have been made to improve and adapt these covers to evolving technological and environmental conditions (Staub et al., 2011; Li et al., 2020; Müller and Wöhlecke, 2019; Chetri et al., 2022; Ojasanya and Dewoolkar, 2024). Due to ongoing climate change, landfills are now under intense observation, as CH_4 emissions from these facilities pose a serious threat to the environment (Askr et al., 2024; Wang et al., 2024). Landfills need to be assessed from a multidisciplinary perspective due to their many impacts and unpredictability. A holistic perspective on the reclamation process is crucial for ensuring the successful implementation of technical measures, as it requires addressing numerous unconventional issues related to geotechnics, civil and sanitary engineering, chemistry, and hydrogeology. Equally important is the biological aspect, wherein the selection of appropriate plant species plays a significant role in restoring areas to their original functionality and can also contribute to the reclamation of contaminated soil.

In order to successfully evaluate the effectiveness of the implemented solutions, long-term observations under real conditions are necessary, since laboratory simulations cannot express the full complexity of the processes taking place at landfill. An interdisciplinary approach to the topic, combining several fields and scientific disciplines, is crucial, and an engineering perspective, as well as an environmental one, provides opportunities for a constructive and scientifically sound solution to the problem of ensuring proper landfill reclamation.

Therefore, the author of this thesis attempted to evaluate the interdisciplinary impact of two different technical reclamation methods using synthetic or mineral liner on selected environmental components in order to determine the more effective method. Two similar MSW landfills in the Czech Republic (CR) reclaimed with high-density polyethylene geomembrane (HDPE GM) and in Poland (PL) reclaimed with compacted clay liner (CCL) were selected for evaluation.

1.2. Study objectives and research hypothesis

The objectives of the thesis are to:

- To conduct a comparative analysis of technical and biological reclamation at the Zakroczym (PL) and Zdounky (CR) MSW landfills,
- To conduct a comparative analysis of the monitoring test results for groundwater, leachate, and CH_4 in LFG at the selected landfills,
- To identify the environmental risks and technical safety of the landfills by creating predictive models of leachate production, LFG generation and performing slope stability analysis,
- To determine the effect of landfill leachate on *Sinapis alba* L. seeds and on the respiration rate of soils used in the reclamation of the studied landfills,
- To develop recommendations for the use of selected liners in landfill reclamation.

To achieve the objectives of the thesis, the following research hypothesis was formulated: *The type of cover used for technical reclamation (a 1 mm thick HDPE geomembrane or mineral liner with $k \leq 10^{-7}$ m/s) influences the quantity and composition of the generated landfill gas and leachate.*

1.3. Scope of the work

The scheme in Fig. 1.2 presents a simplified research methodology that includes four key stages for evaluating the effectiveness of the reclamation methods used: (i) monitoring, (ii) laboratory tests, (iii) modelling and (iv) biomonitoring studies aimed at determining the variability in the composition of landfill leachate and gases, assessing soil properties, evaluating risks and investigating the impact of leachate on plants.

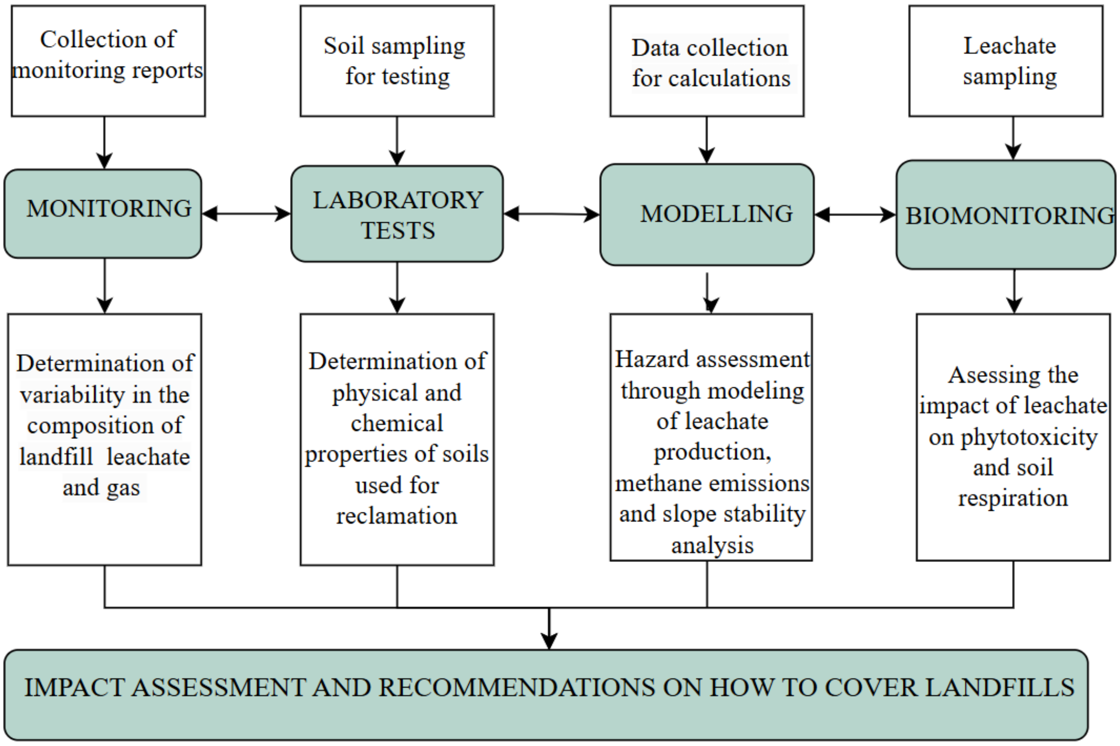


Figure 1.2. Scope of the research work.

Below a summary of the research and analysis conducted in each chapter of the thesis is presented:

Chapter 1. This chapter serves as an introduction to the thesis, outlining the primary issue of waste disposal and emphasizing its increasing relevance in light of the risks associated with landfills. In this chapter, the research objectives are defined, focusing on developing solutions to establish recommendations for an effective method of landfill reclamation. In addition, a research hypothesis is formulated and the scope of the study is outlined, highlighting the key aspects that will be analyzed in subsequent sections of the thesis. A research gap is also identified, which justifies undertaking new, comprehensive research in this field.

Chapter 2. This chapter describes the processes that occur in MSW landfills and the hazards associated with these facilities. In addition, the chapter provides a detailed description of the leachate formation process including its composition, characteristics, and factors influencing its intensity and chemical composition. It provides discussion of landfill gas production, including stages of waste decomposition and the biochemical transformations of organic matter.

Chapter 3. This chapter discusses current regulations for landfill construction and reclamation in various countries, and characterizes engineering methods for landfill sealing and cover in accordance with good engineering practice.

Chapter 4. This chapter characterizes the locations of the two study sites, examining their geological settings, hydrogeological conditions, construction phases, and reclamation processes.

Chapter 5. This chapter presents an integrated methodology for evaluating the effectiveness of landfill closure and reclamation, based on a multi-stage investigation and analysis of the results obtained.

Chapter 6. This chapter presents the results from monitoring studies, laboratory tests, modelling studies, and biomonitoring. The data were analyzed and discussed in detail, including comparisons of different reclamation options and an examination of the potential limitations of the methods used. These considerations were used to formulate conclusions and practical recommendations.

Chapter 7. This chapter discusses the results in the light of available scientific research and presents a comparison of the two cover systems tested. The analysis includes aspects such as GHG emissions, oxidation potential of the cover, gas treatment, leachate production and its contamination level, influence of meteorological conditions, degree of contamination of the reclamation cover, stability limitations, material costs and durability, life cycle assessment of the materials as well as phytotoxicity of the leachate and soil respiration.

Chapter 8. This chapter provides a summary of the research and analyses conducted. It presents the key conclusions regarding the effectiveness of the reclamation solutions examined. It also outlines directions for further work and research aimed at optimizing the processes occurring at landfills and their reclamation methods.

1.4. Research gap

From the literature review, the following research gaps were observed.

- Lack of uniform international guidelines for the landfill reclamation process

Currently, no consistent standards or procedures regulate the reclamation process at the international level. As a result, the methods used vary, leading to irregularities in the implementation of reclamation activities and making it difficult to assess their effectiveness.

- Insufficient knowledge of the effectiveness of the most commonly used cover system (HDPE GM vs CCL) in terms of environmental impact

Despite the widespread use of HDPE GM and CCL, there is a lack of comparative studies of their long-term effectiveness and comprehensive effects on environmental components under real conditions based on comparisons of selected landfills.

- Lack of interdisciplinary research that combines multiple fields and scientific disciplines

Many unique geotechnical, engineering, environmental, and hydrogeological issues must be addressed for successful reclamation. This also applies to biological processes, in which the selection of appropriate plant species plays a critical role in restoring land for agricultural, tourist, forestry, or other uses. An interdisciplinary approach is essential because both engineering and environmental perspectives offer opportunities for constructive and scientific solutions to the challenge of properly reclaiming problematic sites such as landfills.

- Gap in integrated assessment of environmental impact of landfill cover

There is a lack of comprehensive research that integrates monitoring analysis, laboratory testing of liquid and solid samples, modelling, and biomonitoring studies. Current studies do not integrate leachate and LFG production models, monitoring, landfill stability, phytotoxicity, or respiration analyses. Currently, studies focusing on alternative cover methods, such as capillary cover, biochar cover, anisotropic barriers, and exposed GMs, are popular. However, long-term analyses of the environmental impact of most common traditional cover, such as mineral and synthetic cover, are missing. Improper reclamation can lead to air, water and soil pollution; therefore, further research is required to assess the impact of mineral cover and to mitigate potential risks of landfills already reclaimed using traditional reclamation methods.

2. Processes occurring in municipal solid waste landfills

This chapter provides a comprehensive examination of the hazards associated with landfills and the processes inside them. It elaborates on the formation of leachate, detailing its composition, chemical characteristics, and the factors influencing leachate levels. Additionally, the chapter offers an in-depth discussion of LFG production, including the phases of waste decomposition and the biochemical transformation of organic matter.

2.1. Hazards caused by landfills

Landfills emit harmful substances, including GHG (i.e. CH_4 and CO_2), leachate, dust, bioaerosols, particulate matter, odors, and fire hazards, affecting soil quality, surface water, and groundwater sources (Vaverková et al., 2018; Soho et al., 2021; Askr et al., 2024). Solid waste undergoes a series of biological, chemical, and physical transformations. These transformations encompass the dissolution and suspension of materials, evaporation of chemicals and water, biological production of liquid percolating through the waste, and sorption of volatile and semi-volatile organic chemicals within the buried materials. All of these processes unfold across different phases within the landfill (Mor and Ravindra, 2023). During the anaerobic degradation of the organic components of waste by microorganisms, gas is produced, the main component of which is CH_4 (40–60% LFG), CO_2 (40–50%), nitrogen, water vapor, and other innumerable trace gases (Majdinasab et al., 2017; Vaverková, 2019). This process typically begins 1–2 years after waste is placed in a landfill and lasts for 15–25 years, depending on the type of waste, humidity, and temperature. At the same time, precipitation and the flow of water through waste layers generate leachate containing dissolved organic substances, inorganic compounds, and heavy metals (HMs), which pose a serious threat to the environment (Vaverková, 2019). Nevertheless, in well-organized systems, landfills transfer the accumulated leachate (after pre-treatment) to leachate treatment plants (LTP), while LFG control includes a special flare that burns CH_4 and can include beneficial use of the gas, i.e., through electricity generation (Wang et al., 2021) (Fig. 2.1.). Since the study analyzed landfill cover systems, detailed attention has been given to the production of LFG and leachate, which are the largest contributors to landfill hazards and can determine use of particular cover systems.

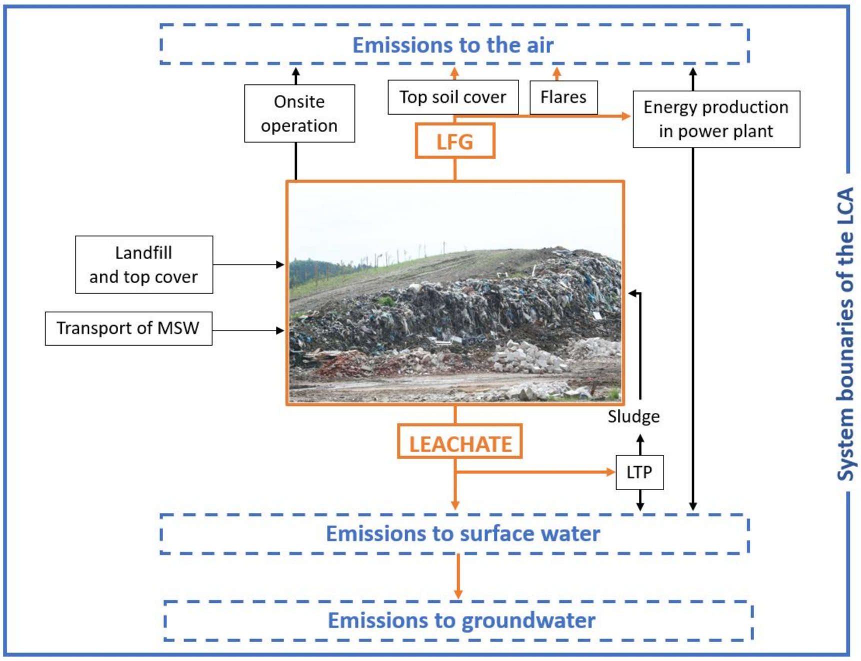


Figure 2.1. Basic processes occurring in landfills acc. Tonini et al. (2018) with author modifications.

2.2. Leachate production

One of the main problems associated with landfill operation is leachate, which is considered a significant threat to the aquatic environment in landfill areas (Podlasek et al., 2023). Landfill leachate is a liquid generated by the percolation of rainwater through solid waste accumulated in a landfill, including the moisture contained in the waste and its degradation products (Costa et al., 2019; Wang et al., 2021). The slow decomposition of materials stored in landfills results in the transport of toxic compounds and formation of landfill leachates, which are referred to as emerging contaminants (Das and Raj, 2025). Leachate infiltrates through the stored waste, concentrates at the bottom layer, and then it is collected by the landfill drainage system (Ma et al., 2022). This liquid is a major pollutant from landfills that significantly endangers the quality of soil, surface water, and groundwater, leading to serious environmental and human health issues. Due to its highly complex composition and characteristics, research focused on predicting leachate output, improving treatment technologies, and understanding its physicochemical properties fundamentally relies on a deep understanding of how leachate is generated (Zhang et al., 2023a). Water is the primary vector for transporting contaminants from landfills. Therefore, understanding the water balance of a landfill and estimating the amount of leachate generated by landfills are critical for designing cover systems and proper management of landfill protection systems (Berger, 2022). The real volume of leachate in the landfill can be calculated on the basis of field and laboratory measurements (Koda and Zakowicz, 1998).

The fundamental premise of the water balance method is that all waste infiltrated by water turns into leachate, as noted by Fenn et al. (1975). The water balance, illustrated in Fig. 2.2, can be determined through various equations.

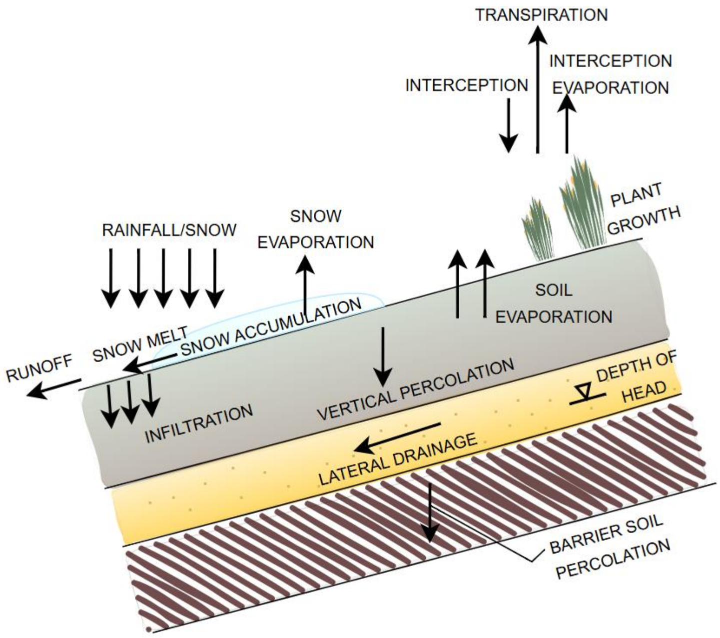


Figure 2.2. Components of water balance on MSW landfill (acc. Peyton and Schroeder, 1993 with author modifications).

The basic equation describing the water balance of a landfill phase is presented below Eq. (1):

$$L = P - S_p - E_t \tag{1}$$

Where: L – leachate, P – precipitation, S_p – surface outflow, E_t – evapotranspiration.

However, leachate production at a reclaimed landfill is more complex. One of the most common equations in the literature is expressed as follows Eq. (2):

$$P + W + H_1 + D_p = E_t + E_w + \Delta R + S_p + H_2 + H_3 \tag{2}$$

Where: P – precipitation, W – water content in landfilled wastes, H_1 – underground water inflow, D_p – surface water inflow, E_t – evapotranspiration from the surface and landfill slopes, E_w – evaporation from the retention reservoirs, ΔR – effective capabilities of water retention by a landfill body, S_p – surface outflow, H_2 – underground water outflow, H_3 – leachate outflow collected by the drainage net.

2.2.1. Forms of water in waste and processes of their transformations

Before understanding the mechanism of leachate production, it is necessary to first analyze the forms of water occurrence and the mechanism of their mutual transformation in the soil connected to the waste. Fig. 2.3 shows that there are three types of water: free, intracellular, and bound water. The water found in organic matter cells is referred to as intracellular water. During waste storage, as a result of compression and degradation, it is released, resulting creation of free and bound water. Bound water, which accounts for approximately 5% of the total moisture, exists in the pores of the waste as a hydrogel, restricting its flow (field water holding capacity). Free water, is divided into gravity and capillary water, the former moves under the influence of gravity, leaving the system as leachate, whereas capillary water, present in the unsaturated zone, is affected by both gravity and matrix suction. When gravity dominates, capillary water moves downward as leachate, but with an increase in suction, its movement is inhibited thus, it remains as the field water holding capacity (WHC) (Zhang et al., 2023).

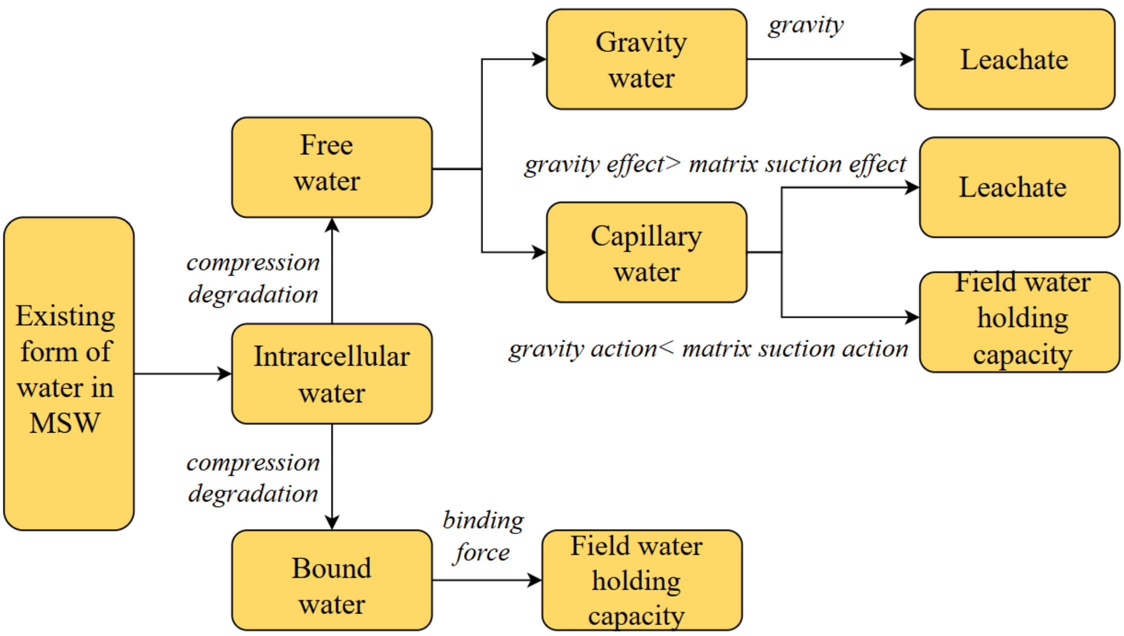


Figure 2.3. Forms of water in waste and mechanisms of their transformations acc. Zhang et al. (2023) with author modifications.

2.2.2. Leachate composition and volume

The amount of leachate generated is mainly influenced by precipitation, evapotranspiration, surface runoff, groundwater infiltration, and degree of waste compaction (Luo et al., 2020; Miao et al., 2019). Leachate composition can vary significantly and is influenced by various factors, such as the physical structure and particle size of the waste, landfill age, moisture content, the internal landfill temperature, the operational and compaction methods used, waste degradation phase as well as the hydrological and climatic conditions at the site (Mohammad et al., 2022). Previous studies have identified nearly 200 hazardous substances present in the leachate. Contaminants can be divided into four groups: (1) dissolved organic matter, including i.e. volatile fatty acids (VFAs) and more persistent compounds such as humic and fulvic acids; (2) inorganic macronutrients such as ammonia (NH_3+), sodium (Na^+), potassium (K^+), calcium (Ca^{2+}), magnesium (Mg^{2+}), manganese (Mn^{2+}), iron (Fe^{2+}), chloride (Cl^-), sulfate (SO_4^{2-}), and bicarbonate (HCO_3^-); (3) HMs such as chromium (Cr^{3+}), nickel (Ni^{2+}), copper (Cu^{2+}), zinc (Zn^{2+}), cadmium (Cd^{2+}), mercury (Hg^{2+}), and lead (Pb^{2+}); and (4) xenobiotic organic compounds such as aromatic hydrocarbons, phenols, chlorinated aliphatic compounds, pesticides, and plasticizers (Luo et al., 2020). Loaded with numerous contaminants, leachate can seep into nearby aquatic and soil environments if a landfill is poorly located, designed or operated. However, there is also a risk of properly maintaining landfills when long-term use causes degradation of sealing materials (e.g. HDPE GM) and clogging of the drainage system, ultimately reducing the integrity of the anti-seepage system and causing leakage into the environment (Ma et al., 2022).

2.3. Landfill gas production

The process of degradation of landfilled waste not only involves the loss of quantity of waste but also the transformation of the solid into liquid and gaseous phases, resulting in the production of LFG. CH_4, a potent GHG that significantly contributes to global warming and serves as the primary precursor to tropospheric ozone, exhibits a greenhouse effect 80 times stronger than CO_2 while being eliminated from the atmosphere more rapidly (Ghosh et al., 2023). In practice, LFG production begins shortly after waste is buried in the landfill and continues until organic matter is present (Andriani and Atmaja, 2019).

2.3.1. Landfill gas generation phases

Waste decomposition is characterized by five phases: aerobic decomposition, acidic fermentation, unstable methanogenesis, stable methanogenesis, and decomposition attenuation. Phase I (aerobic decomposition), usually lasts 7–30 days from the entire decomposition process (Andriani and Atmaja, 2019). In phase I, when the waste is covered, the empty spaces retain oxygen (O_2), which, together with the O_2 dissolved in the moisture in the waste, acts as the main electron acceptor. Soluble sugars are a carbon source for microorganisms, and the decomposing waste reacts quickly with trapped O_2, leading to the formation of CO_2 and H_2O. This phase lasts until the available O_2 is exhausted, that is under aerobic conditions.

In the transition phase (Phase II), the environment changes from aerobic to anaerobic instead of O_2, nitrate and sulfate are used as electron acceptors the same time, O_2 is replaced by CO_2, which promotes reductive conditions. At the end of this phase, elevated concentrations of chemical oxygen demand (COD) and volatile organic acids (VOA) are observed in the leachate. The initiated activity of microorganisms in Phase II intensified in Phase III. Continuous fermentation and hydrolysis result in the intensive production of large amounts of organic acids, with less hydrogen (H_2) release. The hydrolytic process leads to the intensive formation of intermediate volatile VOAs, which allow the microbial transformation of biodegradable compounds by acidogenic bacteria. As a result of acid decomposition, COD, biological oxygen demand (BOD) and conductivity values increase, while the reduced pH mobilizes HMs and eliminates essential nutrients. In the CH_4 fermentation phase (Phase IV), methanogenic bacteria convert intermediate acids into CO_2 and CH_4, with the rate of acid production decreasing and the pH of the landfill increasing to near neutral (6.8–8.0). As a result, a decrease in BOD, COD and leachate conductivity is observed, while gas production increases. At the final stage of maturation (Phase V), the decomposition of readily decomposable organic compounds to CH_4 and CO_2 slows down rapidly, and the process enters a state of relative stagnation, which further hinders biological treatment because of the presence of humic and fulvic acids (Mor and Ravindra, 2023). Fig. 2.4 graphically illustrates the previously described phases of organic matter decomposition and the associated production of LFG.

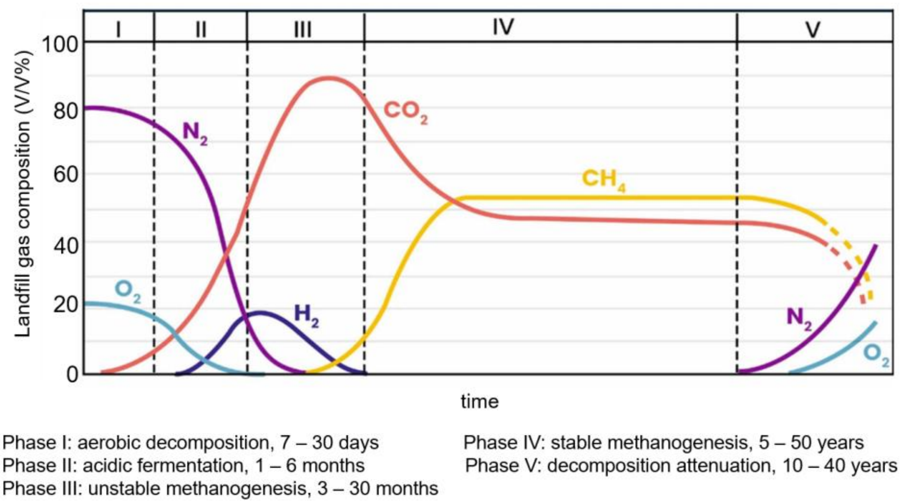


Figure 2.4. Changes in LFG production over time acc.to Mor and Ravindra (2023) with author modifications.

2.3.2. Biochemical processes leading to landfill gas production

The mechanism of LFG formation involves chemical reactions, such as hydrolysis, fermentation, anaerobic oxidation, acidogenesis, acetogenesis, and methanogenesis, with the participation of fermentative bacteria, such as acetogens and methanogens (Fig. 2.5).

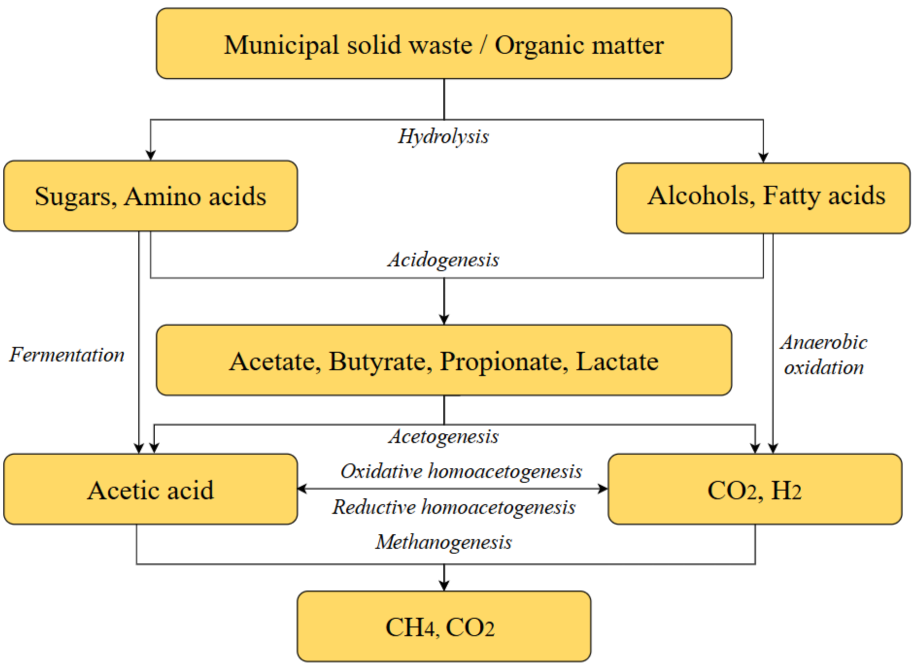


Figure 2.5. Mechanism of landfill gas production from landfill acc. to Nanda and Berruti (2021).

The initial hydrolysis of waste produces sugars, amino acids, and fatty acids, which are converted into intermediate compounds (acetate, butyrate, propionate, and lactate), while acid fermentation, aided by high moisture and organic matter content, leads to the production of VFAs (acetic acid, butyric acid, propionic acid, and lactic acid). Acetate-oxidizing bacteria can convert acetate to H_2 and CO_2 via oxidative homoacetogenesis. Finally, through methanogenesis, acetate-oxidizing bacteria convert acetate into CH_4 and CO_2, which are the main components of the LFG. In addition, competing acetotrophic methanogens convert CO_2 and H_2 into CH_4 (Nanda and Berruti, 2021).

2.3.3. Landfill gas collection and energy production

The LFG produced can be collected through a system of active wells and pipes, and then either burned (resulting in its oxidation to biogenic CO_2) or used to generate electricity. Currently, waste-to-energy (WtE) facilities are common, and include thermal processes (such as incineration, pyrolysis, and gasification), biochemical conversion, and landfill processes. These processes primarily recover electricity, heat, fuel gases, liquids, and solid residue. In practice, two or more of these methods can be combined, although each approach faces different challenges (Beyene et al., 2018). LFG is used for electricity generation, particularly in large power plants, for economic reasons. With a CH_4 content of approximately 40–65% by volume, gas can be used to generate electricity using internal combustion engines (1-3 MW), turbines (above 5 MW), microturbines (30-250 kW), and fuel cells. However, once the CH_4 content falls below 35–40% by volume, the gas must be combusted (Tadesse and Lee, 2024). Fig. 2.6 shows a schematic of the operation of a landfill with power generation capability and CH_4 collection system.

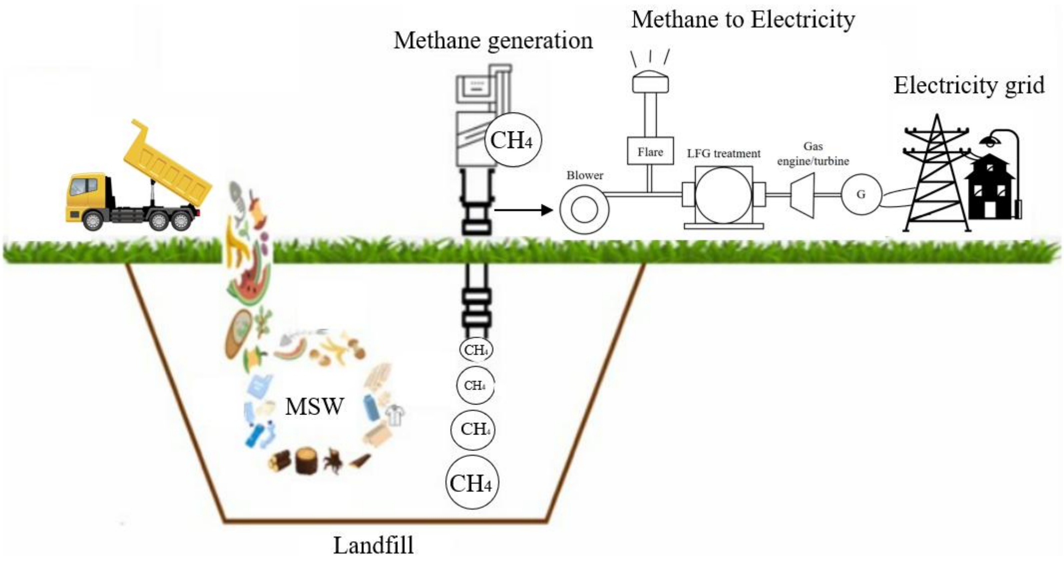


Figure 2.6. Landfill gas production from MSW landfill acc. to. Askr et al. (2024).

2.3.4. Impact of leachate recirculation on landfill gas production

In order to accelerate the production of LFG from landfill, a popular method is leachate recirculation (Reddy et al., 2014; Ali and Yue, 2020; Mohammad et al., 2022; Zhang et al., 2023b). Bioreactors are engineered landfills where waste decomposition is accelerated by adding water to the waste and/or recirculating leachate to stabilize the waste more rapidly (Broun and Sattler, 2016; Tonini et al., 2018). Traditional dry landfills generate LFG slowly and require prolonged stabilization of MSW, posing serious environmental challenges and health risks. Bioreactor landfills allow for faster stabilization through intensive moisture recirculation and faster LFG generation while reducing the organic strength of the leachate (Srivastava and Chakma, 2022). Braun and Sattler (2016) compared two types of landfills: conventional and bioreactors. They demonstrated that a bioreactor landfill produces a higher peak electricity production (75 774 MWh in bioreactor compared to 48 793 MWh for a conventional landfill), however, a conventional landfill produces electricity for 32 years longer compared to a landfill with a bioreactor. As a result, the overall electricity generation from a traditional landfill surpasses that of a bioreactor landfill, and enhancing biogas collection efficiency is the only way to boost electricity output from a bioreactor landfill (Broun and Sattler, 2016). Although landfill energy production is beneficial, maximizing LFG production can lead to toxic leachates associated with high ammonia (NH_4^+) concentrations (Kurniawan et al., 2022). Another limitation in bioreactors is recirculation method. In the tight landfill covers (e.g. with GM), it is required to use special leachate injection through vertical boreholes (Guérin et al., 2004). Benson et al. (2012) concluded that recirculation at a GM-covered landfill could also lead to uncontrolled increase in gas pressure under the GM, which could cause sliding after the GM and loss of stability.

Staub et al. (2011) verified the differences in LFG production in a conventional landfill covered with semipermeable cover ($k < 1\times10^{-6}$ m/s) and in a landfill with impermeable cover (with GM) (Fig. 2.7). The energy recovery time was the same in both cases (15 years), but the amount of CH_4 production was different. The GM-based system has a very low permeability therefore, the liner provides better retention of the gas produced inside the landfill, which promotes efficient recovery and reduces uncontrolled CH_4 emissions to the atmosphere. However, higher tightness may be associated with a higher gas pressure, requiring well-designed degassing systems.

The semi-permeable liner had a higher discontinuity in CH_4 production and a higher permeability. This phenomenon is due to the differences in the gas capture efficiency (CE) of the cover. Although such a lining is economical and beneficial for the water balance and thermal stability of landfills, its properties may result in a partial loss of CH_4 to the atmosphere (for a maximum of 80 m^3 CH_4/h, collection only a maximum of 50 m^3 CH_4/h), and consequently, reduce the efficiency of biogas production. It should also be noted that the semi-permeable liner has a longer gas production time (up to 30 years) than the synthetic cover (approximately 22 years). Nevertheless, after 15 years, regardless of the type of cover, the intensity of gas production decreased, at which time the gas management system switched to a mode in which excess CH_4 was no longer captured, but was directed to the flaring system and burned.

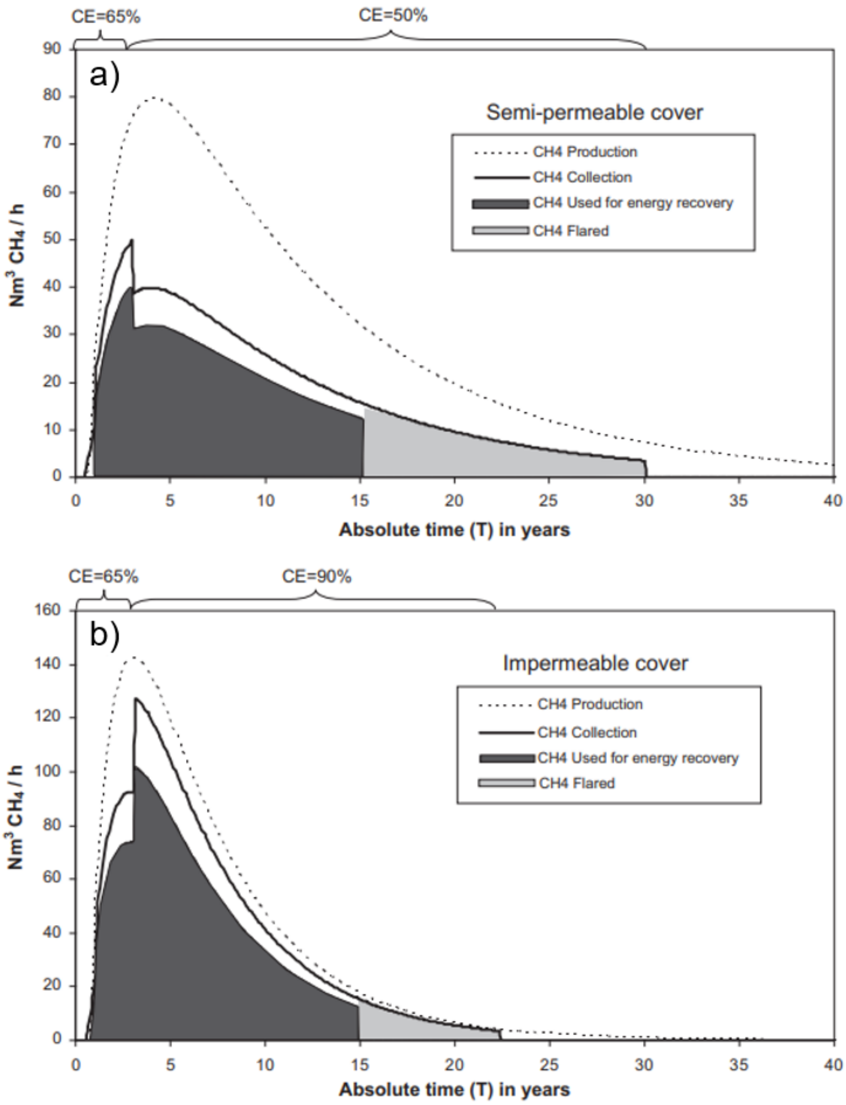


Figure 2.7. Methane production for the landfill: a) with semi-permeable cover, b) impermeable cover acc. Staub et al. (2011).

2.3.5. Modern techniques for landfill gas emissions control

Modern LFG emission control techniques play a key role in reducing the negative environmental impacts of landfills. Emissions of CH_4 (which is a much more potent GHG than CO_2) are difficult to fully capture because unlike landfill leachate, which is collected through drainage pipes, the gas is diffused and its degradation is challenging to estimate. Moreover, laboratory methods often fail to accurately replicate the natural processes occurring in the environment. Traditional methods of gas control often do not guarantee full effectiveness, as shown in a study by Themelis and Ulloa (2007), who examined 25 landfills in California and estimated that the CH_4 capture was 43 Nm^3 per ton of MSW, and the estimated CH_4 loss was 82 Nm^3 per ton of MSW.

To address these challenges, advanced technologies are being developed to monitor LFG emissions. Among the most important are airborne and satellite remote sensing techniques, which provide comprehensive monitoring of emissions. In this area, satellites such as TROPOMI, GHGSat, and EMIT have great potential because they provide frequent and accurate measurements (Wang et al., 2024). The use of satellite data was also tested by Karimi et al. (2021), who used the European Space Agency Sentinel-2 imagery and a random forest algorithm to accurately identify gas emission sites, achieving high agreement ($r > 0.96$) with Landsat 8 data. These methods make it possible not only to detect and locate major sources of CH_4 emissions but also to quantify them and systematically monitor the effectiveness of the implemented corrective measures. This will allow CH_4 emissions to be managed at a site-specific level, improve the reliability of emission inventories, support climate policies, and strengthen the enforcement of environmental regulations (Maasakkers et al. 2022).

Another innovative tool for measuring LFG emissions is the use of drones equipped with specialized sensors. This solution was also tested by Sliusar et al. (2022), who investigated the use of drones in landfill management, focusing on spatial data analysis and the monitoring of LFG emissions. The authors noted the potential of modern remote sensing technologies such as airborne sensors, photogrammetry software, and GIS systems to provide detailed environmental studies and assess compliance with operational standards. In addition, the authors found that drones can provide a more detailed analysis than satellite and aerial imagery, which translates into a great analytical potential for drones.

A study by Jahan et al. (2024) confirmed that such solutions can detect CH_4, especially in hard-to-reach landfill areas. Another alternative to traditional measurement methods is the use of machine and deep learning methods. For example, Askr et al. (2024) presented a deep learning model that can predict CH_4, identify areas with high leakage potential, and optimize gas collection systems based on the analysis of images of waste mixtures. This approach enables the integration of data from multiple sources, including in-situ measurements, which is essential for correcting inventory models that account for both planned and unplanned emissions (e.g., malfunctioning burners or large leaks). The method uses waste detection and a hybrid Inception-ResNet-V2 model to classify wastes into different categories and then estimates their mass, CH_4 production, and energy potential. The results indicated a classification accuracy of 93%. The proposed solution eliminates the need for manual waste classification, and identifies areas with high CH_4 emissions, thus supporting the optimization of gas collection systems.

Thus, it is noted that modern monitoring systems, which combine technologies of remote aerial measurements, satellites, drones, and machine learning algorithms, show some alternatives to conventional measurement systems. Although full gas capture remains a challenge, the integration of these solutions makes it possible to effectively identify leak sources and implement effective intervention strategies. Additionally, predicting CH_4 emissions and producing green electricity from waste can help mitigate climate change and promote sustainable development goals (SDGs) (Themelis and Ulloa 2007; Jahan et al. 2024; Askr et al. 2024; Wang et al. 2024).

3. Landfill construction and reclamation

This chapter discusses current regulations on the landfill construction and reclamation in various countries, and characterizes the technical methods of landfill sealing and cover in accordance to good engineering practices.

3.1. Overview of legal requirements for landfill construction and sealing in selected countries

The construction of landfills, as a key component of the WM system, is subject to strict legislation at the EU, international and national levels. In the EU, the main guidelines are defined by *Council Directive 1999/31/EC of April 26, 1999, on the landfill of waste*, together with its amendment *Directive (EU) 2018/850 of the European Parliament and of the Council of May 30, 2018*. These regulations require that the base and slopes of a landfill consist of a natural geological mineral layer that meets the requirements of hydraulic conductivity (k) and thickness (h), which, for the protection of soil, groundwater, and surface water, has a combined effect at least equal to the effect of the following requirements:

- hazardous waste landfill: $k \leq 1.0 \times 10^{-9}$ m/s, h ≥ 5 m,
- non-hazardous waste landfill: $k \leq 1.0 \times 10^{-9}$ m/s, h ≥ 1 m,
- inert waste landfill: $k \leq 1.0 \times 10^{-7}$ m/s, h ≥ 1 m.

In situations where the natural geological barrier does not meet these criteria, it is possible to supplement it with an artificially constructed barrier with a minimum thickness of 0.5 m (*European Parliament and Council Directive, 1999/31/EC; Directive (EU) 2018/850*). The barrier must also be equipped with a leachate drainage collection system with a minimum thickness of 0.5 m to effectively control and drain leachate to the leachate tank. The Directive also requires regular monitoring of MSW landfill. As part of the monitoring, the landfill should be regularly check in case of chemical properties of leachate, groundwater, surface water, LFG and landfill settlement. In operational phase monitoring is taking place more frequent than during post-operational phase. Tab. 3.1 below shows the frequency with which monitoring should take place at landfills.

Table 3.1. Frequency of landfill monitoring according to EC Council Directive 1999/31.

Parameter	Operational phase	Post-operational phase
Volume of leachate from the leachate tank	monthly	every 6 months
Leachate composition	quarterly	every 6 months
Volume and chemical composition of surface water	quarterly	every 6 months
Potential gas emissions and atmospheric pressure (CH_4, CO_2, O_2, H_2S, H_2 etc.)	monthly	every 6 months
Groundwater level	every 6 months	every 6 months
Landfill settlement level	annually	annually

Under Polish law, in addition to the provisions of the above-mentioned *Council Directive, 1999/31/EC*, the construction of landfills is regulated, inter alia, by the *Regulation of the Minister of the Environment on April 30, 2013 on landfills* (Rozporządzenie Ministra Środowiska z dnia 30 kwietnia 2013 r. w sprawie składowisk odpadów, Dz. U. 2022 poz. 1902) and the *Regulation of the Minister of Climate and Environment dated March 19, 2021, amending the Regulation on landfills* (Rozporządzenie Ministra Klimatu i Środowiska z dnia 19 marca 2021 r. zmieniające Rozporządzenie w sprawie składowisk odpadów) as well as *the Waste Act of December 14, 2012* (Ustawa o odpadach z dnia 14 grudnia 2012 r., Dz. U. 2013, poz. 21), according to which, the construction of a new landfill requires the inclusion of a plan for its construction in the provincial WM plan (Article 127): "*If the construction of a landfill is not provided in the provincial waste management plan, the authority responsible for issuing a landfill construction permit shall refuse to issue the permit". Regulation of the Minister of the Environment on April 30, 2013 on landfills* specifies the minimum requirements for a geological barrier, specifying a thickness ≥ 1 m and $k \leq 1.0 \times 10^{-9}$ m/s for landfills other than hazardous and inert. In cases where natural conditions are insufficient, it is permissible to supplement the barrier with an artificial layer of minimum thickness of 0.5 m, whereby additional insulation may be a synthetic system designed considering the specificity of the waste being stored. The sealing system must be supplemented with a drainage system for seepage water consisting of a drainage layer made of gravel-sand material or other materials with similar properties with $k \geq 1 \times 10^{-4}$ m/s and a thickness ≥ 0.5 m.

In the Czech Republic, in addition to the applicable *European Council Directive 1999/31/EC of April 26, 1999, on the landfill of waste*, when designing the sealing of a landfill, the technical *standard ČSN 83 8032 Landfilling - Landfill sealing* (Skládkování odpadů - Těsnění składek) is also taken into account, which assumes that the appropriate sealing of a landfill should include a mineral layer ≥ 0.5 m thick with $k \leq 1\times10^{-8}$ m/s (or a bentonite mat as an alternative to the mineral layer) with a 1 mm thick HDPE GM placed on it (rough on both sides on slopes). The system should be supplemented by a drainage layer of sand and gravel formations with a thickness of 0.3 m.

In United States, according to the *Resource Conservation and Recovery Act* (RCRA) (1976), regulations related to non-hazardous WM are found in Section 239–259. New facilities should have a liner created from two parts: the upper part in the form of a 0.76 mm GM liner, and the lower part from at least a 0.6 m layer of compacted soil with a $k \leq 1\times10^{-9}$ m/s *(§258.40 Subpart D – Design criteria*). The material must provide adequate abrasion resistance to the top and bottom interfaces to prevent material movement on slopes.

According to the Japanese guidelines for MSW landfills, $k \leq 1\times10^{-8}$ m/s is included in the sealing and soil requirements for the bottom liner (*Ministry of Health and Welfare of Japan, 2001*). In case when the landfill is located on impermeable soil with a thickness of at least 5 m, then the *k* can be equal 1×10^{-5} m/s or less (*Ministry of the Environment of Japan, 2012*). Bottom barrier systems vary depending on the country's adopted regulations, but nevertheless, systems consisting of natural clay, crack-free natural rock, compacted clay with a GM, a layer of asphalt concrete with a GM or a double GM are commonly used in Japan (Katsumi et al., 2021).

In China, the sealing is based on the Chinese specification CJJ113-2007. Technical code for liner system of MSW landfill proposed four types of liner systems: GM+CCL, GM + geosynthetic clay liner (GCL), a single CCL, and a single GM. The GM+CCL liner system is composed of a thickness of 0.75 m covered by the HDPE GM with a thickness of 1.5 mm, and *k* of the CCL should be $\leq 1 \times 10^{-9}$ m/s.

The construction of landfills in accordance with applicable legal standards requires the implementation of multi-layer, often similar, sealing systems that guarantee the minimization of leachate flow and environmental protection. An integrated approach to the design of geological barriers, supplemented with artificial layers, if necessary, is a fundamental element for protecting land and water from potential contamination.

3.2. Technical methods for landfill sealing in accordance with good engineering practices worldwide

In addition to generally applicable regulations, there are also good engineering practices to support the landfill construction process. According to the guidelines of the International Society for Soil Mechanics and Geotechnical Engineering (ISSMGE) a well-made sealing system consists of 4 main components (Fig. 3.1).

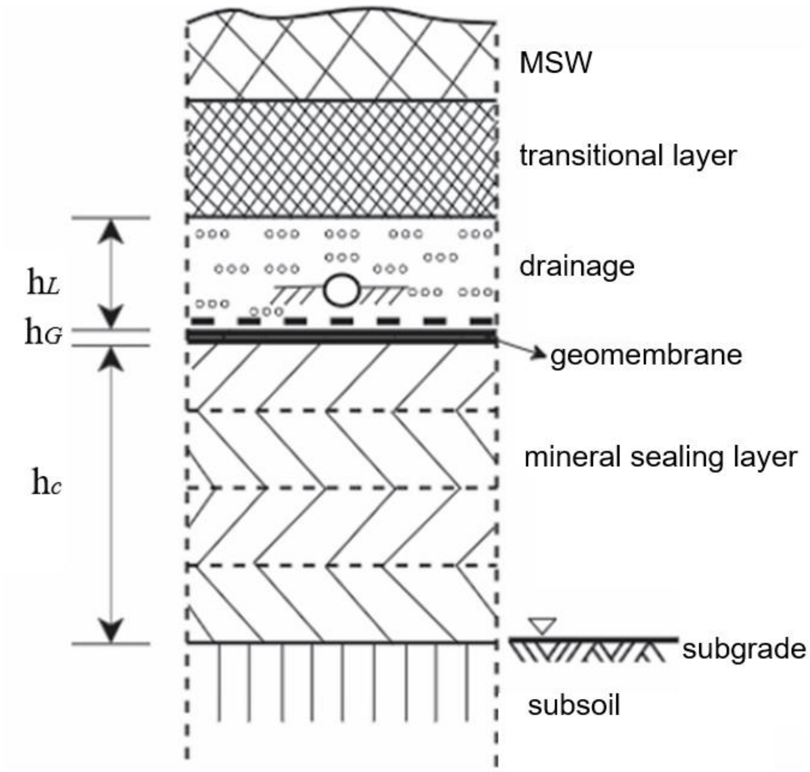


Figure 3.1. Landfill sealing system recommended by ISSMGE (acc. Jessberger et al., 1993, modified by author).

The lowest mineral sealing layer (h_c) must be resistant to erosion and leachate infiltration, thereby minimizing leaks. The next sealing layer (h_G) must be characterized by its resistance to settlement, stemming from its stress and deformation properties, and provide long-term protection against the leakage of undesirable substances (Christensen, Cossu & Stegmann, 1994). A protective layer of geotextiles should be placed on top of the GM to distribute the stresses concentrated in the GM. A drainage system (h_L) collects and drains leachate from the waste. This solution prevents the accumulation of leachate above the liner system. Additionally, an optional transition layer can be placed between the drainage layer and waste (Jessberger et al., 1993).

Depending on the regulations of the countries, the limit standards of the indicators: h_L, h_G, h_c and k_c are different. Tab. 3.2 shows the summary of limit values of the indicated parameters for selected counties like Japan, China, USA, Germany, Poland and the Czech Republic. The most restrictive are German regulations, which establish that the *kc* has to be ≤ 1×10^{-10} m/s, while Japanese and Czech Republic landfills assume that the optimal rate will be ≤ 1×10^{-8} m/s.

Table 3.2. Limit values of the indicated parameters for landfill sealing in selected countries acc. Jingjing (2014) modified by author.

Country	Parameter			
	h_L [m]	h_G [mm]	h_c [m]	kc [m/s]
Japan	0.5	1	0.5	1×10^{-8}
China	0.3	1.5	0.75	1×10^{-9}
USA	0.3	1.5	0.6	1×10^{-9}
Germany	0.3	2	0.75	1×10^{-10}
Poland[a]	0.5	1.5-2.0[b]	0.5	1×10^{-9}
Czech Republic[c]	0.3	1	0.5	1×10^{-8}

[a] *Regulation of the Minister of Environment of 2013 on landfills with amendments*
[b] *Wysokiński (2009)*
[c] *ČSN 83 8032 Landfill sealing*

According to Wysokiński (2009), substrate sealing can be divided into natural (clays, sandy loams), artificial (GM), or polyvinyl chloride (PVC) GM or mixed (clays or loams with plasticizers, asphalt materials or bentonite-synthetic composites). The selection of an appropriate seal depends on the volume of the landfill and geological conditions. Fig. 3.2 shows sealing systems according to the Building Research Institute (Wysokiński, 2009). Municipal landfill base sealing can be divided into four types: (a) single mineral barrier, which is constructed in the order of a mineral layer and a drainage layer, (b) single composite mineral-synthetic barrier, which is constructed of a mineral layer, a synthetic layer and a drainage layer, (c) double synthetic, which is constructed of a mineral layer, which is constructed of alternating two layers of mineral synthetic and drainage. For small objects with favorable geological conditions sealing: a) is recommended, whereas for medium and large objects with unfavorable geological conditions, b) is recommended and for large objects with unfavorable geological conditions sealing, c) or d) should be chosen (Wysokiński, 2009). Considering the sealing systems below, it can be stated that the single composite mineral-synthetic barrier system predominates worldwide for MSW landfills.

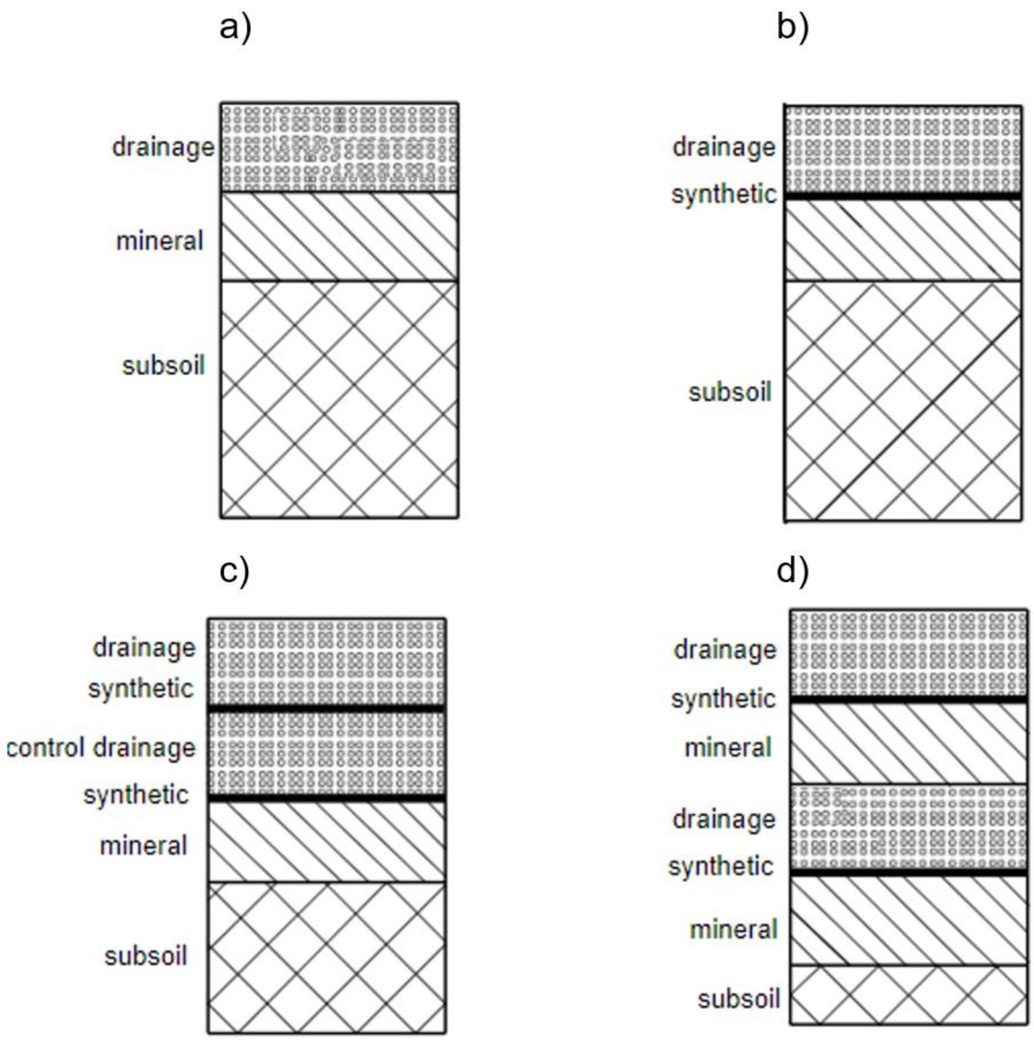


Figure 3.2. Modified landfill sealing schemes: a) single mineral barrier, b) single composite mineral-synthetic barrier, c) double synthetic, d) double composite barrier (acc. Wysokiński, 2009, modified by author).

3.3. Overview of legal requirements for landfill reclamation and cover in selected countries

After waste acceptance has ceased, the slopes and top of the landfill are reclaimed and protected against water and wind erosion by a suitable reclamation cover, which design depends on the characteristics of the waste. *Council Directive 1999/31/EC of April 26, 1999, on the landfill of waste* provides the legal basis for the implementation of mandatory landfill cover requirements in EU members, which are obliged to comply with these requirements and incorporate them into national legislation. This document specifies that the design of the landfill cover must be adapted to the waste category, with differences between landfills for hazardous and non-hazardous waste. For example, hazardous waste landfills require an engineered geosynthetic sealing system in the cover, whereas non-hazardous landfills do not, and the gas collection layer plays a key role owing to the risk of potential ignition. However, regardless of the type of landfill, the cover must also include a low-permeability mineral layer, a drainage layer with a minimum thickness of 0.5 m, and a soil cover with a thickness > 1 m to ensure adequate environmental protection and long-term durability of the cover. Mentioned requirements acc. to Council Directive 1999/31/EC are listed in Tab. 3.3.

Table 3.3. Elements necessary for the construction of a landfill cover acc. to Council Directive 1999/31/EC.

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 m	required	required

Depending on the legal regulations and technical recommendations of a country, differences in the characteristics of the various layers of reclamation cover, including their thickness and type of materials, are significant. Each country considers its specific climatic, geological, and legal conditions, making the requirements for landfill reclamation and cover different (Jakimiuk et al., 2022). In the United States, under the provisions of RCRA Part 258.60 Subpart F, the final reclamation cover must be designed before landfill closure begins. This system ensures that the permeability of the final cover does not exceed the permeability of the sealing system installed at the bottom of the landfill or the natural subgrade layers, with $k \leq 1\times10^{-7}$ m/s. Additionally, to minimize liquid percolation, the infiltration layer must be at least 45.7 cm thick, and the layer forming the vegetation must be at least 15.2 cm thick to allow for stable plant growth (RCRA, 1976; Jakimiuk et al., 2022).

In China, the regulations for the closure of MSW sanitary landfills are contained in the *Technical code for municipal solid waste sanitary landfill* (GB 50869-2013 and GB 51220-2017), which specify the available cover options to ensure adequate environmental protection after reclamation. There are three possible variants of the landfill cover design. The first option involves the use of GM placed on a CCL ≥ 0.45 m thick, with $k \leq 1\times10^{-7}$ m/s. This solution combines the advantages of a GM as a barrier with very low permeability and additional protection in the form of a properly prepared soil layer.

The second option is based on the mineral layer; however, it requires a greater thickness (≥ 0.90 m) and much lower permeability, with k not exceeding 1×10^{-9} m/s. Such thickness and tightness guarantee an effective limitation of the infiltration of rainwater and pollutants into the lower layers. The third option, assumes the use of a GM made of HDPE or low-density polyethylene (LDPE) with a thickness of at least 1–1.5 mm and extremely low permeability, reaching even $k = 1\times10^{-14}$ m/s, at the same time, it is characterized by a durability of at least 30 years. This makes it possible to ensure long-term environmental protection with a relatively small mineral layer thickness or even complete reduction, provided that the design and soil and water conditions allow it.

In Germany, regulations include specific requirements for both Class I and Class II landfills (for MSW with a higher biodegradable fraction), requiring drainage layer with a certain hydraulic conductivity ($k \geq 10^{-3}$ m/s) in both cases (Deponieverordnung, 2009).

According to the Polish regulations specified in the *Regulation of the Minister of Environment of April 30, 2013 on waste landfills*, after the date of cessation of waste acceptance for storage in a landfill of non-hazardous and inert waste or a landfill of inert waste, or their segregated parts, the slopes and the surface of the landfill's top shall be cleaned up and protected from water and wind erosion by making an appropriate reclamation cover, whose construction depends on the properties of the waste. The minimum thickness of the landfills covers for other than hazardous and inert waste is not less than 1 m, which is intended to enable the creation and maintenance of a permanent vegetation cover.

In the CR, technical standards and industry documents are used, including the *ČSN 83 8035 Landfilling – Closure and Reclamation of Landfills* (*Skládkování odpadů – Uzavírání a rekultivace skládek*) standard, which, together with the guidelines issued by the Ministry of Environment, specify the principles of landfill closure and reclamation. Special attention is paid to the selection of the sealing materials. If the bottom of the landfill has been sealed with an HDPE GM, it is recommended that the same material be used for the final closure of the landfill, provided that this solution is technically feasible (*Chapter 7.3.2. ČSN 83 8035*). Tab. 3.4 and on Fig. 3.3 below is a summary of the MSW cover systems used according to regulations in selected countries.

Table 3.4. MSW landfill cover systems worldwide according to different regulations.

Country	Layer characteristics				
	Top soil cover (h_T)	**Drainage layer (h_L)**	**Low permeable layer (h_C)**	**Optional artificial sealing layer (GM)**	**Gas drainage layer (h_G)**
China[a]	Nutrient vegetation layer (≥0.15 m) on the top of supporting soil layer (≥0.45 m, $k \geq 10^{-6}$ m/s).	≥ 0.3 m $k \geq 10^{-3}$ m/s	required	GM on the top of CCL ≥ 0.45 m, $k \leq 10^{-7}$ m/s or CCL ≥ 0.9 m with $k \leq 10^{-9}$ m/s or HDPE or LDPE GM ≥ 1-1.5 mm, $k \leq 10^{-14}$ m/s, service life ≥ 30 years.	≥ 0.3 m (particle size 25-50 mm)
USA[b]	Erosion layer of a min. of 15 cm of earthen material capable of sustaining vegetation.	required	The final cover should consist of an infiltration layer or barrier layer of a minimum of 45 cm which k < than the bottom liner system (if present) or the existing natural subsoils and in no case, should it exceed 1.0×10^{-7} m/s.	If a GM is used in the bottom liner, then it becomes a necessity to use one in the cover to comply with k limits.	Can have various components depending upon the site conditions and anticipated gas generation.
Germany[c] Class I*	required	≥ 0.3 m $k \geq 10^{-3}$ m/s, slope>5%	required 0.5 m low permea-	not mentioned	≥ 0.3 m

Country	**Layer characteristics**				
	Top soil cover (h_T)	**Drainage layer (h_L)**	**Low permeable layer (h_C)**	**Optional artificial sealing layer (GM)**	**Gas drainage layer (h_G)**
			bility soil $k \leq 5\times10^{-9}$ m/s		
Germany[c] Class II	required	$\geq$ 0.3 m $k \geq 10^{-3}$ m/s slope > 5%	0.5 m low permea-bility soil $k \leq 5\times10^{-9}$ m/s	GM 2.5 mm	required in some cases
Poland[d]	Landfill cover with min. 1 m with a minimum slope of 5%.	The layer is created from medium and coarse-grained soil.	CCL	not mentioned	The layer is created from medium and coarse-grained soil.
Czech Republic[e]	Layer thickness $\geq$ 1 m, of which 300 mm humus for agricultural reclamation, surface slope modeled at min. 3%).	Medium and coarse aggregate.	The layer is formed by a combinatio n of natural clay seal, synthetic sealing GM and protective geotextiles.	recommend (HDPE GM of thickness 1 mm)	The layer is created from medium and coarse-grained soil.
Italy[f]	$\geq$ 1 m	$\geq$ 0.5 m	CCL $\geq$ 0.5 m, $k \leq 10^{-8}$ m/s	not mentioned	$\geq$ 0.5 m
UK[g]	$\geq$ 1 m	required	required	not mentioned	required

[a] *Technical code for municipal solid waste sanitary landfill (GB 50869-2013 and GB 51220-2017)*
[b] *Part 258.60 Subpart F of the RCRA*
[c] *Deponieverordnung. DepV/2009*
**Landfill of class I and II are landfills for nonhazardous waste (e.g., MSW). Class II landfills contain a higher amount of biodegradable waste compared to Class I*
[d] *Polish Regulation of the Minister of Environment of 2013 on landfills with amendments*
[e] *CSN 83 8035 Landfilling – Closure and Reclamation of Landfills and Junga et al. (2015).*
[f] *Cossu and Garbo (2018), Attuazione della direttiva 1999/ 31/CE relativa alle discariche di rifiuti*
[g] *The Landfill (England and Wales) Regulations, 1559/2002*

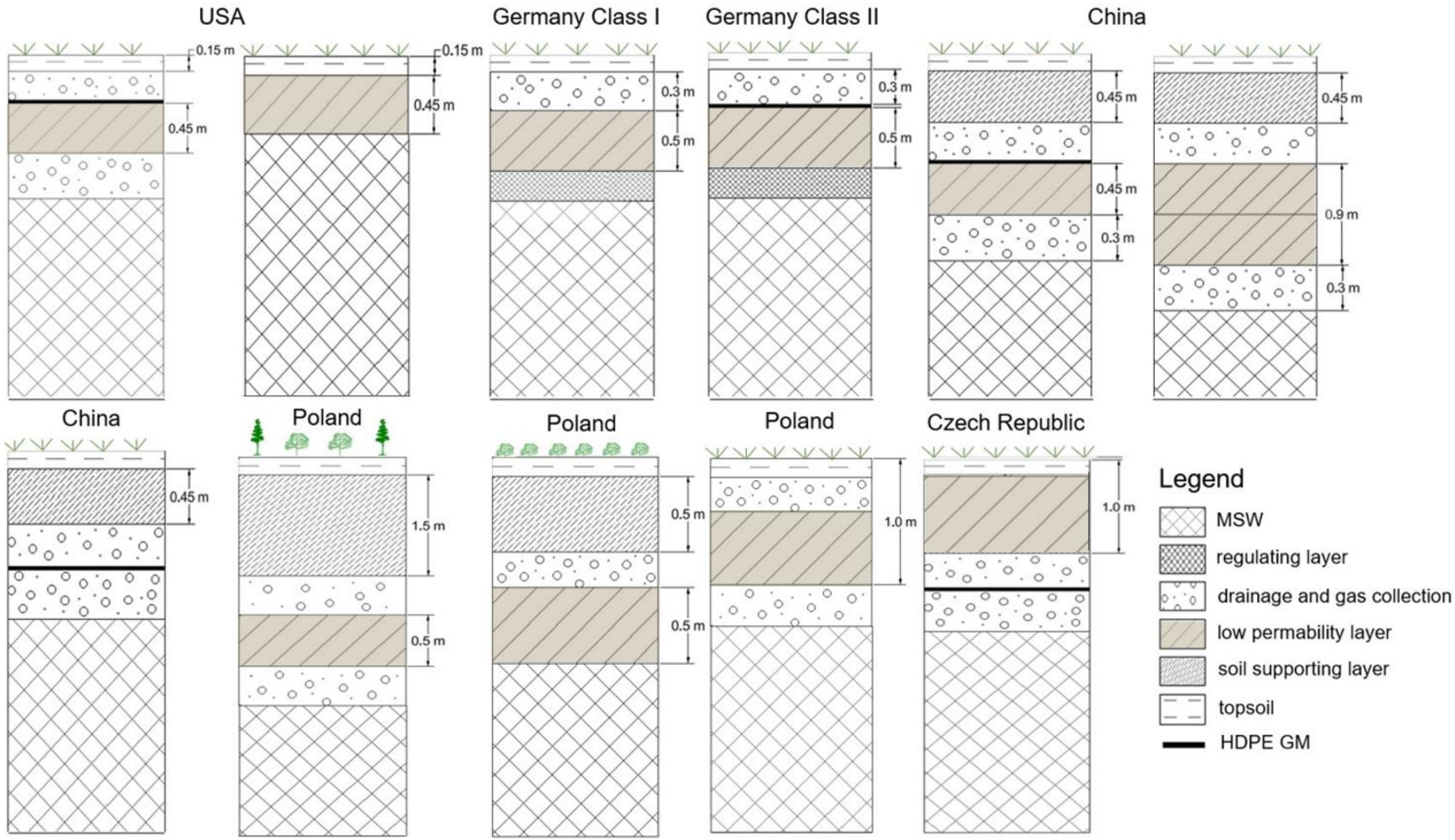


Figure 3.3. Examples of the cover systems in selected countries (own study)

3.4. Technical methods for landfill reclamation in accordance with good engineering practices worldwide

Globally, there is a lack of consistent and detailed guidance regarding how and which specific materials should be used for landfill reclamation. However, there is consistency in the basic layers used in the landfill cover, which provides a protective barrier to control LFG volatilization and minimize rainwater infiltration into the waste. Landfill cover usually consists of several layers of materials carefully designed to perform different functions (Ling et al., 2024). There are five basic layers commonly used in the cover system: leveling and biogas collection, sealing, drainage, protective, and surface layers which are visible on Fig.3.4 and they functions are presented in Tab. 3.5.

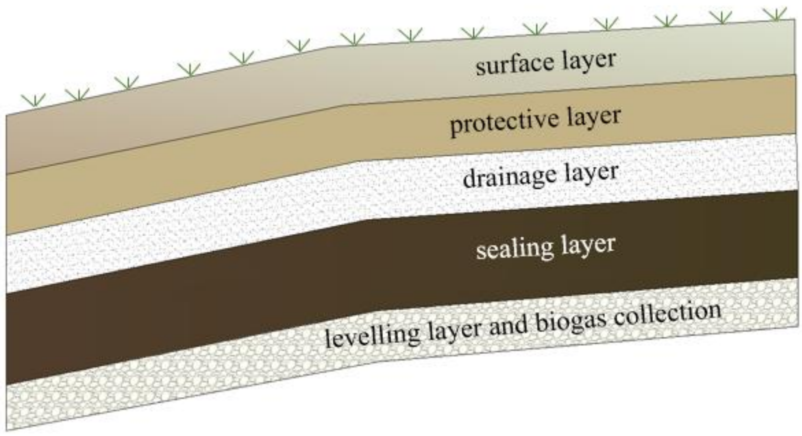


Figure 3.4. Schematic MSW landfill cover system acc. to Wysokiński (2009) and Manassero et al. (1997), modified by author.

Table 3.5. Landfill cover system components and their functions (Wysokiński, 2009; Cossu and Garbo, 2018; Manassero et al., 1996).

Layer	Materials	Function
surface layer	vegetative soil, compost, geosynthetics	provide support for vegetation growth, prevent erosion
protective layer	soil, recycled raw materials	water storage, root puncture protection, overall stability
drainage layer	sand or gravel, geogrids, geocomposites	drainage of water infiltrating through the landfill
sealing layer	compacted clay, GM, bentonite, other sealing materials with $k < 10^{-7}$ m/s	minimize water infiltration through waste, minimize biogas volatilization
gas collection layer and/or leveling layer	sand, gravel, geogrids, geotextiles, recycled materials with $k > 10^{-4}$ m/s	gas production control and transport of gas to storage and further disposal/cogeneration site

According to good engineering practice, the landfill cover system according to ISSMGE should consist of topsoil, drainage system, GM, mineral sealing level, gas venting system and regulating layer. Graphical representation of the cover system acc. to ISSMGE is presented on Fig. 3.5.

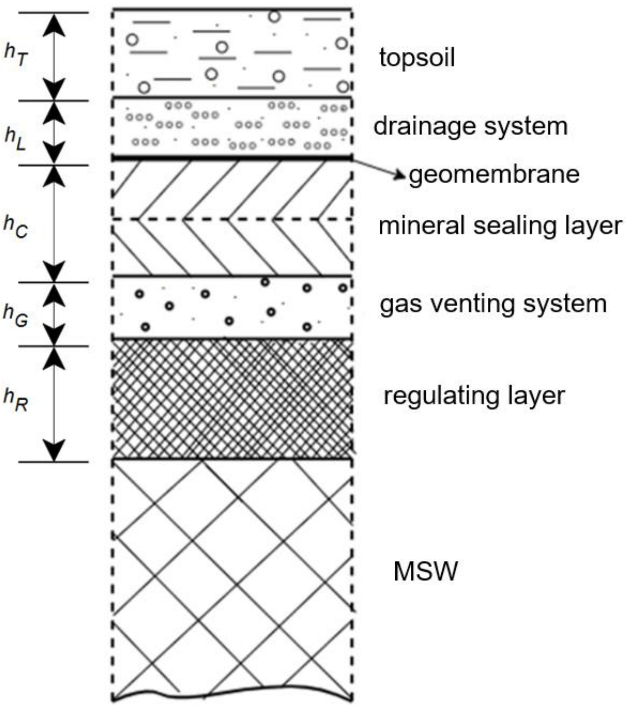


Figure 3.5. European basic cover system recommended by ISSMGE (Jessberger et al., 1993, modified by author).

3.5. Challenges of conventional mineral and synthetic cover systems

Nowadays, the key challenges in maintaining the integrity and performance of conventional mineral and synthetic covers – particularly with regard to infiltration control are becoming increasingly apparent. To further illustrate these challenges, Manassero et al. (1997) compared two most commonly used cover systems: CCL and composite cover with GM to evaluate percolation by these materials. According to their research, in CCL systems the flow through the liner was observed at the beginning of 1990 and based on the matric suction data, the sudden increase of percolation at the end of 1992 seems to be due to the appearance of tension cracks (Fig. 3.6a). In the composite barrier cover system, the percolation remains very low and no tension crack in the clay does not affect the leakage rate (Fig. 3.6b). This raises the question of whether the result is due to the GM preventing cracking which in turn stops vapor water evaporation or if the GM itself is responsible for inhibiting percolation.

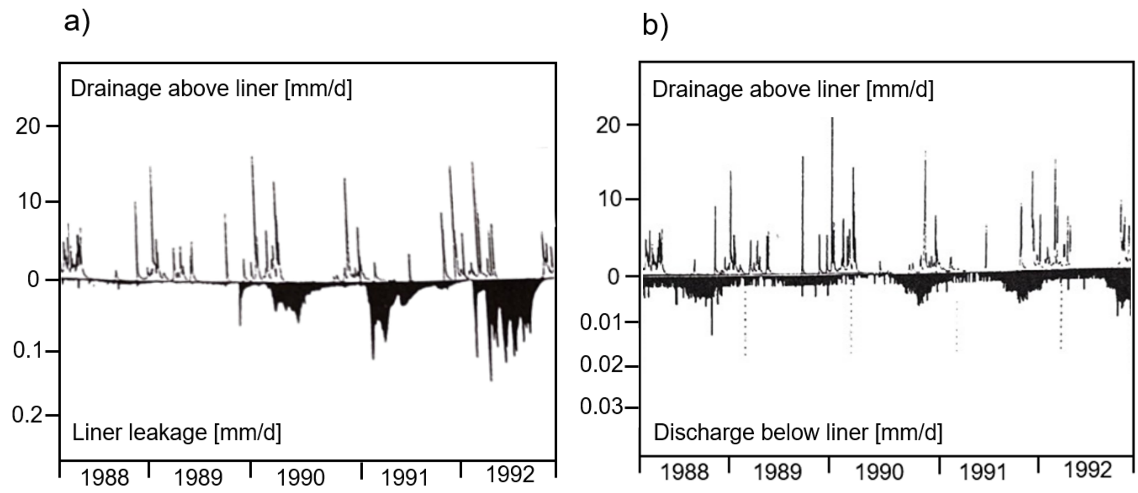


Figure 3.6. Discharges above and below cover systems: a) compacted clay liner, b) composite liner acc. to Manassero et al. (1997).

However, it is important to note that the effectiveness of the CCL barrier depends on the hydraulic conductivity. Studies have suggested that the hydraulic conductivity of the clay in CCL can be weakened by cycles of drying, differential settlement, and thawing, which creates pores in the soil (Ojasanya and Dewoolkar, 2024). In contrast, the GM cover must consider the possibility of defects, the size and shape of which determine the flow rate through the defects. The increase of hydraulic pressure over the GM defect is a key factor in determining the flow rate, and the difference in hydraulic pressure encountered by the GM defect system regulates the flow rate through the defect (Emmanuel, 2014). On the other hand, installing GM on landfill sites is a complex process that requires precise preparation of the substrate (the surface must be stable, uniform, fine-grained, and free of holes) and careful installation. During installation, the aim is to minimize the number of seams using double-wedge seam technology with electronic control of the welding parameters. This process is strictly controlled by models that combine welding parameters with seam geometry and long-term joint behavior, as confirmed by technical standards (e.g., German DVS 2225-4) (Müller and Wöhlecke, 2019). Fig. 3.7 shows the process of installing a GM during the reclamation of a MSW landfill in the CR. During the measurements, an external voltage is introduced into the barrier, ensuring the early and accurate detection of defects. The minimum detectable defect size is usually at least 5 mm in diameter. Thus, LDS enable effective control of the integrity of the cover system and minimize the risk of leaks (Müller and Wöhlecke, 2019), which is particularly important when the material ages over time. LDS are extremely important due to GM's ageing processes.

Figure 3.7. Installation of GM during reclamation work (own photos).

In general, HPDE GM consist of 96–97.5% of polyethylene resin, 2–3% of carbon black and 0.5–1.0% of other additives such as antioxidants and stabilizers and may be may be expected to experience some degradation or aging with time that will lead eventually to its failure (Rowe, 2002). Rowe (2005) estimated HDPE GM lifespan at MSW landfills at ~160 years, but field tests by Sun et al. (2019) have shown that the life can be shorter, even less than 10 years. Maintaining a GM under harsh conditions, such as prolonged ultraviolet (UV) exposure and an insufficient protective layer, significantly accelerates ageing. As a result of material degradation and the development of defects, leachate flow increased from 0.05 m^3/d at the beginning to 32.5 m^3/d in the 100-year simulation (Sun et al., 2019). In view of this, as mentioned earlier, it is recommended to increase the monitoring of the ageing process and use geoelectric defect detection methods to extend the life of the GM and reduce the negative impact on the environment.

From a geotechnical perspective, GM improves the stability of the landfill cover by providing a highly impermeable barrier that prevents leachate infiltration, thereby maintaining the structural integrity of the cover. It also minimizes differential settlement and provides protection against erosion, thereby increasing the overall stability of landfills. However, on the other hand, the use of a GM can also promote sliding of material on the cover, especially on steep slopes (Koerner, 2012). The geotechnical stability of sloped multilayer cover relies on the shear strength at different interfaces (Cortellazo et al., 2022). The final covers of landfills are usually sloped to maximize the capacity of landfilled waste and facilitate surface runoff. Low interface shearing resistance between components of the cover systems limits the steepness of the slopes (Datta, 2009). Depending on the characteristics of the stored waste, the safe slope is

1(V):3(H) to 1(V):2.5(H) (Wysokiński, 2009). However, steeper slopes are also not recommended as the potential for erosion and slope failure. Side slopes are typically steeper, often designed with a ratio of 1(V):2(H) for soil covers and 1(V):3(H) or flatter for covers that include geosynthetics (Chetri, 2021). Using a textured GM instead smooth, improves interface shear strength (Datta, 2009). This ensures that the slopes maintain stability and proper drainage, balancing the potential for erosion and structural integrity. GM cover according to the Louisiana Administrative Code (LAC) is not recommended on slopes steeper than 1(V):4(H) (Romero et al., 2023). Nevertheless, in some countries around the world, including the US, a final cover with a GM placed on a slope of 1(V):4(H) is the cause of a landslide (Benson et al., 2012). A similar case was described by Zhao and Karim (2018), who demonstrated that a cover system consisting of topsoil, vegetative soil, drainage sand, PVC GM, geosynthetic clay liner (GCL), and gas relief layer with a slope of 14° (1V:4H) and height of 60 feet (~18 m) failed owing to downslope movement along the GM interface.

The contributing factors were low shear strength at the GM–soil interface, excessive pore pressure, and gas pressure below the GM. Over one year of seasonal dry-wet cycles, the soil shear strength decreased in bare clay covers but increased in vegetation-covered clay. Over one year of seasonal dry–wet cycles, the soil shear strength decreased in bare clay covers but increased in vegetation-covered clay. These changes primarily influenced soil cohesion, with no notable effect on the internal friction angle. According to Koerner and Daniel (1997), the installation of a GM or CCL on a slope requires special attention because of the possibility of slope instability, which can result from gravitational, seepage, and seismic forces, among others.

As there is a wide range of materials used in MSW landfill reclamation, it is important to adapt the protection technology to local conditions, which affects the durability of the landfill cover and the ability to reuse the reclaimed land. It is essential to use appropriate liner and drainage materials to minimize the negative impact of landfills on the environment, which is important in long-term reclamation processes.

3.6. Alternative landfill covers

An alternative cover system refers to modern solutions for landfill covers that aim to replace or supplement traditional conventional mineral or synthetic systems. The primary objective of these systems is to minimize rainwater infiltration. Scientific research on alternative landfill cover systems has been growing rapidly due to their economic and environmental advantages. One significant advantage of alternative covers is their lower construction and maintenance costs compared with conventional methods (Sharma and Reddy, 2004; Chetri and Reddy, 2021). In designing landfill cover systems, preventing rainwater infiltration is always a key consideration. One promising solution is the capillary cover, which reduces leachate production by utilizing a capillary system. Here, the crucial aspect is the careful selection of materials with different granulometries: the capillary layer is constructed from fine sand (0–1 mm), while the capillary block is built from grains ranging from 0.7–2.0 mm (maximum 3.2 mm) and installed on an appropriate slope (5–15°) to optimize water drainage. Experience from pilot installations indicates a system efficiency of 90% (decreasing with rainfall exceeding 10 mm/d) (Cossu and Garbo, 2018). This is important because the conventional system with mineral cover may not ensure long-term protection against water infiltration because of the formation of desiccation cracks, limited retention of topsoil, and an increase in barrier permeability owing to freeze/thaw cycles and root activity (Hauser et al., 2001). However, the capillary system also has some limitations, including the tendency to clog pores in fine-grained soil after saturation resulting in impeded gas transport, which can lead to LFG accumulation under the barrier layer (Chetri and Reddy, 2021).

Building on these developments, anisotropic barriers have been introduced as another type of capillary barrier. Their layered structure—featuring varying soil properties and compaction techniques—restricts downward water flow while promoting lateral movement (Chetri and Reddy, 2021). Additionally, anisotropic barriers and ET covers are less expensive to install than traditional covers, and their effectiveness is particularly high in dry and semi-arid climates, although they exhibit limitations similar to those of ET covers (USDOE, 2000).

Complementary to the physically engineered systems are those that utilize natural processes through vegetation. ET covers and phytocovers employ the evaporation and transpiration of plants to reduce water infiltration. In practice, these systems involve placing a soil layer over the landfill and planting appropriate vegetation that absorbs water and releases it back into the atmosphere, thereby reducing leachate formation (Rock, 2010; Cossu and Garbo, 2018; Arifuzzaman et al., 2024). Wan et al. (2016) showed that vegetation roots also not only reduced water infiltration but also significantly increased the slope safety factor of the cover compared to bare cover under dry-wet cycles. However, this positive effect decreased with increasing slope length and slope angle.

Another alternative is biocover system, which is based on microbial CH_4 oxidation on the full surface. Various materials—such as soil, compost, and mixtures of sand and compost—are used to facilitate this process (Kriipsalu et al., 2023). In a biocover, CH_4 flowing through the biocover is converted to CO_2 by methanotroph bacteria under aerobic conditions in the processes of microbial oxidation. The entire process is referred to as microbial oxidation of CH_4 (Bajwa et al., 2022). Biocover increase the level of CH_4 oxidation nevertheless, the lack of O_2 at deeper levels can be a limiting factor for CH_4 oxidation (Thomasen et al., 2019). Another issue, is related to the lack of maturity of the compost which may degrade further, increasing oxygen demand and promoting CH_4 production instead of oxidation (Chetri and Reddy, 2021). Kriipsalu et al. (2023) showed that the CH_4 oxidation process stabilizes in biocover in 5–6 years after establishing the cover. Furthermore, a study by Reddy et al. (2021) demonstrated that adding biochar to silty clay is an effective method for reducing CH_4 emissions in landfill covers. Biochar not only aids in CH_4 oxidation but also enhances water-holding capacity, hydraulic conductivity, and the content of organic matter and fixed carbon, thereby creating more favorable conditions for the growth of CH_4 oxidizing bacteria. Finally, a system similar to the conventional synthetic cover is the exposed GM cover, in which the landfill is sealed with a GM without an overlying soil layer. One of the main environmental advantages of exposed GM systems is that there is no infiltration of rainwater into the landfill, and the costs are lower than those with an additional soil layer over the GM (Li et al., 2020). However, such systems are more susceptible to environmental damage such as hail, increased volume of stormwater runoff, limited vehicle access, and greater susceptibility to wind uplift (Perera et al., 2012).

4. Characteristics of the research sites

The following chapter describes the locations of the analyzed research landfills. Their hydrogeological conditions, construction stages, and reclamation phases were characterized based on archival material. The landfilled wastes at the studied landfills are also characterized in this chapter.

4.1. Municipal solid waste landfill in Zakroczym

4.1.1. Site description

Landfill is located at Byłych Więźniów Twierdzy Zakroczymskiej street (52°26'24"N 20°37'27"E) in Zakroczym (Poland), in the municipality of Zakroczym in Masovian Voivodeship, in the northeastern part of the area covered by the administrative borders of Zakroczym, at a distance of about 1 km from the city center. The landfill is located on plot No. 34, precinct 02-11, on provincial road No. 62 from Nowy Dwór Mazowiecki to Płock.

The facility is adjacent to (Fig. 4.1):

- from the north – agricultural land and further, the S7 Warsaw-Gdańsk expressway,
- from the east – industrial areas with concrete production STEFANCO Sp. z o.o. and road construction company WMB-Mostostal Warsaw,
- from the south – national road No. 62Warszawa-Płock and further agricultural land,
- from the west and north-west – areas of the Zakroczym Fort and buildings of the former agricultural cooperative, parking lots, Trzos L. concrete plant.

The closest residential buildings are located 250 meters to the south of the landfill boundary. In the immediate vicinity of the landfill in the town of Zakroczym, there are no areas subject to protection in accordance with the *Act of April 16, 2004, on Nature Protection as amended* (Ustawa z dnia 16 kwietnia 2004 r. o ochronie przyrody, Dz. U. 2004, poz. 880).

Figure 4.1. Location of the Zakroczym landfill.

The functional set-up of the landfill consists of:

- cells: western (1) (1.3 ha), southern (2) (2.0 ha), eastern (3) (0.7 ha);
- social container;
- gas treatment container;
- disinfection paddling pool;
- scale with a capacity of 50 Mg;
- domestic sewage system with a septic tank without outflow;
- leachate tank with capacity 780 m^3;
- leachate drainage system with a length of approx. 670 m;
- leachate water pumping station;
- degassing wells and containerized LFG utilization station;
- water supply connection;
- groundwater control and measurement network (piezometers:P-1, P-3, P-6 i P-7);
- fencing of the area with a height of 2 m;
- isolation greenery with a width of min. 10 m.

Considering the morphological composition of the waste stored in landfills, it can be concluded that it has diverse character. Landfilled waste can be divided into 6 main groups: group 2 (waste from agriculture, horticulture, and food processing), group 4 (waste from the leather and textile industry), group 16 (waste not included in other groups), group 17 (construction waste), group 19 (waste from waste processing and from sewage treatment and water treatment plants), and group 20 (municipal waste) (*Regulation of the Minister of Climate of 2 January 2020 on the waste catalogue*) (Rozporządzenie Ministra Klimatu z dnia 2 stycznia 2020 r. w sprawie katalogu odpadów). Detailed information on the landfilled waste by group is presented in Tab. 4.1.

Table 4.1. Morphological composition of waste deposited at the Zakroczym landfill site.

Waste group	**Description**
02 03	Wastes from fruit, vegetables, cereals, edible oils, cocoa, coffee, tea and tobacco preparation and processing; conserve production; yeast and yeast extract production, molasses preparation and fermentation
02 06	Wastes from the bakery and confectionery industry
04 02	Wastes from the textile industry
16 03	Off-specification batches and unused products
16 11	Wastes linings and refractories
16 81	Wastes generated as a result of accidents and unforeseen events
16 82	Wastes generated as a result of natural disasters
17 01	Concrete, bricks, roof tiles, and ceramics
17 03	Bituminous mixtures, coal tar and tarred products
17 05	Soil (including excavated soil from contaminated sites), stones and dredging spoil
17 08	Gypsum-based construction material
17 09	Other construction and demolition wastes
19 01	Wastes from incineration or pyrolysis of waste
19 05	Wastes from aerobic treatment of solid wastes
19 08	Wastes from waste water treatment plants not otherwise specified
19 09	Wastes from the preparation of water intended for human consumption or water for industrial use
19 12	Wastes from the mechanical treatment of waste (for example sorting, crushing, compacting, pelletizing) not otherwise specified
20 02	Garden and park waste (including cemetery waste)
20 03	Other municipal waste

According to the archive data, the dominant landfilled fraction is fine waste of less than 10 mm, accounting for 40.5% of the total mass. The second largest group was other mineral wastes (39.7%). Paper and cardboard waste accounted for 11.4% and plastic waste accounted for 7.6%. The remaining small portion (0.8%) consisted of other organic waste (Annual landfill report for 2022 by SGS). The composition of the waste showed a predominance of mineral and fine fractions, which may be important for assessing the impact on landfill processes.

4.1.2. Geological structure and hydrogeological conditions

The Zakroczym landfill is located in a former aggregate pit, within glaciofluvial sediments (gravels and sands) characterized by a significant proportion of sand and a variable clay content. The deposit has a seam-like structure, is not affected by water logging, and its base lies at a depth of approximately 8 meters, with a thickness reaching up to several meters. Geological surveys conducted as part of the landfill upgrade project (including exploratory boreholes, well drilling, electrical resistivity soundings, and field mapping) revealed that quaternary formations reach a thickness of up to 50 m. The lower layers are predominantly water-saturated sands, forming an aquifer tapped by drilled wells at depths of 30 to 37 m. The upper layers consist of alternating sands and clays, with the local absence of the clay fraction. In the geotechnical boreholes of the landfill (up to 15 m deep), no water was suspended on overlying layers or clay liners, inhibiting infiltration. The hydrogeological profile (B-B') (Fig. 4.2) shows a cross-section through the soil layers from south (S) to north (N), considering the geological profiles of piezometers P-3 and P-1.

4.1.3. Landfill construction and reclamation

The MSW landfill in Zakroczym was established in the 1970s. The extraction of aggregates was stopped in the early 1980s, after which the landfill was established and operated in an unorganized and unlined manner. The modernization of the landfill began in 1996. Zakroczym landfill is divided into three cells, west, east, and south, with slopes of 1(V):2(H). The first was the western cell which began receiving waste on 18.11.1997 (the cell was sealed in 2008). The second was the eastern cell, which started operation on 10.2011, and the last was the southern cell, which started operation on 07.2014. The eastern cell was built at a ground ordinate of 97.0 m above sea level, while the eastern and southern cells were built slightly higher at ordinates of 109–111 m above sea level. The landfill has a natural cohesive barrier with a thickness of about 1.5–5.0 m and $k = 6.8\times10^{-11}$ m/s but the impermeable layer is discontinuous and does not cover the entire landfill bottom.

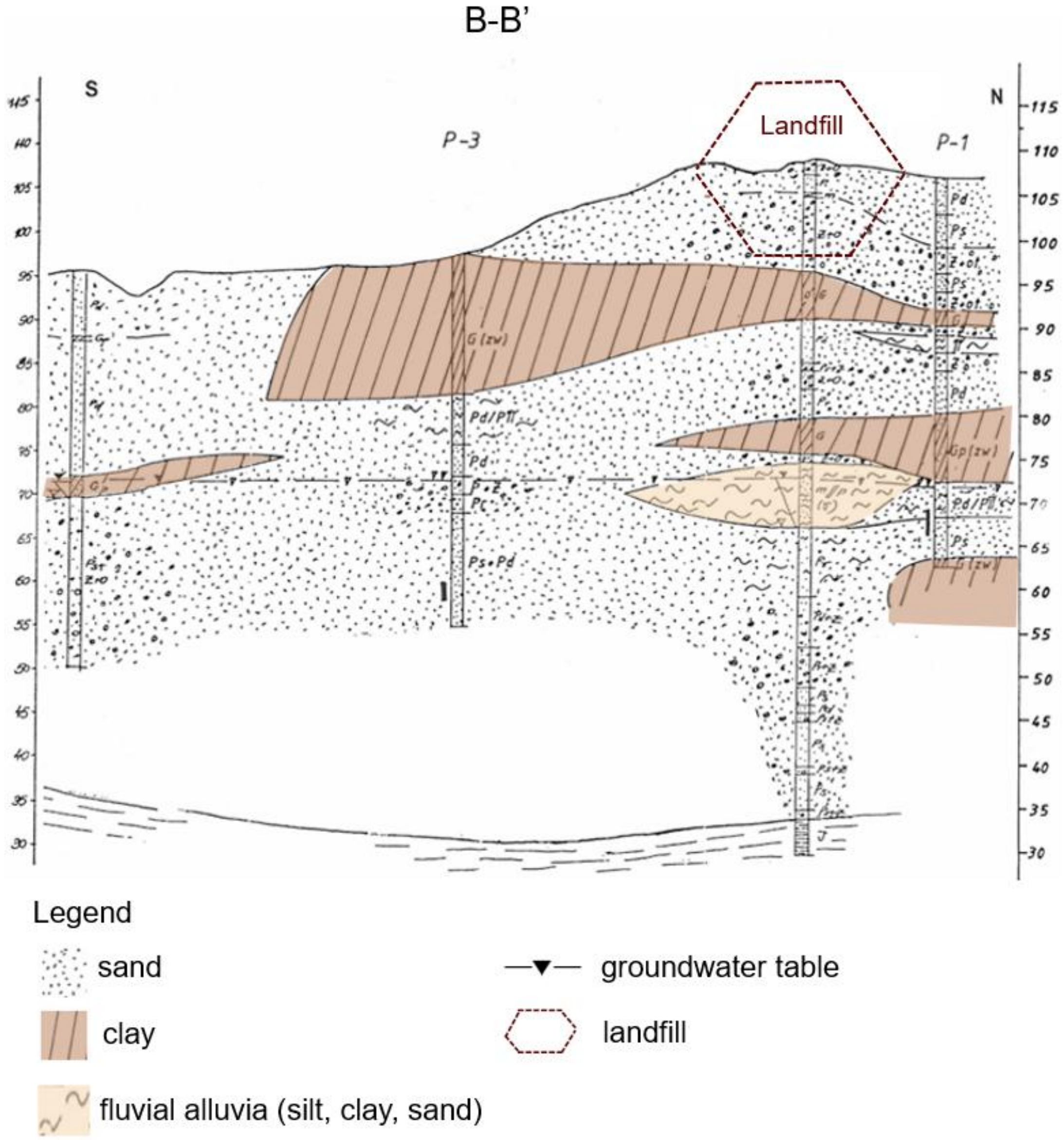


Figure 4.2. Hydrogeological cross-section (source: archival materials of the Zakroczym landfill, modified by author).

In other parts there are permeable formations in the form of medium and coarse sands. Accordingly, an artificial geological barrier with a thickness of 0.5–0.7 m was installed to supplement the cell sealing. To increase the protection of the geological barrier, synthetic insulation was also applied to all cells using HDPE GMs or PLASTPAPA foil with a thickness of 1.5–2.0 mm. The final component of the sealing of the landfill base was a drainage system made of sand ($k > 1\times10^{-4}$ m/s), with a thickness of 0.5–0.7 m, and perforated pipes for the transport of leachate to impermeable tank. Fig. 4.3 shows the sealing schemes for the base of the Zakroczym landfill, with a division into individual cells.

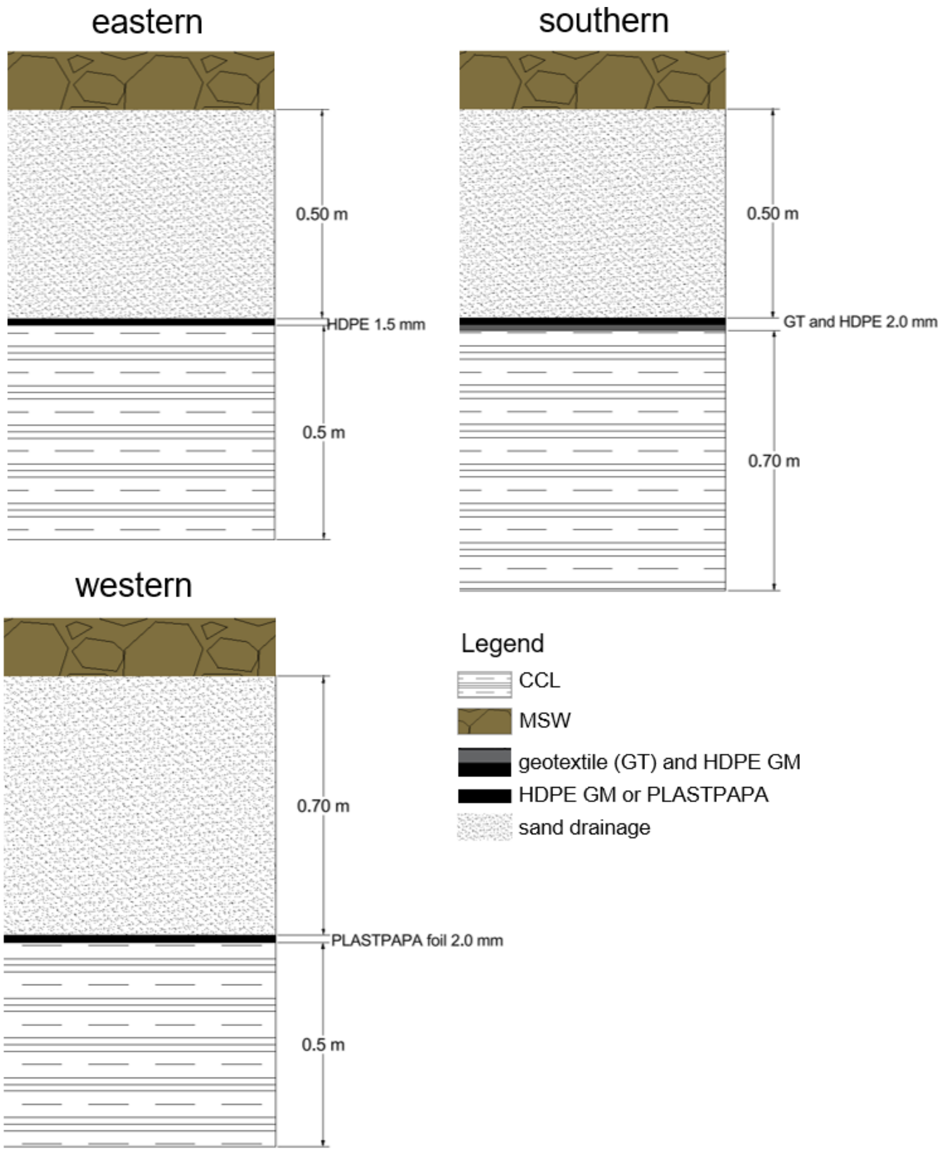


Figure 4.3. Zakroczym landfill sealing in eastern, western and southern cells (own study).

The body of the landfill was formed from delivered waste, which was placed in layers within designated work plots and compacted using a bulldozer and compaction roller. After reaching a thickness of approximately 2.0 m, the waste was covered with a mineral isolation layer thickness of 0.2–0.3 m. The landfill cells have a leachate drainage system made of PVC pipes directed to the collection well of the pumping station. The cells also have a biogas capture system consisting of gas collection wells equipped with a monitoring system. These are connected to a containerized biogas utilization station, where electricity is generated by combustion in an engine coupled to a power generator.

After waste acceptance at the western cell ceased on 31 December 2011 at Zakroczym landfill, technical and biological reclamation works started and lasted until

2017. The first stage of the work involved cleaning the site, compacting the waste, and levelling the slopes. This was followed by levelling and gas collection layer of mineral, organic waste and sand. A 0.5 m layer of clayey sand with $k \leq 10^{-7}$ m/s (according to empirical equation Hazen-Tkaczukowa) was laid on top of the levelling layer. Next, a 0.7 m thick reclamation cover was placed on the top, composed of soil with a humus content of approximately 2%, which is in accordance with §17 of the *Regulation of the Minister of the Environment on April 30, 2013 on landfills*, which assumes that „the minimum thickness of reclamation cover for a landfill of non-hazardous and inert waste is not less than 1 m and allows the formation and maintenance of a permanent vegetation cover". Fig. 4.4 below shows the cover scheme used at the landfill in Zakroczym.

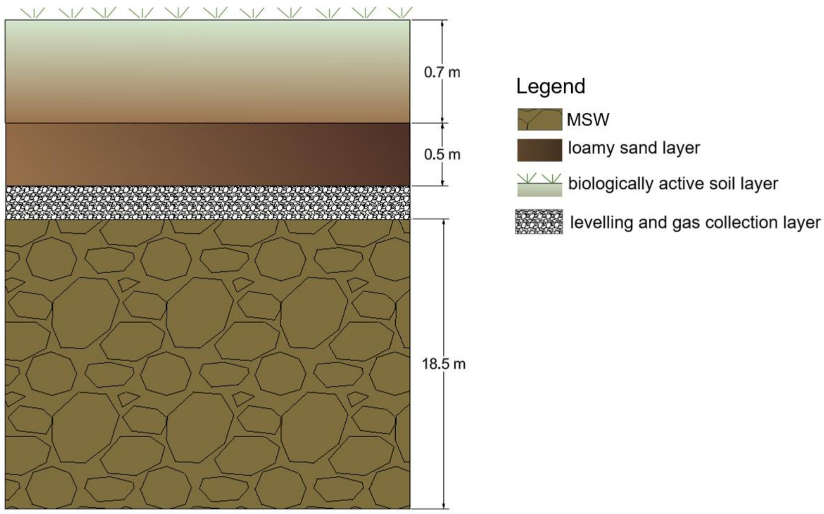


Figure 4.4. Landfill reclamation cover system in Zakroczym (own study).

Analysis of the topography of the reclaimed landfill site indicated significant elevation variation within the study area, with a slope of 1(V):2(H) (Fig. 4.5). The elevation of the site ranges from 109.5 m to 118.5 m above sea level. The highest parts of the landfill (approximately 118 m above sea level) are located mainly in its northern and central parts, suggesting that a thicker reclamation layer has been imposed there, or there is less subsidence. On the other hand, the lowest parts (approximately 109.5–112 m above sea level) are found in the southwestern and southeastern areas, which may be due to more intense subsidence of the soil. The slopes and tops of the reclaimed western cell have been planted with grass mixtures, including *Dactylis glomerata* L. and meadow mix, as part of the biological reclamation process. In 2024, a 500 kWp photovoltaic system was also installed on the reclaimed land, covering an area of approximately 1 hectare (Fig. 4.6).

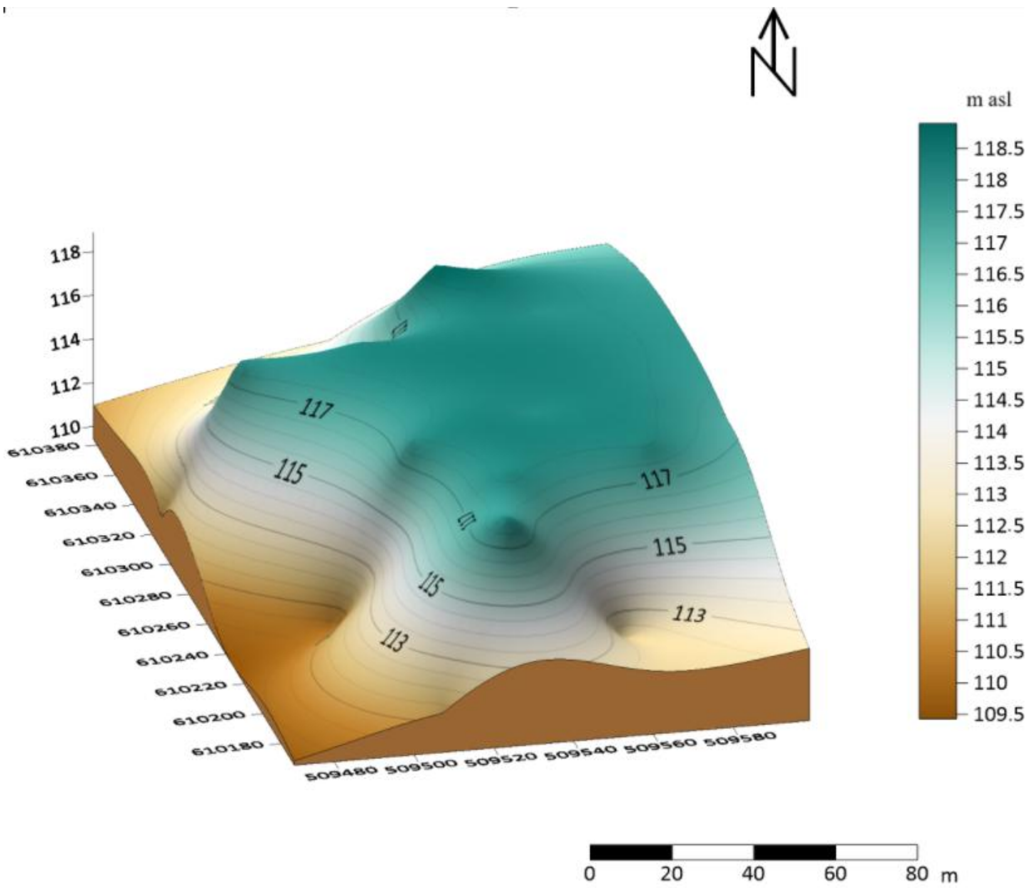


Figure 4.5. Topography of the reclaimed western cell of the Zakroczym landfill (own study).

Figure 4.6. The western cell at the Zakroczym landfill in 2022-2024 (own photo).

4.2. Municipal solid waste landfill in Zdounky

4.2.1. Site description

The DEPOZ landfill in Zdounky (49°14'29" N 17°18'30" E) is located in the Kroměříž district (Zlín region), CR. The landfill is located in a triangular area formed by state roads between the villages of Zdounky, Nětčice, Troubky – Zdislavice, 750 m west of the edge of the village of Zdounky and another 450 m west of the edge of the village of Nětčice. The closest residential buildings are about 430 m to the north of the landfill boundary. There are no protected areas in the close vicinity of the Zdounky landfill.

The site is adjacent to (Fig. 4.7):

- from the north – national road No. 43215, agricultural land and the Nětčice village,
- from the east – national road No. 42817 and agricultural land,
- from the south – agricultural land and further Zdounky village and Olśinka river,
- from the west – agricultural land and Lipina river.

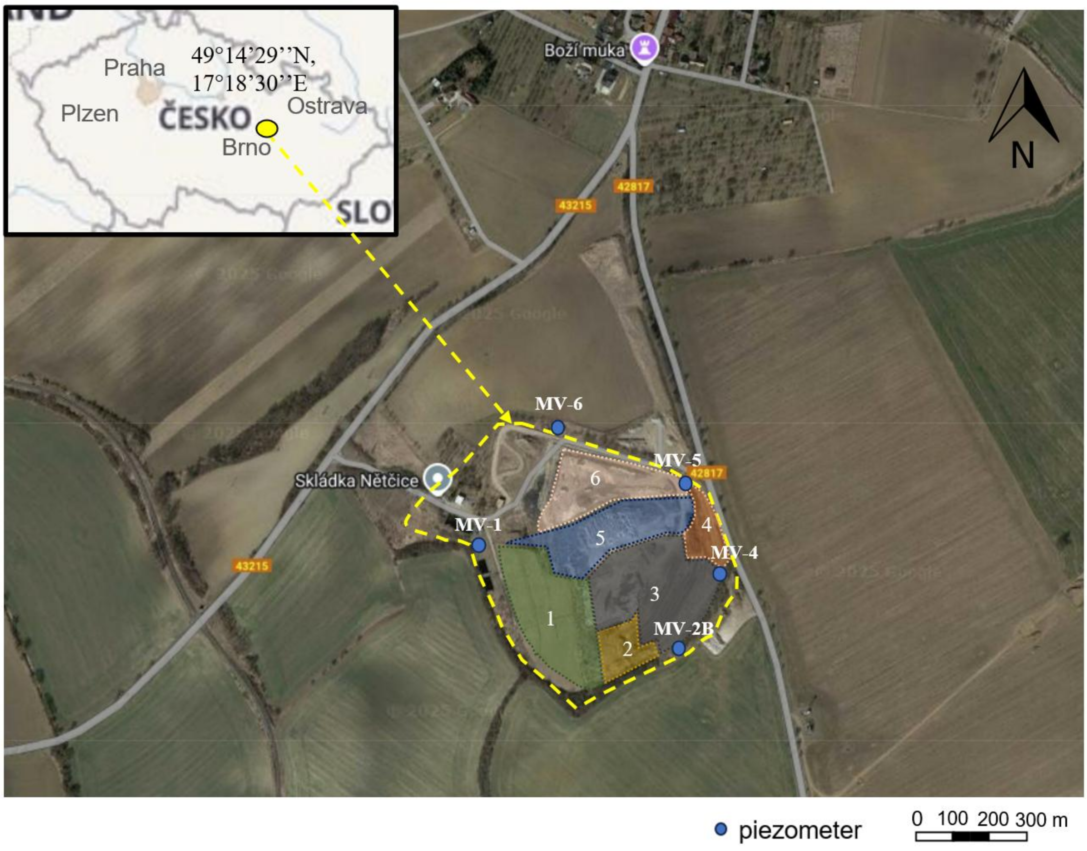


Figure 4.7. Location of the Zdounky landfill: (1) stage 1, (2) stages 2a and 2b, (3) stage 3, (4) stage 4, (5) stage 5, (6) stage 6.

The functional set-up of the landfill consists of:

- cells: stage 1 (1.92 ha), stages 2a and 2b (1.02 ha), stages 3a and 3b (1.46 ha), stage 4 (0.58 ha), stage 5 (0.693 ha) and stage 6 (0.61 ha),
- administration building,
- leachate drainage of about 840 m length,
- leachate tank with capacity 2×430 m^3,
- surface water tank with dimensions 38×16 m,
- collection ditches,
- bridge scale,
- groundwater monitoring system (piezometers: MV-1, MV2B, MV-4, MV-5 and MV-6),
- composting plant with an area of 50×40 m and a capacity of 2450 tons/year (operation began on 04.2012),
- landfill degassing system,
- construction waste disposal site.

The Zdounky landfill is assigned to a controlled group S-OO (“other waste”) landfill, classified as a subgroup S-OO3 in terms of technical security, intended to store MSW from neighboring sites (Podlasek et al., 2022). In the northern part of the landfill, there is also a separate sector of subgroup S-OO1 for the disposal of construction wastes, such as gypsum-based wastes, stabilized wastes, high-sulfur wastes and wastes with elevated metal content or asbestos-containing wastes.

Landfilled waste can be divided into 11 main groups: group 1 (waste from prospecting, mining, physical and chemical processing of ores and other minerals), group 2 (waste from agriculture, horticulture and food processing), group 3 (waste from wood processing and the production of panels and furniture, pulp, paper and cardboard), group 4 (waste from the leather and textile industry), group 10 (waste from thermal processes), group 12 (waste from shaping and physical and mechanical surface treatment of metals and plastics), group 15 (packaging waste; sorbents, wiping cloths, filter materials and protective clothing not included in other groups), group 16 (wastes not included in other groups), group 17 (construction waste), group 19 (waste from waste treatment processes and wastewater and water treatment plants), and group 20 (municipal waste) (Tab. 4.2). A reduced amount of the biodegradable fraction is disposed of at the Zdounky landfill, as there is a composting plant on the site that accepts and processes organic waste, such as sludge from the washing of raw materials (02 03 01), raw materials and products unfit for consumption (02 03 04), waste from the distillation of spirits (02 07 02), sawdust and industrial wood waste (03 01 05), and biodegradable municipal waste (20 02 01), which decreases landfilling and increases the efficiency of organic recycling.

Table 4.2. Morphological composition of waste deposited at the Zdounky landfill site.

Waste group	**Description**
01 04	Wastes from physical and chemical treatment of non-metallic minerals
02 03	Wastes from fruit, vegetables, cereals, edible oils, cocoa, coffee, tea and tobacco preparation and processing; conserve production; yeast and yeast extract production, molasses preparation and fermentation
03 01	Wastes from wood processing and the production of panels and furniture
04 02	Wastes from the textile industry
10 01	Wastes from power stations and other combustion plants (except 19)
10 09	Wastes from casting of ferrous pieces
10 10	Wastes from casting of non-ferrous pieces
10 12	Wastes from manufacture of ceramic goods, bricks, tiles and construction products
10 13	Wastes from manufacture of cement, lime and plaster and articles and products made from them
12 01	Wastes from shaping and physical and mechanical surface treatment of metals and plastics
15 01	Packaging (including separately collected municipal packaging waste)
16 11	Waste linings and refractories
17 01	Concrete, bricks, tiles and ceramics
17 02	Wood, glass and plastic
17 03	Bituminous mixtures, coal tar and tarred products
17 05	Soil (including excavated soil from contaminated sites), stones and dredging spoil
17 06	Insulation materials and asbestos-containing construction materials
17 09	Other construction and demolition wastes
19 05	Wastes from aerobic treatment of solid wastes
19 08	Wastes from waste water treatment plants not otherwise specified
19 09	Wastes from the preparation of water intended for human consumption or water for industrial use
19 10	Wastes from shredding of metal-containing wastes
19 12	Wastes from the mechanical treatment of waste (for example sorting, crushing, compacting, pelletizing) not otherwise specified
20 02	Garden and park waste (including cemetery waste)
20 03	Other municipal wastes

4.2.2. Geological structure and hydrogeological conditions

The area of the Zdounky landfill is characterized by a geology based on Paleogene rocks of marine origin, belonging to the outer and Magurian Carpathian flysch (Fig. 4.8). Hydrogeologically, the area is homogeneous, with the dominant fracture permeability of the flysch rocks, but with a low water yield. The Olšinka depression between Chřiby and Litenčické foothhill is filled with rocks of the outer Carpathian flysch. The local translucent sediments are included in the Ždánice — Hustopeče formation in the facies of calcareous clays, saliva, and sandstones of the mantle. Towards the Litenčické hills, the frontal depth is already formed, and is filled by the predominant saliva calcareous clays and clays with a thickness of about 500–700 m. Claystones, at various degrees of weathering, were identified at the landfill area by drilling. There are no significant differences between the sub-flysch units, and therefore, the area is characterized with a common fractured permeability.

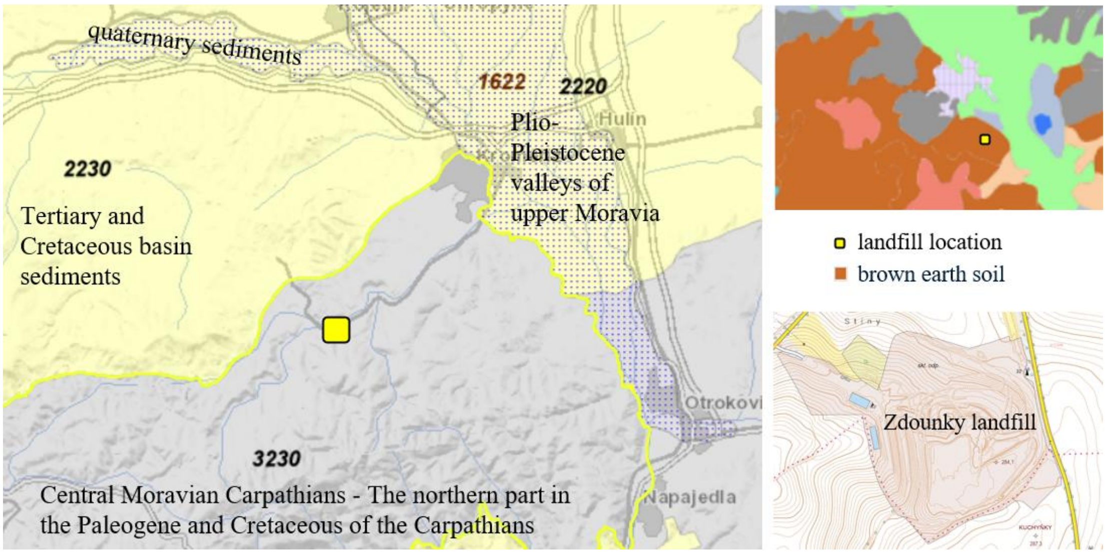


Figure 4.8. Geological conditions around the Zdounky landfill (source: *https://geoportal.gov.cz/web/guest/map?openNode=Soil&keywordList=inspire*, access date: 1.04.2025).

The hydrographic axis of the area is the Lipinka stream, which flows on the western edge of Zdounky into the stream Olšinka with an average flow at the mouth of 0.13 m^3/s. The depth to groundwater table is ranging from 4 to 14 m below the surface level. The hydraulic gradient is almost equal to 70‰. The groundwater level fluctuates significantly. Relatively significant changes are registered over time and year-on-year. A significant fluctuation of groundwater levels is characteristic of a low-permeable environment (with a low active porosity). The piezometric level occurs in the formation of quaternary clayey soils, forming the artesian "ceiling" of the deeper aquifer. The aquifer thickness is ranging from few to several meters (Podlasek et al., 2021).

4.2.3. Landfill construction and reclamation

The MSW landfill in Zdounky was constructed in the 1990s in a significant morphological depression which is described as a local narrow valley open to the west-southwest towards the flat Lipinka valley. Prior to the landfill, the area was used for agricultural purposes, particularly for crop production. The total area of the landfill is between 251 and 280 m above sea level, while the surrounding area is between 240 and 396 m above sea level. The subsoil consisted of claystones with varying degrees of weathering. The area has low permeability owing to its geological structure, described as fracture permeability and, to a lesser extent, interstitial permeability. The landfill consists of several stages: 1 (constructed 12.1995), 2a (constructed 07.2001), 2b (constructed 08.2008), 3a (constructed 09.2004), 3b (constructed 10.2010), 4 (constructed 08.2013), 5 (constructed 12.2015), and 6 (constructed 10.2019), with slopes of 1(V):3(H). For cells sealing in stages 1, 2a, 2b, 3a, an artificial geological barrier in the form of a CCL with a thickness of 0.5–1.0 m was constructed, while for cells 3b, 4, 5 and 6 bentonite mats were laid on the subsoil. In order to increase the protection of the geological barrier, synthetic insulation was also applied in all cells using HDPE GM with a thickness of 1.5–2 mm protected by geotextile. The landfill sealing base was complemented by a drainage system of river gravel 0.3–0.5 m thick with a slope of 5% and perforated pipes to allow the transport of leachate to a special impermeable tank. In stages 4, 5 and 6, the gravel drainage was also complemented by stacked used tires. The sealing implemented in stages 3b,4, 5 and 6 provides the highest level of protection, while variants 2b and 3a provide more basic sealing solutions for the landfill. Fig. 4.9 shows the sealing schemes for the cells at Zdounky landfill.

To prevent damage to the HDPE GM, a 0.5 m layer of construction debris was first laid down on the newly constructed cell or part of it. Then the debris was deposited within the designated working areas, compacted, and covered with inert material (from a sewage treatment plant that has undergone anaerobic stabilization, soil, possibly foundry sand, and compost of insufficient quality) to prevent light parts from flying away and for hygienic reasons.

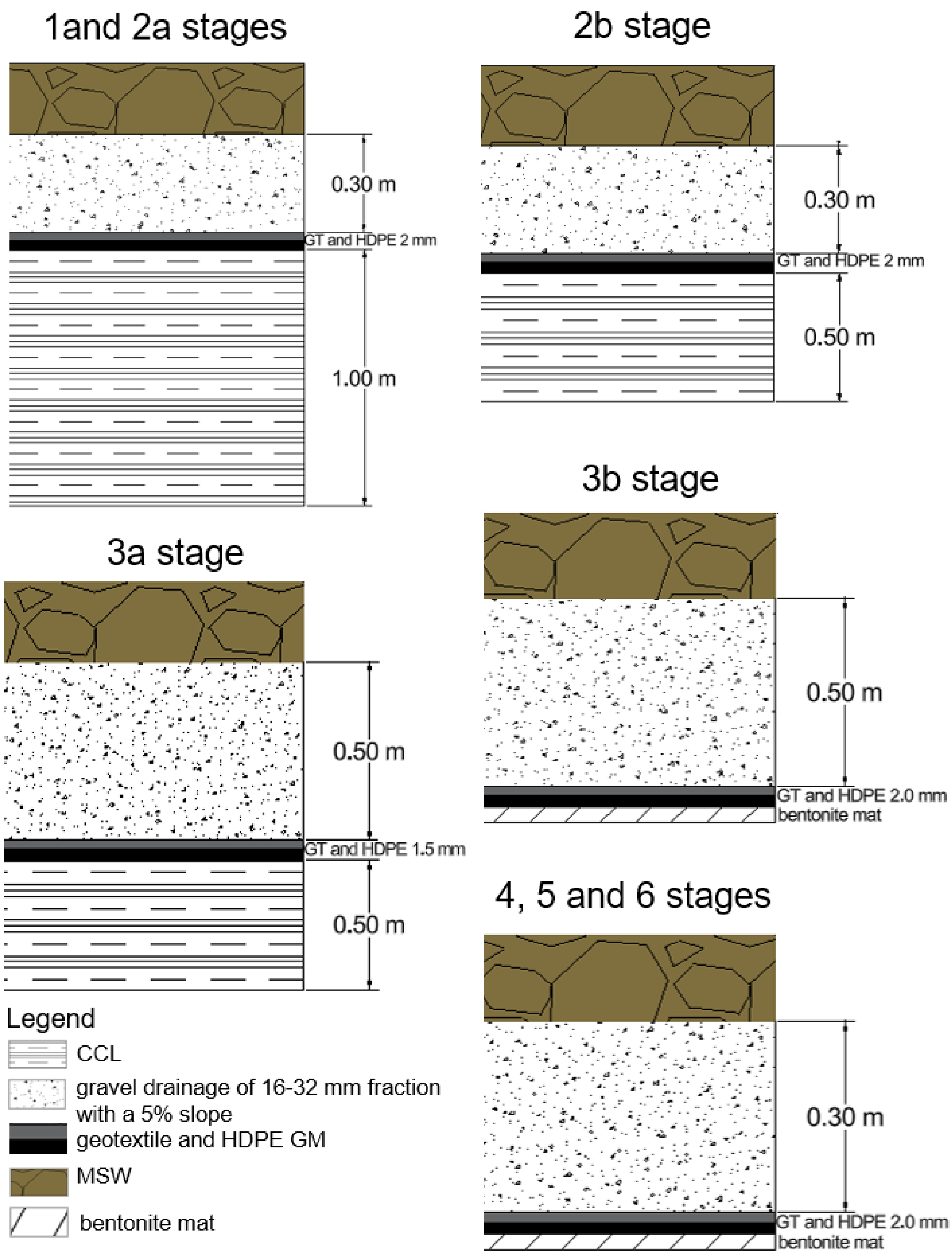


Figure 4.9. Zdounky landfill sealing in studied cells (own study).

According to the *ČSN 83 803*, the Zdounky landfill is classified as class III, which requires the construction and operation of a degassing system. LFG is continuously pumped and burned in a cogeneration unit to produce electricity. Excess electricity not used in landfills is sold to the public grid. Regular pumping and optimization of the amount of biogas pumped from this landfill allows power plants to produce electricity. The leachate is discharged through a surface drainage system and leach pipe to the leachate tank and then transported by a tanker to a LTP for treatment.

After waste acceptance at the stage 1 cell ceased in 2011 at Zdounky landfill, technical and biological reclamation works started and lasted until ~2019. The closure process was carried out in accordance with the *ČSN 83 8035* (Skládkování odpadů - Uzavírání a rekultivace składek), standard for closure and reclamation of landfills (Vaverková et al., 2019). The first stage of the work was to clean and level the landfill using of inert waste (approximately 0.25 m thick). The next stage was to cover the landfill with a synthetic material using 1 mm HDPE GM, protected from below and above by a 500 g/m^2 protective geotextile. A 300 mm drainage layer of gravel was placed on top of the synthetic layer to intercept the rainwater. The cover system was completed with a 400 mm layer of subsoil and 300 mm layer of biologically active soil. According to *ČSN 75 0145*, the most suitable materials for the cover layer are clay and sandy loam soils (Božek et al., 2006). Fig. 4.10 shows the cover scheme used at the Zdounky landfill. The reclaimed site at Zdounky is characterized by a varied relief, with elevations ranging from approximately 258 m above sea level (at the boundary of the landfill) to a maximum of 282.5 m above sea level (Fig. 4.11). The analysis of the elevation data indicates that the terrain slopes gently to the southwest, with a slope of 1(V):3(H).

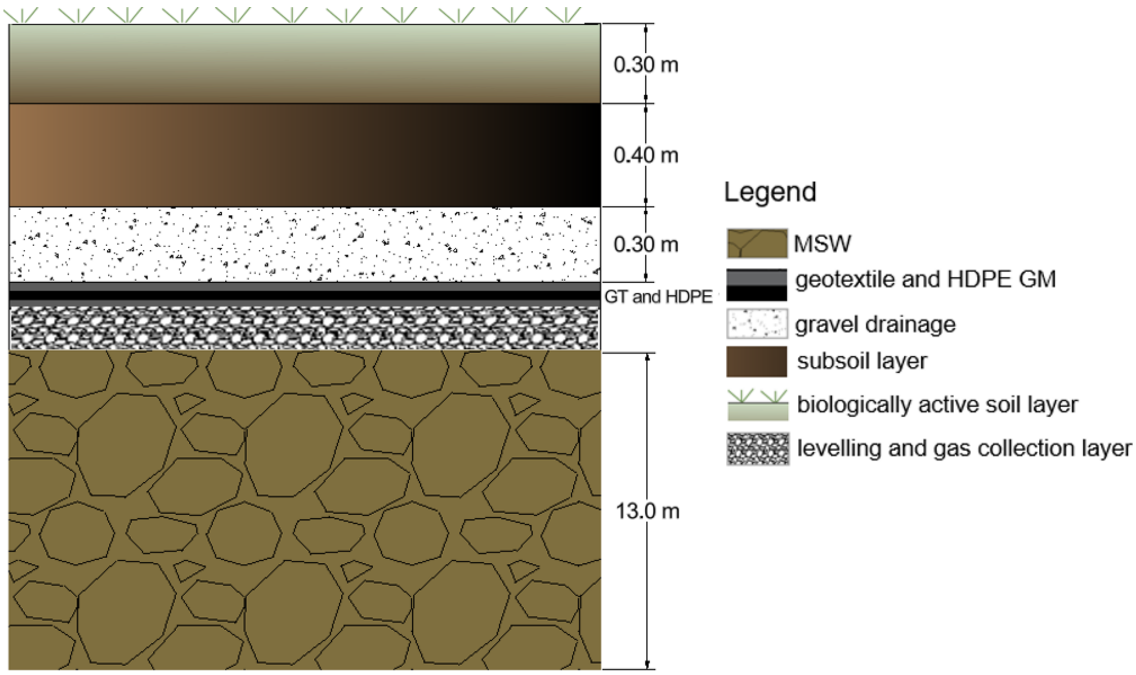


Figure 4.10. Landfill cover used at Zdounky landfill.

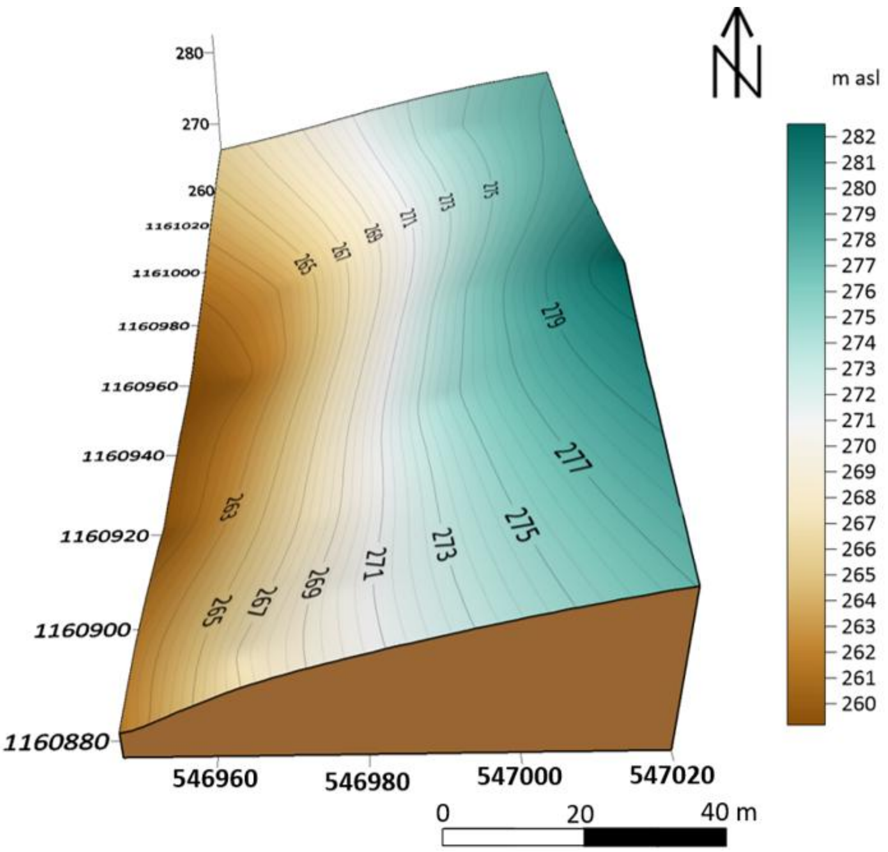


Figure 4.11. Topography of the reclaimed cell of the Zdounky landfill (own study).

Fig. 4.12 shows the changes in time (between 2017-2023) of the Zdounky landfill.

Figure 4.12. Reclaimed cell from stage 1 at Zdounky landfill.

The slopes and top of the reclaimed cell in the process of biological reclamation, were covered with grasses: *Festulolium* L. (23.6%) *Arrhenatherum elatius* L. (20.6%), *Galium album* L. (12.9%), *Calamagrostis epigejos* L. (10.5%), *Elytrigia repens* L. (9.4%).

4.3. Comparison of selected parameters of landfill

The Tab. 4.3. compares the selected landfills, Zakroczym and Zdounky, in terms of the closure and reclamation processes, considering technical and environmental aspects. Both landfills are characterized by a similar construction period (Zakroczym – 1997; Zdounky –1995) and reclamation period, which started in 2011 in both landfills and ended in 2017 in Zakroczym and 2019 in Zdounky, respectively. The areas of the reclaimed cells were comparable (Zakroczym – 1.34 ha, Zdounky – 1.92 ha), as well as the average annual precipitation (500 mm and 490 mm). In both cases, deposited waste belonged to similar categories, including municipal and construction waste. However, despite many similarities, landfills show significant differences, including the structure and slope of the reclamation cover system, the amount of waste deposited, and layers thickness.

Table 4.3. Comparison of selected characteristics of the analyzed landfills.

Comparative examples	**Closing and reclamation of a landfill cell in a selected landfill**	
	Zakroczym landfill	**Zdounky landfill**
Construction date	11.1997	12.1995
Reclamation date	2011–2017	2011–2019
Area of reclaimed cell	1.34 ha	1.92 ha
Slopes	1(V):2(H)	1(V):3(H)
Thickness of stored waste	18.5 m	13 m
Precipitation	500 mm	490 mm
Neighborhood	Expressways, industrial and production areas, parking lots, agricultural fields	National roads, farmlands, rivers
Piezometers	3 piezometers in the outflow direction from the landfill and 1 piezometer in the inflow direction	1 piezometer in the outflow direction from the landfill, 4 piezometers in the inflow direction
Landfill type	Non-hazardous and non-inert waste landfill	Landfill for waste group S-OO (other waste),

Comparative examples	**Closing and reclamation of a landfill cell in a selected landfill**	
	Zakroczym landfill	**Zdounky landfill**
		subgroup S-OO3 (MSW) with specially designated landfill sectors for subgroup S-OO1 (construction waste)
The amount of waste stored in the cell	357 000 m^3 (308 750 t)	249 600 m^3 (215 654 t)
Type of waste stored	5 main groups: group 2 (Wastes from agriculture, horticulture, aquaculture, forestry, hunting and fishing, food preparation and processing), group 4 (Wastes from the leather, fur and textile industries), group 16 (Wastes not otherwise specified in the list), group 17 (Construction and demolition wastes (including excavated soil from contaminated sites) and group 19 (Wastes from waste management facilities, off-site waste water treatment plants and the preparation of water intended for human consumption and water for industrial use), 20 (Municipal wastes (household waste and similar commercial, industrial and institutional wastes) including separately collected fractions)	11 main groups: group 1 (Wastes resulting from exploration, mining, quarrying, physical and chemical treatment of minerals), group 2 (Wastes from agriculture, horticulture, aquaculture, forestry, hunting and fishing, food preparation and processing), group 3 (Wastes from wood processing and the production of panels and furniture, pulp, paper and cardboard), group 4 (Wastes from the leather, fur and textile industries), group 10 (Wastes from thermal processes), group 12 (Wastes from shaping and physical and mechanical surface treatment of metals and plastics), group 15 (Waste packaging; absorbents, wiping cloths, filter materials and protective clothing not otherwise specified), group 16 (Wastes not otherwise specified in the list), group 17 (Construction and demolition wastes (including excavated soil from contaminated sites) and group 19 (Wastes from waste management

Comparative examples	**Closing and reclamation of a landfill cell in a selected landfill**	
	Zakroczym landfill	**Zdounky landfill**
		facilities, off-site waste water treatment plants and the preparation of water intended for human consumption and water for industrial use), group 20 (Municipal wastes (household waste and similar commercial, industrial and institutional wastes) including separately collected fractions)
Cover system	• a 0.7 m thick cover system consisting of soil with a humus content of approx. 2% • 0.5 m of cohesive layer made of clayey sand with a hydraulic conductivity $k \leq 10^{-7}$ m/s. • leveling layer made of mineral and organic waste	• 1 m reclamation cover • gravel drainage layer with thickness 0.3 m • geotextile • HDPE GM 1 mm • geotextile • leveling layer made of inert waste
Final development	The slopes and evergreen covered with grass mixtures, the composition of which included, among others, *Dactylis glomerata* L and meadow mix. A 500 kWp photovoltaic installation was built on the reclaimed cell.	Slopes and top covered with grass mixtures such as *Arrhenatherum elatius* L., *Festuca pratensis* L. and *Festuca rubra* L.
Other important elements affecting the differences between landfills	High content of fine waste, less than 10 mm in size, which occupies 40.5% of the total weight, and mineral waste, accounting for 39.7% of landfilled waste.	Composting plant, separate off-site sector for construction waste.

5. Research methodology

The diagram in Fig. 5.1. illustrates the complex methodology for assessing the effectiveness of landfill closure and reclamation, in which the emphasis is placed on multi-stage, integrated research and analysis of the obtained results. The main goal was to determine the impact of technical methods of landfill covering and reclamation on the environment, which in turn into the development of recommendations for the selection and use of cover materials under various conditions based on the research on two partially reclaimed landfills in the CR and PL. The methodology consisted of four basic groups of studies: (i) monitoring studies, (ii) laboratory analyses, (iii) model studies, and (iv) biomonitoring studies. As part of the monitoring studies, the groundwater quality in the areas adjacent to the studied landfills was directly verified, and the chemical properties of leachates and LFG were validated to track the long and short-term changes caused by the applied cover methods. Laboratory tests were used to assess the physicochemical properties of the soils used for reclamation, which helped determine whether they met the appropriate legal requirements regarding permeability or chemical composition. These analyses were complemented by modelling studies based on simulations under conditions close to real ones, which allowed to predict the changes in leachate production and LFG emissions, as well as an assess landfill stability with different cover systems. The final group of studies was biomonitoring tests, in which it was verified whether the leachate did not exhibit phytotoxic properties towards plants. Additionally, to phytotoxicity, the respiration activity of the soil used for reclamation was evaluated. Plants, owing to their ability to react quickly to stress factors, served as early and valuable bioindicators of environmental changes (Cakaj et al., 2024; Patel et al., 2024). Monitoring changes in species composition enabled the detection of subtle pollution signals that might not have been revealed by purely chemical analyses.

The results of all four types of studies provided a basis for assessing the environmental impact of landfill covers. Based on these results, conclusions and recommendations were formulated as to which landfill covers would best minimize negative environmental impacts while ensuring the safety and durability of the structure.

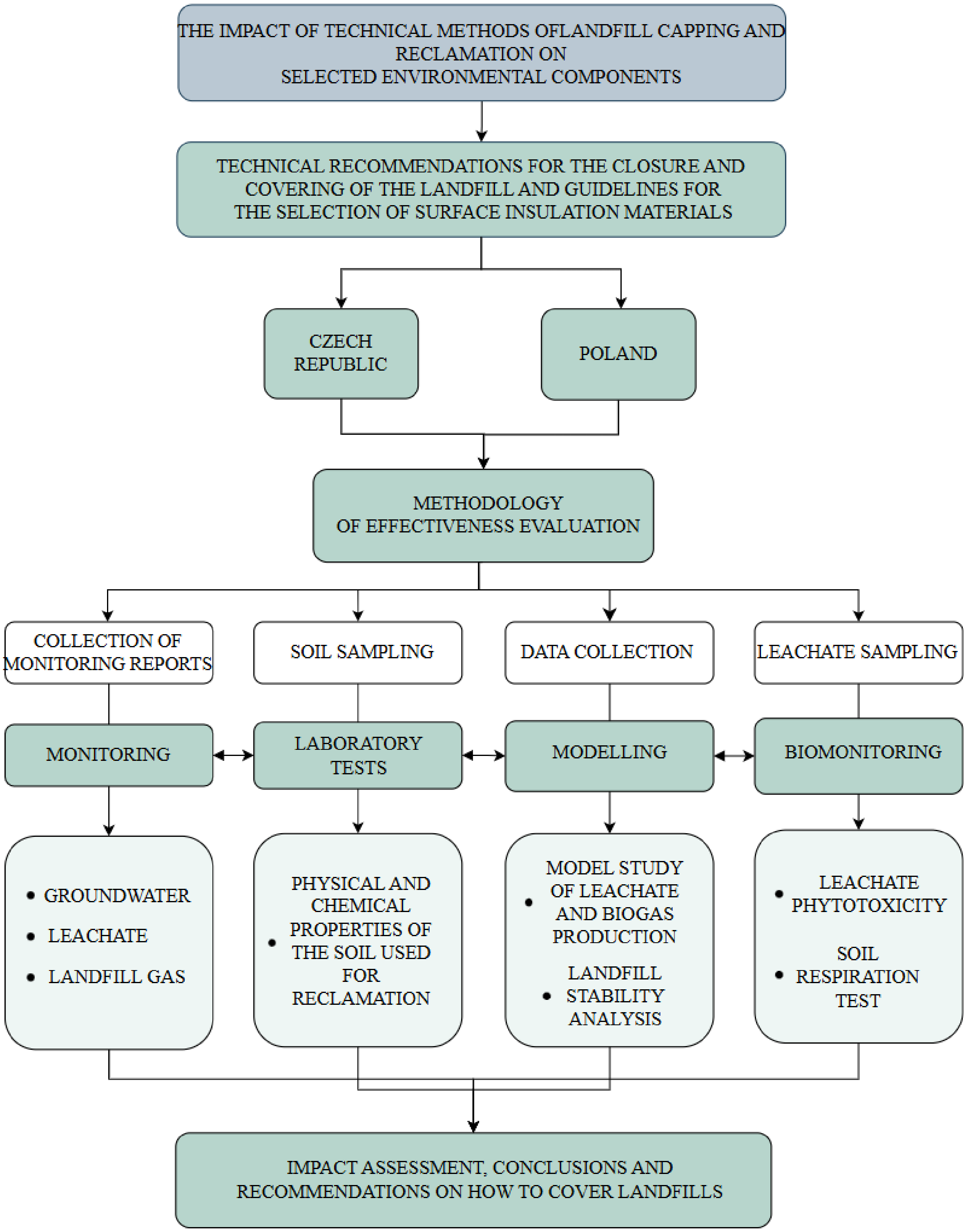


Figure 5.1. Methodological scheme of the research work.

5.1. Monitoring data analysis

5.1.1. Groundwater quality analysis

The study used data from landfills (Zakroczym and Zdounky) monitoring reports. Groundwater samples were collected from piezometers twice a year: in spring (I) and autumn (II) in the years 2008–2022, in accordance with the procedure specified in the standards PN-EN ISO 5667-3:2013-05 and PN-ISO5667-11:2004. The results of groundwater quality monitoring were compared with the limit values established for drinking water and groundwater by the World Health Organization (WHO) (2017), the *Regulation of the Minister of Maritime Affairs and Inland Navigation* of 11 October 2019 *on the criteria and method for assessing the status of groundwater bodies* (Rozporządzenie Ministra Gospodarki Morskiej i Żeglugi Śródlądowej z dnia 11 października 2019 r. w sprawie kryteriów i sposób oceny stanu jednolitych części wód podziemnych, Dz.U. 2019 poz. 2148) and US EPA (2018). Tab. 5.1 presents the water quality standards to which the results of the groundwater monitoring analysis at the studied facilities were compared. In the analysis of groundwater quality, only parameters that were measured at both locations were considered. The analysis also did not include groundwater quality monitoring for Hg, Pb and Cd due to the fact that the values obtained were below the limit of quantification.

Table 5.1. Quality standards for groundwater according to selected regulations.

Standard		**pH [-]**	**EC [μm/cm]**	**Zn [mg/l]**	**Cr_{total} [mg/l]**
US EPA (2018)	Maximum Contaminant Levels	6.5-8.5	-	-	0.1
	Secondary Drinking Water Regulations			5	-
WHO (2017)		6.5-8.5	-	water in pipes is 0.05, but underground water usually exceeds this level	0.05
Polish Regulation (2019) *	I quality class	6.5-9.5	700	0.05	0.01
	II quality class		2500	0.5	0.05
	III quality class		2500	1	0.05

**Regulation of the Minister of Maritime Economy and Inland Navigation of 11 October 2019 on the criteria and method for assessing the status of groundwater bodies*

Groundwater monitoring network at Zakroczym landfill, consists of 4 piezometers: P-1, P-3, P-6 and new piezometer P-7, capturing the quaternary aquifer. A piezometer P-1 is located on the water inflow to the landfill, while piezometers P-3, P-6 and a new piezometer P-7 capture water outflow of the landfill site. In the vicinity of the Zakroczym landfill there are mainly industrial and service areas, including the Modlin Airport, and some agricultural land is also located there.

Zdounky landfill has a groundwater monitoring network consisting of 5 piezometers: MV-1, MV-2B, MV-4, MV-5 and MV-6. Piezometer MV-1 is located in the direction of outflow from the landfill, and is therefore considered a monitoring point that reflects the actual impact of the landfill on groundwater. The other piezometers (MV-2B, MV-4, MV-5 and MV-6), are located in the direction of water inflow, and therefore present the influence of adjacent areas on the status and quality of groundwater in the area. The vicinity of the Zdounky landfill is mostly agricultural land, which could affect the quality of water in the area. Fig. 5.2 shows the location of piezometers on the studied sites. In order to assess the impact of landfills on groundwater quality, monitoring data from 2008 to 2022 were analyzed.

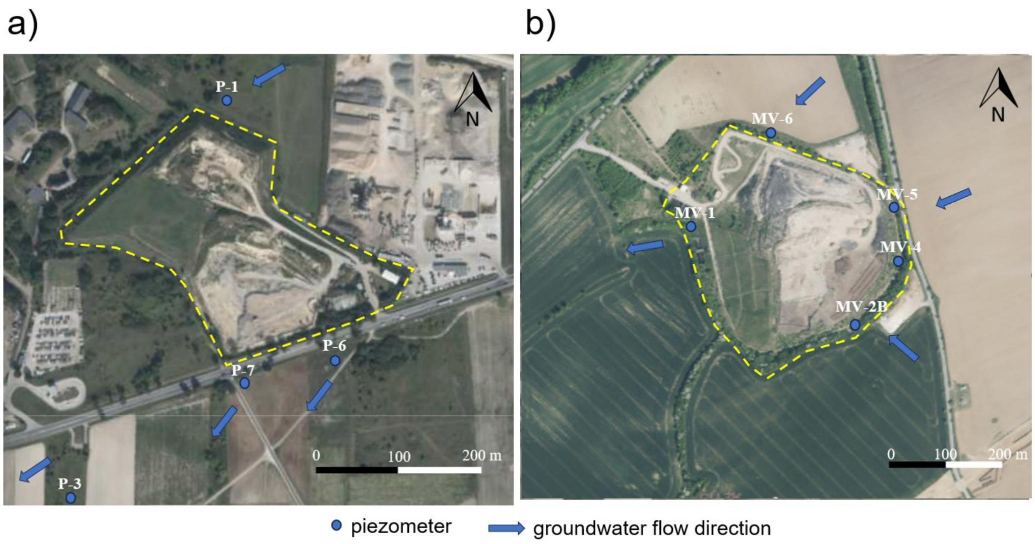


Figure 5.2. Location of piezometers in the studied areas: a) Zakroczym landfill, b) Zdounky landfill.

5.1.2. Leachate composition

In order to assess the level of contamination of leachate from two landfills located in Zdounky and Zakroczym from 2008 to 2022, their chemical properties were analyzed. The following parameters were considered for evaluation: pH, EC, Zn, Cr (VI), Cr_{total}, mercury (Hg), Polycyclic Aromatic Hydrocarbons (PAH), NH_4^+ and phosphorus (P_{total}). The obtained values were compared with Tab. 5.2, which shows the limit of selected values of leachate parameters according to various international regulations.

Table 5.2. Limits of selected values of leachate quality parameters according to different regulations.

Source		**Announcement of the Ministry of Infrastructure and Construction (2016)[5]**	**Regulation of the Ministry of Marine Economy and Inland Navigation (2019)[6]**	**US EPA (2000)**
		Limit values of pollution indicators for some substances particularly harmful to the aquatic environment in industrial wastewater discharged into sewage devices	Limit values of pollutants for wastewater and rainwater or meltwater discharged into water and soil	Effluent limitations regarding non-hazardous waste landfill
Parameters	pH [-]	6.5–9.5 or 8–10**	6.5–9.5	6.7–9.8
	Zn [mg/l]	5.0	2.0	31.8
	Cr (VI) [mg/l]	0.2	0.1	0.247
	Cr_{total}[mg/l]	1.0	0.5	0.24
	Hg [mg/l]	0.06*	0.06*	-
	PAH [mg/l]	0.2	-	-
	NH_4^+[mg/l]	100 [1)] or 200[2)]	10.0	10[3)] or 4.9[4)]
	P_{total}[mg/l]	[5)]	3	6.5

*Notes: * daily average, ** concerns wastewater containing cyanides and sulphides 1) applies to wastewater discharged to treatment plants for agglomerations with an equivalent number of inhabitants <5000, 2) applies to wastewater discharged to treatment plants for agglomerations with an equivalent number of inhabitants ≥5000, 3) daily maximum, 4) maximum monthly average. Concentrations of indicators expressed in mg/L, with the exception of pH (unitless), 5) Announcement of the Minister of Infrastructure and Construction of 28 September 2016 on the publication of the consolidated text of the Regulation of the Minister of Construction concerning the method of fulfilling obligations by industrial wastewater suppliers and the conditions for discharging wastewater into sewerage systems, 6) Regulation of the Minister of Maritime Economy and Inland Navigation of 11 October 2019 on the criteria and method for assessing the status of groundwater bodies.*

Leachate from the investigated landfills are collected in sealed impermeable tanks (Fig. 5.3). The leachate tank at the Zakroczym landfill has a capacity of 780 m^3. The leachate was sampled by a drainage system and pumped into the tank by two pumping stations, P1 and P2. In contrast, the landfill in Zdounky has two leachate tanks, each with a capacity of 630 m^3. The leachate from the Zakroczym landfill was sampled four times a year (in March, June, September, and December) according to PN-ISO 5667-10:2021-11, whereas the leachate from the Zdounky landfill was collected twice a year (in April and October). The procedure for sampling leachate from the leachate tanks followed the standard outlined in ISO 5667-10:1992. The frequency of leachate sampling was in accordance with the monitoring requirements outlined in Polish law (Sampling frequency was determined based on the guidelines of the in the *Regulation of the Minister of Environment of April 30, 2013 on landfills waste with amendments*) and Czech law *(Technical Standards ČSN 83 8036 Waste landfilling - Monitoring of landfills* (Skládkování odpadů - Monitorování skládek).

Figure 5.3. Leachates tanks: a) Zdounky landfill, b) Zakroczym landfill.

The Leachate Pollution Index (*LPI*) was used to further illustrate the pollution potential of landfill leachate. This index is used in landfill monitoring because it allows the tracking of temporal changes in leachate pollution, assessment of the pollution potential of monitored leachate, and comparison of leachate pollution potential between analyzed landfills. *LPI* analysis is a comprehensive assessment that allows leachate pollution levels to be calculated based on the characteristics of 18 defined contaminants, divided into organic, inorganic, and HMs, by assigning them appropriate significance levels (Kumar and Alappat, 2005; Podlasek et al., 2023). The *LPI* can be calculated using the Eq. (3).

$$LPI = \sum_{i=1}^{n} w_i \, p_i \tag{3}$$

Where *LPI* is the weighted additive leachate pollution index, w_i is the weight for the *i*th pollutant variable, p_i is the sub index score of the *i*th leachate pollutant variable, *n* is number of leachate pollutant variables used in calculating *LPI*, and $\sum_{i=1}^{n} w_i = 1$.

Due to the fact that chemical analysis was performed for n ≤ 18 at the studied landfills, *LPI* calculations were performed using the Eq. (4).

$$LPI = \frac{\sum_{i=1}^{m} w_i p_i}{\sum_{i=1}^{m} w_i} \tag{4}$$

Where *m* is the number of leachate pollutant parameters for which data is available, but in analyzed case *m* < 18 and $\sum_{\mathrm{i}=1}^{m} \mathrm{w_i} < 1$.

To calculate the *LPI,* leachate monitoring data from to 2008-2022 were used for the analysis. In the case of the Zakroczym landfill, the available data made it possible to calculate the *LPI* for 27 leachate samples, whereas in the Zdounky landfill, the number of samples was 19.

5.1.3. Landfill gas

In addition to control the level of leachate contamination in landfills, it is extremely important to control LFG emissions, which can be used to assess the level of decomposition of organic matter in landfills. This study used data from the monitoring of landfills from 2008 to 2022. LFG monitoring included the measurement of percentage volume of LFG: CH_4, CO_2 and O_2, which were determined by accredited laboratories. However, only the variability of CH_4, the most representative gas responsible for the decomposition of organic matter in the landfill, was assessed for the analyses. The frequency of measuring the composition and LFG emission was determined on the basis of the guidelines contained in *Council Directive 1999/31/EC of April 26, 1999, on the landfill of waste*, according to which the composition of the landfill gas should be measured every 1 month in the operational phase and every 6 months in the post-operational phase. The monitoring network of the Zakroczym landfill currently (as of 2022) includes 9 degassing wells: W1 and W2 on the active eastern section; Gp1 and Gp2 on the active southern section; Gp3 and Gp4 on the new section; and Z3, Z8, and Z10 on the reclaimed western cell. At the Zdounky landfill, the LFG monitoring network is much more developed. It consists of 13 vertical degassing wells in stage 1 (I.1-I.12), 1 well in a stage 2 (II.2), 6 vertical wells in stage 3 (III.1, III.2, III.4, III.6, III.7, III.9), 2 wells in stage 4 (IV1 and IV2), and 4 wells in stage 5 (V1-V4). Due to the study of the impact of landfill reclamation on the environment, only wells from reclaimed cells of the landfills (the western cell in Zakroczym and stage 1 in Zdounky) were considered in the evaluation of changes in the CH_4 content.

Qualitative statistical tests were performed to assess whether there were statistically significant differences in the CH_4 content of LFG at the partially reclaimed MSW landfills in Zakroczym and Zdounky. Data including medians, standard deviations (SD), minimum and maximum values for the period from 2008 to 2022 were collected and statistical analyses were performed in R Studio. Reclamation of both landfills started at the same time (in 2011), so the data were divided into two periods: before 2011 and after 2011. Accordingly, the following three research questions were formulated as a basis for the hypotheses, which were independently tested using parametric and non-parametric tests depending on the distribution of the data:

I. Are there significant differences between the 2 groups (until 2011 and after 2011) showing the percentage concentration of CH_4 at the Zakroczym landfill? If yes, between which quarters?
II. Are there significant differences between the 2 groups (until 2011 and after 2011) showing the percentage concentration of CH_4 at the Zdounky landfill? If yes, between which quarters?
III. Are there significant differences between the 2 groups (until 2011 Zakroczym landfill and after 2011 Zdounky landfill) showing the percentage concentration of CH_4 at the Zdounky landfill? If yes, between which quarters?

The normal distribution of each group was tested using the Shapiro-Wilk test, while the homogeneity of the data was tested using Levene's test. When the conditions of normality, homogeneity and independence of samples were met, perform an analysis of variance (ANOVA) with 95% confidence interval was chosen (Gautam and Kumar, 2021) for the analysis. On the other hand, if the conditions of normality of the data or homogeneity were not met for further statistics, non-parametric Wilcoxon rank sum tests and Kruskal-Wallis test were used.

5.2. Laboratory tests of soils used for reclamation

In order to check the geotechnical parameters of the soil used for landfill reclamation, samples were collected from both reclaimed landfill sites. In the case of Zakroczym landfill, samples were collected from 6 test boreholes at depths of 0.1–0.2 m, 0.5–0.7 m and approximately 1.0 m in August 2022, while in the case of the Zdounky landfill, samples were collected from 8 test boreholes at depths of 0.1–0.2 m and 0.5–0.7 m in April 2023. The samples (n = 19 samples at Zakroczym and n = 16 samples at Zdounky landfill) were tested to determine the physicochemical properties of the soil. The sampling locations of the studied sites are shown in Fig. 5.4 and Fig. 5.5.

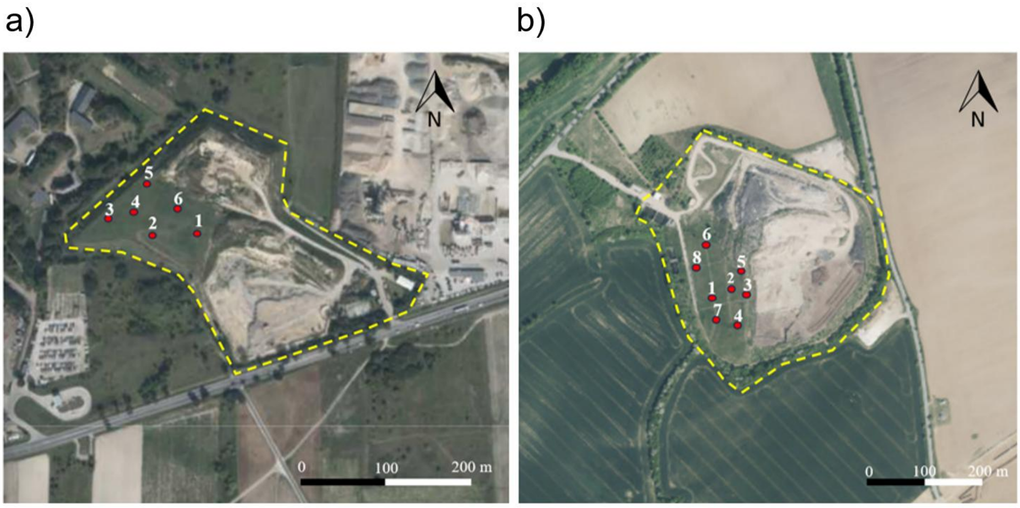


Figure 5.4. Location of sampling points at the studied landfills: a) Zakroczym landfill, b) Zdounky landfill.

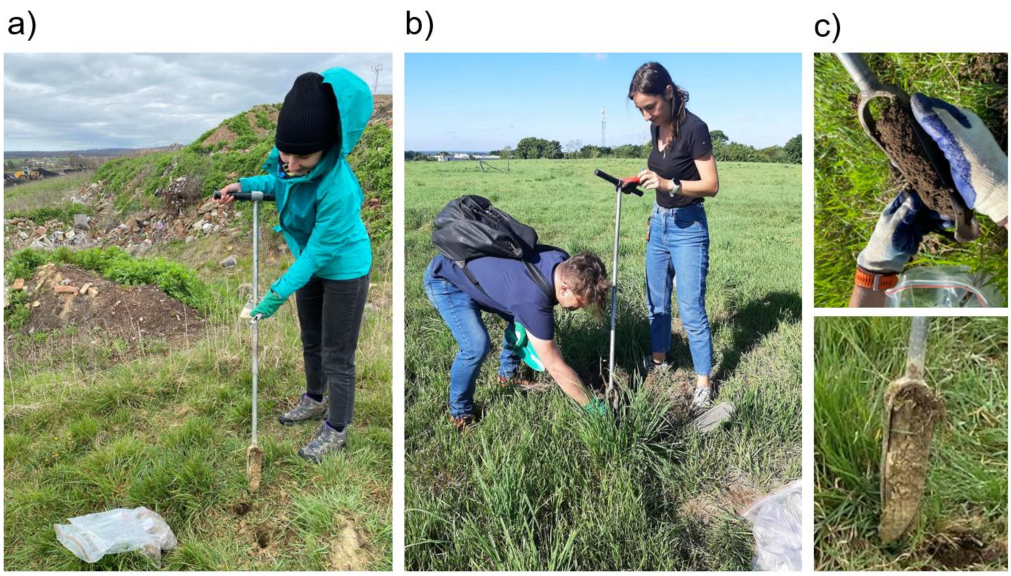


Figure 5.5. Soil sampling at research sites: a) Zdounky landfill, b) Zakroczym landfill, c) (from the top) Zakroczym landfill from the Zdounky landfill (own photos).

5.2.1. Physical properties of soils

The analysis of the granulometric composition was performed using the aerometric method according to ISO 17892-4:2016(E). Wet sieving of soil samples through a 0.063 mm diameter sieve with hexametaphosphate (40g/l) as a dispersion agent was used to determine the grain size of the tested soils. The equivalent particle diameter diameters were calculated using Stokes' law according to the following Eq. (5):

$$d_i = 0.005531\sqrt{\left((\eta \times H_r)/\left((\rho_s - \rho_w) \times t\right)\right)} \tag{5}$$

Where: d_i – the equivalent particle diameter (mm), η – the dynamic viscosity of water mPa×s at the temperature of the test, H_r – the effective depth of hydrometer (mm), ρ_s – the particle density (Mg/m^3), ρ_w – the density of the sedimentation fluid at the temperature of the test (Mg/m^3), t – the time elapsed from the start of sedimentation (min).

Based on the calculated equivalent diameters, the contents of clay fraction ($d \leq 0.002$ mm), silt fraction ($0.002 < d \leq 0.05$ mm), sand fraction ($0.05 < d \leq 2.0$ mm) and gravel fraction (≥ 2.0 mm) were determined in the tested soils, and then verified the type of soil. Fig. 5.6 shows the laboratory determination of the granulometric composition of the soils.

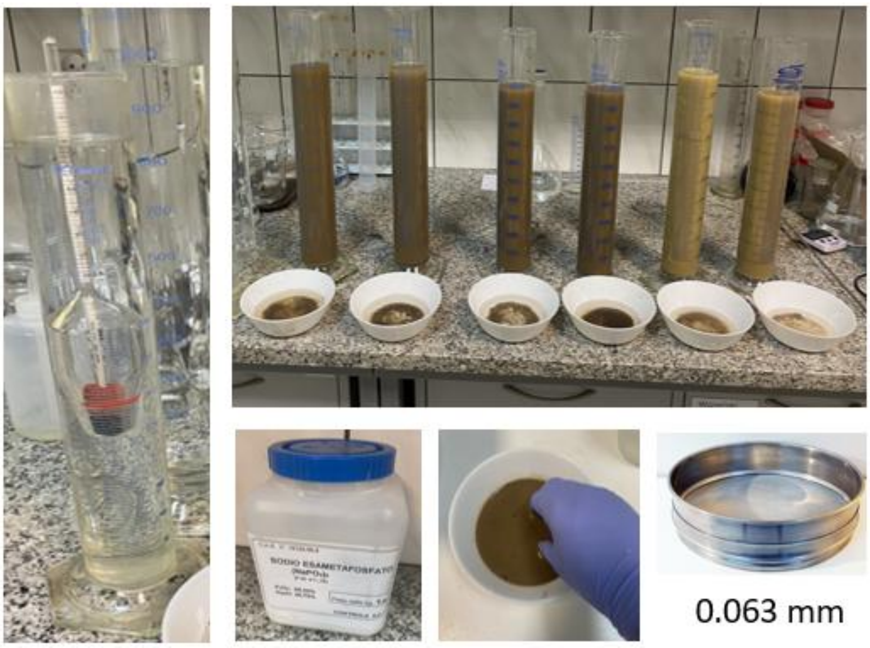


Figure 5.6. Laboratory determination of the granulometric composition of the tested soils (own photos).

To define the soil type more precisely, in addition to analyzing its granulometric composition, consistency limits such as the liquid limit – w_L and the plastic limit – w_p were determined using the Casagrande method and the rolling method, respectively, in accordance with EN 1997-2:2007 Eurocode 7. Subsequently, the soil liquidity index – I_L (Eq. 6), the plasticity index – I_P (Eq.7) and consistency index – I_C (Eq. 8) were calculated (PN-EN 1997-2:2009 Eurocode 7).

$$I_L = \frac{w_n - w_p}{w_L - w_p} \tag{6}$$

$$I_P = w_L - w_p \tag{7}$$

$$I_C = \frac{w_L - w_n}{I_p} \tag{8}$$

Where: w_n – natural moisture content (%), w_p – plastic limit (%), w_L – liquid limit (%), I_p – the plasticity index (%).

Fig. 5.7 shows the laboratory determination of the consistency limits of the soils tested for landfill reclamation in Zakroczym and Zdounky. In total, n = 7 samples were tested to determine the consistency limits of the soils in Zakroczym, whereas n = 16 samples were tested in Zdounky. The difference in the number of tests performed was due to the difficulty in determining the degree of plasticity owing to the low content of cohesive fraction. To complement the physical tests with information on the origin of the soils studied, which can influence the ability to accumulate HMs and the stability of the cover, an organic content test was performed. For this purpose, the loss on ignition (LOI) method was used, in which the organic parts were annealed in an electric furnace at 600-800°C (Eq. 9), according to EN 1997-2:2007 Eurocode 7.

$$\mathrm{LOI} = \left(\frac{w_2 - w_3}{w_2 - w_1}\right) \times 100 \tag{9}$$

Where: w_1 – weight of the crucible (g), w_2 – weight of crucible plus oven-dry sample (g), w_3 – weight of crucible plus oven-dry sample after ignition (g).

Figure 5.7. Determination of plasticity and liquidity of the tested soils (own photograph).

5.2.2. Analysis of chemical properties of soils

Because there are concerns about potential environmental and health problems related to microorganisms, HMs, and organic pollutants in waste (Yaashikaa et al., 2022; Pisharody et al., 2022), it is important to verify the chemical properties of soils used in landfill cover systems. To determine the chemical properties, parameters such as electrolytic conductivity (EC), pH, and the content of HMs such as zinc (Zn), lead (Pb), copper (Cu), nickel (Ni), and cadmium (Cd) were determined in the soil samples. A total of n = 19 samples were analyzed for HMs in soils in Zakroczym, whereas n = 16 samples were analyzed in Zdounky. The EC of the soil samples was determined using the conductometric method for soil solutions prepared at a soil-to-distilled water volume ratio of 1:2.

The pH was determined by a potentiometric method using 1 mol KCl solution. Atomic Absorption Spectroscopy (AAS) was used to determine the HMs content of the soil. AAS is a quantitative analytical technique used to estimate the concentration of specific HMs in a sample by analyzing the radiation absorbed by the analyte (Bings et al., 2010). In the AAS technique, an atom in the ground state absorbs light at a specific wavelength, causing it to move to a higher energy level, and an electron is transferred from the ground state to the excited state. The photon energy raises the electron from the energy level $E0$ to the energy level $E1$. The absorption level was used to calculate analyte concentration. Precise identification of individual elements is possible through the use of specialized light sources (appropriate elemental lamps) and appropriate wavelength matching. The main advantages of AAS are their high sensitivity, ability to counteract interference, lower limit of detection (LOD), and wide range of analysis (Rai et al., 2023). In order to determine the HMs content in soils used for landfills covers in Zakroczym and Zdounky, the soil samples taken from depths of 0.1–0.2 m, 0.5–0.7 m and approximately 1.0 m were previously sieved through a 2 mm sieve and weighed at 1 g each. They were then mineralized in a Milestone microwave oven (Start D, Italy) according to the method 3051A (EPA, 2007) using concentrated HNO_3. The soil solutions obtained after mineralization were filtered and diluted to 100 ml with deionized water. The HMs content of the tested soils was analyzed using an ICE 3000 spectrometer (Thermo Scientific, USA), and Thermo SOLAR software was used to optimize the method.

For each analysis, a cathode ray tube with the corresponding metal operating at a specific wavelength was used. A calibration curve method was used to determine each heavy metal (Farrukh, 2012). In this method, standard solutions of six concentrations were prepared: 0.125 mg/l, 0.25 mg/l, 0.5 mg/l, 1.0 mg/l, 2.0 mg/l, 4.0 mg/l. The absorbance of the standards was measured at a specific wavelength using a spectrophotometer, and the resulting absorbance values were used to prepare a calibration curve. Based on the data points, a regression line was fitted to determine the relationship between the concentration and absorbance. The test sample solution was adjusted such that its absorbance fell within the measurable range of the calibration curve. The absorbance of this solution was measured at the same wavelength as the standards, and the concentration of HMs in the sample was determined from the calibration curve (Podlasek et al., 2024). Fig. 5.8 shows the test stands during the ongoing HMs determination in the studied soils.

To assess soil contamination with the above-mentioned selected HMs, the obtained results were compared to the limit values specified in the *Regulation of the Minister of Environment of September 1, 2016, on the manner of conducting the assessment of the contamination of the earth's surface* (Rozporządzenie Ministra Środowiska z dnia 1 września 2016 r. w sprawie sposobu prowadzenia oceny zanieczyszczenia powierzchni ziemi, Dz. U. 2016 poz. 1395).

In the absence of hydraulic conductivity tests, the worst-case scenario was considered with $k \geq 1\times10^{-7}$ m/s in Group IV (group of industrial sites with a sampling depth > than 0.25 m below ground level) was used to determine the maximum allowable concentration. In Tab. 5.3. the permissible maximum concentrations of HMs in the soil in group IV are presented.

Figure 5.8. Mineralization of soil samples and determination of HMs content, from left – mineralization, from right – determination of metals by AAS (own photos).

Table 5.3. Permissible maximum concentrations of HMs in in the soil in group IV acc. to the *Regulation of the Minister of Environment of September 1, 2016, on the manner of conducting the assessment of the contamination of the earth's surface.*

No.	Name of the pollutant	Limits for risk-causing substances (mg/kg d.m.)
1.	Zn	300
2.	Pb	200
3.	Cu	200
4.	Ni	100
5.	Cd	6

5.3. Modelling studies

In the following subchapter, the methods presented serve as a valuable complement to traditional monitoring and laboratory analyses, enabling a more accurate prediction of long-term landfill processes like leachate production and GHG emission. This integrated approach not only facilitates the prediction of variations in leachate production and landfill gas emissions but also allows for a thorough assessment of the landfill slope stability under different cover system configurations.

5.3.1. Modelling leachate generation using the HELP model

The purpose of the model study using the Hydrologic Evaluation of Landfill Performance (HELP) model was to simulate leachate production under realistic conditions in the Zakroczym and Zdounky landfill. The HELP model incorporated in the UnSat Suite Plus program is a tool that enables the simulation of landfill operations and is suitable for modelling both operating and designing landfills (Schroeder et al. 1994, Podlasek, 2023). The model was designed to assess the water balance or hydrological performance of landfill components, such as covers or barrier liners, which allows for comparing and selecting a solution that is more technically, economically, and environmentally efficient (Sinnathamby et al., 2024). HELP includes the modelling of flow through the vertical cross-section of the landfill and its defined layers, including vertical percolation, lateral drainage, barrier soil, and GM (Xu et al., 2012). Tab. 5.4 presents the function of each layer in the model.

Table 5.4. Categories of landfill layers available in HELP.

Layer category	Function
Vertical percolation layer	Serves primarily as a moisture storage zone; typically includes topsoil and waste layers.
Lateral drainage layer	A layer of moderate to high permeability material that is underlain by a liner with a lateral drainage collection and removal system.
Barrier soil liner	A layer of low permeability soil designed to limit percolation/leakage.
GM liner	A synthetic flexible membrane liner designed to prevent vertical leakage and minimize infiltration.

HELP program uses the saturated and unsaturated conductivity of soil and waste to model the vertical drainage, lateral drainage and soil liner percolation. Saturated hydraulic conductivity characterizes the flow through porous materials where the pore spaces are completely occupied by a wetting fluid, such as water. The saturated hydraulic conductivity for each layer was provided as part of the input data. In the calculations, it was assumed that the barrier soil liners are saturated at all times and leak only when a positive head on the top surface of the liner exists. Percolation through the soil barrier was treated as vertical flow, following Darcy's law. This allowed the model to simulate water percolating through the liner only when water reaches the liner's surface (Podlasek, 2023). Percolation through the GM was considered as the flow through pinhole defects or as vapor diffusion. Leachate outflow was modeled as lateral drainage assuming a saturated flow. The processes that are modeled in HELP can be divided into surface processes (snowmelt, interception or rainfall by vegetation, surface runoff, and evaporation) and subsurface processes (soil-water evaporation, plant transpiration, vertical drainage, liner leakage, and lateral drainage). The soil water storage in HELP model in measured per volume basis (θ), volume of water (V_t) per total (bulk-soil, water and air) which is characteristic of practice in agronomy and soil physics (Eq. 10):

$$V_t = V_s + V_w + V_a \tag{10}$$

Where: V_t – total soil volume (m^3), V_s – volume of soil (m^3), V_w – volume of water (m^3), V_a – volume of air (m^3).

For the runoff calculation, the SCS curve number was used (USDA, 1985) due to the fact that it can be applied to different types of soils and is computationally efficient. Accordingly, the formula for runoff is (Eq. 11):

$$Q = P' - S' \tag{11}$$

Where: Q – actual runoff (mm), P' – maximum potential runoff (mm), S' – maximum potential retention after runoff starts (mm).

HELP model requires data on climate: precipitation, temperature, soil radiation; average wind speed, relative humidity; soil: type of material, total porosity, field capacity, wilting point, saturated hydraulic conductivity, subsurface inflow; vegetation: condition of vegetation on cover layers (vegetation class), maximum leaf area index (LAI), evaporative zone depth; design data: slope parameters, thickness of layers, number of layers, landfill area, drainage length, leachate recirculation (Fig. 5.9). Data concerning meteorological conditions of the studied landfills locations is presented in Appendix 1.

The most important group of model input data is the soil data, such as soil type, porosity, field capacity, wilting point, and hydraulic conductivity, which allow for the differentiation of the landfill layers used or designed.

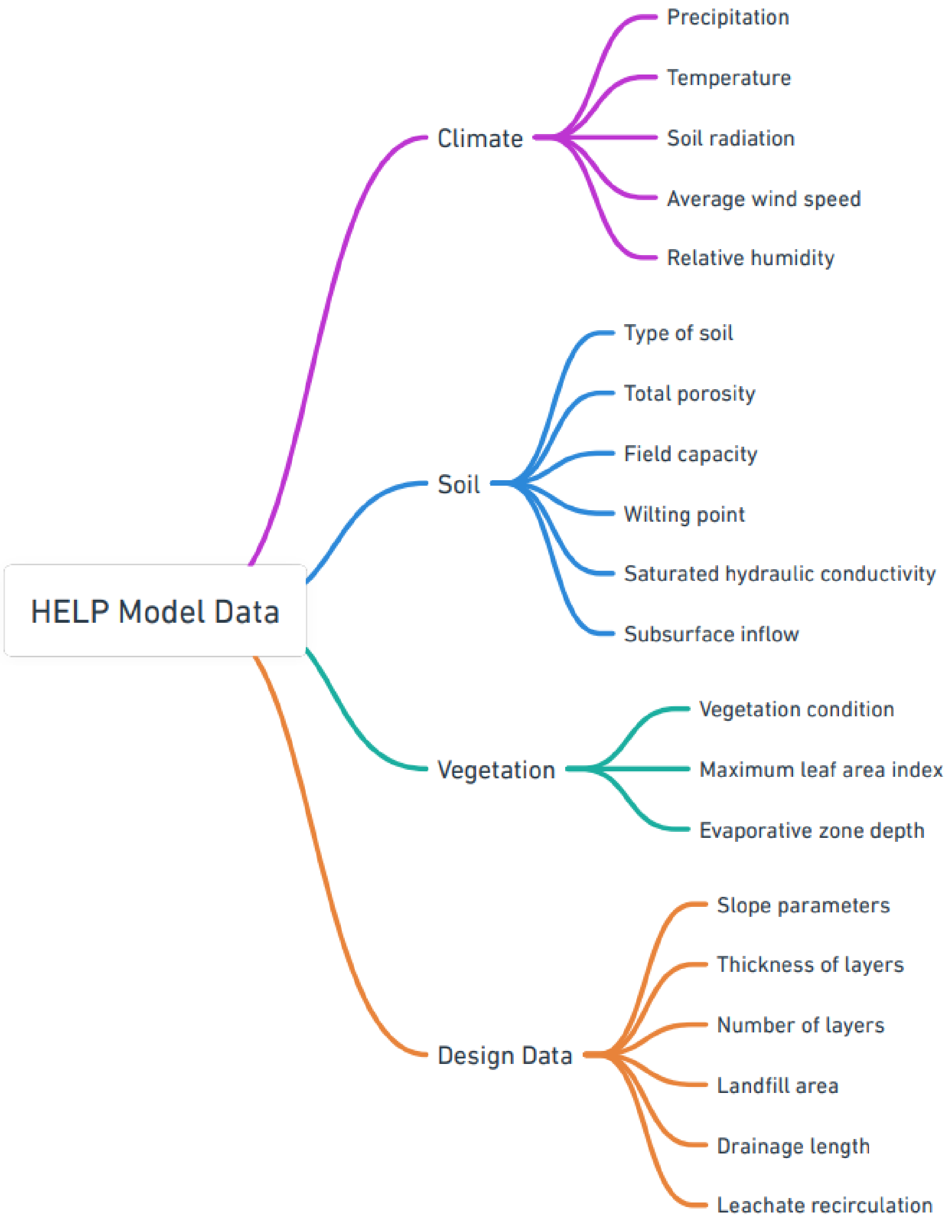


Figure 5.9. HELP Model input data.

In Tab. 5.5, the group of soils used in the modelling along with their properties is presented. The composition and quantity of landfill leachate also vary depending on the type and age of the landfill and the morphology of the stored waste (Jamrah et al., 2024). To evaluate the effectiveness of the applied landfill layers in the models performed for both landfills, the same default data for MSW were used.

Table 5.5. Materials used in HELP model.

Material	Category	Total porosity (vol/vol)	Field capacity (vol/vol)	Wilting point (vol/vol)	Saturated hydraulic conductivity (m/s)	Subsurface inflow (m/s)
Clayey sand	Vertical percolation layer	0.437	0.105	0.047	1.7×10^{-6}	0.000
MSW	Vertical percolation layer	0.671	0.292	0.077	1.0×10^{-5}	0.000
Fine sand	Lateral drainage layer	0.457	0.083	0.033	3.1×10^{-5}	0.000
Geotextile and geonets	Drainage net	0.850	0.010	0.005	1.0×10^{-1}	0.000
HDPE GM	GM liner	0.000	0.000	0.000	2.0×10^{-15}	0.000
Clay	Barrier soil liner	0.475	0.378	0.251	2.5×10^{-7}	0.000
Sandy clay loam	Barrier soil liner	0.398	0.244	0.136	1.2×10^{-6}	0.000
Coarse sand	Lateral drainage layer	0.417	0.045	0.018	1.0×10^{-4}	0.000

Six scenarios of MSW landfill construction were considered in calculations of leachate generation (Fig. 5.10, Tab. 5.6). Scenarios 1–3 represented the MSW in Zakroczym, while scenarios 4–6 represented the MSW in Zdounky. Scenarios 1 and 4 represented landfills already reclaimed, whereas the others were in operation phase. The modelling was based on the results of physical properties of the soils in the cover of the studied landfills. In all cases, the bottom liners consisted of sandy clay loam soil with a hydraulic conductivity of $k < 1.2\times10^{-7}$ m/s and a 2 mm thick HDPE GM liner. The main differences between the scenarios were in the cover liner, where in Scenario 1, clayey sand with a thickness of approximately 1.5 m was placed on top of the waste layer, whereas in Scenario 4, it was an HDPE GM with a thickness of 1 mm and a clay layer with a thickness of 1 m. The slopes of the landfill in Scenarios 1–3 were 1(V):2(H), while in scenarios 4–6 the slope was 1(V):3(H).

To assess the vegetation condition, a good grass stand (LAI=4) was established in the reclaimed cells. A drainage gradient of ≥ 1% was used in all scenarios studied.

Percolation through the GM is considered as flow through hole defects or vapor diffusion. Two defect locations per hectare for an HDPE GM e with $k = 2\times10^{-15}$ m/s and a placement quality of 4 was assumed in the study. The assigned quality results in poor installation on a site with a less well-prepared soil surface and/or wrinkling of the GM, resulting in poor contact between the GM and the adjacent soil and a large diffusion gap and greater leakage (Waterloo Hydrogeologic, 2004).

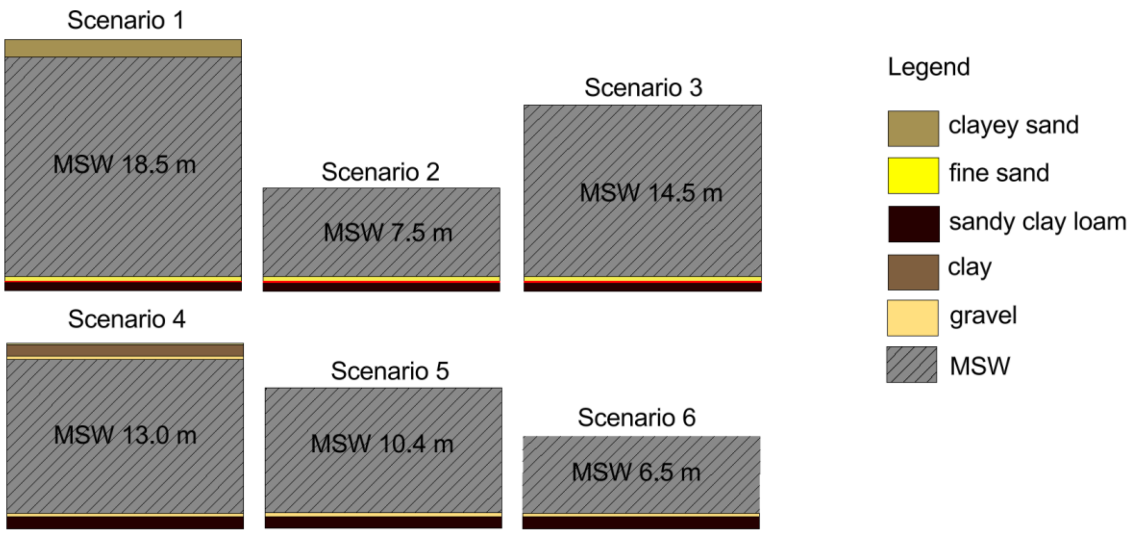


Figure 5.10. Graphical representation of six modelling scenarios.

A Pearson correlation matrix was prepared to indicate the significant correlations between the water balance at the landfill sites. Because simulations of leachate production were performed for constant precipitation, and different variants of weather conditions were not considered, correlations were calculated for all scenarios simultaneously. The correlation scale was expressed in numbers from -1 to 1. Values close to -1 indicate a strong negative correlation, which results in an increase in one variable, causing a decrease in the other variable, and vice versa. By contrast, values close to 1 indicate a strong positive correlation, indicating that if one variable increases, the other variable also increases. The coefficient $r = 0$ indicates that there is no linear relationship between the variables, and the relationship becomes stronger (i.e., the scatter decreases) as the absolute value of r increases, and eventually approaches a straight line when the coefficient approaches -1 (or +1). The interpretation of the calculated r coefficient was performed using the following scale (Schober et al., 2018) acc. to which, r = 1.00–0.9 means a very strong correlation; r = 0.89–0.7, strong correlation; r = 0.69–0.4, moderate correlation; r = 0.39–0.10, weak correlation; and r = 0.09–0.00, negligible correlation.

Table 5.6. Description of landfill layers according different scenarios.

Landfill part	Scenarios*					
	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5	Scenario 6
Cover layers	Clayey sand 1.5 m	-	-	Clay 1.0 m	-	-
		-	-	Gravel 0.3 m	-	-
	-	-	-	HDPE 1 mm	-	-
Wastes	MSW 18.5 m	MSW 7.5 m	MSW 14.5 m	MSW 13.0 m	MSW 10.4 m	MSW 6.5 m
Bottom liners	Fine sand 0.5 m	Fine sand 0.5 m	Fine sand 0.5 m	Gravel 0.3 m	Gravel 0.3 m	Gravel 0.3 m
	Drainage net	Drainage net	Drainage net	Drainage net	Drainage net	Drainage net
	HDPE 2 mm	HDPE 2 mm	HDPE 1.5 mm	HDPE 2 mm	HDPE 2 mm	HDPE 2 mm
	Sandy clay loam 0.7 m	Sandy clay loam 0.7 m	Sandy clay loam 0.7 m	Sandy clay loam 1.0 m	Sandy clay loam 1.0 m	Sandy clay loam 1.0 m

** Scenario 1 – reclaimed western cell at Zakroczym landfill with an area of 1.34 ha*
Scenario 2 – operating southern cell at Zakroczym with an area of 2.02 ha
Scenario 3 – operating eastern cell at Zakroczym with an area 0.7 ha
Scenario 4 – reclaimed stage 1 cell at Zdounky landfill with an area of 1.92 ha
Scenario 5 – operating stages 2, 3 and 4 cells at Zdounky landfill with an area of 3.06 ha
Scenario 6 – operating stages 5 and 6 cells at Zdounky landfill with an area of 1.303 ha

5.3.2. Modelling of landfill gas emissions from reclaimed landfill cells using the LandGEM Tool

The modelling studies were complemented by simulations of LFG emissions from different cover systems over a 140-year period. The U.S. Environmental Protection Agency's (EPA) Landfill Gas Emissions Model (LandGEM) (EPA, 2005) was used to estimate the amount and environmental impact of LFG at two partially reclaimed landfills. The use of the U.S. numerical model LandGEM provides a wide range of possibilities for the quantitative estimation of emissions from organic waste decomposition, which at the same time makes it possible to predict the amount of energy produced (Bouzaidi and Ouazzani, 2024). The amount of LFG evaporation and the rate of its biological degradation are determined by the amount of organic fraction and its distribution, the availability of nutrients, the moisture content and the initial density of the waste deposited in the landfill. It is also assumed that under normal conditions the decomposition rate of organic matter in a landfill is measured by gas production (Goushki et al., 2023). With a model it is possible to estimate the amount and change of CH_4, CO_2, and other air pollutants in landfills over a long time (EPA, 2005; Kale and Gökçek, 2020; Delgado et al., 2023). The tool requires inputs such as the year the landfill was opened and closed and the design capacity of the waste (Delgado et al., 2023). In addition, the model determines parameters such as the CH_4 generation rate (k), the potential CH_4 generation capacity (L_0), and the average percentage of CH_4 in LFG. Because the model was able to use average monitoring data (percentage of CH_4 in LFG) from specific years at the landfills studied to determine trends, it was not necessary to estimate them through modelling, which could have a significant impact on the results. As a result, the LandGEM model was chosen as the basis, in part because of its general acceptance by the scientific community. LandGEM defines CH_4 emissions using the first-order decomposition equation (Eq. 12) (EPA, 2005):

$$Q_{CH_4} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} \mathrm{k} L_o \left(\frac{M_i}{10}\right) e^{-kt_{ij}} \tag{12}$$

Where: Q_{CH_4} is the annual CH_4 production in the calculation year (m^3/y), L_o is the CH_4 production capacity (m^3/t), k is the CH_4 production rate (y^{-1}), i is the 1-year increase, j is the 0.1-year increase, M_i is the amount of MSW buried in the i^{th} year (t), n is the calculation year, t_{ij} is the age of the j^{th} section of MSW buried in the i^{th} year.

The LandGEM model uses existing monitoring data on the percentage composition of LFG for the period 2008–2022 for the Zakroczym and Zdounky landfills. For the modelling of LFG emissions, only the reclaimed cells, were considered. In the case of the Zakroczym landfill, the western cell was built in 1997 with a capacity of 357 000 m^3 (308 750 t) and an area of 1.34 ha, where MSW was accepted until 2011. The annual weight of waste was estimated to be 20 583 tons, and 1 m^3 of MSW weighed 864 kg. The study of the Zdounky landfill, considered a reclaimed cell from stage 1, built in 1996, with an area of 1.92 ha. Waste was placed into the cell system with a capacity of approximately 249 600 m^3 (215 654 t) until 2011. The annual weight of waste was estimated to be 13 478 tons. In order to make the comparison between the two landfills as close as possible to the type of material stored, it was assumed for the study of emissions from the reclaimed landfills in Zdounky that 1 m^3 of MSW weighs the same as in the Zakroczym landfill, i.e. 864 kg. In accordance with the fact that the average precipitation at both study sites was < 635 mm, k = 0.02 was assumed for modelling LFG production, which is consistent with US EPA (2005). Estimating the amount of precipitation was necessary because, according to a study by Chanton and Liptay (2000), moisture reduces oxidation by limiting airflow to methanotrophic bacteria, thereby affecting CH_4 production. According to the model instructions, the volumetric content of CH_4 should be 40–60%. CH_4 production is highly dependent on the morphological composition of the waste, so to better compare the facilities, the same waste composition was assumed at both sites, resulting in CH_4 of 40%. An important fact in modelling CH_4 emissions from landfills was also to consider the degree of CH_4 oxidation. According to Bian et al. (2021b), the amount of CH_4 produced in reclaimed landfills varies with the type of cover used, and the level of oxidation may range from less than 10% to as much as 100%. Fig. 5.11 shows the mechanism of CH_4 oxidation in landfill soil covers. CH_4 generated in the waste migrates upward through the soil cover and is oxidized to CO_2 and H_2O by biochemical oxidation mediated by methanotrophs.

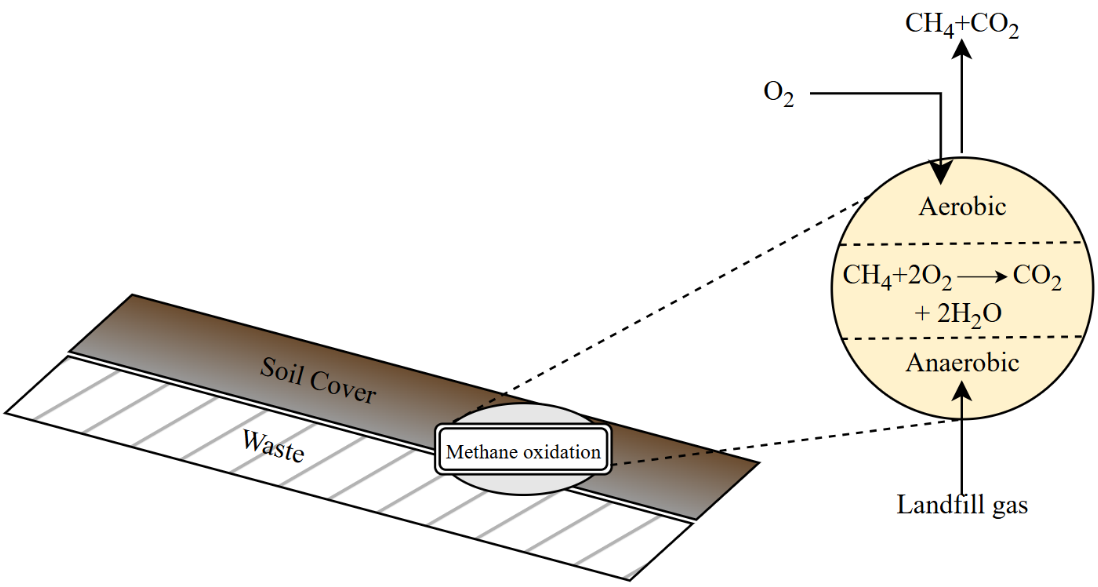


Figure 5.11. Landfill methane oxidation in a cover soil system acc. to Sadasivam and Reddy (2014).

At the Zakroczym landfill, due to the mineral material used in the cover (clayey sand), 7% of its oxidation level was included in the calculations, consistent with the column studies performed by Sadasivam and Reddy (2014). On the other hand, for the Zdounky landfill, due to the assumption of a tight synthetic cover, the oxidation level was not included in the determination of emissions. The estimated emissions of LFG including CH_4 and CO_2, for the period 1997–2137 are presented in Chapter 6.3.2.

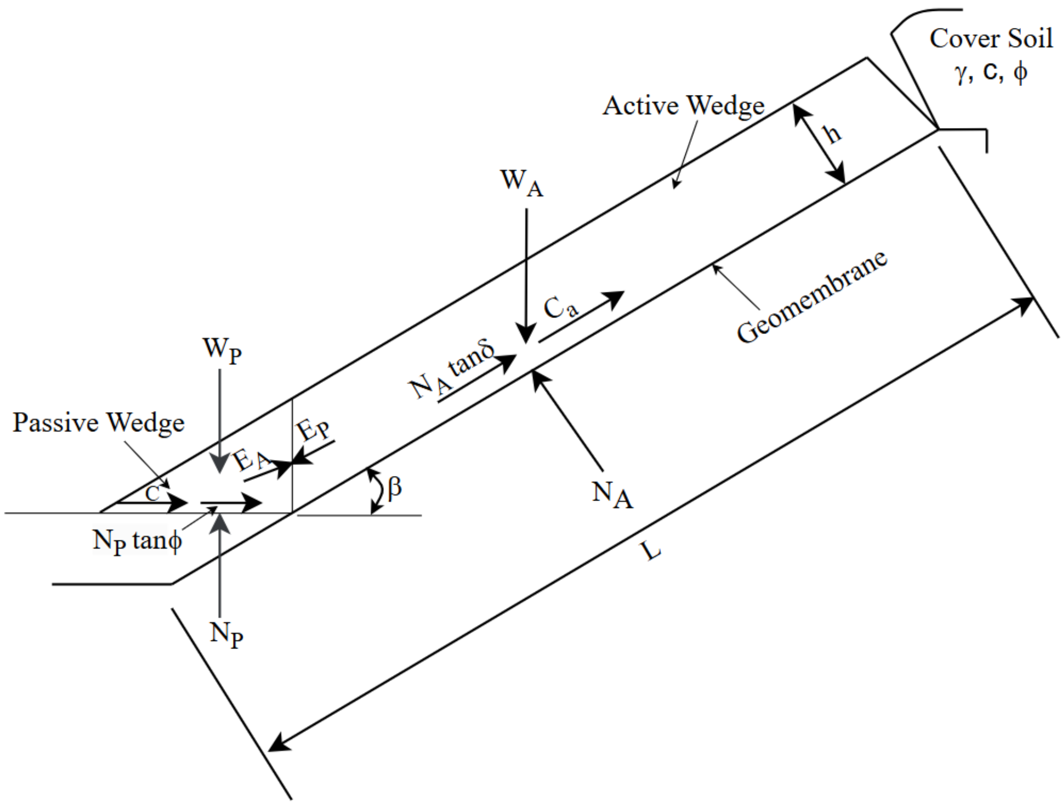
5.3.3. Landfill stability analysis

The stability of the landfill slopes was evaluated from two perspectives. The first involved assessing the overall stability of the landfill using the Bishop method. The second focused on analyzing the risk of sliding at the interface between the layers of the landfill cover, which is particularly important when synthetic materials are used (Koda, 2014). For the Zdounky landfill, both GM sliding and overall stability were studied, whereas for the Zakroczym landfill variant stability was assessed solely based on overall stability calculations because of the lack of GM in the cover system.

5.3.3.1. Sliding of the final cover

Due to the failures observed over the past several decades in the stability of landfill slopes and the sliding of final covers, there is a recognized need to conduct stability analyses that can prevent such failures (Romero et al., 2023) or allow for early intervention to reinforce the slope—for example, through the use of geogrids to maintain an appropriate level of safety (Cortellazo et al., 2022). In light of this, attempts have been made to assess the sliding of the cover using a GM applied to the reclaimed cell of the Zdounky landfill, which is the subject of this study. For this purpose, calculations were performed to determine the factor of safety (FS) using the limit equilibrium method, assuming a slope of finite length. Fig. 5.12 presents a schematic cover consisting of a thin layer of soil and a GM with a slope angle β, which has a finite length.

Calculations were initially carried out for the active soil wedge, which depended on the selected unit weight of the cover layer (γ), effective force normal to the failure plane of the active wedge (N_A), and the adhesive force between cover soil of the active wedge and the GM (C_a), assuming the worst-case scenario in which the adhesion between cover soil of the active wedge and the GM is 7 kPa (Eq. 13–15).



h – thickness of the cover soil (m)
L – length of slope measured along the GM (m)
β – soil slope angle beneath the GM (°)
ϕ – friction angle of the cover soil (°)
δ – interface friction angle between cover soil and GM (°)
C_a – adhesive force between cover soil of the active wedge and the GM (kN)
c_a – adhesion between cover soil of the active wedge and the GM (kPa)
C – cohesive force along the failure plane of the passive wedge (kN)
c – cohesion of the cover soil (kPa)
E_A – interwedge force acting on the active wedge from the passive wedge (kN)
E_P – interwedge force acting on the passive wedge from the active wedge (kN)
W_A – total weight of the active wedge (kN)
W_P – total weight of the passive wedge (kN)
N_A – effective force normal to the failure plane of the active wedge (kN)
N_P – effective force normal to the failure plane of the passive wedge (kN)
γ – unit weight of the cover soil (kN/m^3)
FS – factor of safety against cover soil sliding on the GM (-)

Figure 5.12. Limit equilibrium forces involved in finite length slope analysis acc. to Koerner and Daniel (1997).

Using vertical force equilibrium, the force acting on the passive wedge from the active wedge (E_A) was determined. In this case, a worst-case scenario was also assumed for δ, which was assigned a value of 22° (Eq. 16). Next, the passive soil wedge was considered, for which the total weight of the passive wedge (W_P), effective force normal to the failure plane of the passive wedge (N_P), and cohesive force along the failure plane of the passive wedge (C) were calculated (Eq. 17-19). The horizontal force equilibrium equation was used to determine force acting on the passive wedge from the active wedge (E_P) (Eq. 20), and the assumption that the forces from the active and passive soil wedges are equal ($E_A = E_P$) (Eq. 21) allowed for the calculation of the equation components (Eq. 22–24) and the derivation of a relationship used to determine the factor of safety FS (Eq. 25) (Koerner and Daniel, 1997).

Considering an active wedge:

$$W_A = \gamma h^2 \left(\frac{L}{h} - \frac{1}{\sin\beta} - \frac{\tan\beta}{2}\right) \tag{13}$$

$$N_A = W_A \cos\beta \tag{14}$$

$$C_a = c_a\left(L - \frac{h}{sin\beta}\right) \tag{15}$$

The interwedge force acting on the active wedge is:

$$E_{\mathrm{A}} = \frac{FS(W_{\mathrm{A}} - N_{\mathrm{A}}\cos\beta) - (N_{\mathrm{A}}\tan\delta + C_a)\sin\beta}{\sin\beta\,(\mathrm{FS})} \tag{16}$$

Considering a passive wedge:

$$W_P = \frac{\gamma h^2}{sin2\beta} \tag{17}$$

$$N_p = W_p + E_p \sin\beta \tag{18}$$

$$C = \frac{(c)(h)}{sin\beta} \tag{19}$$

By balancing the forces in the horizontal direction, the following formulation results:

$$E_p = \frac{C + W_p \tan \phi}{\cos \beta \ (FS) - \sin \beta \ \tan \phi} \tag{20}$$

Equality assumption E_A and E_P.

$$E_A = E_p \tag{21}$$

Calculation the equations components:

$$a = (W_A - N_A \cos \beta)\cos \beta \tag{22}$$

$$b = -\left[(W_A - N_A \cos \beta) \sin \beta \tan \phi + (N_A \tan \delta + C_a) \sin \beta \cos \beta + \sin \beta)\left(C + W_\rho \tan \phi\right)\right] \tag{23}$$

$$c = (N_A \tan \delta + C_a) \sin^2 \beta \tan \phi \tag{24}$$

Calculation of factor of safety (FS):

$$\mathrm{FS} = \frac{-b \pm \sqrt{b^2 - 4ac}}{2a} \tag{25}$$

The geotechnical parameters like: unit weight of the cover soil (γ), friction angle of the cover soil (φ), interface friction angle between cover soil and GM (δ), cohesion of the cover soil (c) and adhesion between cover soil of the active wedge and the GM (c_a) in Tab. 5.7 were used to determine the FS of the Zdounky landfill cover and estimate the risk of slippage.

Table 5.7. Geotechnical parameters of the soils used to assess the slippage on the GM at the Zdounky landfill.

Layer	γ **(kN/m^3)**	φ/δ **(°)**	c/c_a **(kPa)**
Reclamation layer (Cl)	20	21	12
Drainage layer (Gr)	21	37	0
Slippage area (GM-soil)	-	22	7

5.3.3.2. Overall slope stability analysis

In order to assess the overall stability at the studied landfills, the Bishop computational method was used. This method considers both the vertical and horizontal interactions between adjacent slices and applies the moment equilibrium condition on a circular-cylindrical failure surface (Wysokiński, 2009; Pisarczyk, 2017). The calculations were repeated several times to identify the failure surface with the lowest factor of safety (i.e., the critical failure surface), which is defined as the ratio of the characteristic sustaining forces along the failure surface to the shear forces (Zabielska-Adamska and Sulewska, 2019). Since the Bishop method is well-suited for computer-based calculations, the GEOSTUDIO software (Slope Stability Module) was used to implement the stability analysis. Calculations were performed for two reclaimed landfills to compare the FS for slopes covered with a synthetic cover (Zdounky landfill) and a mineral cover (Zakroczym landfill). The geotechnical characteristics of the soils used are presented in Tab. 5.8.

Table 5.8. Geotechnical parameters of soil used in the stability models.

Type of soil	***h*** **(m)**	**γ (kN/m³)**	***ϕ'*** **[°]**	***c'*** **(kPa)**
Zakroczym				
Clayey sand	1.2	18.5	27.0	8.0
Fine sand	0.3	20.0	36.5	0.0
MSW*	18.5	10.20	30.0	3.0
Fine sand	0.5	20.0	36.5	0.0
Sandy clay loam	0.7	20.0	15.0	20.0
Zdounky				
Clay	1.0	20.0	21.0	12.0
Gravel	0.3	21.0	37.0	0.0
MSW*	13.0	10.20	30.0	3.0
Sandy clay loam	1.0	20.0	15.0	20.0

**acc. to Zabielska-Adamska and Sulewska (2019)*

In both landfills, HDPE GM with a thickness of 2 mm and a tensile strength of 29 kN/m was used in the base sealing layer. Meanwhile, the geotechnical properties of the soils were taken from literature data.

5.4. Biomonitoring studies

The last group of studies was a biomonitoring study to determine whether leachate from the landfill has phytotoxic properties in early-stage plants, and a study to assess CO_2 levels in the soil used for reclamation to check soil health which has potential effect on plant development.

5.4.1. Phytotoxicity of leachate

To determine phytotoxicity, leachates from two sites were collected in August according to ISO 5667-10:1992 and brought into cold storage. The principle of the phytotoxicity test is to grow germinating seeds of selected plants on contaminated (or test) medium and to evaluate the effect of the sample on the growth of root length in higher plants. The most suitable plant for such tests is *Sinapis alba* L. This cultivated plant is commonly used in phytotoxicity tests because of its high sensitivity to changes in environmental conditions, which manifests as rapid growth retardation or inhibition and leaf necrosis (Šindelář et al., 2020; MicroBioTests Inc., 2004). To assess the effects of leachate from the investigated landfills on vegetation, a phytotoxicity test, PhytoxkitTM (for solid samples) and Phytotestkit (for liquid samples), was performed using the leachate generated at the Zdounky and Zakroczym landfills. Both tests allow the direct measurement of root and shoot lengths in special transparent plates using an image analysis method that eliminates the multiple steps required to perform traditional phytotoxicity determinations (MicroBioTests Inc., 2004). This was in accordance with the ISO 187663:2016.

The phytotoxicity test uses shallow and flat transparent test plates with two compartments with total dimensions of 21×15.5×0.8 cm. The lower compartment contains soil saturated to the level of water holding capacity, and the upper compartment is empty (Adamcová et al., 2016). The results obtained from the study were used to calculate the germination inhibition (IG) (Eq. 26) (Palm et al., 2022) and the root inhibition (IR) (Baran and Tarnowski, 2013).

$$IG\ or\ IR = \frac{\mathrm{A\ \text{-}\ B}}{\mathrm{A}} \times 100 \tag{26}$$

Where: A – means root length or amount of germinated seeds in the control, B – means root length or amount of germinated seeds in the test sample.

When the percentage of *IR* < 0, the growth of roots in plant species is stimulated and the substrate is not considered phytotoxic. In the opposite way, when the percentage of *IR* > 0, the growth of roots is inhibited and the substrate is considered phytotoxic (Šourková and Adamcová, 2023). If I is less than 10% (I < 10%), the sample is non-toxic or slightly toxic, if 10% < I < 50%, the sample is toxic, and if I < 50%, the sample is highly toxic to plants (Adamcová, et al., 2016).

Qualitative statistical tests were also conducted to assess whether there were statistically significant differences in the length of roots exposed to leachate between the two study sites. Data, including medians, SDs, and minimum and maximum values, were collected, and statistical analyses were performed using R Studio. Normality of the data distribution was tested using the Shapiro-Wilk test. As the p-value was < 0.05, non-parametric Kruskal-Wallis and Wilcoxon rank-sum tests were used to compare differences between specified groups.

5.4.1.1. Phytoxkit for solid samples

The phytotoxicity test for solid samples is based on growing seeds in contaminated material and monitoring the inhibition or stimulation of root growth in germinating plants (Zloch et al., 2020). This microtest can also be used to determine the effects of chemicals on plants by introducing them into a reference soil (MicroBioTests Inc., 2004). The root lengths of young plants were compared with those from a control experiment, in which seeds were grown in a reference soil composed of 74% sand, 20% kaolinite, 5% peat, and 1% $CaCO_3$. The use of OECD soil as a medium has been recommended in several ecotoxicological tests. Each sample consisted of 90 ml of OECD control soil. Seeds were arranged to allow their roots to grow in the direction of gravity. The tests were performed in triplicate. After a 72 h incubation period, the samples were photographed, and the root lengths of all samples were measured using Image Tool 3.0 for Windows (UTHSCSA – The University of Texas Health Science Center at San Antonio, San Antonio, USA). Fig. 5.13 shows the preparation to the leachate phytotoxicity tests.

5.4.1.2. Phytotestkit for liquid samples

In contrast to the Phytoxkit test, the Phytotestkit measures the direct effect of chemical compounds on plants without introducing these compounds into the soil (therefore without using the OECD soil). One advantage of this test is its technology, which enables the determination of the direct impact of chemical compounds on seed germination and early plant growth without interference from soil components. In this test procedure, the lower compartment is filled with a foam insert, a parafilm insert, and a thin cellulose filter that is saturated with the appropriate chemical compound (20 ml of the chemical compound or 20 ml of water in the control samples) during the analysis. The Phytotestkit, like the Phytoxkit, was performed in three replicates (I–III). After a 72 h incubation period, the samples were photographed and the root lengths of the plants were measured in all samples. The details of the procedure are described by MicroBioTests Inc. (2004).

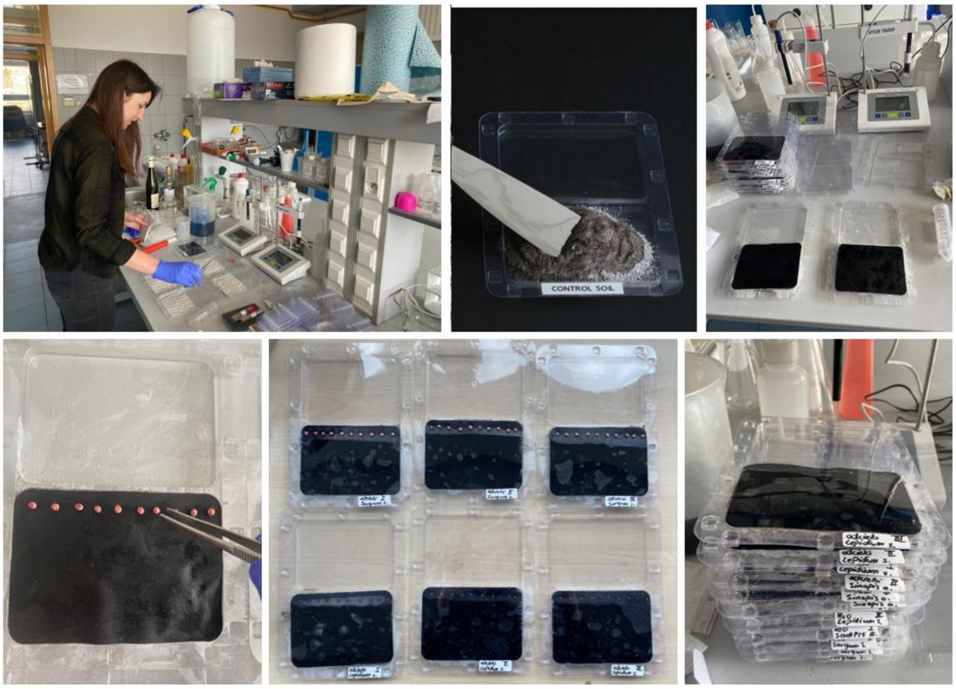


Figure 5.13. Ongoing phytotoxicity testing of leachate in solid sample test (own photos).

5.4.2. Soil respiration test

Measuring CO_2 respiration from soil is directly related to the amount of microorganisms present. As living organisms, many soil microorganisms, such as bacteria and fungi, breathe oxygen to carry out metabolic processes and release CO_2. Since microorganisms have key roles in soil health, such as forming soil structure, decomposing plant debris into organic matter and mineralizing nutrients, it can be inferred that the higher the amount of CO_2 released from the soil, the more microorganisms are present and the healthier the soil should be (Dorsey, 2019). For this purpose, soil samples were taken from the top of the cover system from the both landfills at depths of 0–0.2 m and 0.2–0.5 m. To assess soil health, which can affect plant growth, a soil respiration test was conducted. Based on studies and experiments conducted by various authors, a methodology using the SOLVITA® test kit was selected for baseline respiration testing (Moore et al., 2019; Rogers et al., 2019; Sciarappa et al., 2017). The SOLVITA® test provides a simple and quick way to assess the activity of soil microorganisms by analyzing the amount of CO_2 absorbed by the gel indicator. Microorganisms present in the soil produce CO_2, and the amount of CO_2 increases with their abundance. They are crucial to soil quality and health because they decompose organic matter and act as major decomposers.

The activity of microorganisms provides nutrients required by plants and other organisms, and their level of biological activity reflects the decomposition processes of organic matter in the soil. The method is based on the instructions of the Soil CO_2-Burst Respiration Test kit, which is the most suited for processed, dry and stored soils, and is ecologically relevant for regions with intermittent drying-wetting soil cycles (SOLVITA®, 2019). The collected soil samples were sieved through a 6 mm mesh sieve to remove cobbles, plant debris, and other contaminants. The sieve was cleaned each time to avoid sample contamination. Sterile containers with capacities of 475 cm^3 and 30 cm^3 were filled with sieved soil. Then, 9 ml of water or leachate was added to each container to fill 50% of the available pore space. A gel indicator was placed inside, and the containers were tightly sealed (Fig. 5.14).

Figure 5.14. Preparation of soil samples for respiration measurement using the SOLVITA® Digital Color Reader.

The samples were stored at a constant temperature of 20°C for 24 hours. The indicators were then removed and their color changes, indicating the amount of CO_2, were evaluated using the SOLVITA® Digital Color Reader (DCR). The results obtained were interpreted using the manufacturer's application or the tables in the manual. The limit ranges according to the manufacturer's instructions are shown in the Tab. 5.9.

Table 5.9. Range of limit values for respiration levels of tested soils.

Color	CO_2-C Range	Level	Description
0	0–3	Low	Soil low in microbes, low N-min potential, limited biological activity with low carbon levels.
1	4–9		
2	10–14		
2.5	15–22		
3	23–34	Medium low	Medium active – accumulating carbon, low N-min potential.
3.5	35–55		
4	56–85	Medium high	Ideal activity, active microbes with carbon supply, medium-high N-min Potential.
4.5	86–125		
5	126–195	High	Very active biologically with very high carbon emissions, strong N-min Potential.

6. Research results

This chapter presents the results of the monitoring studies, laboratory tests of the soils, modelling studies and biomonitoring studies at two landfill sites. Changes in heavy metal concentrations, electrical conductivity, and pH in groundwater and leachates composition are discussed. Results of leachate production modeling using UnsatSuite+HELP, as well as methane and carbon dioxide emission simulations with LandGEM, are presented. Slope stability analyses were conducted. Further sections discuss the physicochemical properties of soils covering the closed landfill cells. The phytotoxicity of landfill leachates and the biological activity of soils were assessed through germination tests and soil respiration measurements. Biomonitoring results were compared with chemical analyses to provide a comprehensive view of landfill impacts on the environment.

6.1. Monitoring studies

The subchapter below presents the results of a groundwater quality analysis based on monitoring data to assess the impact of landfill operations on the groundwater quality in Zakroczym and Zdounky landfill surroundings.

6.1.1. Groundwater quality

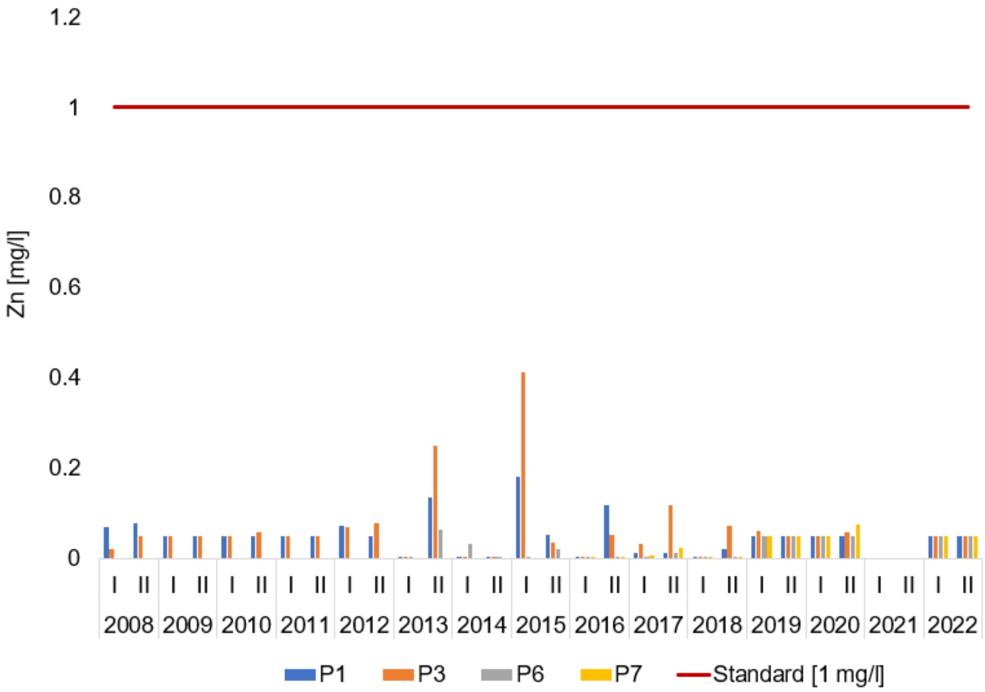
Fig. 6.1 shows groundwater monitoring results of selected parameters measured in the Zakroczym landfill. It was demonstrated that Zn and Cr (VI) concentrations in groundwater were lower than the permissible values for drinking water according to WHO guidelines (2017).

In the case of Zn, the highest value was recorded in 2015 for P-7 and amounted to 0.414 mg Zn/l, in other cases the values oscillated within 0.05 mg Zn/l, which suggests class I groundwater quality according to *Regulation of the Minister of Maritime Affairs and Inland Navigation*, 2019. In contrast, the content of Cr (VI) in groundwater was at the same level within the limit of 0.01 mg Cr (VI)/l with standard of 0.05 mg Cr (VI)/l according to WHO (2017) for drinking water and Class II groundwater quality in accordance with the *Regulation of the Minister of Maritime Affairs and Inland Navigation* *dated October 11, 2019 on the criteria and method of assessing the state of groundwater bodies*.

According to the WHO (2017) and US EPA (2018), the pH of most drinking water is in the range of 6.5–8.5. The results obtained, with pH < 8 in each case, meet the assumption of good groundwater quality. Areas located in limestone areas may contribute to increased water quality. The *Regulation of the Minister of Maritime Affairs and Inland Navigation dated October 11, 2019 on the criteria and method of assessing the state of groundwater bodies* established that for the first class of groundwater quality, the EC should be less than 700 μS/cm, while for the second class of quality 2500 μS/cm. During the monitoring analysis, it was noted that the EC of groundwater in Zakroczym in the first years of analysis 2008–2011 exceeded the limits for Class III groundwater quality, nevertheless after 2011 the situation stabilized, changing the water to less saline, what is well seen in Fig. 6.1d. Exceedance of the permissible limit was observed only in the case of piezometer P-1 where EC in 2008-2011 was above 2500 μS/cm. Nevertheless, it is worth noting that this piezometer is located in the direction of water inflow to the landfill, and therefore the exceedance may have been caused by the impact of neighboring areas. Additionally, the EC index even before the completion of reclamation work improved and has been below the acceptable standard since 2011. The average EC of groundwater in 2008–2022 was 1174 μS/cm, which generally corresponds to Class II groundwater quality, additionally it was noted that in piezometers: P-3, P-6 and P-7 since 2017, EC was below 700 μS/cm in most cases, which suggests I class groundwater quality. Considering the above, the threat of polluting groundwater related to waste landfilling and reclamation of the western cell in the Zakroczym landfill has not been recognized.

Fig. 6.2 and 6.3 shows the spatial distribution of the averaged Zn, Cr, pH and EC contamination of the groundwater for two periods: 2008–2015 and 2016–2022, in order to better illustrate the critical locations that could affect the deterioration of the groundwater quality. It can be seen that in the case of Zn, pH, and EC, the locations of the highest concentrations were the same in 2008–2015 and 2016–2022, however, it cannot be seen that the locations of the highest concentrations are the same for all the identified parameters, which may indicate a lack of accumulation of risk affecting the significant deterioration of water quality.

a)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0.2
0
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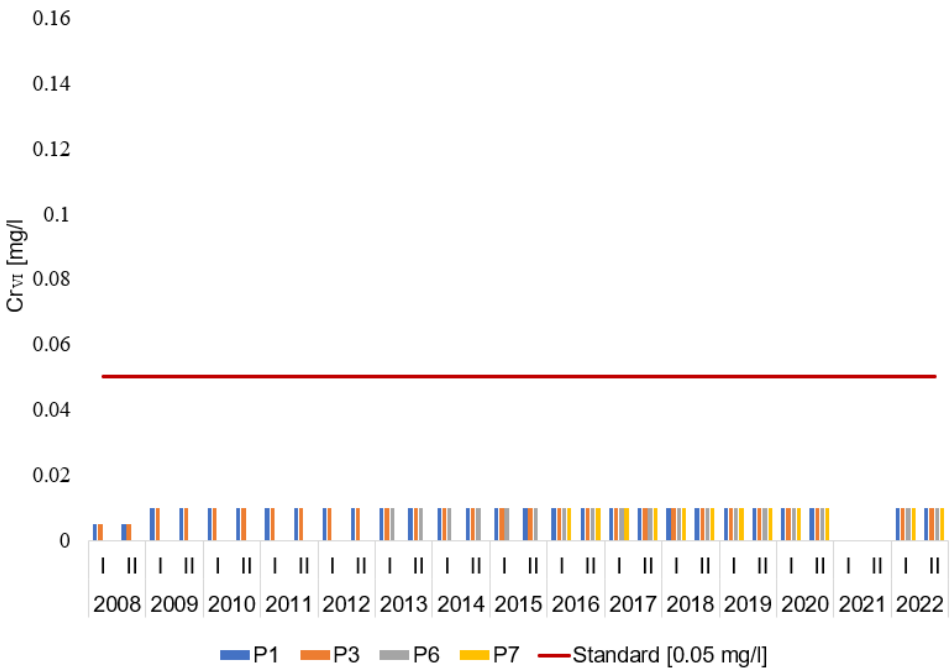
P1 P3 P6 P7 Standard [1 mg/l]

b)


0.16
0.14
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CrVI [mg/l
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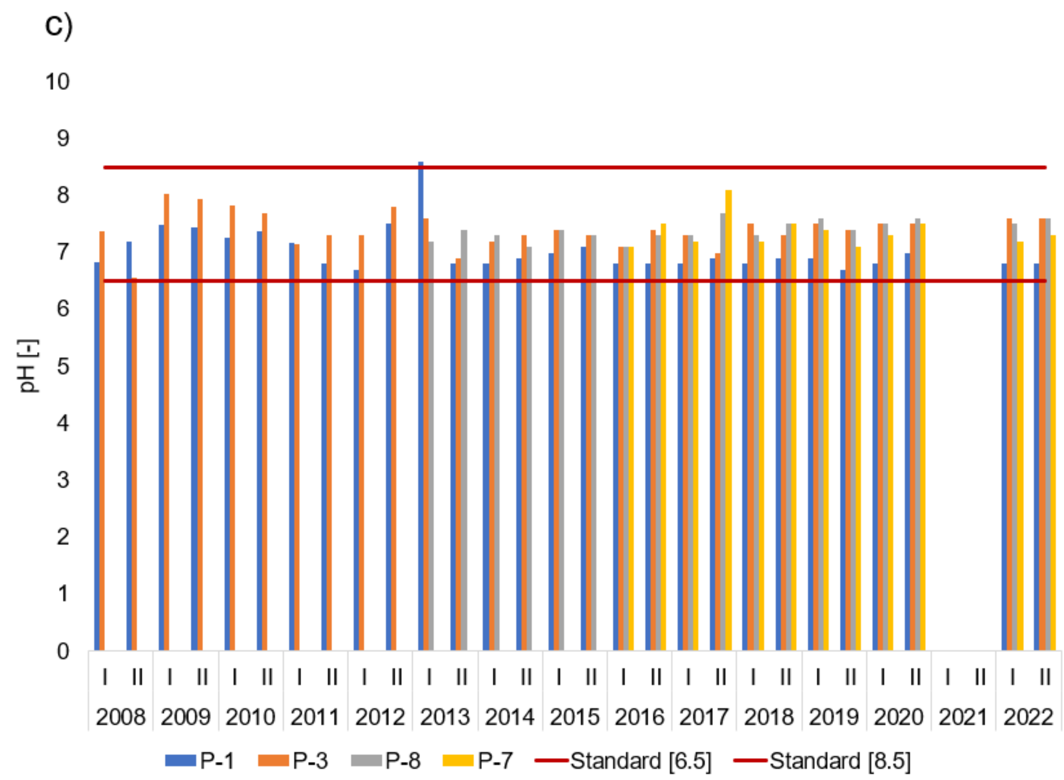
P1 P3 P6 P7 Standard [0.05 mg/l]



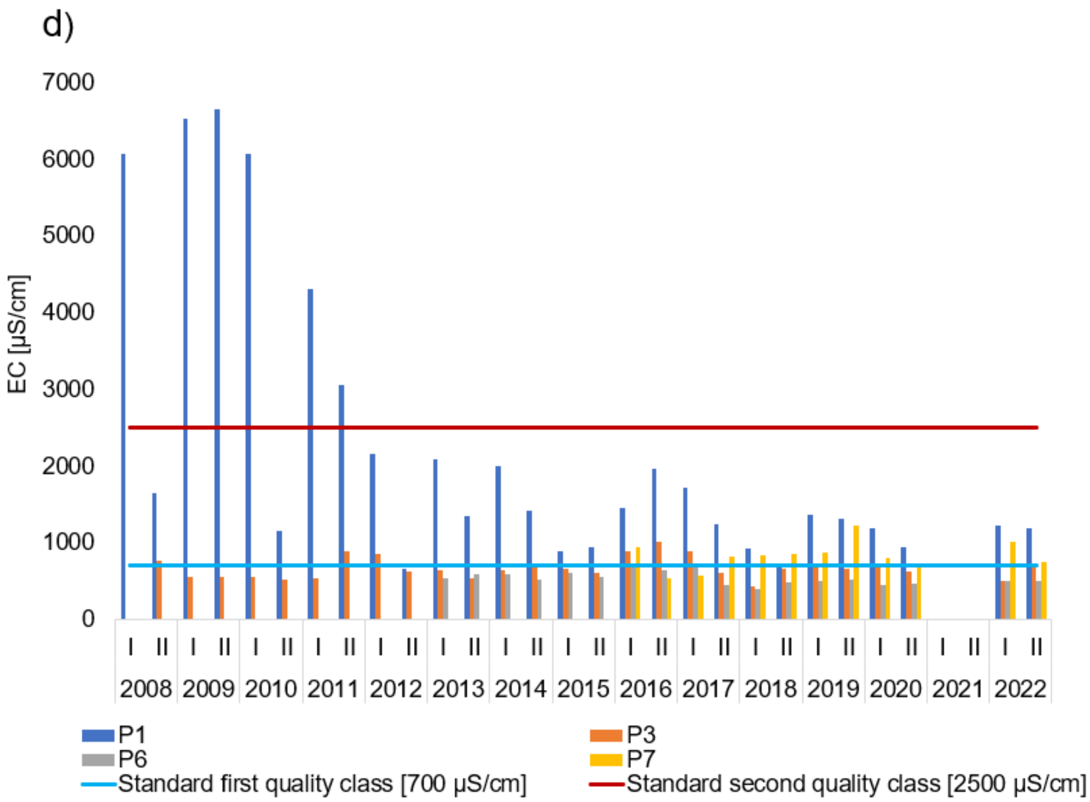
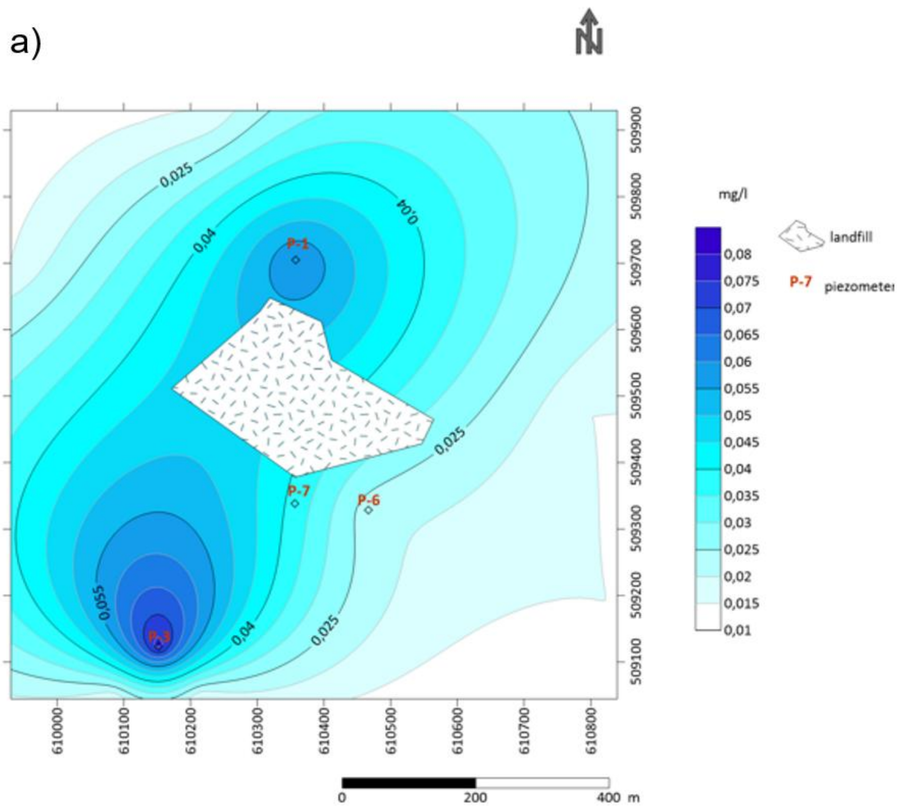


Figure 6.1. Groundwater monitoring in 2008-2022 at Zakroczym: a) Zn [mg/l], b) Cr(VI) [mg/l], c) pH [-], d) EC [µS/cm].

a)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mg/l
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landfill
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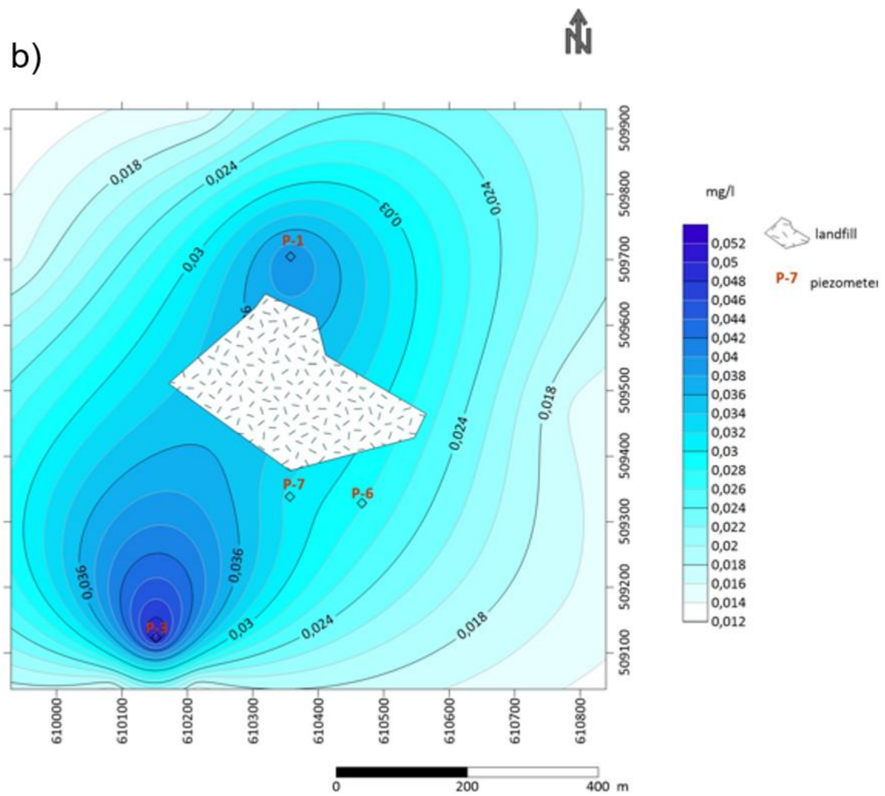
b)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mg/l
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0,048
0,046
0,044
0,042
0,04
0,038
0,036
0,034
0,032
0,03
0,028
0,026
0,024
0,022
0,02
0,018
0,016
0,014
0,012
landfill
P-7 piezometer
P-1
P-7
P-6
P-3
0,018
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0,03
0,036
0 200 400 m

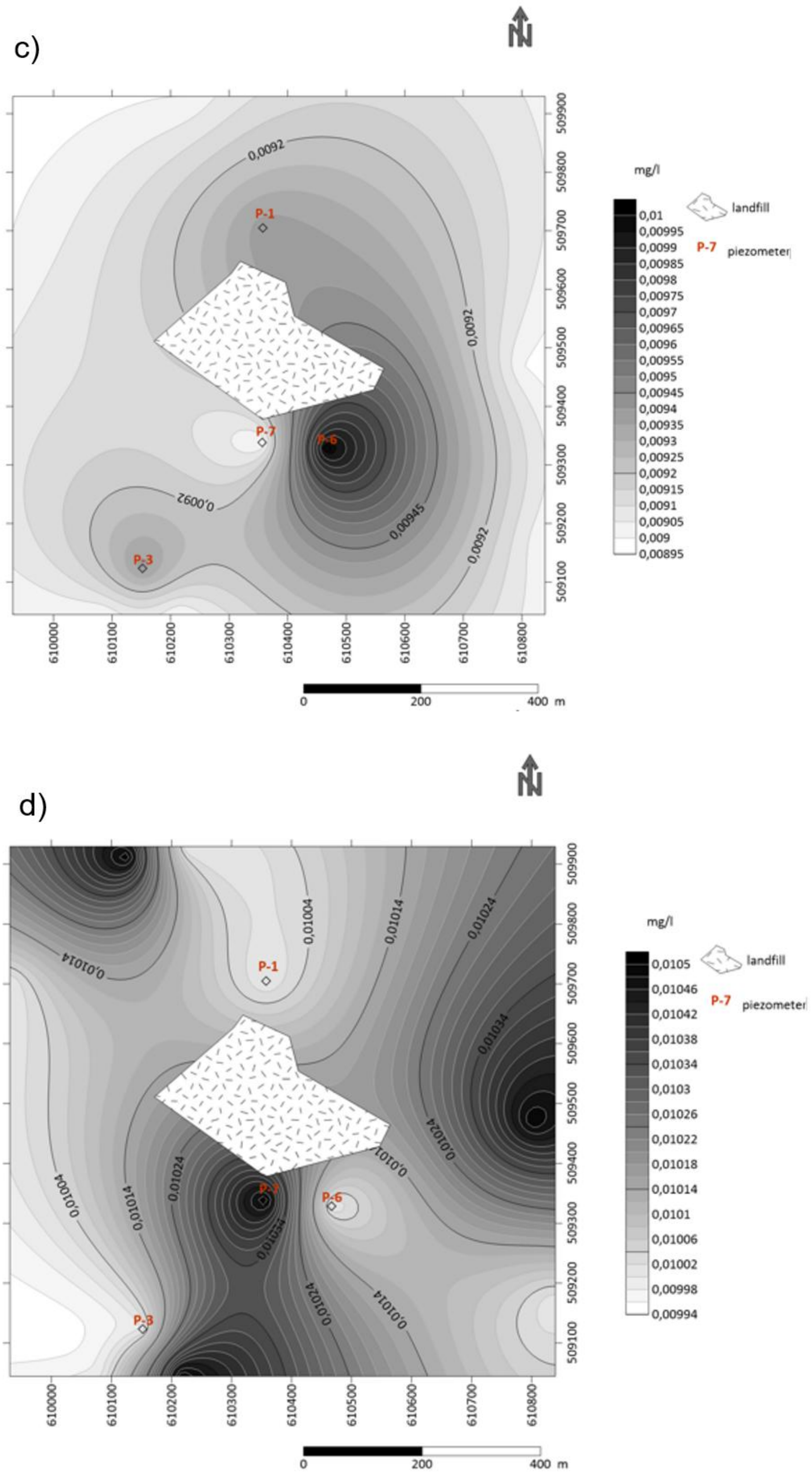
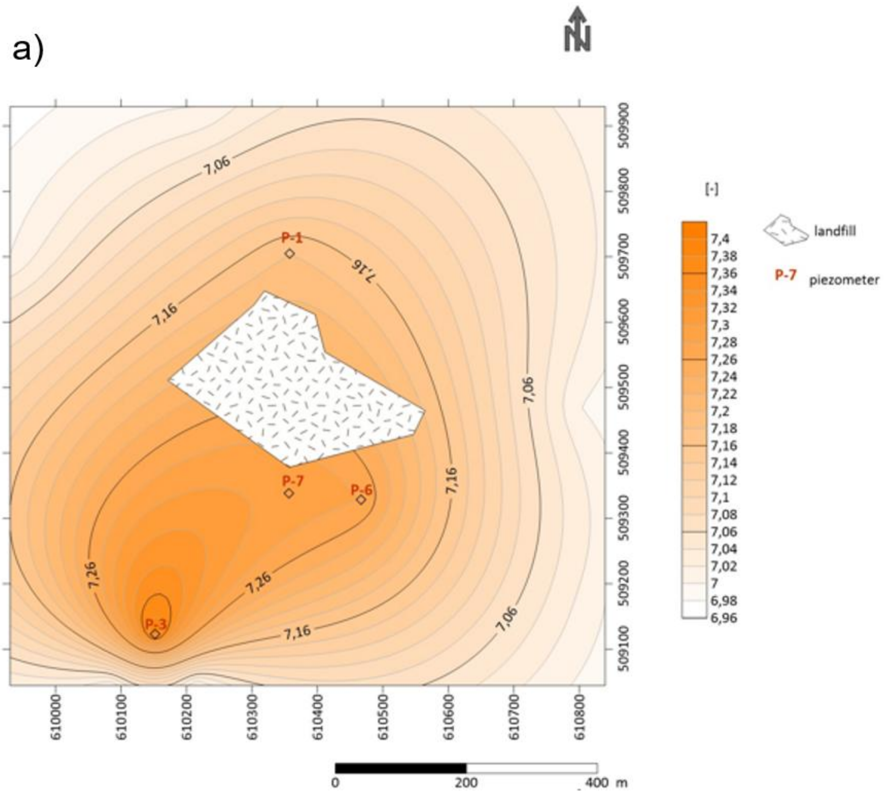


Figure 6.2. Content of selected heavy metals in groundwater in the vicinity of the landfill in Zakroczym: a) Zn [mg/l] 2008–2015, b) Zn [mg/l] 2016–2022, c) Cr (VI) [mg/l] 2008–2015, d) Cr (VI) [mg/l] 2016–2022.

a)


N
7,06
7,16
7,26
P-1
P-3
P-6
P-7
509100
509200
509300
509400
509500
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610200
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200
400 m
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7,4
7,38
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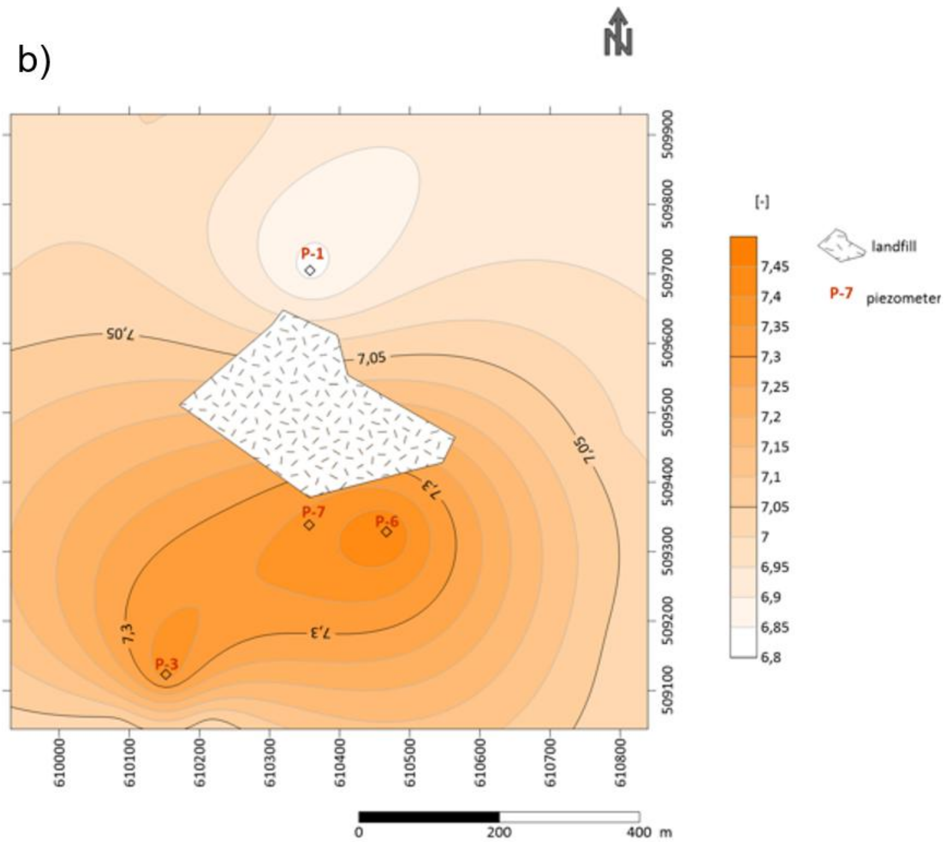
b)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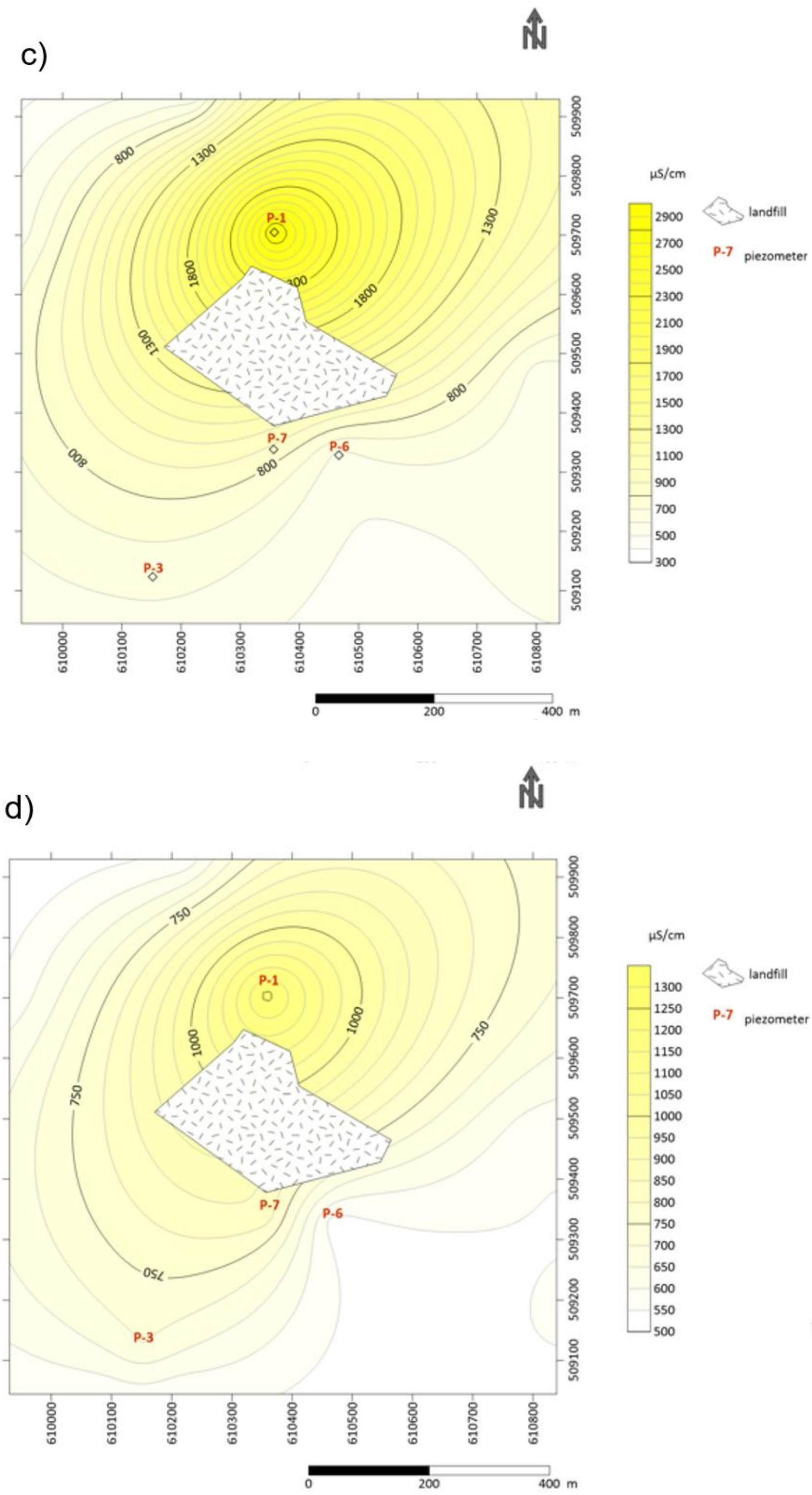
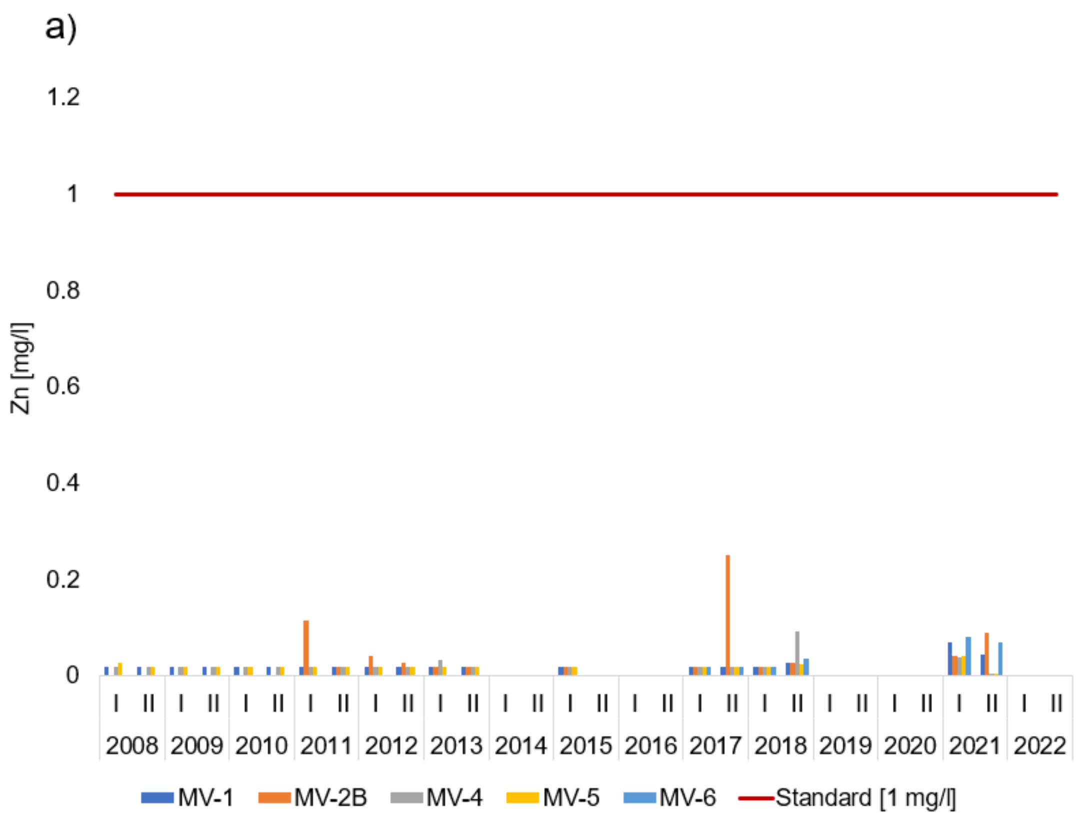


Figure 6.3. pH and EC levels in groundwater in the vicinity of the Zakroczym landfill: a) pH [-] 2008–2015, b) pH [-] 2016–2022, c) EC [μS/cm] 2008–2015, d) EC [μS/cm] 2016–2022.

In the case of the Zdounky landfill (Fig. 6.4), it was shown that Zn concentrations in the groundwater were many times lower than the WHO (2017) limit. The highest value was recorded once in 2017 for MV-2B and was 0.25 mg Zn/l, while in other cases the average Zn content was 0.03 mg Zn/l, which suggests class I groundwater quality in accordance with *Regulation of the Minister of Maritime Economy and Inland Navigation of 11 October 2019 on the criteria and method for assessing the status of groundwater bodies*.

Monitoring results indicate incidental peaks in Cr_{total} concentrations. Exceedances were particularly evident in 2017 and 2018 in all piezometers. The elevated Cr_{total} concentrations in this case could be the result of pollutant runoff from the tire or demolition waste storage sector as evidenced by the study of Eckbo et al. (2022), which mentions the environmental problem of recycling concrete due to leaching and spreading of Cr. Many scientific studies have confirmed the occurrence of HMs in groundwater as a result of poor technical sealing of landfills (Bilardi et al., 2018; Ahmad et al., 2021; Teta et al., 2017), including the failure of leachate collection systems or GM ruptures (Koda et al., 2019). Therefore, it cannot be ruled out that the excess Cr concentrations may have been related to the failure of the leachate collection system. Cr is a contaminant of groundwater and soil, originating mainly from anthropogenic activities (Ceballos et al., 2023).

Tumolo et al. (2020) found that thermal power plants and other combustion installations, as well as waste and wastewater management facilities, are also significant industrial contributors to Cr emissions from water. Nevertheless, it is noted that as of 2018, Cr content in groundwater is already below the permissible limit and corresponds to Class II groundwater quality according to the *Regulation of the Minister of Maritime Economy and Inland Navigation of 11 October 2019 on the criteria and method for assessing the status of groundwater bodies*. For the other quality parameters, pH and EC, as shown in Fig. 6.4 c–d the results of pH (< 8.5) and EC (< 700 μS/cm), which meet the assumption of good groundwater quality (quality class I), and there were no exceedances in any of the case studies from to 2008–2022.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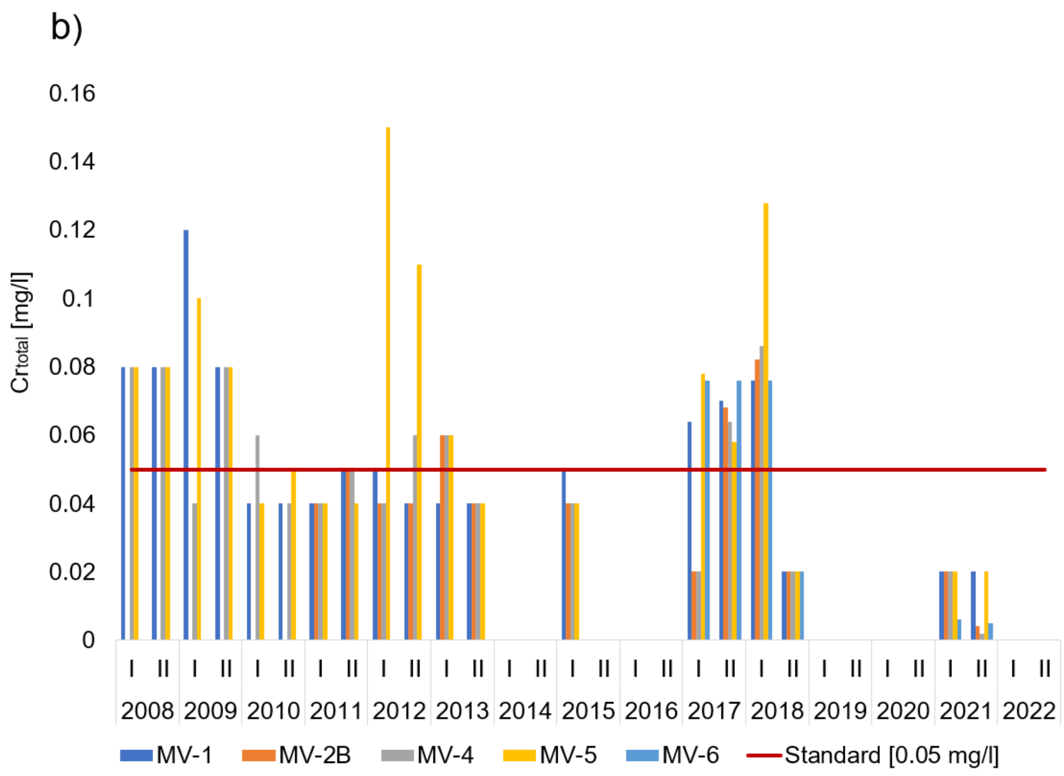
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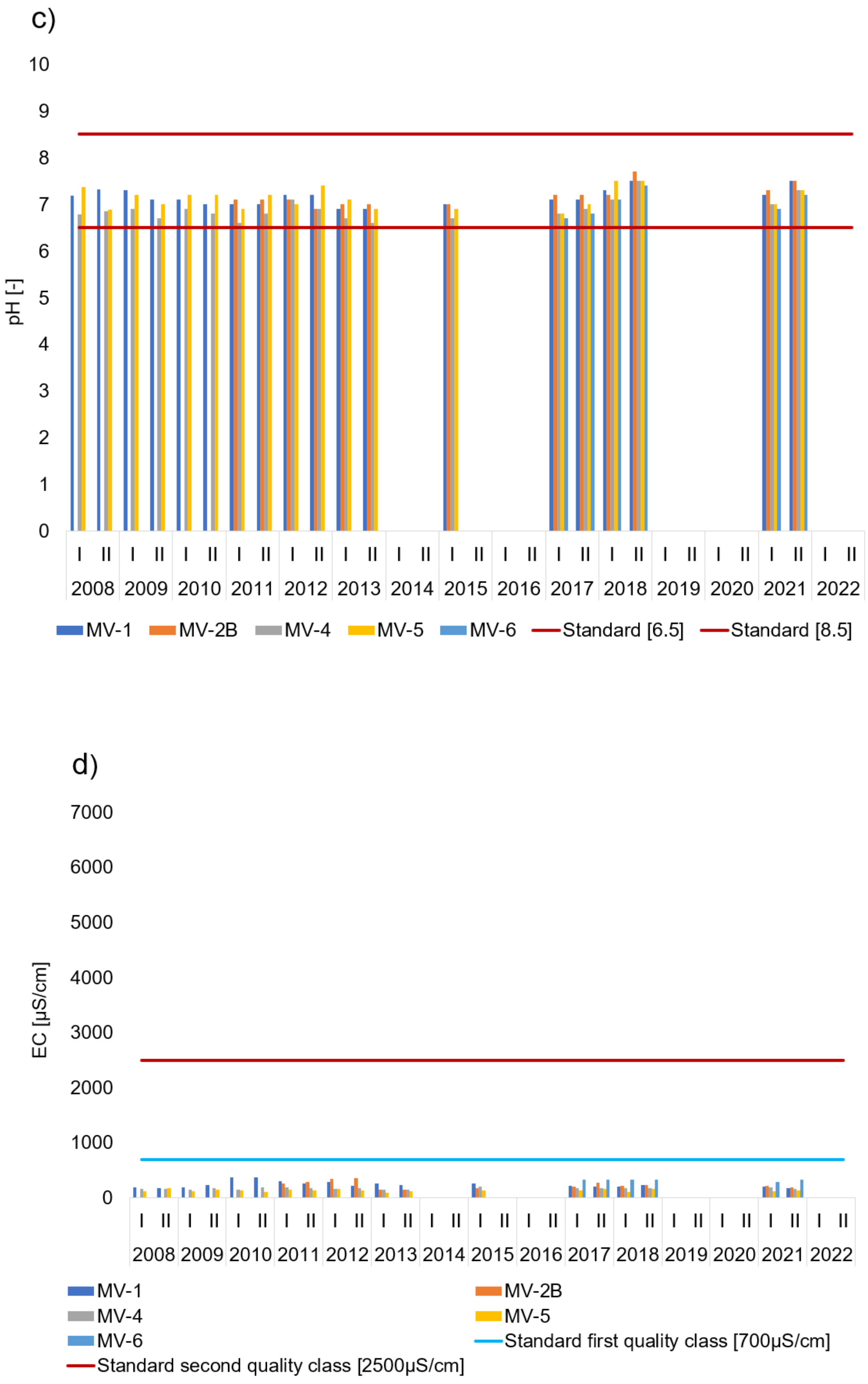


Figure 6.4. Groundwater monitoring at Zdounky: a) Zn [mg/l], b) Cr_{total} [mg/l], c) pH [-], d) EC [µS/cm].

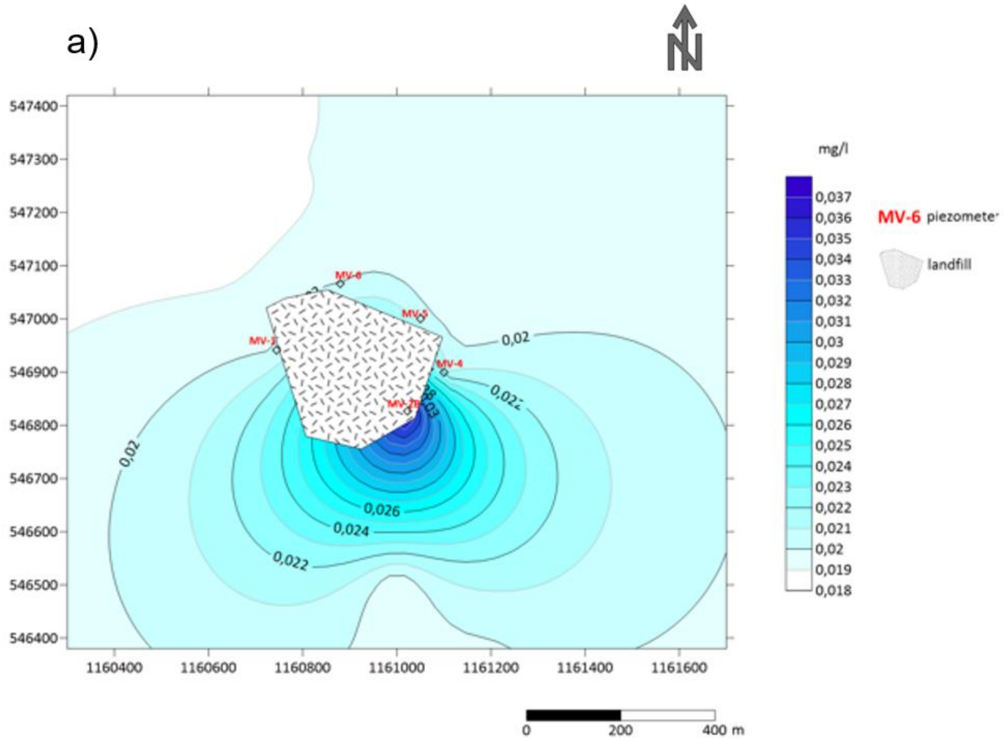
Fig. 6.5 shows the spatial distribution of the average Zn and Cr contamination of groundwater in two periods: 2008–2015 and 2016–2022 in Zdounky. It is noted that the location of the highest Zn concentrations in 2008-2015 and in 2016-2022 was the same, i.e. at piezometer MV-2B, and amounted to 0.037 mg Zn/l (Fig. 6.5 a-b). On the other hand, it is noted that in 2008-2015 the contaminant isolines reach lower values than in 2016-2022, resulting in a decrease in groundwater quality in terms of Zn. Nevertheless, Zn concentrations are low enough that there is still a large reserve compared to the WHO (2017) standard of 1 mg Zn/l. The maximum concentrations of Cr_{total} at the Zdounky landfill also have a consistent location, as in both 2008-2015 and 2016-2022 the highest concentrations were observed in the vicinity of the MV-5 piezometer and were 0.07 mg Cr_{total}/l (2008-2015), and 0.054 mg Cr_{total}/l (2016-2022), respectively causing the standard of 0.05 mg/L to be exceeded.

Fig 6.5 c-d, in turn, shows the spatial distribution of pH and EC in 2008-2015 and 2016-2022. A similar trend to that of Zn is observed, as the range of maximum values for pH and EC increased in 2016-2022 compared to previous years. Nevertheless, in the case of pH as well as EC, the results obtained can be further assigned to Class I of groundwater quality.

It is also noted that, due to the lack of data in 2019 and 2020, it is not possible to state clearly about improvement or deterioration, but the results obtained provide a basis for more frequent testing of groundwater quality, especially in MV-5, where its highest exceedances occurred most often, and MV-1, as it is the most representative piezometer located at the landfill outflow.

Monitoring studies show that the groundwater quality in landfill vicinity in Zakroczym and Zdounky is generally good. In Zakroczym, Zn and Cr (VI) concentrations are below WHO standards, and previous anomalies in EC observed in 2008-2015 have stabilized since 2011, indicating improvement of water quality. In Zdounky, despite very low Zn concentrations and pH and EC values of Class I water quality standards, elevated Cr_{total} content was observed between 2008 and 2018, which has decreased to acceptable levels for Class II water quality since the second half of 2018.

a)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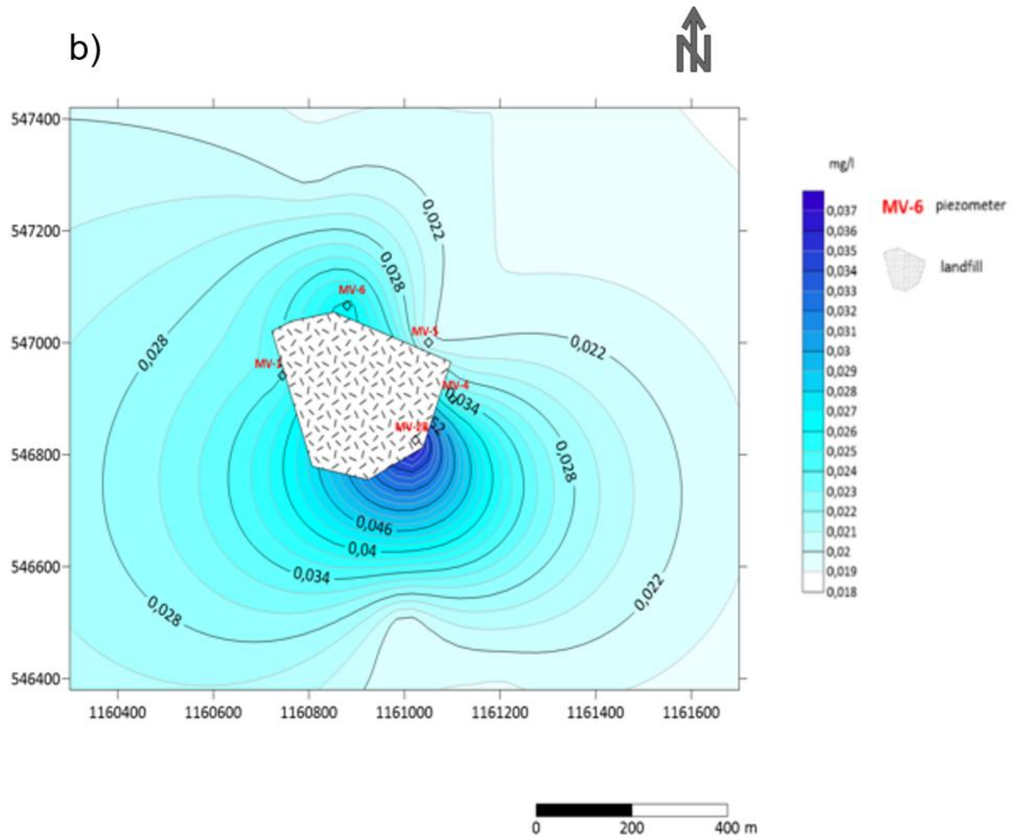
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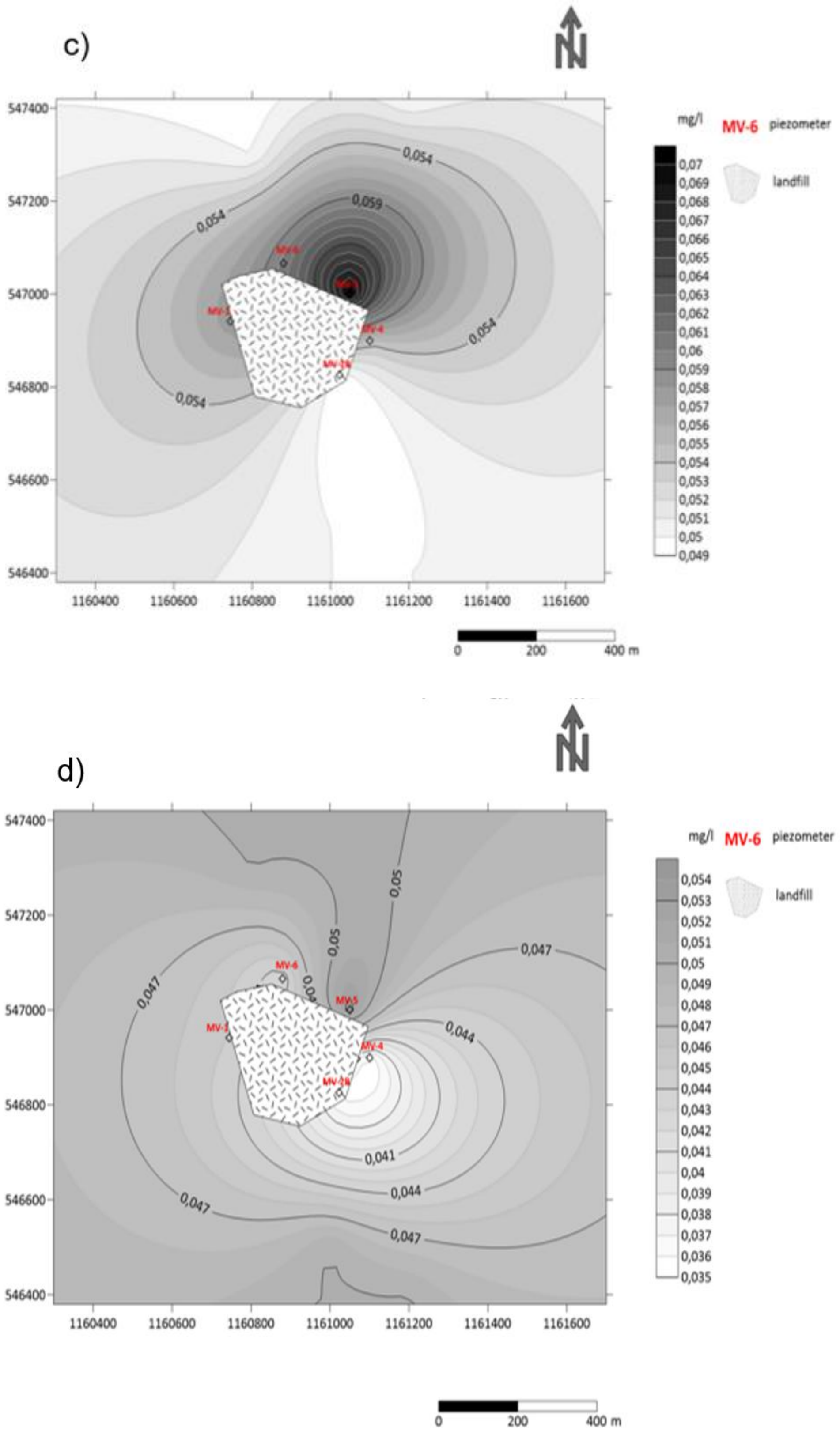
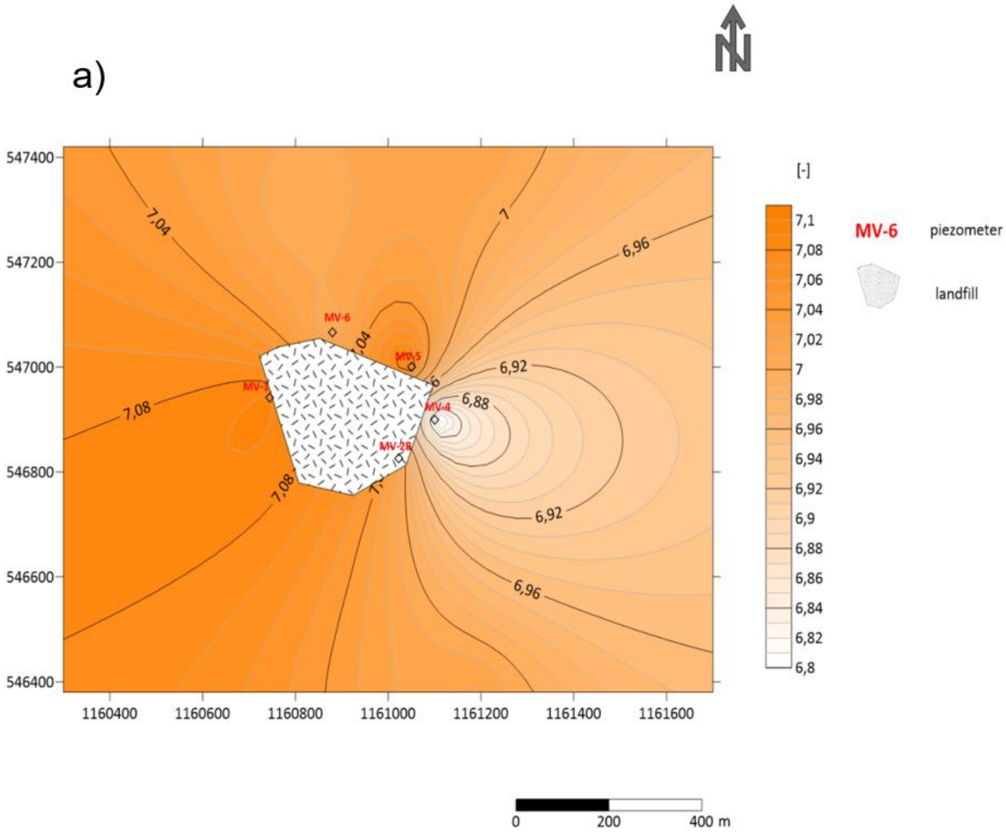


Figure 6.5. Content of selected HMs in groundwater in the vicinity of the Zdounky landfill site: a) Zn [mg/l] 2008–2015, b) Zn [mg/l] 2016–2022, c) Cr_{total} [mg/l] 2008–2015, d) Cr_{total} [mg/l] 2016–2022.

a)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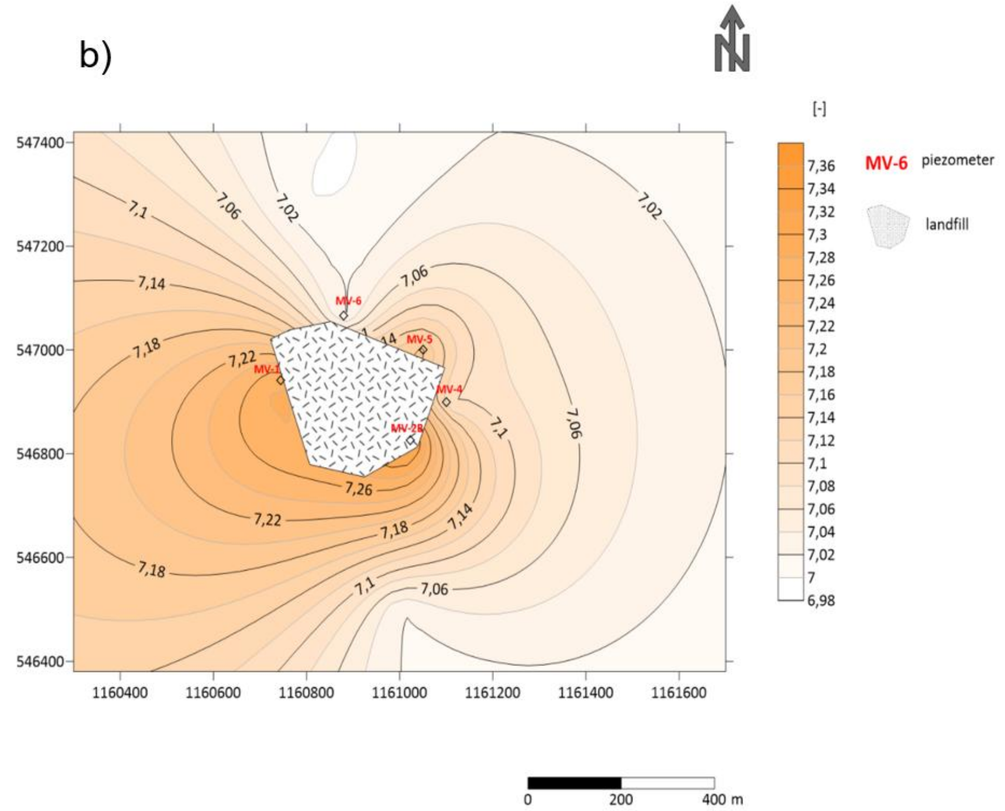
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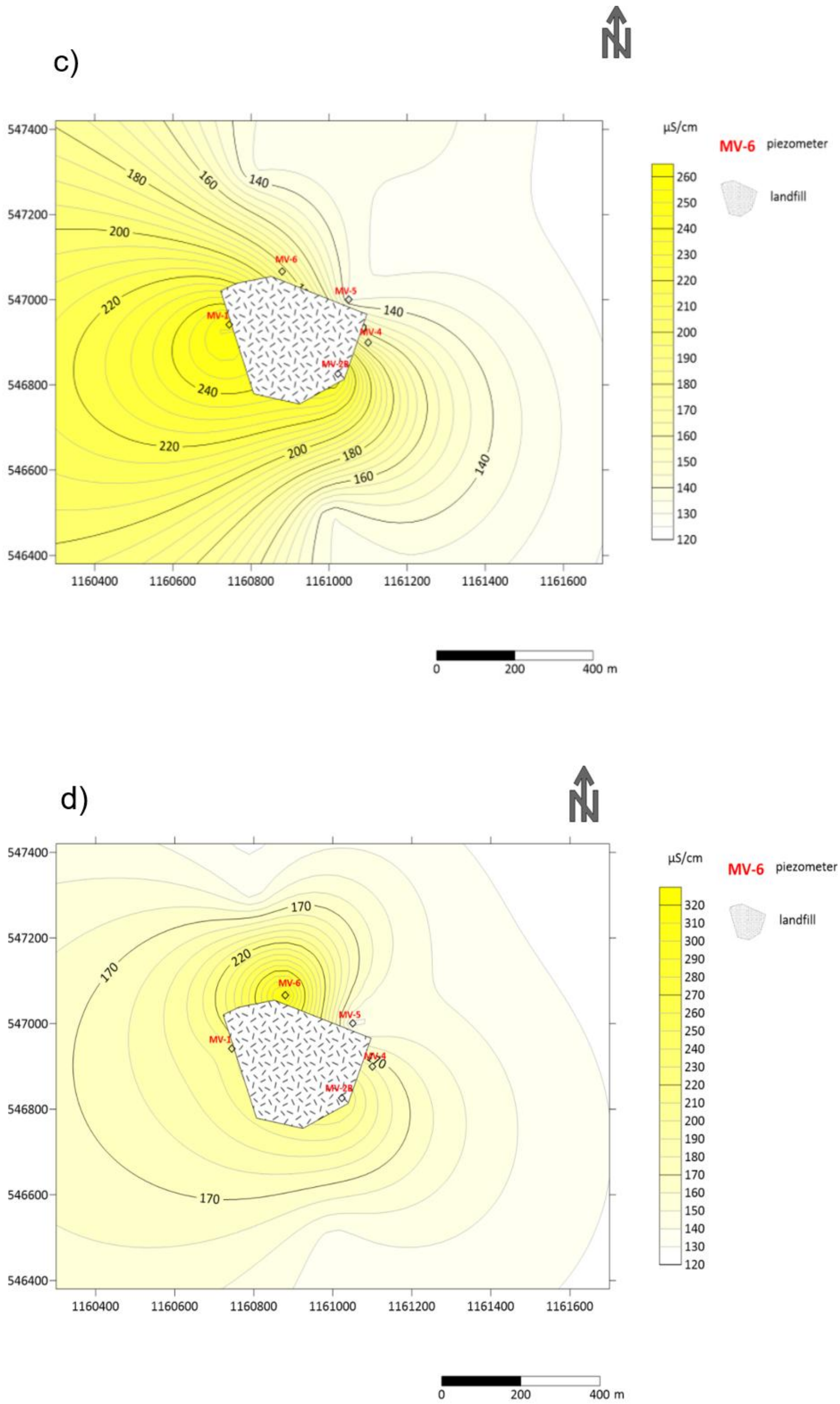


Figure 6.6. pH and EC levels in groundwater in the vicinity of the landfill in Zdounky: a) pH [-] 2008–2015, b) pH [-] 2016–2022, c) EC [µS/cm] 2008–2015, d) EC [µS/cm] 2016–2022.

6.1.2. Monitoring of leachate production and composition

The following subsection analyzes the leachate from two landfills located in Zdounky and Zakroczym from 2008 to 2022. For the Zdounky landfill, a trend of increasing pH was observed over time, and the measured pH values were higher than those for the Zakroczym landfill (Fig. 6.7a). The average pH value for Zdounky was 8.38, whereas that for Zakroczym was 8.32. According to the *Regulation of the Minister of the Environment on December 21, 2015, on the criteria and methods for assessing the status of groundwater bodies*, waters with a pH between 6.5 and 9.5 are considered to have good chemical status. For both Zdounky and Zakroczym, the values obtained allowed the determination of the good chemical status of the leachate. An increase in leachate pH was also noted by Lindamulla et al. (2022), who found that leachate pH increased over time from slightly acidic to alkaline as a result of waste stabilization. Much greater discrepancies were observed in the case of electrical conductivity, where the average EC value from 2008 to 2022 for Zdounky was 10037 μS/cm, while for Zakroczym it was 6802.7 μS/cm. For the EC parameter, there are no limits set for leachate discharged into sewage facilities or the ground. The electrical conductivity of the solid waste itself is generally very low causing a marked contrast with leachate, whose electrical conductivity is high and according to Fetter (1994) ranges from 480 to 72500 μS/cm, with an average value of 5000 μS/cm (Fig. 6.7b). The observed values of pH and EC in this study are consistent with ranges reported in other studies, including those by Kjeldsen et al. (2002) and Aziz et al. (2008), which confirm that aging landfills tend to produce leachate with neutral to alkaline pH and elevated electrical conductivity due to ongoing decomposition and ion accumulation.

Fig. 6.8 shows the contents of Cr and Zn in leachates at the studied landfills. For both Cr (Cr (VI) and Cr_{total}) and Zn, higher concentrations were noted at the Zdounky landfill where the average content of Cr_{total} for 2008–2022 was 0.77 mg/l, and Zn was 0.27 mg/l. In the case of Zakroczym, the average Cr (VI) content reached 0.02 mg/l, and Zn 0.08 mg/l. The significantly higher Cr content in the leachate at the Zdounky landfill is due to the two different forms of Cr measured. Nevertheless, there has been a noticeable improvement in the quality of leachate in terms of Cr_{total} over the past 10 years for the Zdounky landfill. According to the *Announcement of the Minister of Infrastructure and Construction of 28 September 2016 on the publication of the consolidated text of the*

Regulation of the Minister of Construction concerning the method of fulfilling obligations by industrial wastewater suppliers and the conditions for discharging wastewater into sewerage systems, the limits on permissible concentrations for Cr_{total} (max = 1 mg Cr/l), Cr(VI) (max = 0.2 mg Cr(VI)/l) and Zn (max = 5mg/l) have been maintained for both study sites.

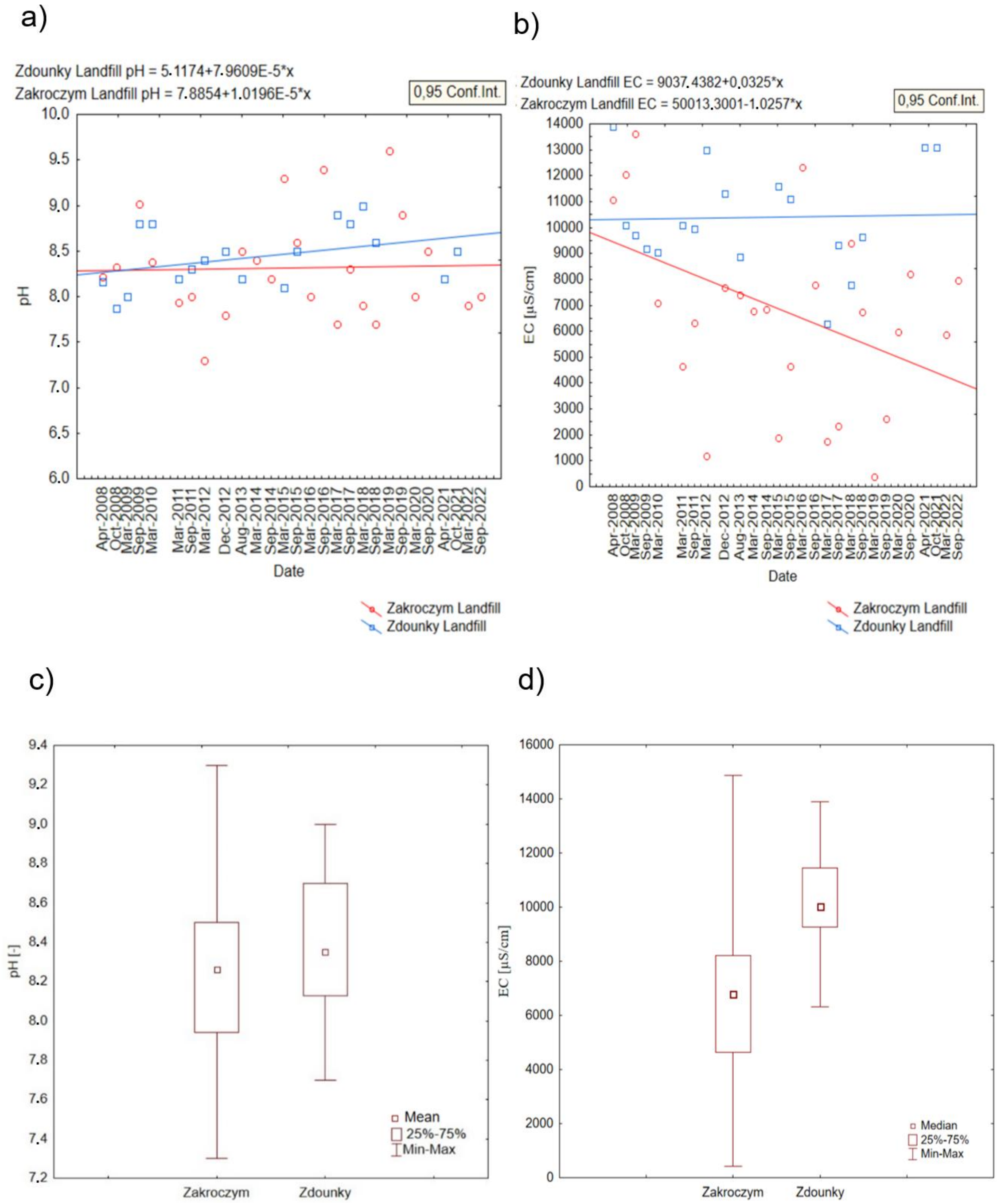


Figure 6.7. Changes in leachate parameters in Zakroczym and Zdounky landfill: a) changes in pH over time, b) changes in EC over time, c) pH box plot, d) EC box plot.

In the case of Zn content in leachates at the Zdounky landfill, no upward or downward trend in the results obtained was noted. The situation was similar for the landfill in Zakroczym, where the obtained Zn concentrations were very close to each other, which was confirmed by the small scatter in the box plot in Fig 6.8b. The recorded Zn concentrations at both of the studied landfills are <2 mg/l, which makes them clean enough that, according to the *Regulation of the Ministry of Marine Economy and Inland Navigation* (2019), they could be treated as rainwater and snowmelt and discharged to water or land.

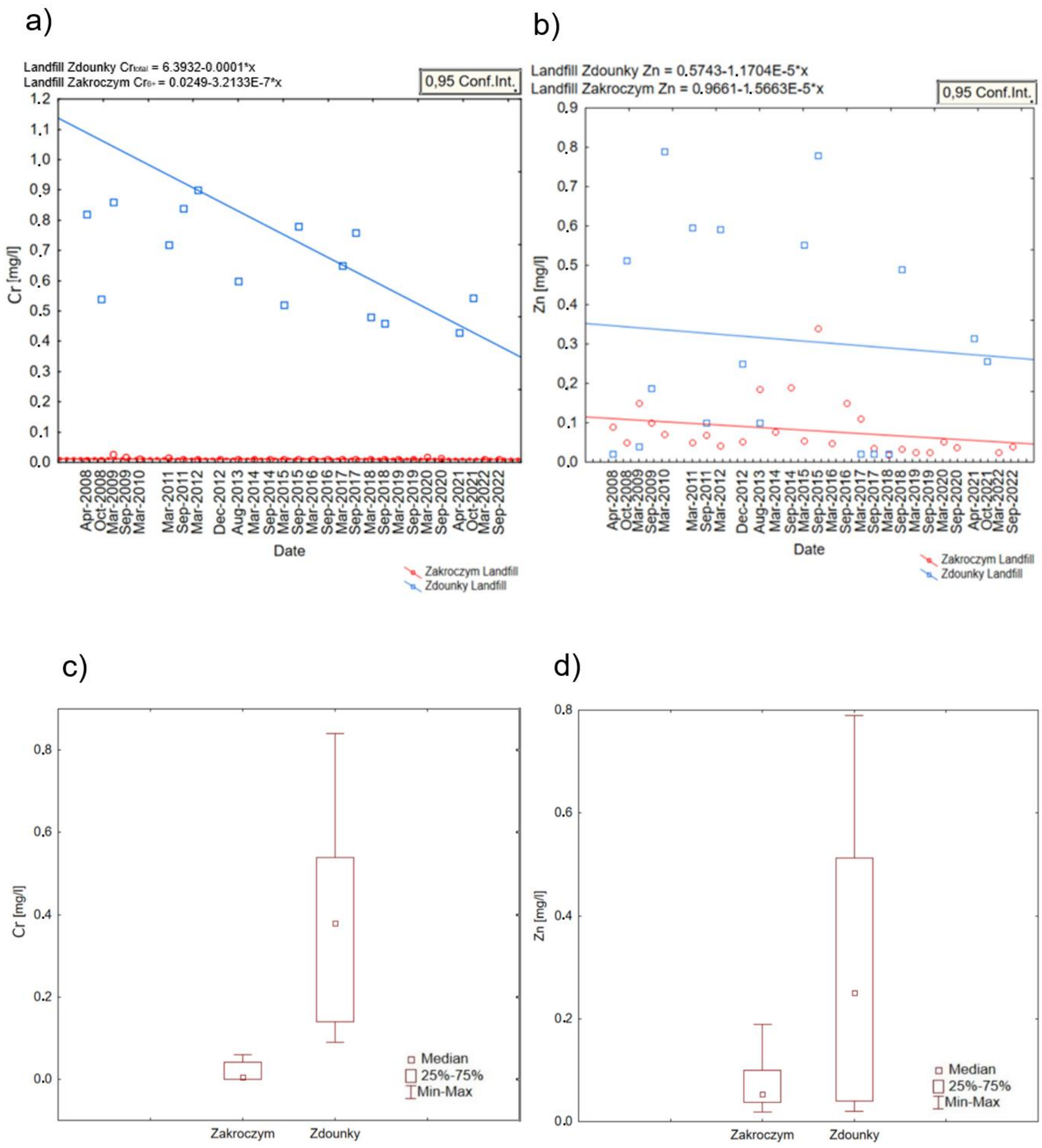


Figure 6.8. Changes in HMs in leachate parameters in Zakroczym and Zdounky landfill: a) changes in Cr over time Cr, b) changes in Zn over time, c) Cr box plot, d) Zn box plot.

Fig 6.9 however shows the observed concentrations of Hg and PAHs at the studied landfills. In both landfills, Hg content is negligible, with a maximum of 0.001 mg/l against a permissible standard of 0.06 mg/l applied to the discharge of industrial wastewater into sewage facilities. Fig. 6.9 b shows that the concentrations of PAHs in the Zakroczym landfill exceed the standard of 0.2 mg/l in the initial years of measurement, but decrease significantly over time (the average values were 0.05 mg/l). For the Zdounky landfill, concentrations were consistently low and hard to detect (average 0.0004 mg/l). In the case of both Hg and PAHs, there is a greater scatter in the values obtained in the case of the Zakroczym landfill, which is due to occasional deviations from the curve. It may be the result of measurement inaccuracy in the case of extreme values obtained in March 2018 (Hg) and August 2013 (PAHs).

An important factor affecting the quality of leachate to a large extent is the vicinity of the areas, which can cause changes in the chemical properties of leachate. The Zdounky landfill is surrounded on all sides by agricultural land and is therefore exposed to nitrogen and phosphorus compounds found in fertilisers. As shown in Fig. 6.10, significantly higher concentrations of NH_4+ and Ptotal were recorded compared with the Zakroczym landfill, which is surrounded by industrial areas.

Based on the analysis of leachate water samples, an exceedance of the permissible values specified in the *Announcement of the Minister of Infrastructure and Construction of 28 September 2016 on the publication of the consolidated text of the Regulation of the Minister of Construction concerning the method of fulfilling obligations by industrial wastewater suppliers and the conditions for discharging wastewater into sewerage systems* was found with respect to NH_4^+ levels at the Zdounky landfill in nearly all measurement series. The recorded NH_4^+ values remained stable over time but exceeded the permissible limit of 200 mg/l (with average NH_4^+ concentrations at Zdounky reaching 408.6 mg/l, and at Zakroczym 114.3 mg/l).

According to Talalaj (2015), high NH_4^+ concentrations can occur in stabilized leachate and may indicate that biological treatment methods are ineffective for their removal. Gómez et al. (2019) defined typical NH_4^+ concentration ranges as 2200–5200 mg/l for young landfills, 1200–3600 mg/l for intermediate landfills, and 400–900 mg/l for old landfills, meaning the obtained results most closely resemble those of the old landfills.

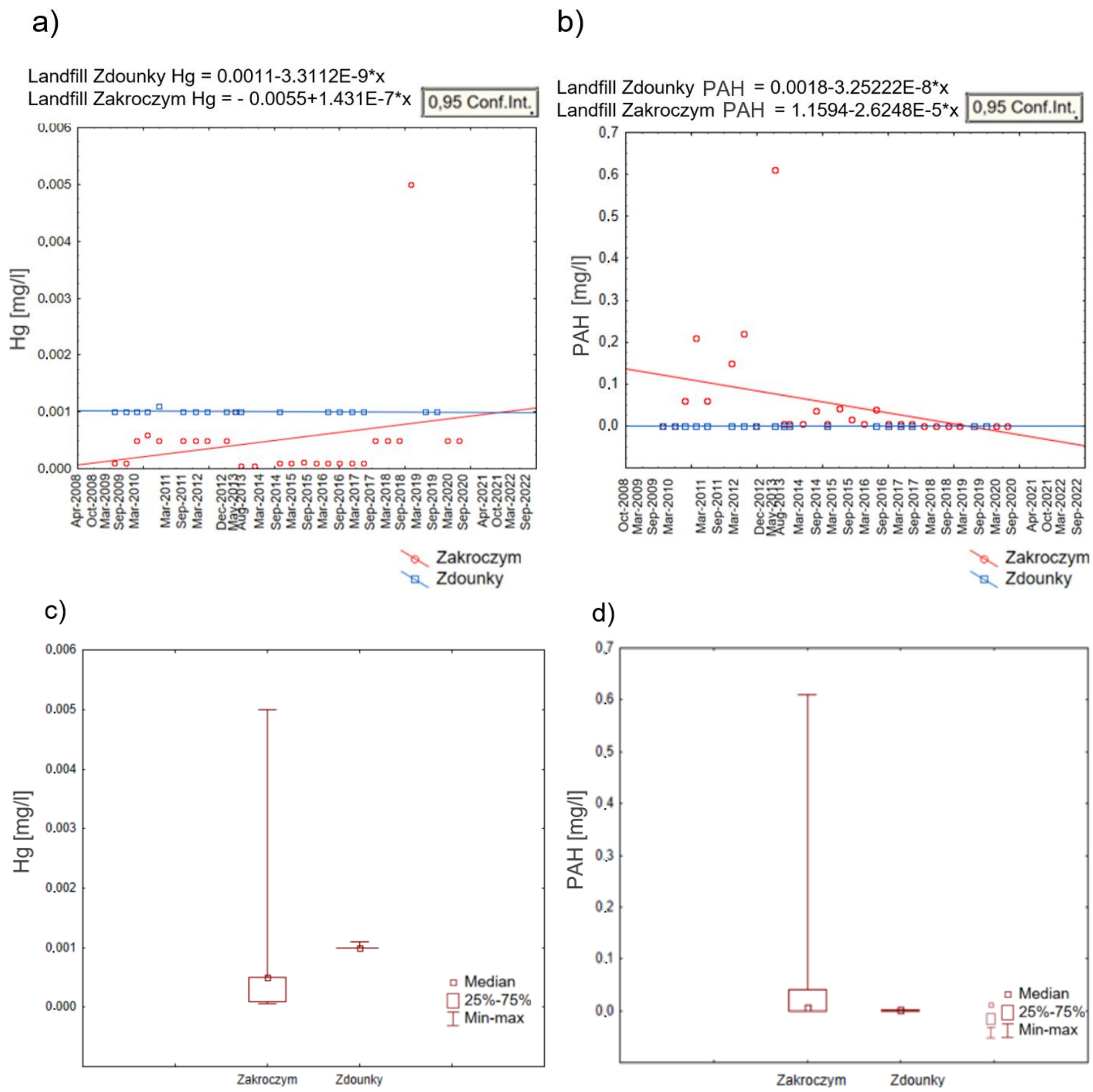


Figure 6.9. Changes in leachate parameters in Zakroczym and Zdounky landfill: a) changes in Hg over time, b) changes in PAHs over time, c) Hg box plot, d) PAH box plot.

Nevertheless, it should be noted that in the case of the Zdounky landfill, a 3-fold exceedance of the permissible limit set by the above-mentioned regulation was observed, while in the case of the Zakroczym landfill, a slight exceedance of the permissible concentration of NH_4^+ in leachate was recorded in only 2 measurements (in Q1 2020 and Q4 2022). Nevertheless, the monitoring of NH_4^+ and P_{total} at the Zakroczym landfill started in 2018, due to the measurement gap between 2008 and 2017, which could also lead to values deviating from the standard, among other things because of the already noted upward trend in the results obtained. A similar trend can be seen for the concentrations of P_{total} in leachate. According to the EPA (2000a), the concentration of P_{total} in leachate from landfills for non-hazardous waste should not exceed 6.5 mg/l, which is largely consistent with the results obtained, except for the two cases recorded in

Zdounky in 2021, where the values obtained were 7.08 mg/l and 7.55 mg/l, respectively (the average P_{total} values in Zakroczym were 2.92 mg/l, while in Zdounky 6.30 mg/l).

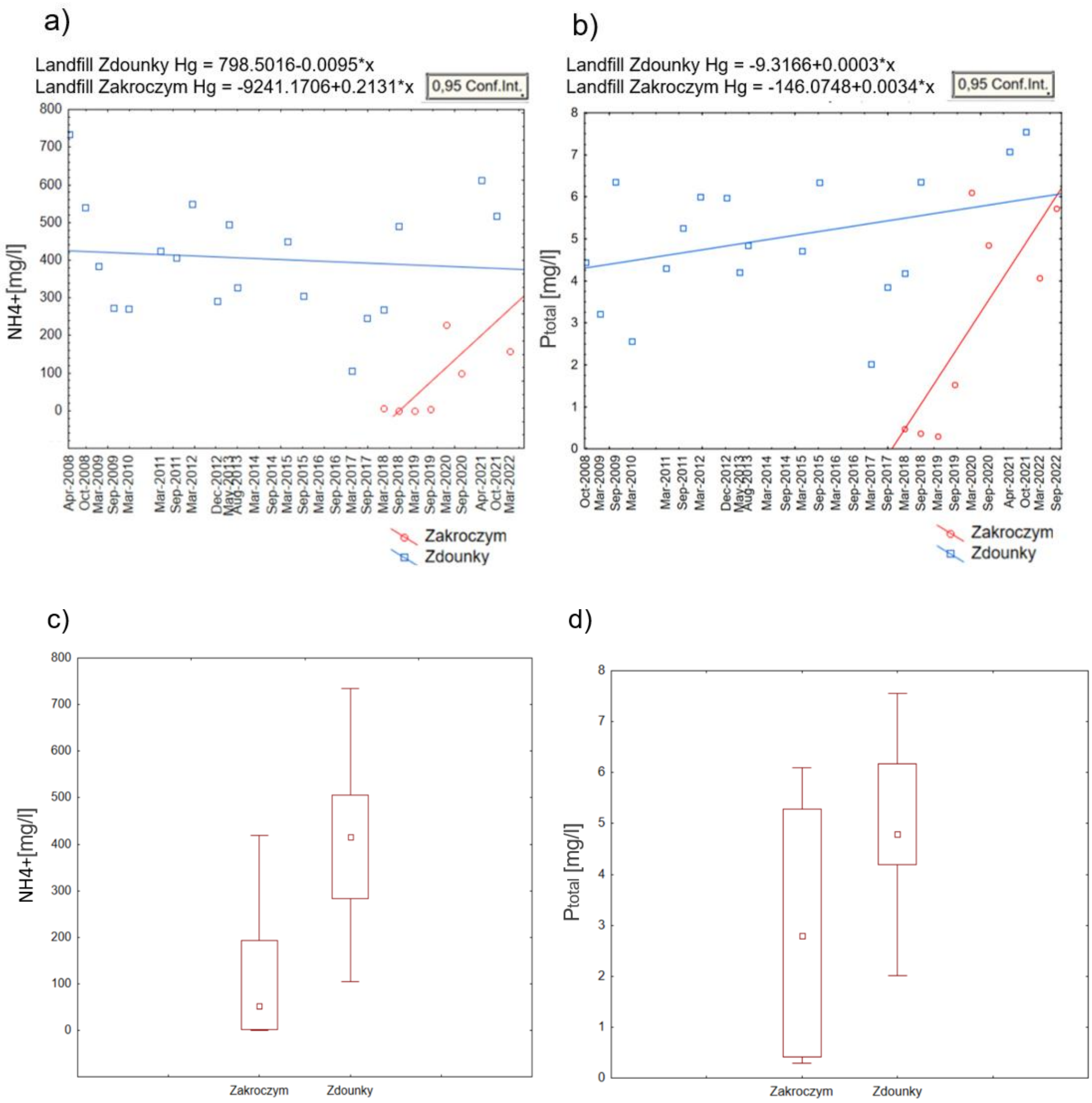


Figure 6.10. Changes in leachate parameters and box plots in Zakroczym and Zdounky landfill: a) changes in NH_4^+ over time, b) changes in P_{total} over time, c) NH_4^+ boxplot, d) P_{total} boxplot.

In order to quantify the leachate pollution potential of the studied landfills, LPIs were calculated, which provided quantitative and comprehensive information on the level of leachate pollution (Podlasek et al., 2023). For the Zdounky landfill, the minimum LPI = 7.5, average LPI = 10.07, and maximum LPI = 13.5, while in Zakroczym, the minimum LPI = 3.59, average LPI = 4.43, and maximum LPI = 7.70, were recorded (Fig. 6.11). According to Kumar and Alappat (2005), LPI should theoretically range between 5 and

100, nevertheless, in the case of the Zakroczym landfill, the lowest recorded LPI was 3.59.

This finding is consistent with the results presented by Tenodi et al. (2020), who estimated the LPI at the studied landfills during the rainy season in 2014 at levels of 4.81, 4.74, and 4.66. The low LPI values obtained for the Zakroczym landfill may be related to the fact that only two inorganic parameters: pH and NH_4^+ were included in the calculation due to the lack of monitoring data. In contrast, for the Zdounky landfill, Total Kjeldahl Nitrogen (TKN) and Total Dissolved Solids were also considered. As Mor et al. (2018) pointed out, each parameter in leachate properties has a significant impact on the LPI calculations. Survey results by Kumar and Alappat (2005) showed that HMs are the most important in LPI assessment (51.1%), followed by inorganic elements (25.7%) and organic components (23.2%). Nonetheless, it is important to note that for active landfills, the leachate discharge standard for surface water in terms of LPI should not exceed 5.696, according to the Acceptable Conditions for Discharge of Leachate, Environmental Quality Regulation (Hussein et al., 2019). In leachate from the Zakroczym landfill, there were situations (September 2016, March 2019, September 2022) when this value was exceeded, and in the case of the Zdounky landfill at each test point. This suggests that landfill leachate may pose a pollution source in the surrounding environment. A different approach was taken by Saghi et al. (2024), who believed that environmental degradation is likely if the LPI remains above the threshold of 35. Another important factor is that the age of the landfill also affects the LPI level. For closed landfills, the LPI value tends to be lower than for active ones, due to the stabilization of decomposition processes (Wdowczyk and Szymańska-Pulikowska, 2021). Abunama et al. (2021) further stated that for immature leachate (<5 years), LPI values are typically around 26.5 for moderately mature leachate (5–10 years) – around 23.6 and for mature leachate (>10 years), approximately 17.5. Given that both studied landfills are over 10 years old, the observed trend of low LPI values is consistent with findings from other research on this topic.

Landfill leachate is a key source of information about the processes occurring within landfills and the effectiveness of applied reclamation methods in mitigating their environmental impact. An analysis of leachate from 2008–2022 conducted for the landfills in Zdounky and Zakroczym revealed significant differences in leachate quality.

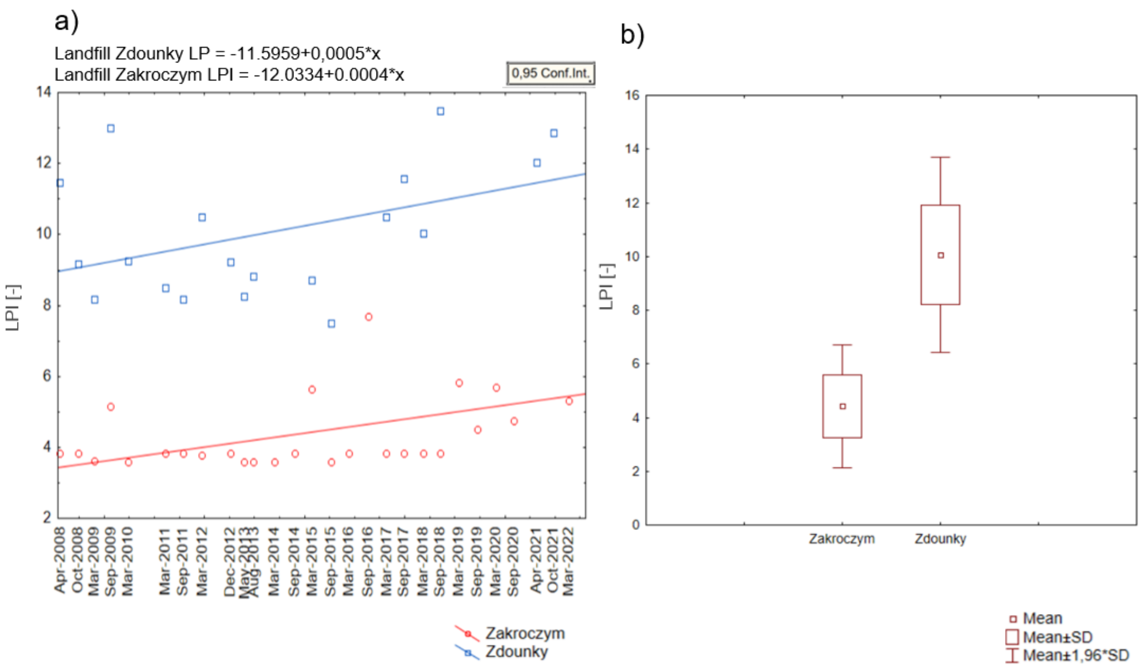


Figure 6.11. Temporal changes of LPI: a) Zakroczym and Zdounky landfills, b) LPI box and whisker plots.

The LPI, which measures the level of leachate contamination, was higher in Zdounky (max. 13.5) than in Zakroczym (max. 7.7), indicating a greater pollution potential in the first case. In Zdounky, the use of an HDPE GM to cover the landfill cell effectively limited rainwater infiltration, resulting in higher concentrations of pollutants such as HMs and inorganic compounds, which require advanced treatment methods. Conversely, at the reclaimed western cell in Zakroczym, the use of a mineral cover layer facilitated greater infiltration of rainwater, which in turn diluted the leachate, reducing pollutant concentrations but also increasing the risk of their migration into groundwater. Morita et al. (2023) demonstrated that covering a landfill with a GM creates a barrier that limits oxygen ingress and the influence of atmospheric conditions, leading to more anaerobic conditions inside the landfill. As a result, redox potential (ORP) decreases, supporting the development of microorganisms adapted to anaerobic environments, such as methanogens and sulfate-reducing bacteria. This type of environment also facilitates the immobilization of metals through their precipitation as sulfides. On the other hand, uncovered landfills are more exposed to oxygen and rainfall, which leads to greater microbial diversity (e.g., a higher presence of aerobic bacteria such as those from the Acidobacteria, Proteobacteria, Bacteroidetes, and Actinobacteria groups) and greater variability in chemical parameters such as pH and moisture. This, in turn, may increase the mobility of pollutants. The differences in the applied reclamation methods affect not only the quality of leachate but also the potential environmental impact of the landfills. Although both sites are equipped with pollution control systems, failure of these systems particularly in Zdounky could increase the risk of contaminating surrounding areas. Therefore, regular monitoring of leachate quality and maintaining the efficiency of protective systems is crucial. The results highlight the importance of tailoring reclamation methods to local environmental conditions and the specific characteristics of leachate generated by each landfill

6.1.3. Landfill gas monitoring

The following subsection presents LFG composition monitoring studies and statistical tests of variability in CH_4 production in groups before the reclamation period (before 2011) and after reclamation (after 2011) at two partially reclaimed MSW landfills in Zakroczym and Zdounky. Statistical tests were performed to assess the variability and level of significance of the CH_4 content in LFG at the reclaimed cells of the studied landfills from 2008 to 2022. Tab. 6.1 shows the basic statistics for the data used to verify the research hypotheses. It should be noted that both the lowest and highest values were observed for the Zakroczym landfill. The median of CH_4 content in all the cases oscillated at approximately 35%. Nevertheless, this level is much lower than that reported by Bouzaudim and Ouazzani (2024), who found that 84% of the landfills studied generated CH_4 at levels between 50% ± 2% and 70% ± 2%. A CH_4 content of 60% in landfills was also found by Nematollahi et al. (2024), who reported that the rest was taken up by CO_2, and Bian et al. (2021b), who estimated CH_4 emissions after landfill closure at 48%. It should be noted that in the studied landfills after reclamation there are situations where the maximum content of CH_4 produced is greater than 30%, reaching 66.5% in the case of the Zakroczym landfill and 39.70% in the case of the Zdounky landfill. Nevertheless, it must be considered that the data from the Zakroczym landfill had a higher SD than the data from the Zdounky landfill.

The first group tested for statistically significant differences between groups was data from the Zakroczym landfill (until 2011 and after 2011). In the first group, 24 data samples were considered, whereas in the second group, 26. Before the selection of statistical tests, the normality of the distribution in each studied set was tested using the Shapiro-Wilk test. The data in the Zakroczym group before reclamation did not have a normal distribution ($p = 0.0009$), while the data in the Zakroczym group after reclamation had a normal distribution (p = 0.2388) therefore, a non-parametric Wilcoxon rank sum test was used to test the hypothesis of differences between the groups, which showed that there was no statistically significant difference between the data groups tested at the Zakroczym landfill before and after the reclamation period (p-value = 0.1425).

Table 6.1. Descriptive statistics on the studied CH_4 before and after reclamation at the Zdounky and Zakroczym landfill.

Period	**Min**	**Max**	**95% confidence interval**		**n**
			Median	**SD**	
Zakroczym before reclamation	0.3	65.7	48.5	23.4	24
Zakroczym after reclamation	0.3	66.5	27.0	20.4	26
Zdounky before reclamation	22.5	34.8	31.5	3.6	18
Zdounky after reclamation	23.2	39.7	33.2	5.1	19

Fig. 6.12 shows box plots of the studied groups in Zakroczym landfill. It is noted that both groups have a very similar range. Nevertheless, one notices completely different medians amounting respectively until 2011 – 48.5% and after 2011 – 27%, which indicates a reduction in the potential CH_4 content of LFG after reclamation, which also has an impact on diffuse emissions.

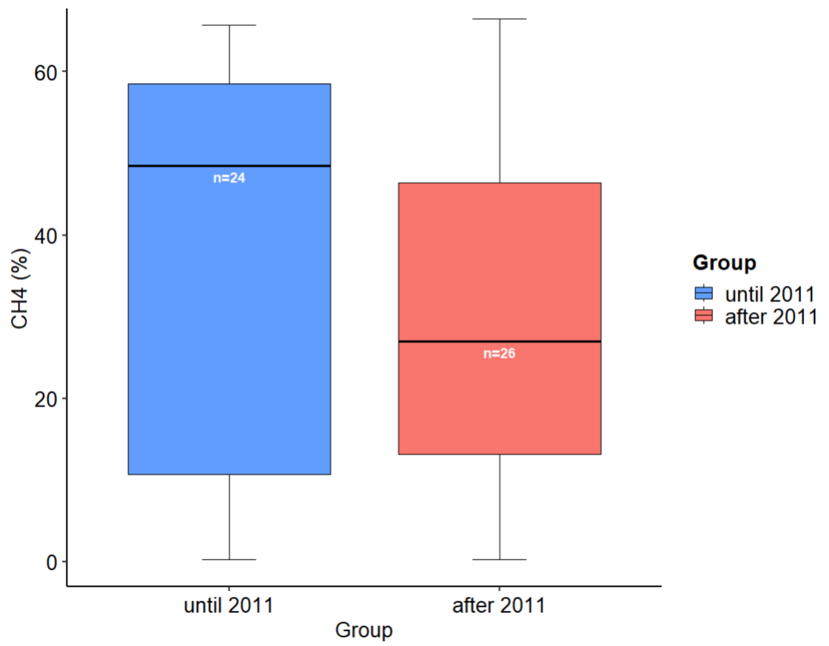


Figure 6.12. Box and whisker plots of CH_4 content at the reclaimed landfill site in Zakroczym before and after the reclamation.

The next groups tested were data from the Zdounky landfill (until 2011 and after 2011). In this case, the Shapiro-Wilk normality test showed that both groups of data had a normal distribution (p-value until 2011 = 0.1844, p-value after 2011 = 0.0967). In addition, the Levene test showed homogeneity of the data (p-value = 0.2492), a parametric two-way ANOVA test was conducted. Tab. 6.2 shows the results of the test, which present no statistically significant differences between groups (p = 0.4220) and no statistically significant differences between quarter (p = 0.1020) was seen.

Table 6.2. ANOVA results.

Statistical characteristic	Df	Sum Sq	Mean Sq	F value	Pr (>F)
Group	1	12.9	12.94	0.662	0.422
Quarter	2	90.1	45.05	2.455	0.102
Group*Quarter	1	7.5	7.52	0.410	0.527
Residuals	32	587.2	18.35		

$* \ p \leq 0.05$

Fig. 6.13 shows box plots of the studied groups. It is noted that the two groups with similar numbers (n = 18 and n = 19) have different ranges. After reclamation of the Zdounky landfill, the level of CH_4 in the LFG increased to 39.7%, which illustrates the rapid effect of the synthetic cover on increasing CH_4 production. Nevertheless, in contrast to the CH_4 content in Zakroczym, the medians in Zdounky were close to each other, amounting to 31.50% until 2011 and 33.20% after, respectively, which may have influenced the observed lack of statistically significant correlations between the groups before and after reclamation.

The third data set tested was the Zdounky after 2011 and Zakroczym after 2011 groups, which represented the percentage concentration of CH_4 in the studied landfills after the reclamation process. The Shapiro-Wilk test showed that the Zdounky after 2011 group (p-value = 0.0968) and the Zakroczym after 2011 group (p-value = 0.2388) had a normal distribution, however, the homogeneity of variance was not met (p-value < 0.05). To compare the groups and verify whether there are statistically significant differences between them, it was advisable to conduct non-parametric tests.

Since the data had a normal distribution, the groups were tested using two non-parametric tests (Wilcoxon rank sum test and Kruskal-Wallis test) and a parametric ANOVA.

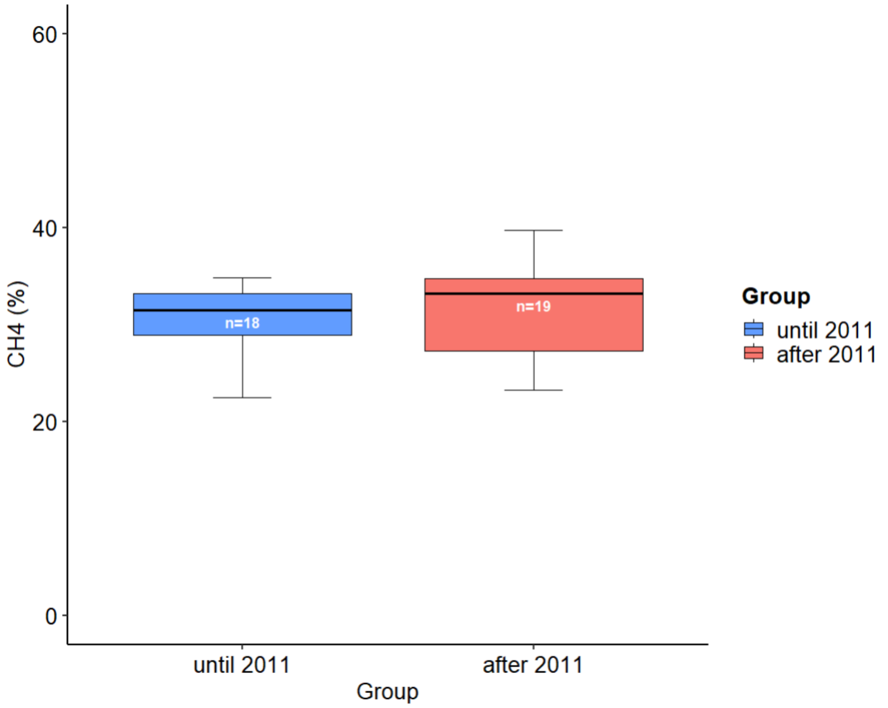


Figure 6.13. Box and whisker plots of CH_4 content at the reclaimed landfill site in Zdounky before and after the reclamation.

None of the tests showed any significant differences between the groups (p = 0.5734, p-value Kruskal-Wallis = 0.5656, p-value ANOVA = 0.669) (Tab. 6.3). Accordingly, the CH_4 content of the cover at the two studied landfills after reclamation is not statistically different.

Table 6.3. Results of the nonparametric Wilcoxon rank sum test and Kruskal-Wallis rank sum test for Zdounky after 2011 and Zakroczym after 2011 groups.

Wilcoxon rank sum test	Kruskal-Wallis rank sum test	ANOVA
W = 222	chi-squared = 0.33018	F-value = 0.1860
p-value = 0.5734	df = 1, p-value = 0.5656	df = 1, p-value = 0.6690

Fig. 6.14 shows the box plots of the studied groups of the two landfills. The two groups have very different ranges from each other. The Zakroczym group showed greater variation in CH_4 levels, with a range of values (from a minimum of approximately 0% to a maximum of over 60%). The Zdounky group, has a much lower variability (from 23.20% to 39.70%), which indicates more homogeneous results in this group. Looking at the graphical representation of the results obtained, it can be concluded that the CH_4 levels vary with different coverages, but the statistical analysis did not show any significant differences. This could be due to the large dispersion of the data, which caused the lack

of differences, as well as the overlap of the CH_4 ranges from Zdounky to the range of values from Zakroczym. Another reason could be the difference in population between the groups (n = 26 and n = 19). However, the results of the Wilcoxon rank sum test (p-value = 0.5592) and Kruskal-Wallis test (p-value = 0.5494) were not significantly different from the variables.

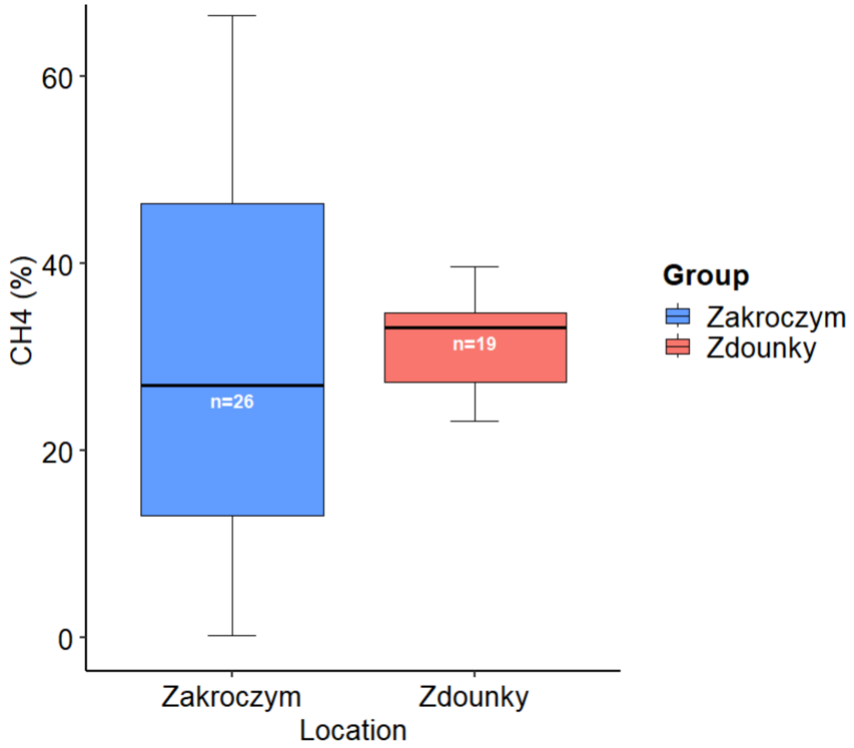


Figure 6.14. Box and whisker plot of CH_4 content in Zakroczym and Zdounky landfill after the reclamation.

In conclusion, the CH_4 content in LFG at the studied landfills did not differ from a statistical point of view. Nevertheless, the absence of a statistically significant difference does not mean that the groups were identical. This means that with the current sample size and level of variability, there is insufficient evidence to reject the null hypothesis of no difference. In practice, this may be due to a large variance, small sample size, or small difference between groups. It is also possible that methodological limitations, such as measurement uncertainty or sampling frequency, contributed to the inability to detect a significant effect. Therefore, the results should be interpreted with caution, and not as definitive proof of equivalence between the sites.

6.1.4. Summary of the monitoring study results

Monitoring studies show that the groundwater quality in landfill vicinity in Zakroczym and Zdounky is generally good. In Zakroczym, Zn and Cr (VI) concentrations are below WHO standards, and previous anomalies in EC observed in 2008-2015 have stabilized since 2011, indicating improvement of water quality. In Zdounky, despite very low Zn concentrations and pH and EC values of Class I water quality standards, elevated Cr_{total} content was observed between 2008 and 2018, which has decreased to acceptable levels for Class II water quality since the second half of 2018. Analysis of the landfill leachate over the period 2008–2022 shows that Zdounky, where an HDPE GM was used (to limit rainwater infiltration) has higher concentrations of HMs and consequently higher LPI (max. 13.5) compared to Zakroczym, where the mineral semipermeable cover system resulted in the dilution of pollutants (max. LPI 7.7). LFG monitoring shows that the mean CH_4 content fluctuates around 35% at both sites; the range of values is wider in Zakroczym (from almost 0% to 66.5%) than in Zdounky (from 22.5% to 39.7%), but the differences between the operational and post-reclamation periods are not statistically significant. The results underline the effectiveness of the reclamation methods used and the need for further monitoring of the environmental impacts of landfills.

6.2. Laboratory testing of the soil

The following chapter presents physical and chemical tests of the soils used for the reclamation of the MSW landfills in Zakroczym and Zdounky.

6.2.1. Physical tests of the soil

Fig. 6.15 and Fig. 6.16 shows selected grain size curves of soils used for reclamation of landfills in Zakroczym (Fig. 6.15) and Zdounky (Fig. 6.16). The grain size curves were plotted on a semi-logarithmic grid, where the diameters of the grains and particles were given on the axis on a logarithmic scale and their percentages on the ordinate axis on a decimal scale (Pisarczyk, 2017). The results of aerometric tests on soils collected from the Zakroczym landfill indicate that the soils forming the cover of the reclaimed landfill cell—down to a depth of approximately 1 m are, according to PN-88/B-04481, cohesive soils, primarily in the form of clayey sand (clSa) (samples 1a, 1b, 2a, 2b, 2c, 3b, 4a, 4b, 4c, 5a, 5b, 5c, 6a, 6b). These soils contain a gravel and cobbles fraction of less than 3%, a sand fraction ranging from 63% to 82.5%, a silt fraction between 12% and 29%, and a Cl fraction between 3.32% and 10%. In addition to clSa, sample 1c of the cover layer contained sandy clay silt (saclSi), which has a sand fraction of 50%, a Si fraction of 37%, and a Cl fraction of 13%, as well as sandy silt (saSi) in sample 4b, whose Sa, Si, and Cl fractions were 63.5%, 31%, and 5.5%, respectively. The elevated sand fraction observed in the soils of the landfill cover may be due to the mixing of soils with the leveling and/or drainage layer used (in the case of the Zdounky landfill).

The results of the aerometric studies however allowed to conclude that the soils present in the cover of the reclaimed landfill in Zdounky up to 1 m depth are cohesive (according to PN-88/B-04481), mainly in the form of clay (Cl) (samples 1a, 3a, 3b, 4a, 4b, 5a, 5b, 6b, 7a, 7b), containing no gravel (Gr) or cobbles fraction, sand (Sa) fraction from 10.62% to 40.10%, silt (Si) fraction from 18.83% to 44.82%, and Cl fraction from 31.9% to 46.47%. In addition to Cl, sandy silty clay (sasiCl) was found in the cover (samples 2a, 2b, 6a, 8a, 8b) with Sa fractions of 30.35% to 44.29%, Si fractions of 32.19% to 43.28%, and Cl fractions of 20.74% to 29.76%. Sample 1b also contained silty clay (siCl) with a sand fraction of 34.09%, a Si fraction of 39.28%, and a Cl fraction of 26.63%. The granulometric contrasts between the soils used in covering the Zakroczym and Zdounky landfill highlight their different orgins what may affect on cover stability.

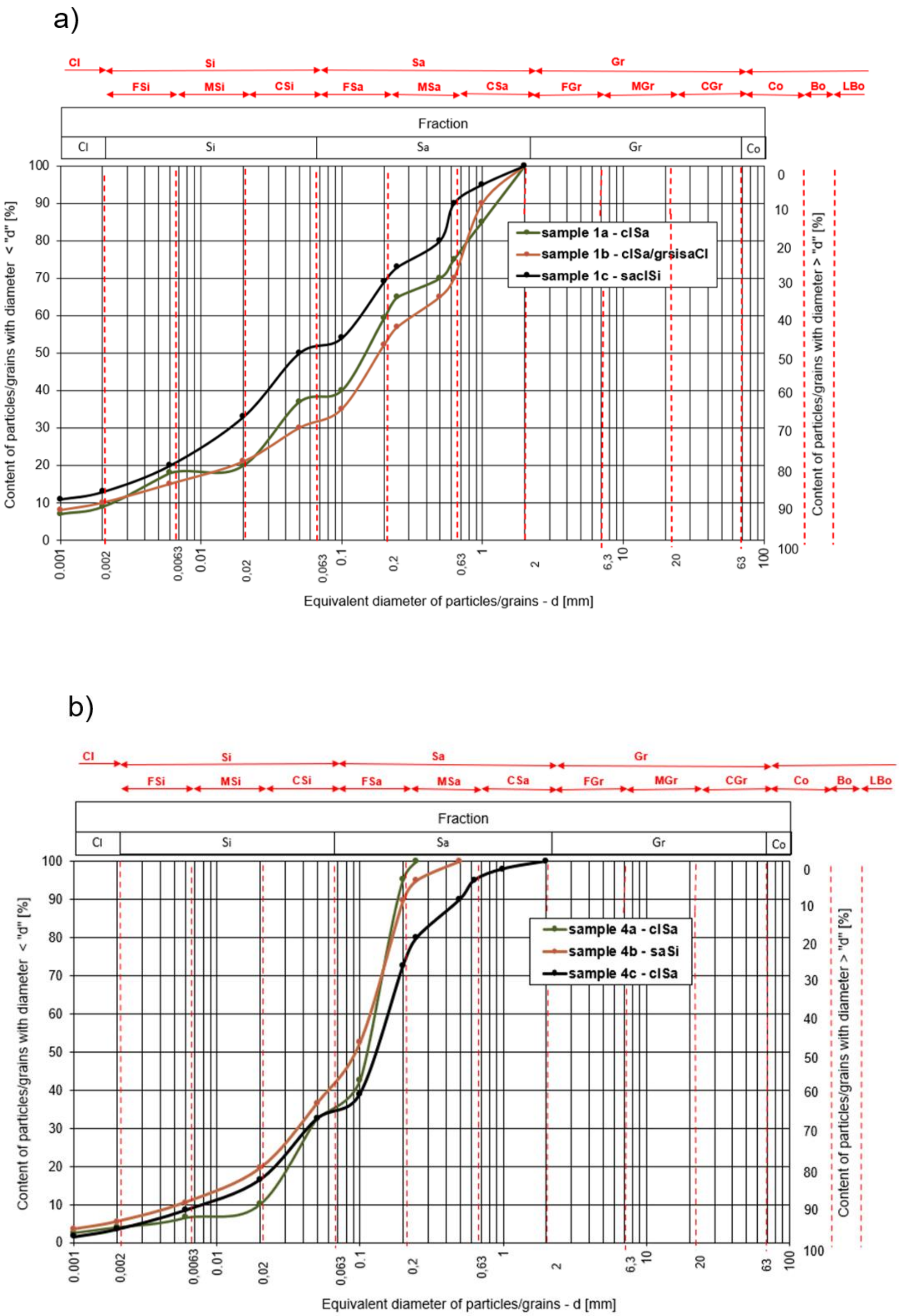
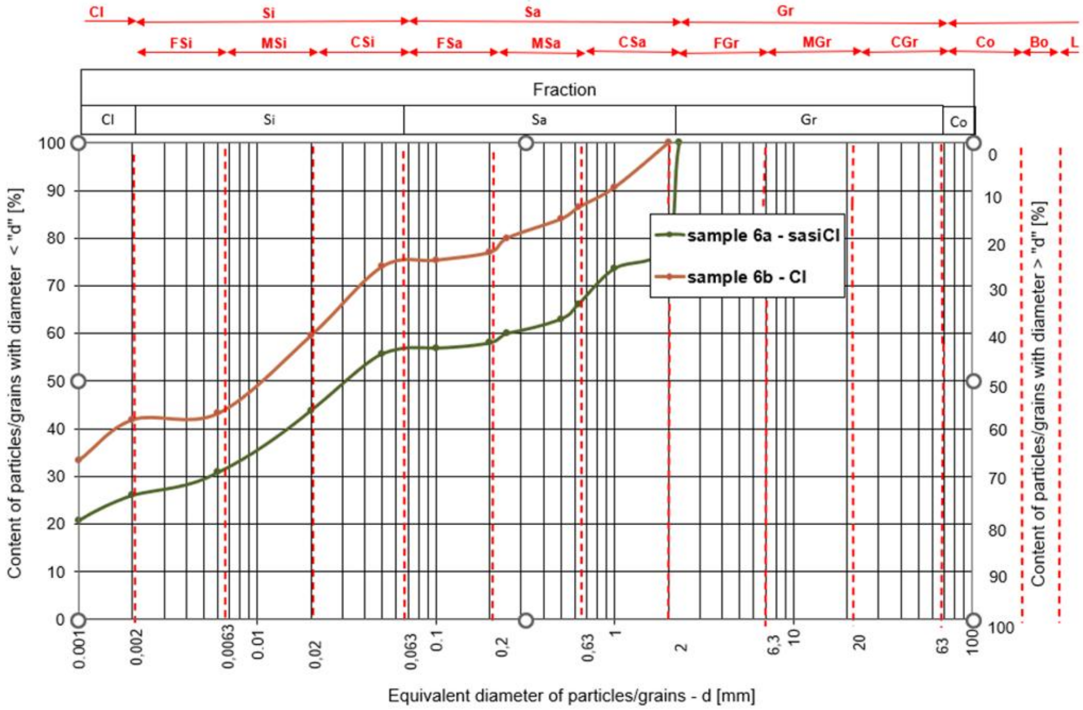


Figure 6.15. Grain-size distribution curves of soils: a) samples 1a, 1b, 1c, b) samples 4a, 4b, and 4c, collected from the landfill in Zakroczym.

a)



b)

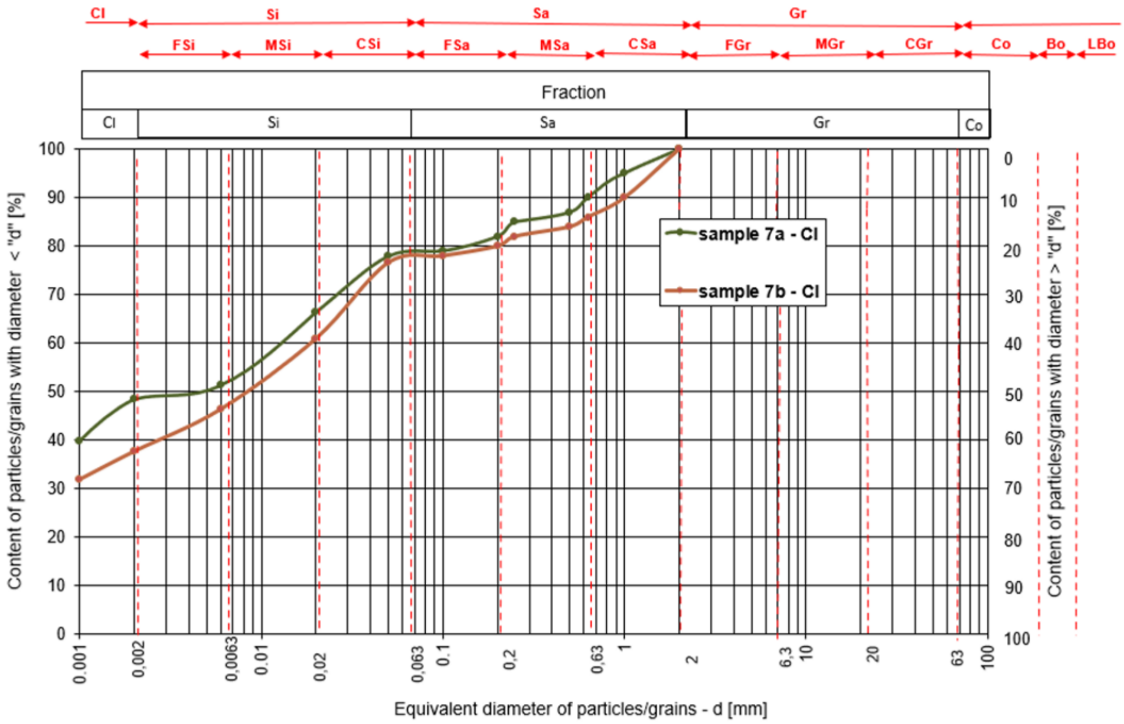


Figure 6.16. Grain size curves of soils a) 6a and 6b, b) 7a and 7b taken from Zdounky landfill site.

The remaining grain-size distribution curves for the tested soils have been attached in Appendix 2. The percentage contents of the individual fractions and classification of the tested soils, along with the sampling depths, are presented in Tab. 6.4.

Table 6.4. Types of soils studied along with the sample collection depths and percentage granulometric composition.

Sample	**Depth (m)**	**Soil type**	**Sa (%)**	**Si (%)**	**Cl (%)**	**Gr (%)**
Zakroczym landfill						
1a	0.1	clSa	63	28	9	0
1b	0.5	clSa/grsisaCl	70	20	10	0
1c	0.9	saclSi	50	37	13	0
2a	0.1	clSa	79.5	15.5	5	0
2b	0.5	clSa	77.5	16.5	6	0
2c	0.9	clSa	75	18.5	6.5	0
3b	0.5	clSa	82.5	12	5.5	0
4a	0.1	clSa	67.5	28.5	4	0
4b	0.5	saSi	63.5	31	5.5	0
4c	1	clSa	67.5	29	3.5	0
5a	0.1	clSa	75.37	17.5	4.38	2.75
5b	0.5	clSa	78.74	15.19	3.32	2.75
5c	1	clSa	73.5	22.5	4	0
6a	0.1	clSa	63	28	9	0
6b	0.5	clSa	70	20	10	0
Zdounky landfill						
1a	0.1	Cl	14.45	42.42	43.13	0
1b	0.5	SiCl	34.09	39.28	26.63	0
2a	0.1	sasiCl	30.35	43.28	26.37	0
2b	0.5	sasiCl	35.76	37.96	26.28	0
3a	0.1	Cl	40.10	18.83	41.07	0
3b	0.5	Cl	10.62	42.90	46.47	0
4a	0.1	Cl	22.27	39.21	38.51	0
4b	0.5	Cl	17.08	37.97	44.95	0
5a	0.1	Cl	21.57	44.82	33.61	0
5b	0.5	Cl	15.32	41.29	43.39	0
6a	0.1	sasiCl	44.29	34.97	20.74	0
6b	0.5	Cl	25.97	40.62	33.41	0
7a	0.1	Cl	22.12	38.07	39.80	0
7b	0.5	Cl	23.44	44.56	31.9	0
8a	0.1	sasiCl	38.04	32.19	29.76	0
8b	0.5	sasiCl	38.67	32.88	28.45	0

Tests on samples of cohesive soils taken from the investigated boreholes showed that the soils were mostly in a semi-solid or solid state ($I_L < 0$). Samples from the Zakroczym landfill have lower I_L ranges ($-3.57 \div 0.09$) than those from the Zdounky landfill ($-0.28 \div 0.06$) and a lower plasticity index, which classifies the Zakroczym soils as low plasticity ($1\% < I_p \leq 10\%$), whereas the Zdounky soils belong to medium plasticity ($20\% < I_p \leq 30\%$) and high plasticity ($30\% < I_p$) soils. There was also no correlation between the cohesiveness of the soils and the depth of the soils sampled at Zdounky. It was also not possible to verify the relationship in the samples taken from the Zakroczym landfill because the sample was too small, and the plasticity were determined only for the soil from the first level (at a depth of 0.1 m). The lack of a comprehensive determination of soil conditions for the remaining samples was due to the inability to determine the plasticity index of the soil owing to the non-cohesive nature of the sampled material.

A study on the physical properties of the soil also examined the organic matter content. The soils sampled from the Zdounky landfill had a fairly constant organic matter content, ranging from 2.10% to 5.91%, which can be classified as low organic matter according to PN-EN ISO 14688-2:2006. In contrast, the soils sampled from the Zakroczym landfill had a wider range of organic matter content (LOI = 0–9.5%), which indicates a more varied cover layer. The highest values were observed in samples from the deepest locations (approximately 1 m below sea level), which may also be related to the mixing of the soil with the deposited waste material, which contained an organic fraction in its morphological composition. However, owing to the small sample size ($n = 6$), no firm conclusions can be drawn, and further analysis is required. Tab. 6.5 shows the results of laboratory tests to determine the organic matter content (LOI), natural moisture content (w_n), consistency limits (w_p and w_L) and corelated indexes (I_p and I_L).

Overall, the cover soils at Zakroczym landfill have lower plasticity, whereas those at Zdounky display medium-to-high plasticity coupled with a more uniform organic-matter content. These contrasts point to different engineering behaviours of the two landfill cover systems and highlight the need for more extensive sampling—especially at Zakroczym—to confirm the observed variability. The different plasticity of the soils used in covering the Zakroczym and Zdounky landfills underscore their different origins and indicate the different properties affecting stability for each landfill cover.

Table 6.5. Organic matter content of soil samples tested and consistency limits.

Sample	Depth [m]	LOI (%)	w_n (%)	w_p (%)	w_L (%)	I_p (%)	I_L (%)
Zakroczym landfill							
1a	0.1		11.48				
1b	0.5		10.91				
1c	0.9	1.36	14.39	13.65	22	8.35	0.09
2a	0.1	1.72	9.03	15.40	19.4	4	-1.59
2b	0.5	0.0	27.13				
2c	0.9	1.94	7.21	13.53	17.5	3.97	-1.59
3b	0.5		10.53				
4a	0.1		8.01				
4b	0.5		13.84				
4c	1	6.0	19.53	21.66	25.5	3.84	-1
5a	0.1		3.28				
5b	0.5		3.98				
5c	1	9.5	14.91	24.12	26.7	2.58	-3.57
6a	0.1		11.48				
6b	0.5		10.91				
Zdounky landfill							
1a	0.1	5.91	28.80	30.11	59.5	29.39	-0.04
1b	0.5	3.98	22.27	26.88	50	23.12	-0.20
2a	0.1	5.34	24.50	24.44	54	29.56	0.00
2b	0.5	3.69	16.70	25.57	57	31.43	-0.28
3a	0.1	-	26.93	31.42	70	38.58	-0.12
3b	0.5	4.11	25.97	29.20	63	33.8	-0.10
4a	0.1	3.68	29.70	27.52	63.8	36.28	0.06
4b	0.5	3.04	24.53	23.50	56	32.5	0.03
5a	0.1	4.87	26.65	26.45	59.80	33.35	0.01
5b	0.5	5.62	24.29	25.83	55.1	29.27	-0.05
6a	0.1	5.06	27.50	28.84	52.5	23.66	-0.06
6b	0.5	5.64	21.46	22.16	53	30.84	-0.02
7a	0.1	4.00	26.90	27.14	60.3	33.16	-0.01
7b	0.5	2.10	18.98	20.78	55	34.22	-0.05
8a	0.1	5.62	28.01	30.75	61.5	30.75	-0.09
8b	0.5	4.36	24.34	27.66	57.2	29.54	-0.11

6.2.2. Chemical properties of the tested soils

Soils used for reclamation purposes on the Zdounky landfill were slightly alkaline with a pH range from 7.04 to 7.53 according to pH scale (Hartemink and Barrow, 2023). No significant deviation from the average of pH = 7.32 was observed at any of the sample tested. In contrast, the soils used for the reclamation of the Zakroczym landfill have a wider range of pH values from 7.34 to 8.2 with an average of 7.81 (Fig. 6.17). The observed pH values cause the soil reaction to vary between slightly alkaline (pH = 7–8) and moderately alkaline (pH = 8–9).

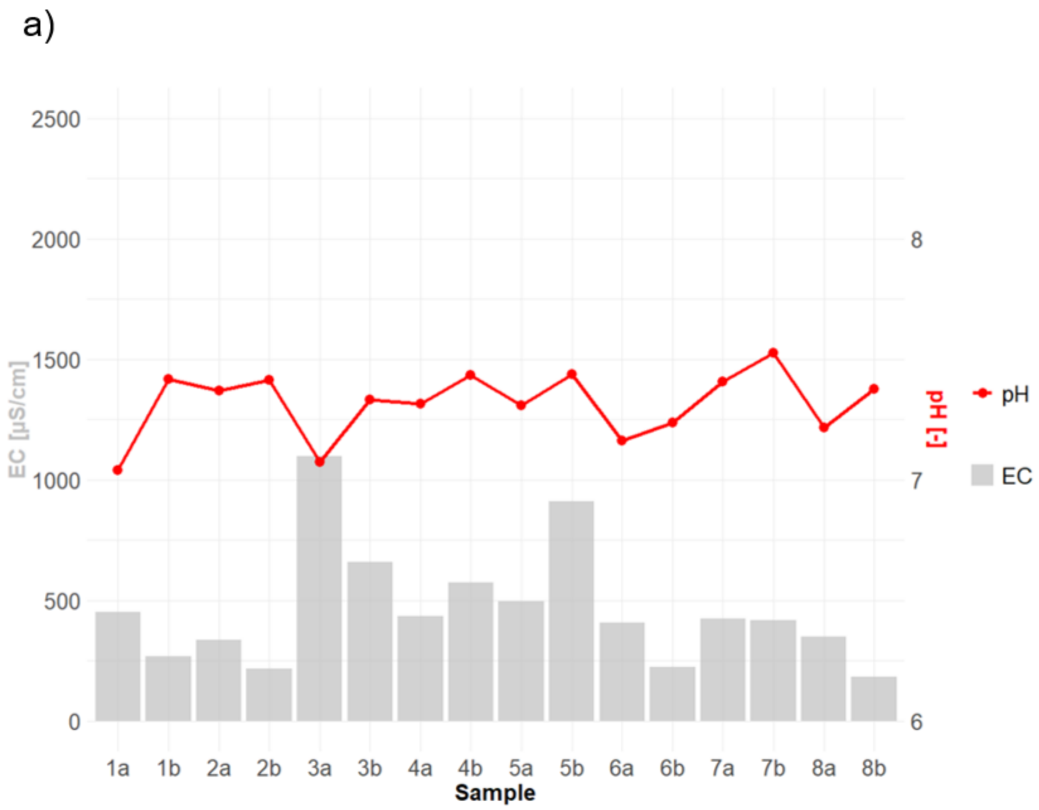
It should be noted that soil samples taken from greater depths in both landfills had a higher pH than those taken from 0.1–0.2 m a.s.l. This is consistent with the study by Lee et al. (2022), who examined waste samples taken from depths of 3–9 m, 23–30.5 m and 48–55 m and found the highest pH for the deepest sample taken. Similar conclusions were reached by Choudhury et al. (2021), who estimated the pH of the surface layer of landfill soil at all sites to be 5.6 ± 0.7, while the pH of soil samples taken at 0.15 m and 0.30 m depths was 6.0 ± 0.7 and 6.2 ± 0.7, respectively. According to Hartemik and Barrow (2023), a pH range of 7–8 achieved for the Zdounky landfill has a positive effect on the availability of nutrients such as calcium (Ca), potassium (K), nitrates (NO_3^-), magnesium (Mg), and phosphates (PO_4^{3-}), with the most significant effect of this range being observed for the assimilability of Ca. However, it should be noted that for most crops, a soil pH of 6 to 7.5 is optimal (NRCS, 2022). This is extremely important for maintaining the vegetation in reclaimed landfills. It is well known that pH plays a key role in the accumulation of metal(loid)s. It is also important to note that pH and EC heights affect the absorption of nutrients by plants.

The EC of the soils used for reclamation was measured in all samples taken to determine the possible effects of salinity. The average EC of the soils taken from the reclaimed areas of the Zdounky landfill was 465.27 μs/cm. The lowest EC of 182.22 μs/cm was recorded for soil sample 8b, while the highest of 1096.67 μs/cm was recorded for sample 3a. The average EC for the Zakroczym landfill was 666.17 μs/cm. The lowest EC of 122.18 μs/cm was measured for sample 2b, while the highest EC of 1987.3 μs/cm was measured for sample 4b. According to the NRCS classification (2022), soils with EC < 2000 μs/cm are considered as non-saline. Individual plant species will respond differently to salinity. Ornamental and vegetable plants are known to be more sensitive to saline conditions, while most grain crops and turfgrasses are considered more tolerant to salinity (Liu et al., 2023). The optimal EC value is crop specific and depends on environmental conditions, however, it is assumed that higher EC hinders nutrient uptake by increasing the osmotic pressure of the nutrient solution, wastes nutrients and increases the release of nutrients into the environment, causing environmental pollution (Ding et al., 2018).

Most vegetables show high productivity when grown at EC ranging from 1000 to 2500 μS/cm (Carmo et al., 2024), whereas some grasses are able to thrive in the range of 700-1190 μS/cm (Tola et al., 2017). According to Harivandi et al. (1992) turfgrass salinity tolerance, species such as *Poa annua*, *Agrostis capillaris*, *Poa pratensis*, *Poa trivialis* and *Eremochloa ophiuroides* are classed as sensitive, tolerating EC up to about 3 000 μS/cm. Field data from the two studied landfills show that the maximum EC in the cover-soil never exceeded 2 000 μS/cm, which is well below the 3 000 μS/cm threshold for even the most salt-sensitive turf species.

Considering the above, it can be concluded that the level of soil salinity at the studied landfills definitely does not affect vegetation, which is important when planning reclamation cover. Considering the results achieved, there was no obvious relationship between the recorded pH values and EC for the two studied landfills, however, significant differences in EC were observed for the shallowest and deepest layers, with higher values recorded in the deepest samples. However, at the Zdounky landfill, the above trend was not observed, which may be related to the lack of impact of the landfilled waste due to the effective GM cover. Husson et al. (2018) proved that EC is correlated with soil texture, especially Cl and organic matter content (the higher the content, the higher the EC).

Fig. 6.19a shows the Pb content of the soil used for the reclamation of the Zakroczym landfill, where the values range from 9.05 mg/kg d.m. to 108.83 mg/kg d.m., with an average of 31.61 mg/kg d.m. The difference between the minimum and maximum Pb values was 99.78 mg/kg d.m., indicating a high variability between samples, especially for samples 5a, 5b, and 5c, which had the highest values. However, all samples were below the established threshold of 200 mg/kg d.m. Fig. 6.19b shows the Pb content of the soil from the Zdounky landfill, where the values are much lower, ranging from 7.17 mg/kg d.m. to 15.65 mg/kg d.m., with an average of 11.81 mg/kg d.m., with an average of 11.81 mg/kg d.m. and a difference of only 8.48 mg/kg d.m. Compared to Zakroczym, the soils in Zdounky showed a much lower Pb content and less variability between samples, which may indicate their lower environmental impact.



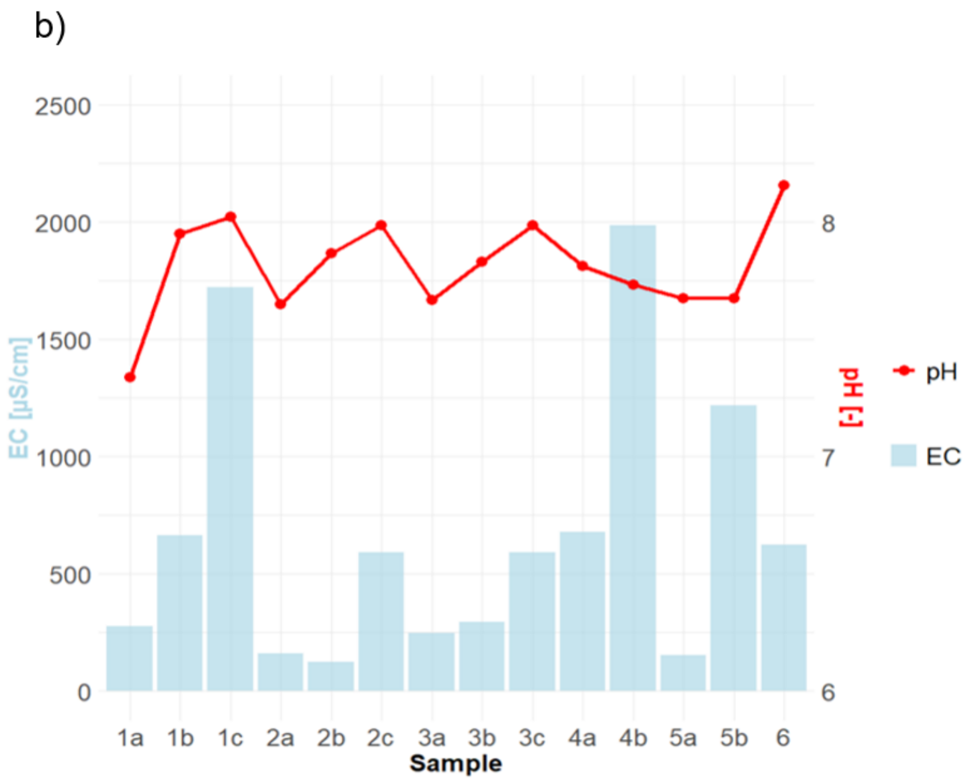


Figure 6.17. EC and pH of samples taken from landfills: a) Zdounky landfill, b) Zakroczym landfill.

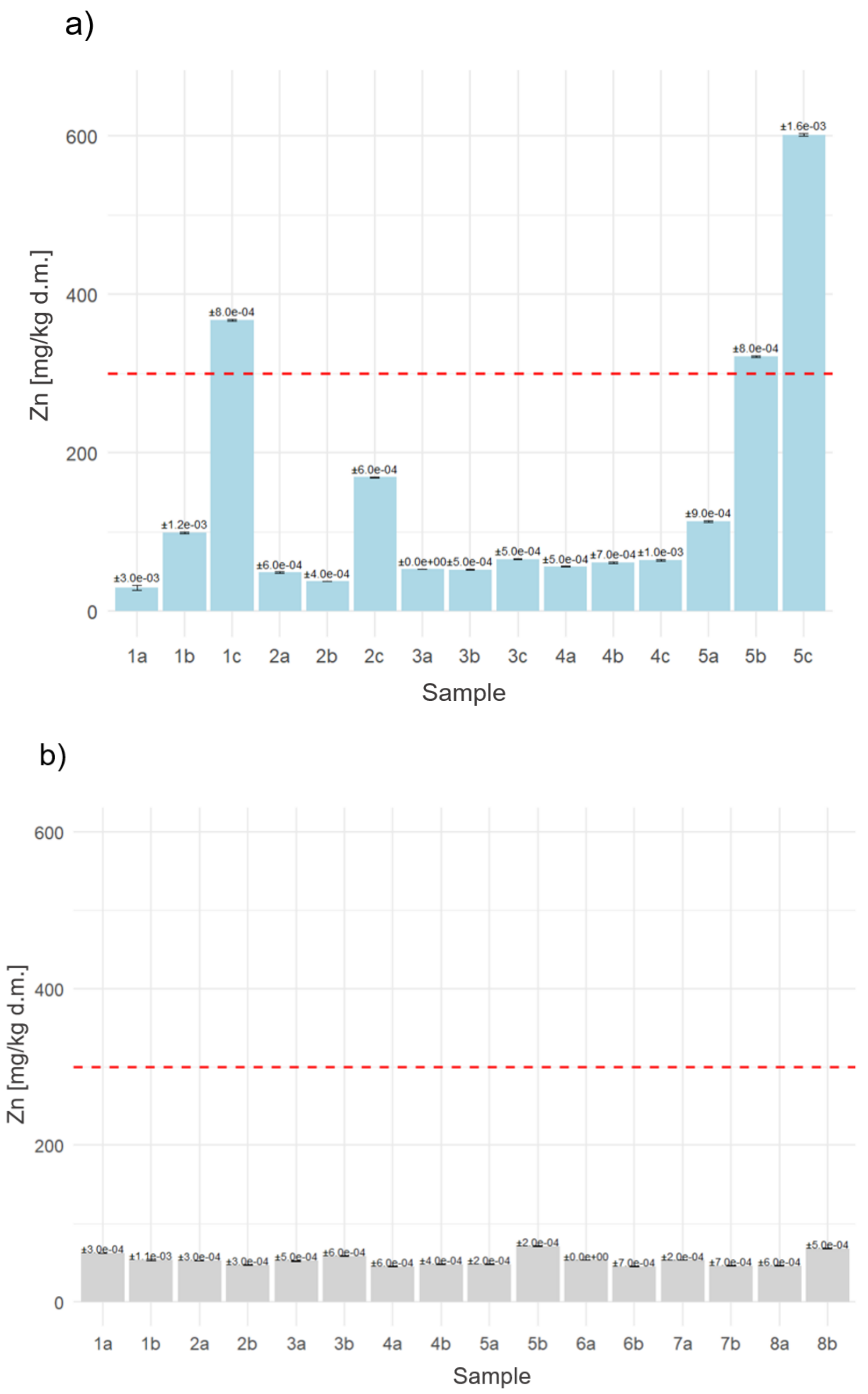


Figure 6.18. Zn concentration in the tested samples: a) Zakroczym landfill (Zn characteristic concentration = 0.0103), b) Zdounky landfill (Zn characteristic concentration = 0.0387).

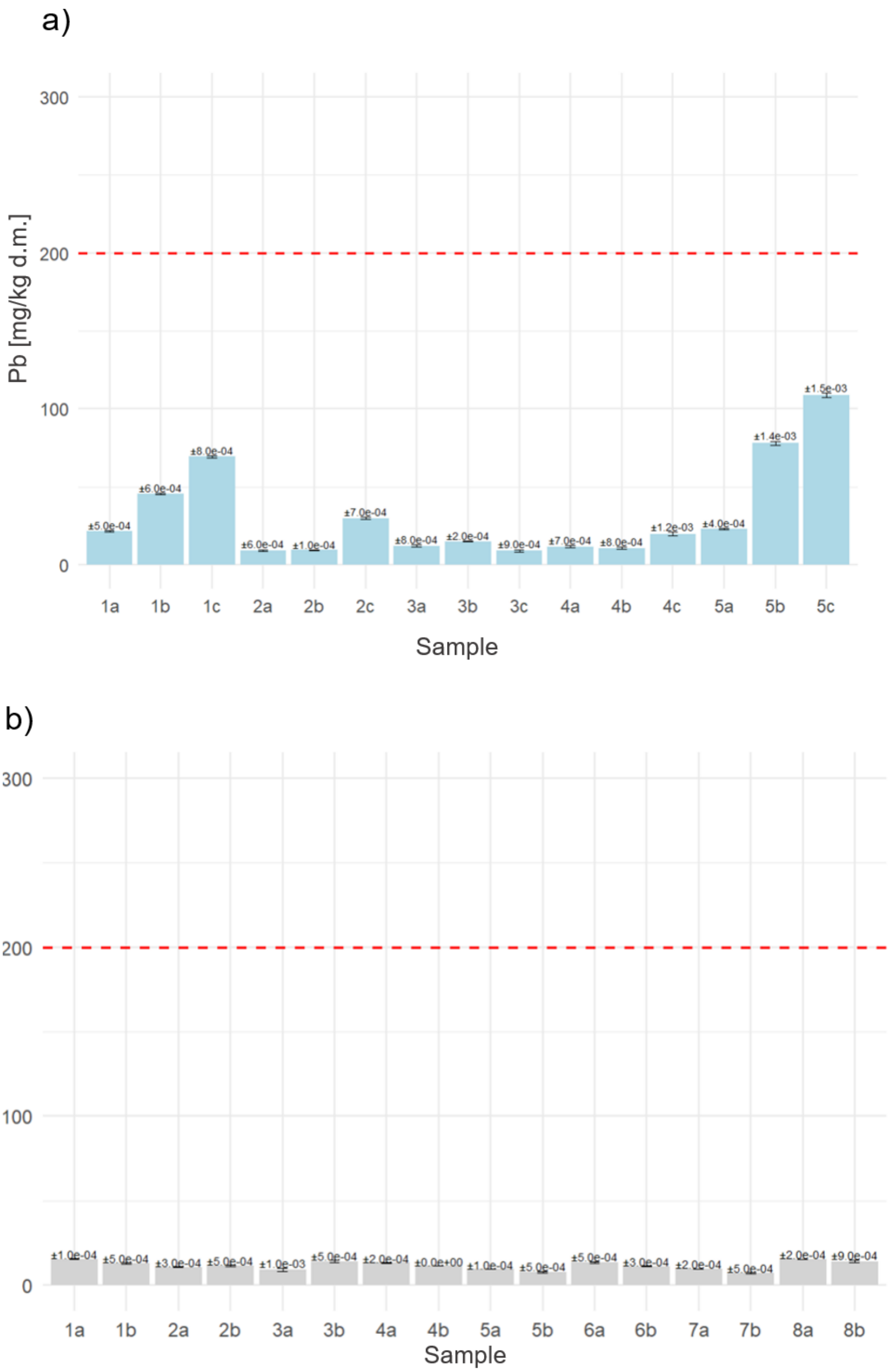


Figure 6.19. Pb concentration in the soil: a) from Zakroczym landfill (Pb characteristic concentration = 0.0576), b) from Zdounky landfill (Pb characteristic concentration = 0.063).

Fig. 6.20a shows the concentration of Cu, which in soil samples at the Zakroczym landfill ranges from 7.9 mg/kg d.m. to 159.86 mg/kg d.m., with an average value of 30.91 mg/kg d.m. and a difference between the minimum and maximum values equal to 151.96 mg/kg d.m., indicating significant soil heterogeneity. The highest Cu concentrations were observed in sample 1c, and samples 5b and 5c, which showed significantly elevated values compared to the other samples. This may be related to contact with landfilled waste, which may also indicate that the landfill cover was not of equal thickness in all places. Nevertheless, the Cu content in all soil samples remained below the permissible level of 200 mg/kg d.m.

In Zdounky (Fig 6.20b), the Cu content ranges from 19.66 mg/kg d.m. to 51.47 mg/kg d.m., with an average of 29.50 mg/kg d.m. and a difference between the minimum and maximum values of 31.81 mg/kg d.m. Such a slight difference indicates a more uniform soil chemical structure in terms of Cu concentration compared with the samples from Zakroczym. In none of the samples from Zdounky, the Cu content exceeded the threshold of 200 mg/kg d.m., which, as in the case of Zakroczym, suggests that the soil met the standards for Cu content.

Fig. 6.21a shows the Ni content of the soil in Zakroczym, which ranges from 5.39 mg/kg d.m. to 24.16 mg/kg d.m., with an average of 10.03 mg/kg d.m. and low variability. All samples were below the limit concentration of 100 mg/kg. Nevertheless, as with Zn, the highest concentrations were recorded in samples from the deepest layers.

In contrast, Ni content was higher in Zdounky (Fig. 6.21b) with values ranging from 23.35 mg/kg d.m. to 66.2 mg/kg d.m., with an average of 36.92 mg/kg d.m., but also below the permissible limit. This is an anomaly, as the concentrations of all other HMs were lower at the Zdounky landfill than at the Zakroczym landfill. This may be due to the weaker phytoremediation potential of plants for Ni and their ability to absorb and store metals in vegetative systems (Borah et al., 2023). Another rationale may be the correlation with the clay fraction content of the soil, as evidenced by the study of Hamner et al. (2013), who showed that there is a strong positive correlation between the proportion of clay fraction and Ni concentration in the soil, causing clay soils to accumulate more Ni than clayey sands of the same origin. This mechanism is due to the greater sorption area of clay minerals, high exchange capacity and Ni-binding Fe/Mn oxides.

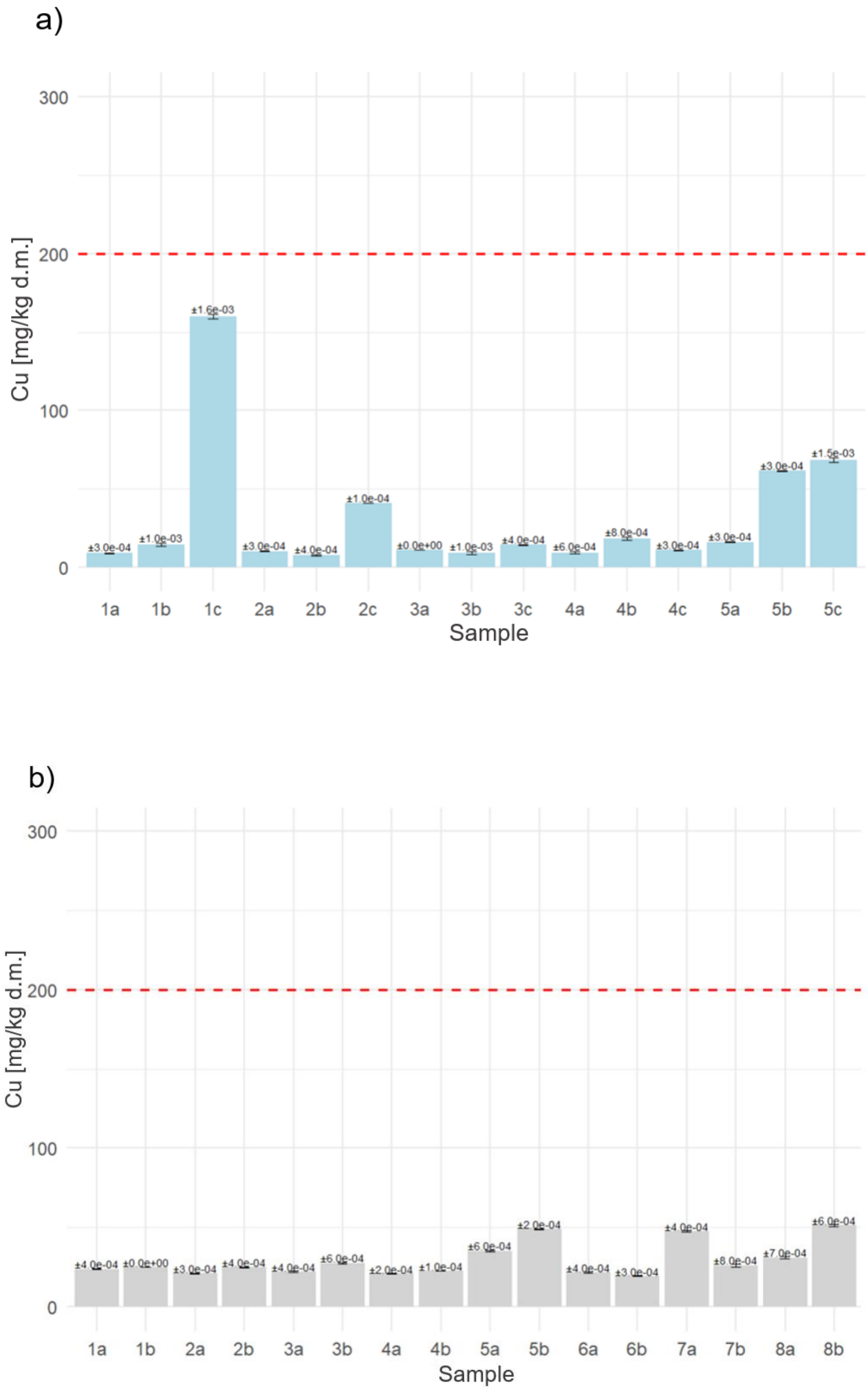


Figure 6.20. Cu concentration in the soil: a) from Zakroczym landfill (Cu characteristic concentration = 0.0103), b) Zdounky (Cu characteristic concentration = 0.0387).

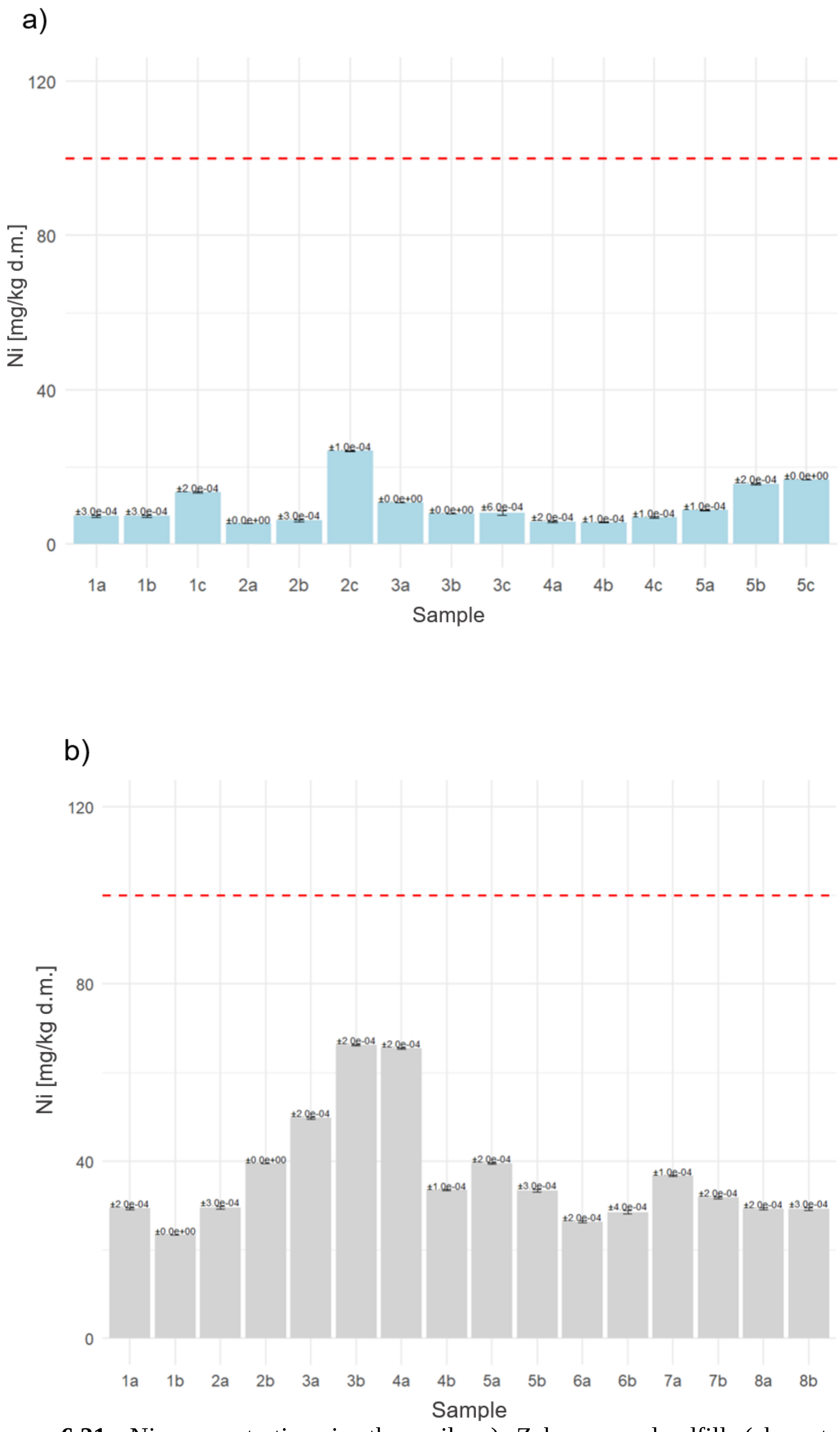


Figure 6.21. Ni concentration in the soil: a) Zakroczym landfill (characteristic concentration = 0.0508), b) Zdounky landfill (characteristic concentration = 0.0568).

In the case of Cd (Fig. 6.22), in Zakroczym the values range from 1.87 mg/kg d.m. to 3.25 mg/kg d.m., with an average of 2.09 mg/kg d.m., while in Zdounky they range from 0.07 mg/kg d.m. to 1.44 mg/kg d.m., with an average of 0.71 mg/kg d.m. The limit of quantification for Cd was higher (1.87 mg/kg d.m.) for the samples tested in Zakroczym hence the results appear to be overestimated compared to Zdounky where there was a lower detection limit and there were no measurements below the limit of quantification. Low concentrations and low variability of Cd at both locations suggest a limited influence of this contaminant from municipal waste morphology. All measured Cd levels are also far below the limit value of 6 mg/kg d.m. However, the divergent limits of quantification highlight the importance of harmonising analytical protocols when comparing datasets from different laboratories. Follow-up monitoring, using identical detection thresholds, is therefore recommended to confirm whether the present depth-related trends persist. Cd carries the strictest soil guideline of all the metals considered—only 6 mg/kg d.m. in Polish and many EU regulations, versus 100 mg/kg for Ni or 200 mg kg for Cu—because of its high toxicity, carcinogenic chatactet and decades-long residence time in the human body (Charkiewicz et al, 2023). Consequently, even the sub-3 mg/kg levels recorded at both landfills deserve ongoing scrutiny to ensure they never approach the action threshold.

Based on the above analysis, the soil used for reclamation of the Zdounky landfill has a lower HMs content than the soil used for reclamation of the Zakroczym landfill. In addition, it was noted that the concentrations of HMs increased with depth at the landfill in Zakroczym, which may indicate the inhomogeneity of the cover, which affects closer contact with the landfilled waste, consequently contributing to an increase in the concentration of HMs. Another reason may be the occasional sprinkling of leachate on the reclaimed cell to reduce its load. The means of the observed concentrations of HMs for the Zakroczym landfill varied considerably with depth, while at the Zdounky landfill, regardless of depth, the mean HMs concentrations remained close to each other, confirming the influence of sampling depth on the HMs content in the Zakroczym landfill. Changes in the observed concentrations with depth are shown in Fig. 6.23 for Zakroczym and Fig. 6.24 for Zdounky below.

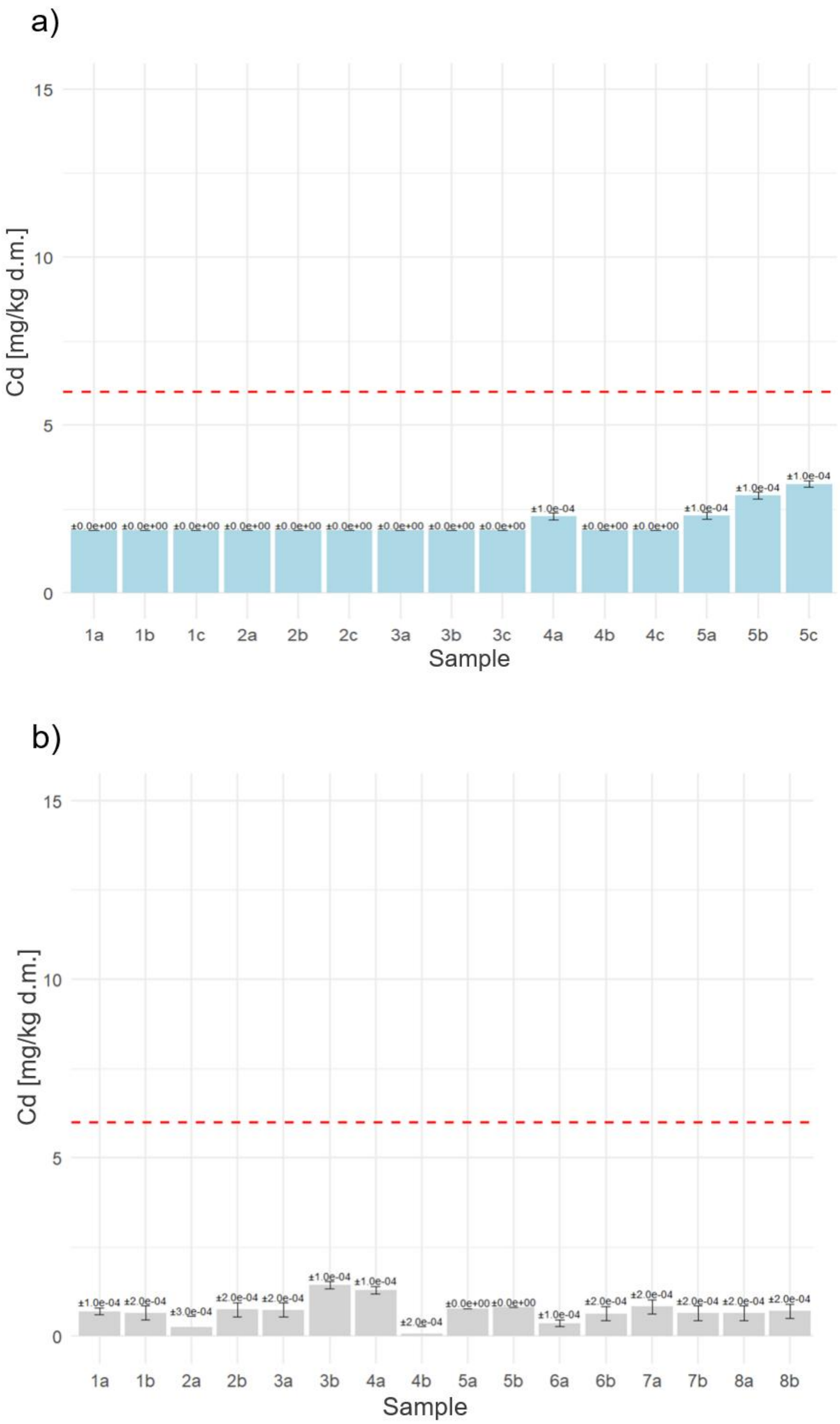


Figure 6.22. Cd concentration in the soil: a) Zakroczym landfill (characteristic concentration = 0.0508), b) Zdounky landfill (characteristic concentration = 0.0568).

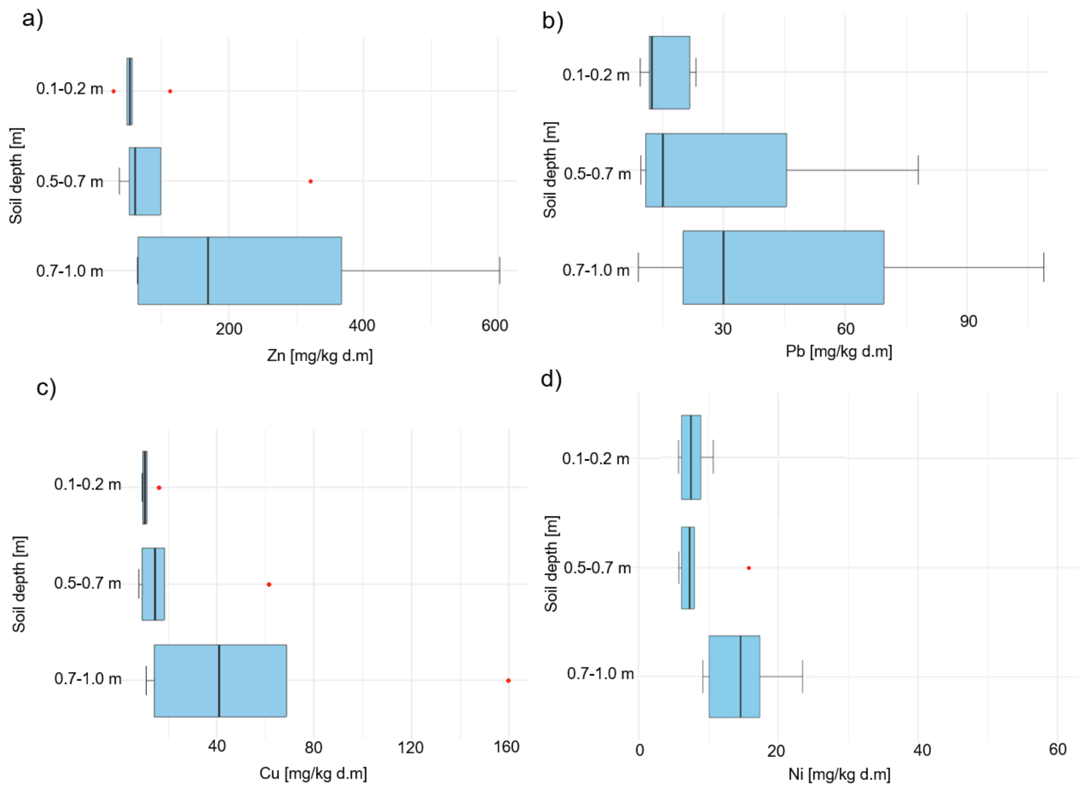


Figure 6.23. Concentrations of selected HMs depending on depth of samples taken from the reclaimed landfill in Zakroczym: a) Zn, b) Pb, c) Cu, d) Ni.

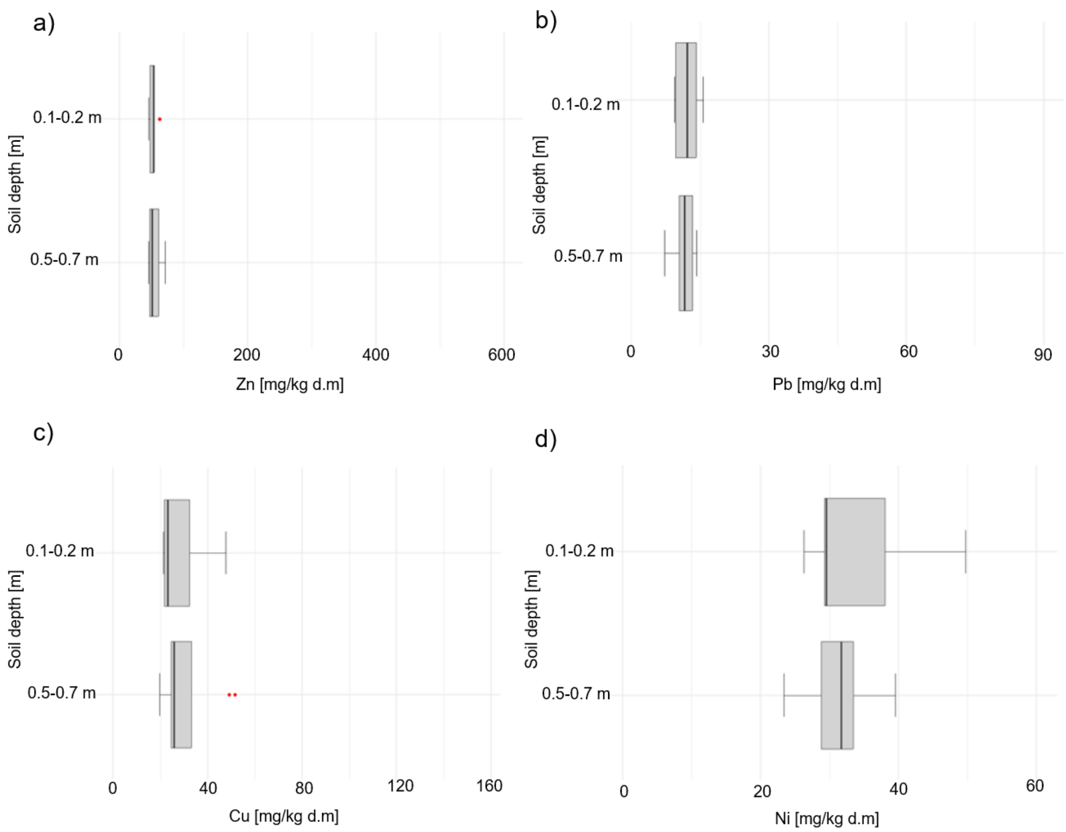


Figure 6.24. Concentrations of selected HMs depending on depth of samples taken from the reclaimed landfill in Zdounky: a) Zn, b) Pb, c) Cu, d) Ni.

6.2.3. Summary of physicochemical test results

Analysis of soils from reclaimed landfills in Zdounky and Zakroczym have revealed significant differences in their physical and chemical properties. In Zdounky, cohesive soils in the form of Cl and sasiCl predominated, with Sa fraction of up to 44%, Si fraction of up to 45%, and Cl fraction of up to 46%. In Zakroczym, on the other hand, ClSa predominate, with a Sa fraction of up to 82.5% and a low Cl fraction (maximum 10%). These differences are also reflected in the plasticity and consistency, where the soils from Zakroczym are classified as low plasticity ($I_p \leq 10\%$), whereas those from Zdounky are classified as medium plasticity and high plasticity ($I_p > 20\%$). At the same time, it should be emphasized that the high proportion of clay fraction observed in Zdounky soils promotes stronger sorption of HMs (Chalermyanont et al., 2009), which is also confirmed by the recent results of Rebi et al. (2024), where soils of a more clayey nature showed a higher adsorption capacity than sandy loam soils.

The pH measurement showed a range from slightly alkaline (pH ≈ 7) to moderately alkaline (pH ≈ 8–9) at the landfill in Zakroczym and slightly alkaline at the landfill in Zdounky, with a tendency to increase with depth. Lee et al. (2022) indicate that pH can affect the mobility and concentration of HMs in soils, with lower pH resulting in a greater release of HMs. Nevertheless, the conducted studies did not detect this trend, possibly due to the small differences in pH between the samples, which in all cases ranged from pH 7 to pH 8. The EC of the soils from Zdounky averaged 465.27 µs/cm, and at Zakroczym 666.17 µs/cm. In both cases, these values do not indicate salinity levels that could restrict vegetation growth. Analyses of HMs content (Zn, Pb, Cu, Ni, Cd) in the cover layer generally revealed lower and more homogeneous concentrations at Zdounky, whereas at Zakroczym there were greater fluctuations and occasional exceedances of standards (mainly in the case of Zn). HMs concentrations in the soils from the Zakroczym landfill are arranged in the following order: Zn > Pb > Cu > Ni > Cd, while at Zdounky: Zn > Ni > Cu > Pb > Cd. It is noted that at both studied landfills the greatest concern is the concentration of Zn in the soil, while Cd has the lowest impact, with its concentrations being below the detection limit (in the case of the Zakroczym landfill) or very low in case of Zdounky landfill.

Average concentrations of HMs in soils in both locations remain below acceptable limits, although samples from Zakroczym exhibit greater variability and local exceedances in the deepest layers sampled, which may indicate insufficient isolation of the reclamation layer from the waste layer. Additionally, at the Zakroczym landfill a trend was observed in which HMs concentrations increased with depth, a trend that was not observed at the Zdounky landfill. This is consistent with the observations of Makuleke and Ngole-Jeme (2020) and Wu et al. (2022), who reported that samples taken from greater depths often exhibit the highest contamination levels and, consequently, the highest environmental risk. These studies underscore the necessity of regular soil quality monitoring to enable a prompt response in the event of significant exceedances and the emergence of risks associated with negative environmental impacts.

6.3. Modelling studies

In the following chapter, simulations of leachate production under near-real conditions were performed using the UnsatSuite +HELP software, and CH_4 and CO_2 emissions were simulated from to 1997-2137 using the LandGEM model. Furthermore, the slope stability of the studied landfills was evaluated.

6.3.1. Leachate generation using HELP model

Fig. 6.25 shows bar charts illustrating the distribution of the main components of the water balance (precipitation, ET, and surface runoff) on an annual basis as well as the daily peak for six scenarios representing the cells at the examined landfills. This study indicates that precipitation has the greatest influence on the water balance of the landfill, which is also supported by the findings of Krause et al. (2023), Beck-Broichsitter et al. (2018) and Podlasek (2023). It was also observed that there was no surface runoff for the operated cells (Scenarios 2 and 3, and 5 and 6). In the case of the reclaimed cell in Zakroczym (Scenario 1), a smaller surface runoff (3.59 mm) was observed compared to the landfill in Zdounky (Scenario 4), where the surface runoff amounted to 6.12 mm. The greater runoff in Zakroczym is attributed to the clayey sand in the landfill cover, which infiltrates the substrate more easily, thereby reducing the volume of runoff compared to the clay layer present at the Zdounky landfill. Another reason may be the lack of an impermeable cover, which effectively limits the infiltration of precipitation into stored waste, similar to the GM used at the Zdounky landfill. From a practical point of view, clay have a high-water retention capacity, however, they are characterized by low permeability and limited gas exchange, which can potentially result in less effective plant transpiration. In contrast, clayey sands have better permeability and may therefore lead to higher ET than clay.

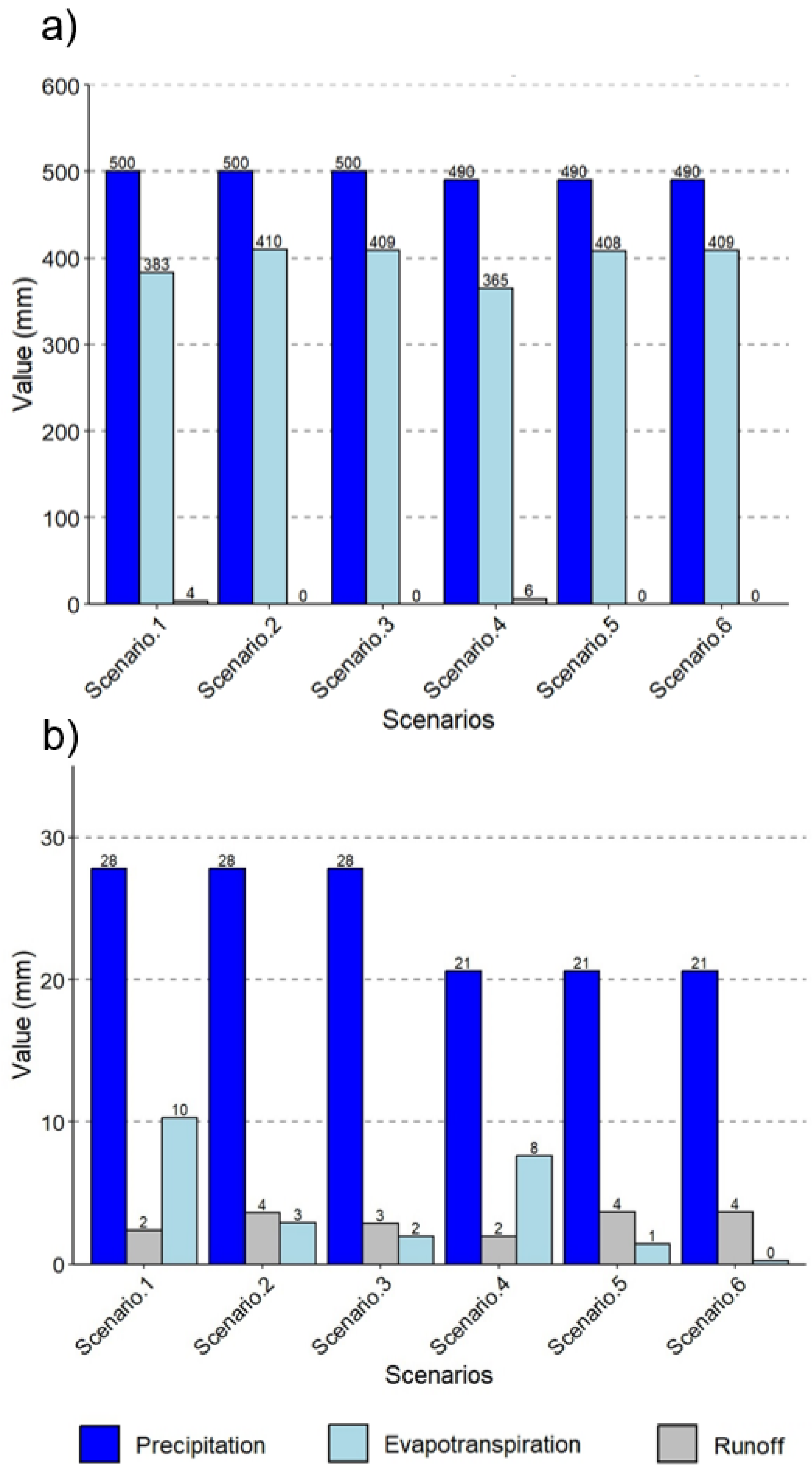


Figure 6.25. The distribution of water balance components in: a) 1-year period of analysis, b) daily peak in 1-year period of analysis.

It has also been found that the amount of water stored in the soil is affected not only by the amount of precipitation, but also by the type and thickness of the soil. Various additives, including additives such as biochar, are often used in landfill covers to make them more effective in preventing infiltration and percolation (Chen et al., 2022). A survey of landfills shows that the highest water holding capacity is in reclaimed landfills (Fig. 6.26), where this is evident in Scenarios 1 (6045.9 mm) and 4 (4963.2 mm). In contrast, the lowest values are observed in Scenarios 2 and 6, which are 2579.6 mm and 2388.5 mm, respectively, due to the lower thickness of the stored MSW compared to the other scenarios.

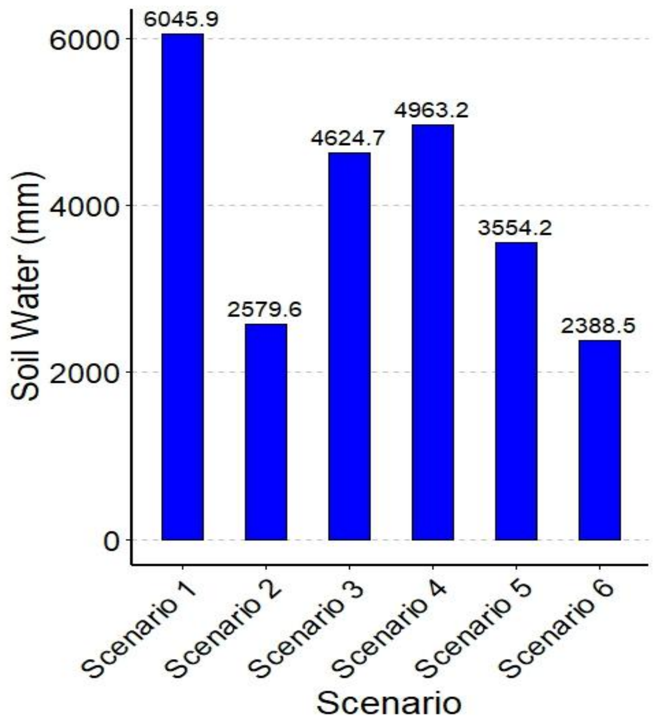


Figure 6.26. Water accumulation in soil according different scenarios.

A Pearson correlation matrix was created to identify significant correlations among the landfill water balance components. The analysis revealed the following significant correlations among the landfill water balance components:

1. Runoff and ET *(r* = -1.0), very strong negative correlation,
2. Runoff and bottom drainage (*r* = -0.97), very strong negative correlation,
3. Runoff and change in water storage *(r* = 1.0), very strong positive correlation,
4. ET and change in water storage (*r* = -1.0), very strong negative correlation,
5. ET and bottom drainage (*r* = -0.97), very strong negative correlation.

These results suggest strong interdependencies between specific components of the water balance, particularly runoff, ET, and water storage changes, which are tightly interconnected. High ET reduces runoff, as indicated by a strong negative correlation (r = -1.0), because more water evaporates and less is left for runoff. At the same time, increased runoff reduces the amount of water infiltrating the drainage system, resulting in a very strong negative correlation (r = -0.97). When the water storage in the active layer increased (change in water storage), a concomitant increase in runoff was observed (r = 1.0). Increased ET reduced both water storage (r = -1.0) and the amount of water entering the drainage (r = -0.97). Ultimately, all elements of the water budget compete for the same water, and correlation values close to ±1 indicate a strong coupling in the model. However, it should be noted that there was no strong relationship between precipitation and soil drainage (r = 0.38), but this could be due to the design of the scenarios used in the analysis. Scenarios 1–3 had the same precipitation levels as Scenarios 4–6 due to the use of 1-year simulations for both landfills. This limited the variability of the precipitation data, with the correlation only reflecting differences between the two groups of landfills where the precipitation levels varied. Fig. 6.27 graphically shows the correlation matrix of the landfill water balance along with the calculated correlations.

These findings align with earlier research emphasizing the critical role of meteorological and hydrological components in leachate production at landfills. Pazoki and Ghasemzadeh (2020) found that the amount of precipitation, surface runoff, evapotranspiration, and infiltration have a strong influence on leachate production in landfills. Similary, Yu et al. (2021) showed that precipitation impacts leachate quality by increasing the concentration of pharmaceuticals and personal care products in leachate, particularly during the wet season. In analyses related to estimating leachate generation at waste landfills, it is common practice to express leachate production as a percentage of annual rainfall. The amount of leachate produced at operating landfills is site-dependent and ranges from 0 in arid conditions to almost 100% of rainfall in humid climates (Petchsri et al., 2006). Choden et al. (2022) showed in their study at a landfill in Bhutan that landfill leachate is equivalent to 30.13% of precipitation. Similar results were obtained by Frikha et al. (2017), where modelling in HELP showed that leachate generation at a landfill in Tunisia is equal to 30–40% precipitation. In a study of leachate production from MSW landfill, Choden et al. (2022) also showed that the amount of leachate can exceed 75% of the precipitation during the active phase of a landfill and that this proportion can fall below 10% during the closed phase, depending on the safeguards of the landfill.

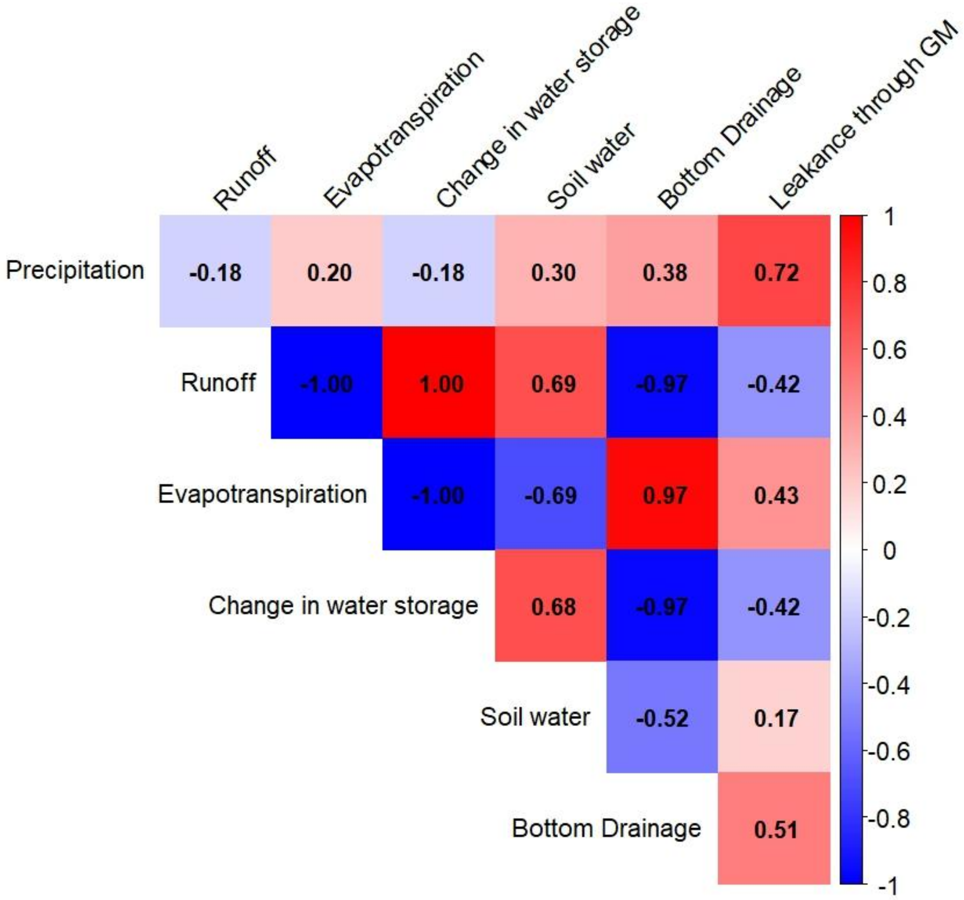


Figure 6.27. Correlation matrix of landfill water balance.

In this study, the amount of leachate produced at the Zakroczym landfill (Scenarios 1-3 with one reclaimed and two active cells) was equal to 45.76% of the precipitation (228.80 mm), while at the Zdounky landfill (Scenarios 4-6 with one reclaimed and two active cells), the amount of leachate produced was 37.31% (182.82 mm). The low leachate production at the Zdounky landfill is most likely the result of a tight cover in the form of a GM installed on stage 1 reclaimed cell, which effectively prevents the infiltration of rainwater as it is collected in a separate rainwater tank. Considering the rainwater collected from the drainage over the GM at the Zdounky landfill, the collected leachate would amount to 41.29% of the precipitation. Jain et al. (2023) also found that the amount of leachate produced decreases with landfill age, as older facilities are likely to have covered and vegetated slopes to promote rainwater runoff and minimize rainfall infiltration into the landfill. An important element that can change the amount of leachate is the degree of waste compaction, which reduces the filtration rate (Hussein et al., 2023), thereby reducing leachate production (Beck-Broichsitter et al., 2020). Nevertheless, Ko et al. (2016) showed that compaction of decomposing MSW through the application of pressure loading can enhance or inhibit landfill processes. Therefore, the frequency and extent of the use of specific machinery for waste compaction must also be considered when comprehensively assessing leachate volume. Tchobanoglous et al. (1993) demonstrated that the density of uncompacted MSW in landfills is 100–150 kg/m^3, whereas the density of compacted MSW is in the range of 300–600 kg/m^3. HELP program assumes a uniform density of landfilled MSW of 312 kg/m^3, which, according to Tchobanoglous et al. (1993), classifies the waste after the compaction process with a compaction ratio in the range of 2:1.

For the 6 Scenarios of annual leachate production per hectare of landfill analysed in the study, it was shown that the lowest amount of leachate (141.2 m^3/year) was recorded for Scenario 4 (Fig. 6.28). It characterizes a reclaimed landfill site where a 1 mm impermeable HDPE GM cover was used, with a 1 m layer of clay on top, effectively blocking the effect of precipitation on leachate production. The cover layer is also often used to protect the base of the landfill, known as a geosynthetic clay liner (GCL), which has low hydraulic conductivity and effectively prevents the risk of leachate mixing with the groundwater (Özçoban et al., 2022; Stark and Newman, 2010). Comparing Scenarios 1 and 4, which analyzed the cells of landfills already reclaimed, it can be seen that the amount of leachate generated is more than four times higher (626.4 m^3) per 1 ha of landfill when using mineral cover than when comparing synthetic cover (141.2 m^3). However, when comparing the two scenarios, it should be noted that in Scenario 4, the rainfall that did not infiltrate from the cover layer was discharged into a separate reservoir through a separate rainwater drainage system, and its volume per 1 ha of landfill area was estimated at 166.5 m^3/year. In addition, it should be noted that the clay layer in the landfill cover allowed only 14% of the rainfall to enter the drainage above GM, which may explain the small amount of leachate entering the lower drainage below GM. In the other cells (scenarios 2, 3, 5, and 6), there were no significant differences in leachate production per ha of area. The increased amount of leachate observed in Scenarios 2 and 3 may be due to higher rainfall in the study area. Considering the operating and closed areas, the Zakroczym landfill (Scenarios 1–3) produces an average of 815 m^3 of leachate per ha/year, while the Zdounky landfill (Scenarios 4–6) produces 603 m^3 of leachate per ha/year.

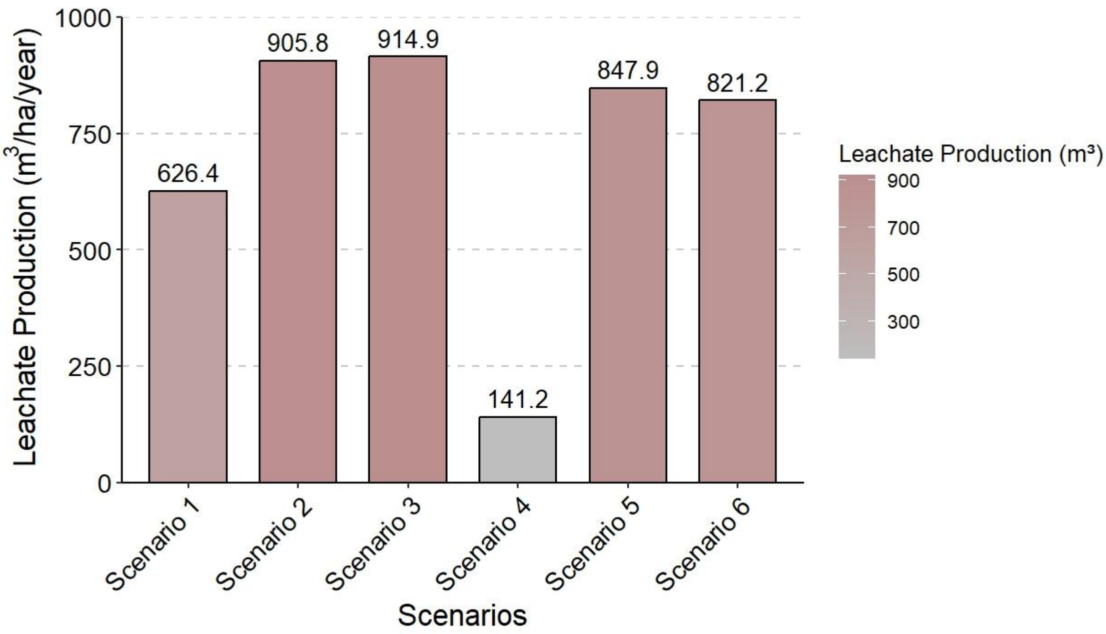


Figure 6.28. Annual leachate production captured by leachates drainage.

Similar amounts of leachate were generated in the study by Podlasek (2023), where the amount of leachate generated per hectare at another Polish landfill reached the level 803-818 m^3 per ha/year. On the other hand, in a study by Alslaibi et al. (2013), it was shown that in a Mediterranean climate with an annual rainfall of 322 mm, the leachate production per 15 ha of landfill in the period 1997–2007 was of the order of 6800 m^3/year, which corresponds to only 453 m^3 of leachate per ha/year. Ng et al. (2019) compared landfills with composite cover with GM and conventional CCL and found that with synthetic cover, leachate infiltration was in the range of 26.0 mm representing 2% of annual rainfall and 28.0 mm representing 3.1% of annual rainfall, while CCL infiltration increased to 65 mm, representing 7.5% of annual rainfall. This is in line with research by Ehrig (1983), who showed that for a young landfill, the amount of leachate produced can be in the range of 3.3–7.2% of the annual rainfall (770 mm). Taking this into account, the leachate generated in the case of a landfill covered with GM a percolation of 14.12 mm/year, corresponding to 2.9% of the rainfall. However, in the case of a landfill covered with mineral cover, 90.58 mm/year, 18% of the rainfall is infiltrated.

In the study carried out in all 6 scenarios, the base of the landfill was sealed with cohesive soil of approximately 1 m thickness and 1.5-2 mm HDPE GM. However, this does not provide 100% effectiveness for groundwater protection because as the landfill ages, the strength of the liner decreases. The study showed that leaching through HDPE GM in the liner was in the order of 0.07–2.8 m^3/ha/year for all scenarios (Fig. 6.29). The highest percolation was observed for Scenario 3, but this was most likely due to the thinnest GM at 1.5 mm (the other cases were 2 mm) and twice the thickness of the landfilled waste. A similar conclusion can be drawn from a comparison of Scenarios 5 (10.4 m MSW) and Scenario 6 (8.5 m MSW), which shows percolation at the Zdounky landfill. The smallest percolation was observed in the reclaimed cell of the Zdounky landfill (Scenario 4) with a level of 0.07 m^3 of leachate per 1 ha. This indicates that the landfill was effectively protected with layers to prevent percolation, which minimizes leachate production and does not significantly increase leachate production despite the installed defects on the GM. However, this may be strongly related to the fact that the HELP model does not specify the age of the landfill or aging rate of the material. The modelling assumes of two defects per hectare of HDPE GM, corresponding to installation quality level 4. Furthermore, a hydraulic conductivity k of approximately 2.0×10^{-15} m/s is assigned to the HDPE GM, which can significantly reduce potential seepage, as indicated by Podlasek (2023).

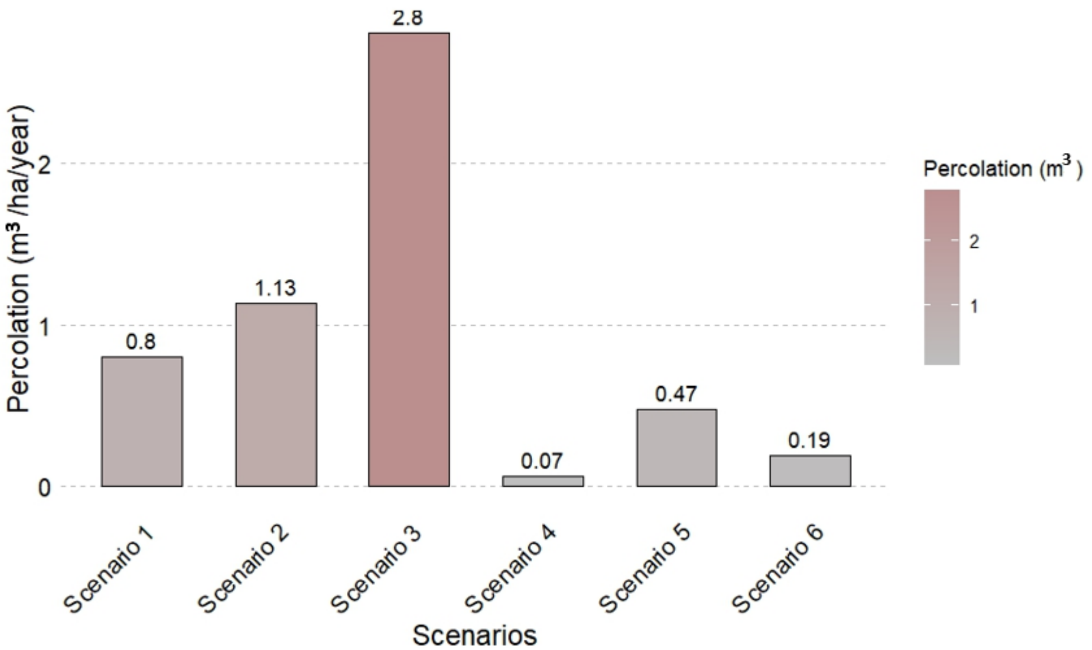


Figure 6.29. Percolation or leakance through HDPE GM in the landfill bottom.

Rowe (2005) reported that, based on currently available data, the life of HDPE GM in an MSW landfill is estimated to be approximately 160 years for the original liner at 35°C or 200 years if the temperature drops to 20°C. However, the aging process of a GM is not only physical but also chemical, which includes the breaking of bonds in the backbone or chemical reactions that lead to mechanical degradation (Schnabel, 1981). Therefore, the degradation process is largely dependent on the chemical composition of the stored waste. On the other hand, Sun et al. (2019) showed that HDPE GMs reach their life expectancy after only eight years of landfill operation when exposed to UV radiation, after which their hydraulic capacity begins to decline rapidly and can be as low as 1325 m^3/ha per year. Considering the results obtained it is allowed to conclude a fairly good technical condition of the GM used.

6.3.2. Landfill gas emission modelling

In addition to landfill leachate, another environmental hazard caused by landfills is GHG emission. Therefore, it is essential to monitor the composition and quantity of the LFG produced to effectively capture it and prevent its diffuse release. Because statistical tests did not reveal significant differences in the composition of LFG, a quantitative simulation of the produced LFG (CH_4 and CO_2) was undertaken for the studied landfills over time horizon of approximately 140 years. Fig. 6.30 and 6.31 present the quantitative production of CH_4 and CO_2 at landfills where different cover systems were used and where the density of the stored waste varied. Fig. 6.30 presents the CH_4 model from the reclaimed western cell in Zakroczym, considering the CH_4 oxidation effect after 2011 due to the applied mineral cover. The graph shows a clear increase in emissions during the initial years of modelling (1997–2012), followed by a gradual decrease, primarily due to a reduced amount of readily biodegradable organic fractions. The study results revealed that the production rate of both gases peaked after cell closure in 2011 (855 421 m^3 CH_4/year and 1 283 000 m^3 CO_2/year) and then gradually declined until the year 2137, reaching values of 69 300 m^3 CH_4/year and 104 000 m^3 CO_2/year.

However, Fig. 6.31 shows a model of LFG production (CH_4 and CO_2) from the reclaimed landfill in Zdounky. In this case, no oxidation was assumed due to the 1-mm-thick HDPE GM used in the cover, which provides a protective barrier against gas emission outside the cover system. As in the case of Zakroczym, the graph shows a clear increase in emissions in the first years of modelling (around 1997-2013), followed by a gradual but significant decrease in CH_4 and CO_2 emissions.

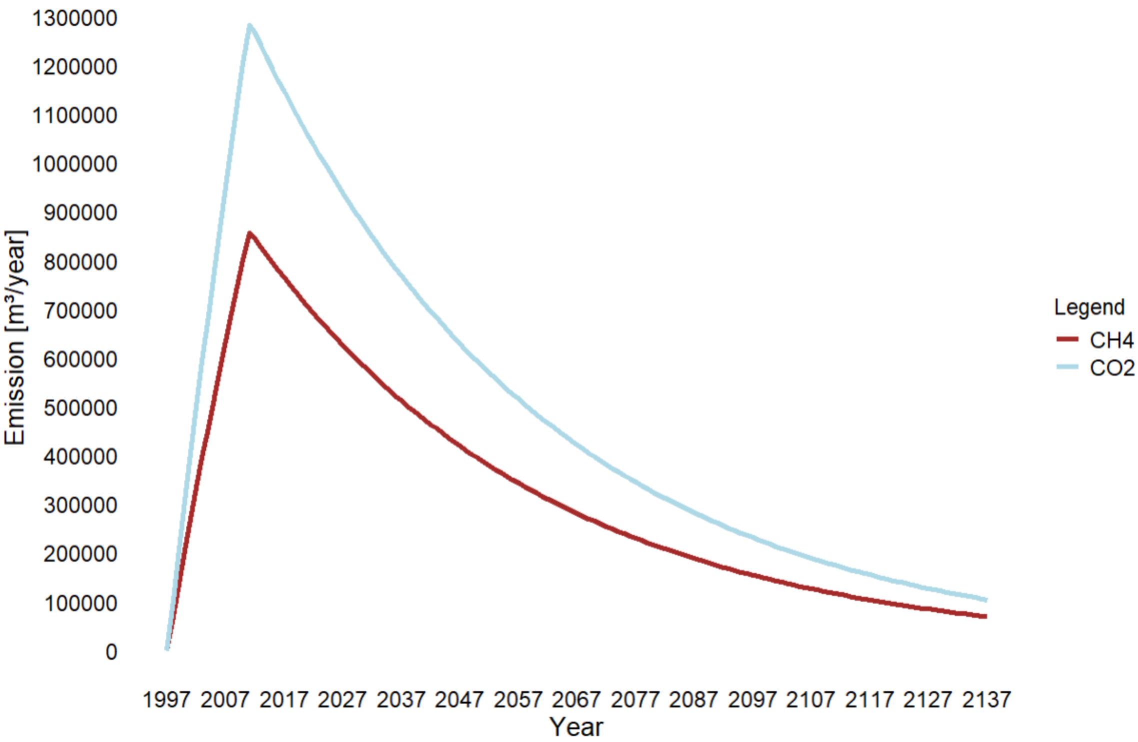


Figure 6.30. Modelling of LFG generation from the western cell in Zakroczym, covering the period 1997–2137.

The results show that the production rates of both gases reached their peak in 2012 (628 100 m^3 CH_4/year and 942 150 m^3 CO_2/year) and then gradually decreased in 2136 year, reaching 52 660 m^3 CH_4/year and 78 899 m^3 CO_2/year, respectively.

The graph also shows a proportional decrease in CO_2 in relation to CH_4, which is related to the stabilization of the anaerobic phase, which is consistent with the above analysis carried out for the Zakroczym landfill. It should also be noted that the maximum amounts of CH_4 and CO_2 produced by the reclaimed landfill in Zdounky were approximately 27% lower than those in the case of the reclaimed landfill in Zakroczym, which is also due to the difference in the amount of waste deposited.

In both analyzed landfills closed in 2011 the highest amounts of generated CH_4 were observed 1–2 years after the facility closure. The results are consistent with Rodrigue et al. (2018) study who stated that the highest amounts of CH_4 are observed three years after the closure of a landfill. The both graphs shows a proportional decrease in CO_2 relative to CH_4, which is related to the stabilization of the anaerobic phase (Borisova et al. 2023; Mohsen et al. 2019; Haro et al. 2019).

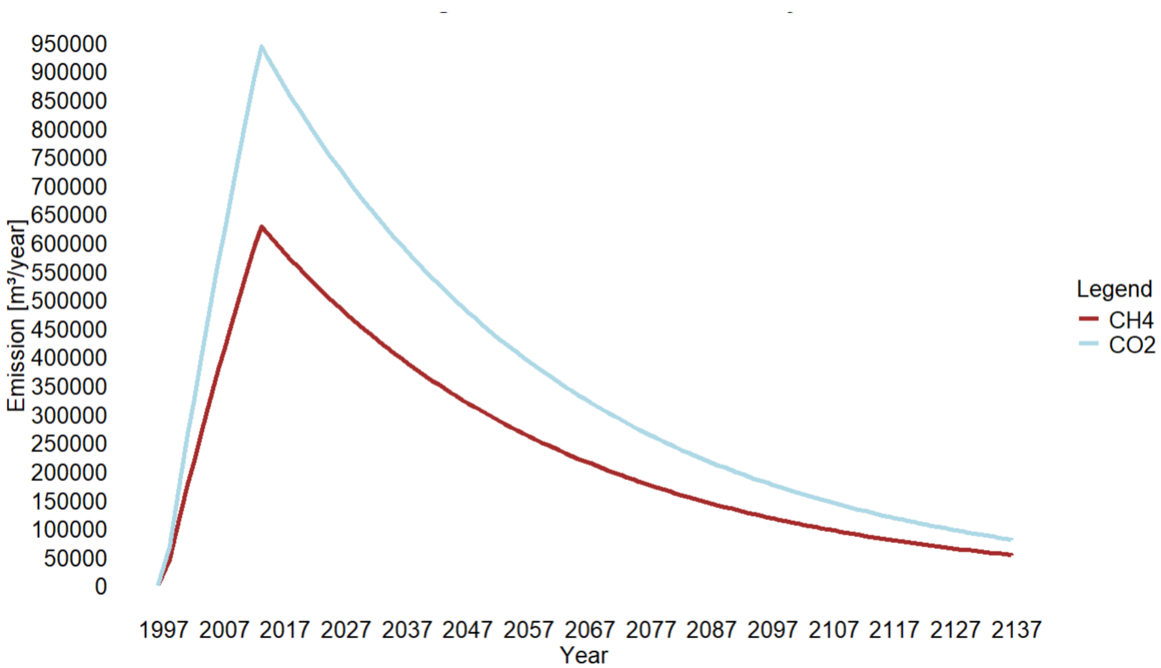


Figure 6.31. Modelling of LFG generation from the western cell in Zdounky, covering the period 1997–2137.

To more precisely compare LFG generation in the two reclaimed landfills using different methods, the obtained results were recalculated to unit emissions of CH_4 and CO_2 per 1 ton of deposited MSW. At the reclaimed cell in Zakroczym a minimum unit emission was approximately 3.37 m^3 CH_4, an average of about 15.81 m^3 CH_4, and a maximum of 41.56 m^3 CH_4 per 1 ton of MSW of deposited waste were observed. In the case of CO_2, the minimum unit emission reaches 5.05 m^3 CO_2, the average is approximately 23.72 m^3 CO_2/year, and the maximum is 62.34 m^3 CO_2 per ton of deposited waste (Fig. 6.32).

Based on Fig. 6.33, the amount of unit LFG generated from the rehabilitated Zdounky site was slightly higher than that of the Zakroczym landfill. For the evaluation, it was assumed that an average of 13 478 tons of MSW was landfilled annually in the Zdounky landfill. It was observed that the minimum specific emission of CH_4 was 3.37 m^3 CH_4, while the average was of the order of 18.17 m^3 CH_4 and the maximum 46.60 m^3 CH_4 from 1 ton of landfilled waste. For CO_2, the minimum specific emission was of the order of 2.73 CO_2, while the average was of the order of 27.26 m^3 CO_2 and the maximum of 69.90 m^3 CO_2 from 1 ton of landfilled waste.

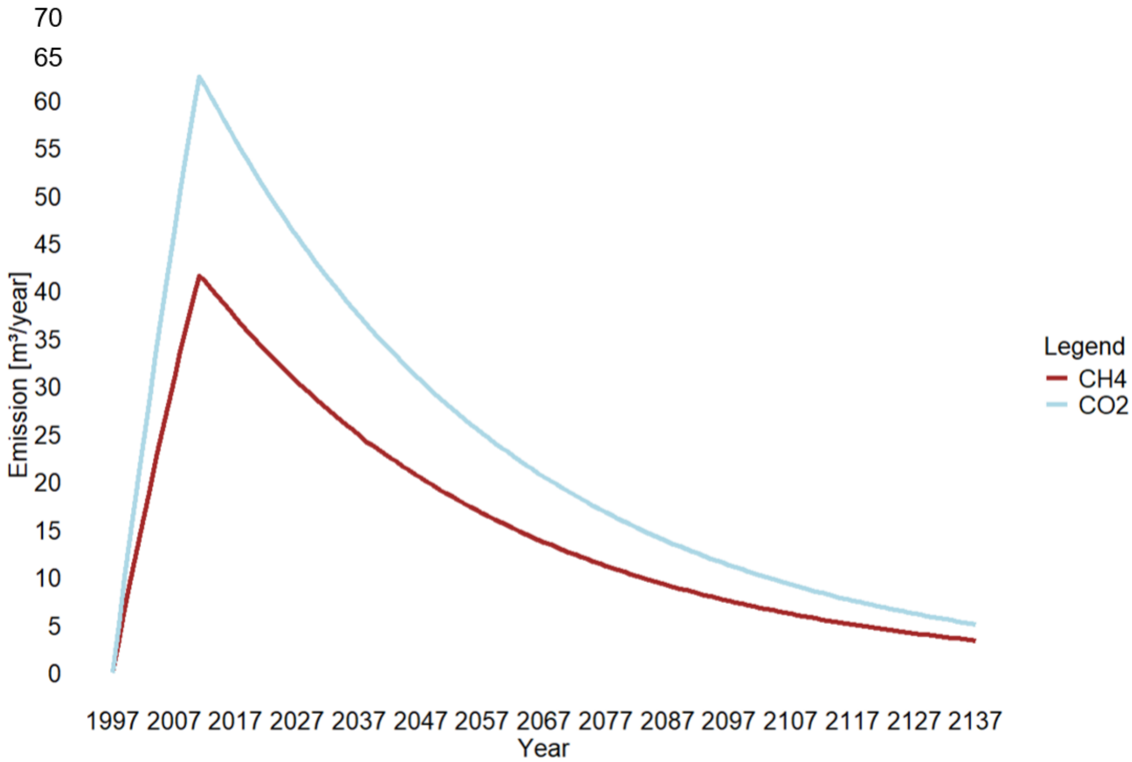


Figure 6.32. Landfill gas generation modelling per 1 ton of MSW from the western cell in Zakroczym.

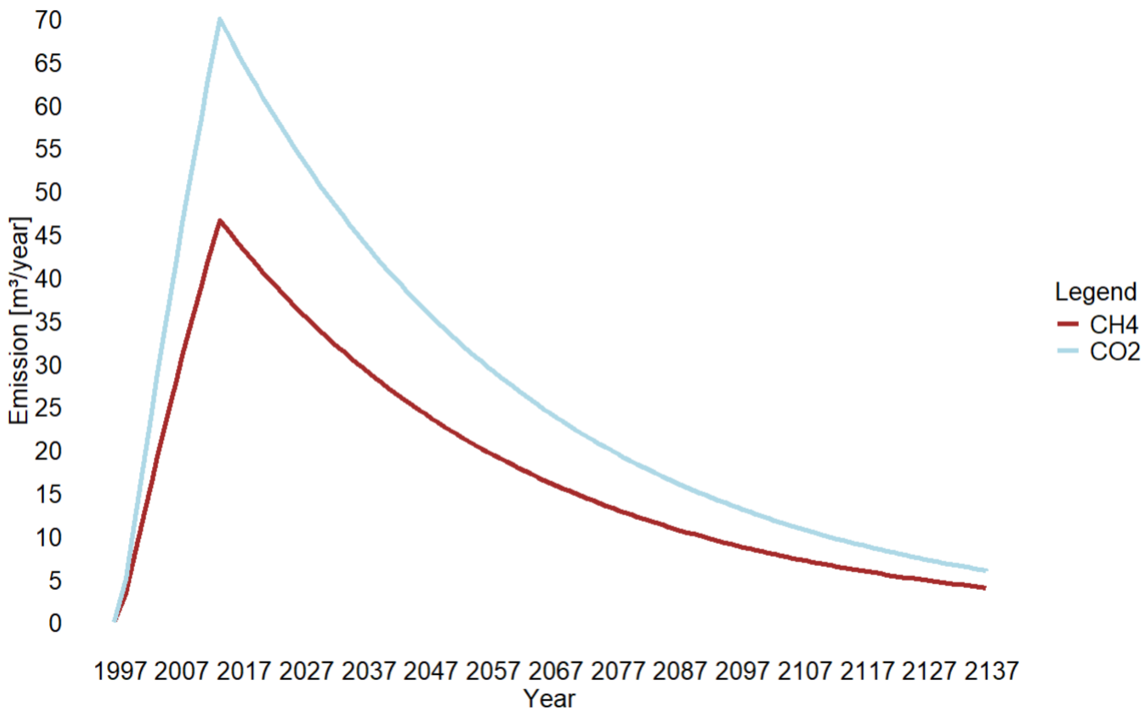


Figure 6.33. Landfill gas generation modelling per 1 ton of MSW in Zdounky.

The differences in the amount of LFG produced between the sites were probably due to CH_4 oxidation in the case of the mineral cover used at Zakroczym. Fig. 6.34 shows a graphical representation of the results obtained from modelling the unit production of LFG at the two sites studied after the reclamation period. It can be seen that the tight synthetic cover reduces the access of oxygen and creates anaerobic conditions, which increases the production of CH_4. However, it should be noted that the CH_4 production reported by the LandGEM model may be higher than that obtained in the field tests conducted, since the numerical model assumes ideal conditions regarding the sealing of the landfill and does not consider all the parameters that affect the gas emissions from the landfill.

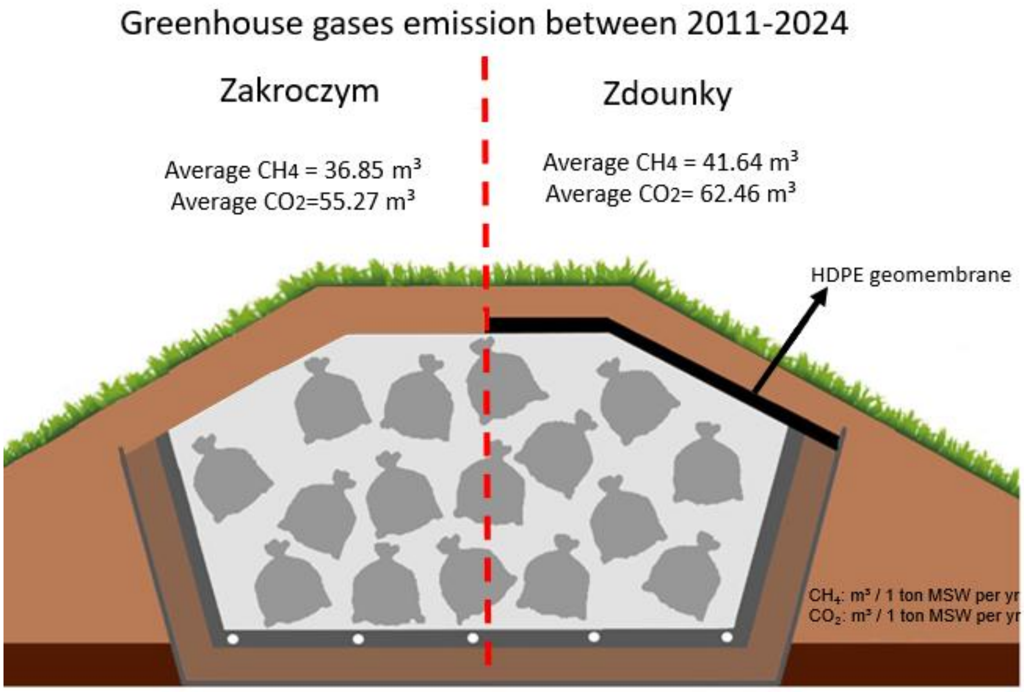


Figure 6.34. Graphical representation of the results obtained from modelling the unit LFG production after landfills reclamation.

Model assessments of LFG emissions are common in literature. Kale and Gökçek (2020) found that the values of emitted CH_4 per ton according should range from 6.2 to 270 m^3/ton, but this depends almost entirely on the type of waste, so as the organic fraction is higher there is more CH_4. Yaman (2020) showed that, assuming a 50% CH_4 content in LFG, the average CH_4 generation from 2018-2028 was 59.3 m^3 CH_4 per ton of MSW. An important study was conducted by Jain et al. (2021), who showed a clear effect of precipitation on the CH_4 potential of landfills. For regions with precipitation below 635 mm per year, the average unit amount of CH_4 produced is 49 m^3 CH_4 per 1 ton MSW, which is significantly ($p < 0.002$) lower than the 73 m^3 CH_4 per 1 ton MSW in areas with precipitation above 635 mm. Similarly, at a threshold of 508 mm per year, unit LFG production was 47 m^3 CH_4 per 1 ton MSW (for < 508 mm) and 75 m^3 CH_4 per 1 ton MSW (for 508–1016 mm), respectively. This suggests that in dry climates with lower precipitation, some of the biodegradable carbon is not fully degraded, resulting in lower CH_4 production. Similar results were also obtained by Kale and Gökçek (2020), who for one of the analyzed landfills (operated for 19 years) estimated production at 73.8 m^3 of CH_4 from 1 ton of MSW.

The results obtained from this study can be usefully applied in planning energy production using CH_4 as an alternative energy source (Alam et al., 2022). Some scientists refer to landfills as power plants, which are a suitable way to ensure environmental sustainability and economic contribution by generating clean energy from waste (Kale and Gökçek, 2020). CH_4 is an energy source whose calorific value from 1 m^3 of CH_4 is 36 MJ (Ajaero et al. 2023). Yaman (2020), on the other hand, indicated that LFG has an energy content of 500 Btu per cubic foot, which is equal to 5.17 kWh from 1 m^3 of LFG. On the other hand, the global warming potential of CH_4 is 21 times higher than that of CO_2, and its generation is the highest (60%) compared to other gases (Mathur et al., 2020). It is also important to note that emissions from the waste sector account for approximately 18% of global anthropogenic emissions, next to the emissions from the oil and gas sector, which have received much attention (Maasakkers et al., 2022).

The question remains, whether it is more sustainable to use CH_4 for energy production and utilize the so-called by-product of landfilling or to try to minimize its emissions in landfills, thus reducing the potential impact of emissions on the GHG effect. Unfortunately, installed gas collection systems cannot capture all the gases produced by waste. Their LFG collection efficiency ranges from 50 to 100% (75% on average), depending on the type and coverage of the collection system (Chetri and Reddy, 2021). Recently, there has been much discussion in the literature on the oxidation of CH_4 in landfills by covering them with composting materials, whether yard waste and leaf compost or biosolids compost (Niemczyk et al., 2021). According to Fraser-McDonald et al. (2022), approximately 40% of CH_4 generated in landfills is oxidized by the presence of aerobic surface soils. In the above study, the level of oxidation of the soil used to cover the Zakroczym landfill was approximately 7%, but this still resulted in visible differences in the LFG. Chetri et al. (2022) evaluated the performance and environmental impact of a proposed biogeochemical cover system composed of steel slag and biochar-amended soil, and compared it to a conventional soil cover composed of CCL. They used parameters such as gas emissions, slope stability, infiltration, and material life cycle assessment using the LCA method to evaluate the performance of the system. The biogeochemical cover system proposed by researchers can provide better or equivalent performance against infiltration and slippage, and significantly reduce CH_4 emissions compared to a conventional cover system.

The higher CH_4 oxidation potential of the biocarbon system (400 μg per 1 g) led to a significant reduction in CH_4 emissions, resulting in the lowest CH_4 emissions of about 8 g/m^2/day. In addition, the use of waste materials such as steel slag and waste wood reduces the overall environmental impact and global carbon footprint, although the variability of CH_4 oxidation rates and other uncertainties under real-world conditions must be considered. In this regard, it should be noted that there are various approaches to LFG emissions and management around the world, including both gas treatment and oxidation to minimize emissions. These technologies, which are being implemented locally and globally, are becoming increasingly important in reducing the environmental impacts associated with GHG emissions. In addition, their development can significantly improve emissions management and promote sustainability through the use of WtE.

6.3.3. Slope stability analysis

6.3.3.1. Geomembrane sliding analysis

The installation of GMs in landfills with steep slopes or the use of unsuitable materials in the landfill cover over the GM are often the cause of the risk of sliding and loss of landfill stability. Therefore, the following analysis were made to evaluate the risk of sliding at the interface between the layers of landfill cover and synthetic material (HDPE GM 1 mm) using the limit equilibrium method at the reclaimed landfill site in Zdounky (Czech Republic). The analysis focused on evaluating the stability of the landfill cover, particularly in relation to the effect of slope gradient on structural safety. Fig.6.35 shows a scheme of the landfill cover, along with the strength parameters of the soil used.

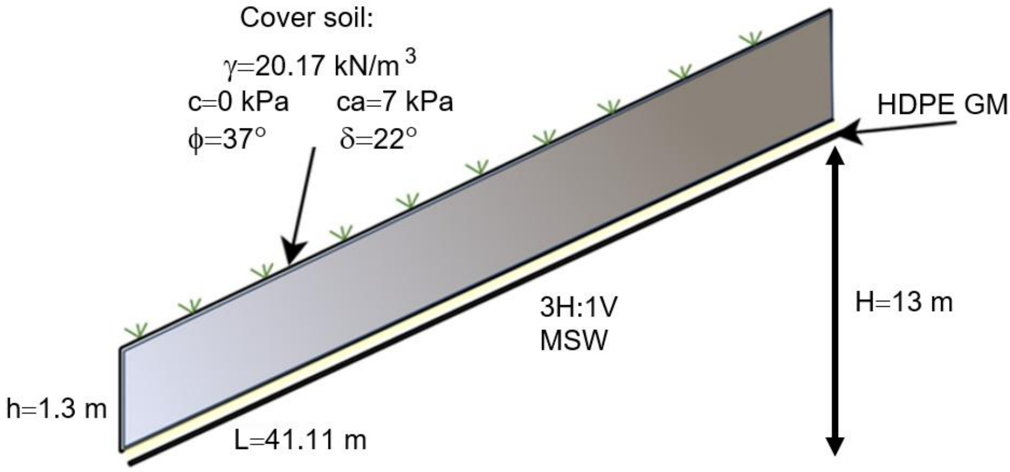


Figure 6.35. Strength parameters of soil in cover at Zdounky landfill.

Considering an active wedge:

$$W_A = 20.17 \times 1.3^2 \left(\frac{41.11}{1.3} - \frac{1}{\sin 18.4} - \frac{\tan 18.4}{2}\right) = 964.26 \text{ kN}$$

$$N_A = 964.26 \times \cos 18.4 = 914.83 \text{ kN}$$

$$C_a = 7\left(41.1 - \frac{1.3}{sin\ 18.4}\right) = 259.0 \text{ KPa}$$

Considering a passive wedge:

$$W_P = \frac{20.17 \times 1.3^2}{sin2 \times 18.4} = 56.83 \text{ kN}$$

$$C = \frac{0 \times 1.3}{sin18.4} = 0 \text{ kPa}$$

$$a = (964.26 – 914.83 \cos 18.4) \cos 18.4 = 91.39$$

$$b = -[(964.26 - 914.83 \cos 18.4) \sin 18.4 \tan 37 + (914.83 \tan 22 + 259.00) \sin 18.4 \cos 18.4 + \sin 18.4)(0 + 56.83 \times tan37)] = -224.89$$

$$c = (914.83 \tan 22 + 258.98) \sin^2 18.4 \tan 37 = 47.27$$

The resulting FS value is then obtained from the following equation:

$$\mathrm{FS} = \frac{-(-224.89) \pm \sqrt{-224.89^2 - 4 \times 91.39 \times 47.27}}{2 \times 91.39} = 2.23$$

The calculated FS was 2.23, indicating that the cover was stable with a considerable margin. Koerner and Daniel (1997) stated that when the calculated FS falls below 1.0, a failure in cover stability is expected, which may be associated with soil sliding. An FS of approximately 1.5 is commonly used for designing landfill slopes in cover systems (Knochenmus et al. 1998). Soil covers with a slope of 1(V):3(H) covered with geosynthetics are not particularly prone to loss of stability (Chetri and Reddy, 2021). In addition to the actual case with a waste prism height of 13 m and a slope of 1(V):3(H), an alternative variant was tested with a slope of 1(V):2(H), the same as that at the Zakroczym landfill, where a mineral material was used for the cover. From the calculations performed, the FS for the alternative variant is equal to 1.52, which indicates that there is still a very low probability of slope sliding, although it is noted that the obtained result is 47% lower than that for the slope with a ratio of 1(V):3(H). Nevertheless, the appropriate FS for the cover slope should be selected individually rather than being fixed at 1.5. The main sources of uncertainty are the shear strength at the interface of the materials and the fluid pressure (water and gas). Shear strength may vary depending on the batch of materials, and changes in fluid pressure affect slope stability (Liu et al., 1997). Benson et al. (2012) in their studies described cases in which the use of GM on slopes of 1(V) and 4(H) caused cover failures. A similar situation occurred in the case described by Zhao and Karim (2018), where a cover system consisting of a topsoil layer, vegetative soil, drainage sand, PVC GM, GCL, and a gas venting layer with a slope of 14° (1V:4H) and a height of 60 feet failed due to downslope movement along the GM interface.

Therefore, when selecting soil parameters for a cover using geosynthetics, it is essential to carefully analyze the detailed geotechnical conditions of the landfill and, in addition to the risk of sliding, perform overall stability calculations for the entire structure.

6.3.3.2. Overall stability analysis

This study focused on assessing the overall stability of two partially reclaimed landfills using the Bishop method to compare the FS factors for slopes covered with synthetic (Zdounky landfill) and mineral cover (Zakroczym landfill). The overall slope stability was examined to assess the stability of the entire landfill geometry, including the evaluation of potential landslides caused by soil mass or waste. MSW disposed of in landfills and the materials used for landfill construction are morphologically very diverse; therefore, there is a high susceptibility to slope stability problems.

Analyzing the calculations, it was found that with the assumed design and geometric dimensions of both municipal landfills, the slope of the landfill can be considered stable (FS > 1.5) at both the storage heights H = 13 meters and H = 18.5 m and the slope of the waste mass slope of 1(V):3(H) and 1(V):2(H). Nevertheless, the stability analysis showed clear differences in the calculated safety factor between the landfill in Zdounky covered with GM (slope of 1(V):3(H)) and the landfill in Zakroczym covered with a mineral cover (slope of 1(V):2(H)). The Zdounky landfill showed higher stability (FS = 2.27) owing to a lower slope angle, which reduces the risk of landslides, and a smaller thickness of stored waste (13.0 m). The landfill in Zakroczym, where the mineral cover was used, is more prone to destabilization due to the steep slope, the large thickness of the landfilled waste (18.5 m), and the possibility of weakening the mechanical properties of the cover due to water saturation.

Nevertheless, the modelling results showed that there was no slope stability problem in this case, as the FS in the worst-case scenario was 1.503. Based on this analysis, it can be concluded that a GM-covered landfill is a more stable solution. However, a key design element is to include a detailed analysis of the contact between the layers, including their shear strength, and to design an effective drainage system. This minimizes the risk of failure and ensures the long-term stability of the structure. Nevertheless, the given values of the geometric parameters of landfilled waste should be treated only as indicative, owing to the large variation in the physical-mechanical parameters of municipal waste and their heterogeneity. Fig. 6.36 shows the graphical results of stability analysis performed using the Bishop model.

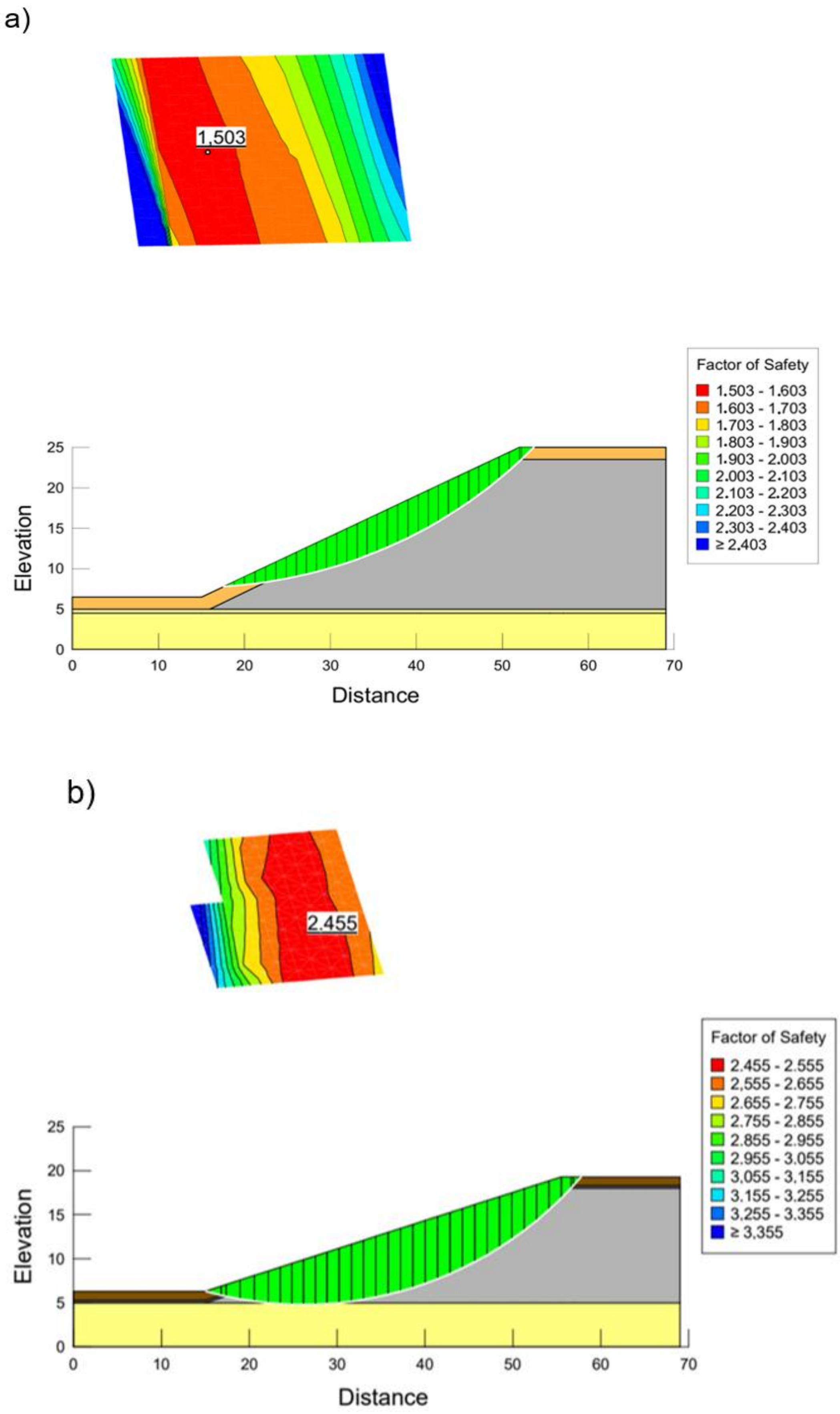


Figure 6.36. Evaluation of overall stability using the Bishop method: a) Zakroczym landfill in Zakroczym, b) Zdounky landfill.

6.3.4. Summary of model studies

This chapter focuses on three aspects of partially reclaimed landfills: water balance and leachate production modelling (UnsatSuite+ HELP), LFG emission estimation (LandGEM), and slope stability analysis (limit equilibrium and Bishop methods). The results of the leachate production simulation showed that the amount of leachate generated, in addition to the intensity of rainfall, also depended on the type and thickness of the cover layers. Higher ET was observed in layers composed of mineral materials with higher permeability (e.g., clayey sands), which reduced surface runoff compared with clay layers. The use of an impermeable GM (1 mm thick) in the cover of the Zdounky landfill promoted a significant reduction in rainwater infiltration deep into the waste body, allowing rainwater to be separated from the leachate and discharged into a separate rainwater tank. For closed cells, it was confirmed that adequate sealing of the surface layers significantly reduced leachate production (to less than 10% of annual precipitation). At the same time, HELP has been shown to be a useful tool in the context of water balance simulations, although it does not fully consider factors such as the degree of material aging or the widely varying density and composition of the waste stored.

The results of simulations of CH_4 and CO_2 emissions using the LandGEM model confirmed that the highest emission values were observed in the first few years after landfill operation, which is associated with the decomposition of the easily biodegradable organic fraction. Thereafter, CH_4 and CO_2 emissions successively decreased until they reached a low level in the final phase of biological stabilization. The use of synthetic cover (HDPE GM) promotes the creation of anaerobic conditions and thus increased CH_4 production, while the use of mineral cover is associated with partial oxidation of CH_4 in the soil layers.

In contrast, the slope stability analysis of the studied landfills showed that in all studied variants (which varied in terms of the height of the waste pile, type of cover, and slope angle), FS remained above 1.5. The Zdounky landfill, covered with HDPE GM and having a lower slope gradient of 1(V):3(H) and a lower height of the waste pile (13 m), where FS > 2.2, was the most stable. The Zdounky landfill also did not see any risk associated with sliding at the geosynthetic-soil interface (FS = 2.23). On the other hand, when the slope parameters were changed to a ratio of 1(V):2(H), the FS decreased to 1.52, indicating the limitations of using GM in landfills with slopes less than 1(V):3(H).

The Zakroczym landfill, despite its steeper slopes (1V:2H) and greater thickness (18.5 m), also maintained safe stability conditions (FS≈1.5). However, in terms of structural safety, there is not much provision and failure to meet any of the strength parameters, which could result in a loss of stability.

In conclusion, simulations have confirmed the importance of a properly designed cover system (both synthetic and mineral) for reducing water infiltration, controlling GHG emissions, and ensuring landfill stability. The most important factors include the type and geotechnical parameters of the sealing layers, slope angle, and the density and thickness of the landfilled waste. In the long term, it is also important to consider the aging and possible degradation of the geosynthetic as well as the biological and physical degradation of waste. Thus, the results provide valuable guidance for landfill designers and managers, while emphasizing the need for further detailed in situ research on the dynamic properties of landfill covers.

6.4. Biomonitoring studies

The chapter below presents the results of the phytotoxicity of leachate based on its effect on the germination of *Sinapis alba* L. seeds using the Phytoxkit test for solid samples and Phytotestkit for liquid samples, and evaluates the respiration potential of the soils used for reclamation at the studied landfills in Zakroczym and Zdounky.

6.4.1. Phytotoxicity testing of leachates

After an incubation time of 72 h, samplers from both test sets were photographed, and plant root length was measured in all samples using Image Tool 3.0, for Windows. The root length of *Sinapis alba* L. was measured in six variants: Zakroczym liquid, Zdounky liquid, Zakroczym solid, Zdounky solid, control liquid, and control solid. The apparent effect of leachate on the inhibition of germination of *Sinapis alba* L. was confirmed by the highest germination in the control sample in liquid form (n = 44), which supports the reliability of the results in this group, as well as in solid form (n = 25). In tests on liquid samples, roots tended to be longer than those in the solid medium. Tab. 6.6 shows the measured minimum and maximum root values, the 95% confidence interval with median and SD, and the number of germinated seeds (n), which is also the number of measured roots.

Table 6.6. Basic statistics of the studied data groups.

Group*	Min (mm)	Max (mm)	95% confidence interval		n
			Median	SD	
Zakroczym liquid	0.837	40.8	9.96	12.3	27
Zdounky liquid	1.76	73.6	31.5	24.3	41
Zakroczym solid	2.03	82.0	7.63	18.2	22
Zdounky solid	0.4	25.4	4.36	7.95	12
Control liquid	1.28	99.6	28.4	34.2	44
Control solid	1.21	90.3	18.1	22.3	25

** liquid means a completed Phytotestkit test for liquid samples, while solid means a completed Phytoxkit test for solid samples*

The highest median was achieved with Zdounky liquid (31.5 mm), exceeded the control by 10.9%. However, for leachate samples from Zakroczym in the liquid test, the median was 31.6% lower than that of the control sample. In contrast, for the solid variant test, samples taken from both locations (Zakroczym – 7.63 mm, Zdounky – 4.36 mm) recorded lower root length values compared to the control (18.1 mm), suggesting potentially greater toxicity of leachate in contact with the reference soil than on leachate alone, and greater sensitivity of the Phytotoxkit test than Phytotestkit in contact with leachate. This is also observed in detail in the box plot shown in Fig. 6.37.

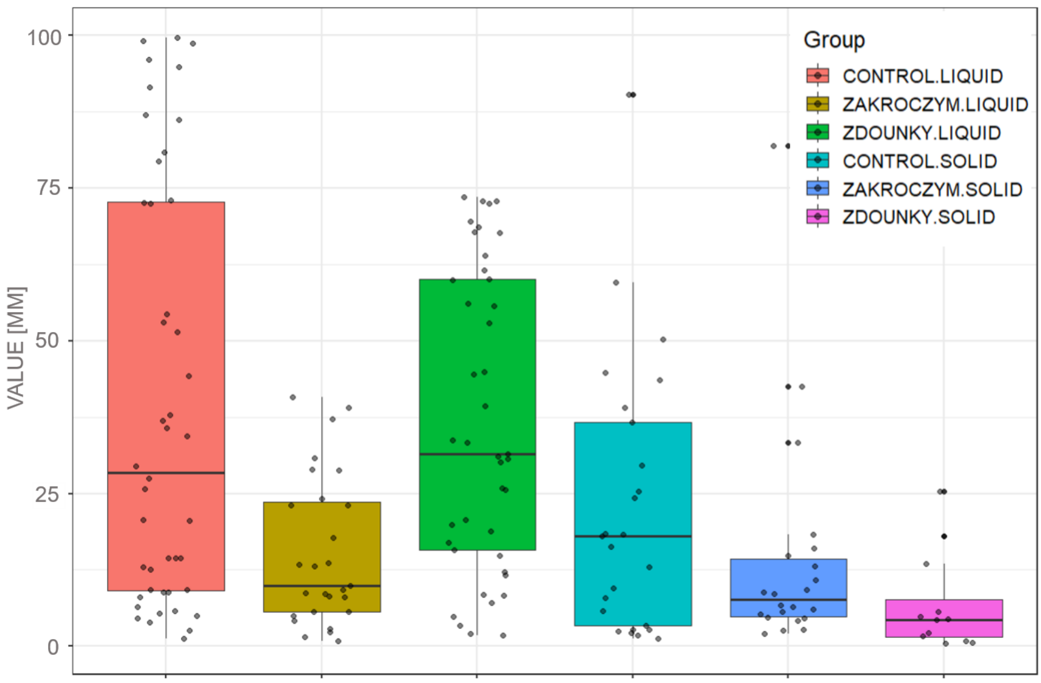


Figure 6.37. Root length of *Sinapis alba* L. after Phytoxkit (solid) and Phytotestkit (liquid) tests.

Statistical analyses were performed to confirm statistically significant differences between the study sites and test types. The data in any study group did not have a normal distribution, as confirmed by the Shapiro-Wilk test, and in all cases, the p-value was < 0.05. Therefore, a nonparametric test was performed. Because the number of study groups was > 2, the Kruskal-Wallis test was used. Table 6.7 shows the results of the normality test of the data distribution and the results of the non-parametric Kruskal-Wallis test, which showed a fairly clear significant difference (p-value = 0.0000007568 < 0.05), indicating that at least one of the study groups was significantly different from the others.

Table 6.7. Results from the Shapiro-Wilk and Kruskal-Wallis tests.

Group	Shapiro-Wilk normality test	Kruskal-Wallis rank sum test
Zakroczym liquid	W = 0.89107, p-value = 0.008404	ch-squared = 46.493 df = 5 p-value = 0.0000007568
Zdounky liquid	W = 0.91237, p-value = 0.003925	
Zakroczym solid	W = 0.61813, p-value = 0.000002	
Zdounky solid	W = 0.78261, p-value = 0.005983	
Control liquid	W = 0.85941, p-value = 0.000076	
Control solid	W = 0.8586, p-value = 0.002576	

The Wilcoxon rank-sum test with Benjamini-Hochberg error correction was used to determine which groups were specifically different. A significant difference between groups was observed in Zakroczym and Zdounky liquid samples (p = 0.001), confirming the higher phytotoxicity of Zakroczym leachate samples, whereas no significant difference was observed between Zakroczym solid and Zdounky solid samples (p = 0.061), so that the phytotoxicity of the leachates did not differ between sites despite the higher median root length (7.63 mm) associated with the Zakroczym leachate samples than with the Zdounky samples (4.36 mm). Other significant differences between the groups are shown in the correlation matrix in Fig 6.38.

Due to the observed significant difference between the control and test samples (control liquid - Zakroczym liquid; p-value = 0.01, and control solid - Zdounky solid; p-value = 0.02), the *IR* index was calculated. The highest IG and IR were recorded in the solid test on the leachate from Zdounky (*IG*= 88.89% - Trial I and IR=79.44% - Trial II).

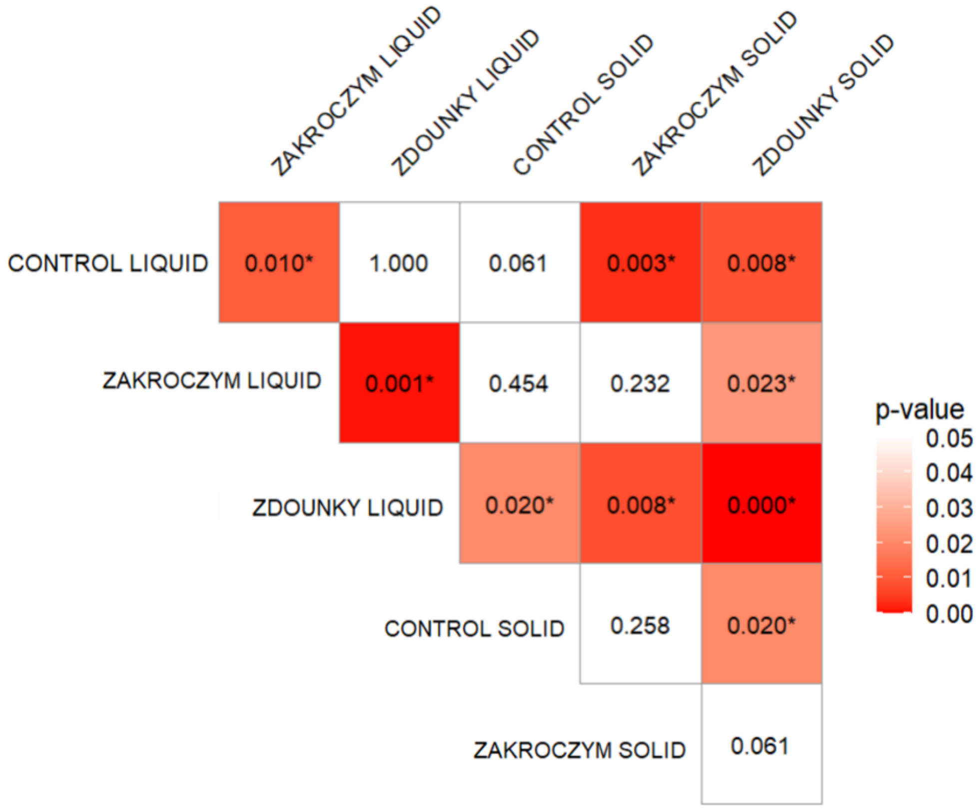


Figure 6.38. Triangular matrix of dependencies resulting from the Wilcoxon rank sum test.

At the same time, the greatest promoter of root growth was trial II liquid on the leachate from the landfill in Zdounky, amounting to -95.7%. Nevertheless, in the case of the liquid test in Zdounky, *IG* improved germination compared with the test trial in only one time. In the case where *IR* equaled -95.7%, IG inhibition still amounted to 12.5% relative to the test trial, suggesting an anomaly related to a positive effect of the leachate on some of the seeds. The leachate from the landfill in Zdounky exhibited more extreme values than those from the landfill in Zakroczym. Nevertheless, despite the extreme cases where the *IR* level in Zdounky was negative and suggested improved germination, when the type of test was disregarded, the average *IR* in Zdounky was estimated at 24.12%, whereas in Zakroczym, it was 15.83%, suggesting the toxic nature of the leachates from both landfill sites. Comparing the toxicity levels causing *IR* for both sites, it can be concluded that three out of six examined samples of leachate from Zdounky were highly toxic, whereas in the case of the leachate from the landfill in Zakroczym, only one out of six samples was highly toxic. For *IG*, the index was 7.22% at the landfill in Zakroczym and 28.47% in Zdounky indicating a slightly toxic and toxic character of the examined leachates, respectively.

It was also observed that regardless of the leachate examined, both in Zakroczym and Zdounky, the highest *IR* and *IG* were observed in the solid tests using the reference substrate, which may indicate a greater sensitivity of the test in which a substrate simulation was conducted. Tab. 6.8 presents the summary of *IG* and *IR* results for the studied sites.

Table 6.8. Gemination and root inhibition caused by landfill leachate from Zakroczym and Zdounky.

Test type	Sample No.	Zakroczym		Zdounky	
		IG (%)	IR (%)	IG (%)	IR (%)
Solid	I	11.11	61.11	88.89	61.99
	II	22.22	31.45	44.44	79.43
	III	0	3.26	0	50.20
Liquid	I	-12.5	39.42	-25	-14.03
	II	12.5	-51.17	12.5	-95.74
	III	10	10.92	50	62.85

non-toxic or slightly toxic   toxic   highly toxic

A similar study of the phytotoxicity of leachates from the landfill in Zdounky was conducted by Zloch et al. (2018). Their research showed that the leachates from the landfill in Zdounky did not show significant changes in toxicity over time (February–June 2017), and that the tested samples showed higher IR values with increasing leachate concentration. However, the leachate concentration and HMs content do not always have a negative impact on plants. In fact, lower concentrations of leachate can stimulate plant growth, as confirmed by previous studies. Palm et al. (2022) conducted a biotest using *Sinapis alba* L. and *Triticum aestivum* L., which showed that a moderate proportion - up to 50–70% of the leachate solution – had a positive effect on the growth parameters of cultivated plants, including leaf growth and total shoot biomass. In view of the above, further studies should be conducted on other plant species to assess the toxicity of leachates and the effect of their dilution on vegetation, as this may influence the potential use of leachate recirculation at landfills. Vaverková et al. (2023) demonstrated that regular biomonitoring studies also allow early detection of pollution incidents, identification of migration pathways for toxic substances, and effective management of reclamation efforts.

6.4.2. Respiration studies of soils used for reclamation

Soil health testing is extremely important and provides an additional control indicator of quality and impact on vegetation, as well as information on the impact of potential landfill soil uses, such as leachate recirculation. The following results were obtained from respiration measurements of soils taken from the covers of reclaimed landfills that were watered to achieve a 50% water holding capacity (WHC) with water or landfill leachate. Based on this study, the visible effect of leachate on the health of soil used for reclamation was noted. For soil samples from the Zakroczym landfill watered with leachate, values even exceeded 70 ppm CO_2-C, suggesting high biological activity in some of these samples and high fertility. The average CO_2-C content of soil from Zakroczym watered with leachate is 63.80 ppm CO_2-C, which indicates medium high microbial activity and medium fertility (Fig. 6.39a). In contrast, soil samples irrigated with H_2O most often rank in the lower range, often below 20 ppm CO_2-C (average 15.3 ppm CO_2-C), which corresponds to low microbial activity, as well as low fertility, which is most likely due to the much lower nutritional value of H_2O compared to leachates with high NH_4^+ contents.

Soils from the Zdounky landfill watered with leachate also achieved higher CO_2-C values in all samples tested than those flooded with H_2O, however, the differences were not as noticeable as in the case of the Zakroczym landfill, as shown in Fig. 6.39b. For the soil samples watered with leachate, the values averaged up to 19.17 ppm CO_2-C, suggesting a low level of soil biological activity. This may be due to the excessive concentration and density of leachate, as confirmed by the analyses conducted in Section 6.1.2. Nevertheless, 44% of the leachate-watered samples in Zdounky had already reached a lower level of medium fertility, which is considered the most suitable for plant growth. In the case of watered samples, as in the case of the Zakroczym landfill, the average values were much lower than when the land was watered with leachate, and in this case amounted to 9.14 ppm CO_2-C, which corresponds to low biological activity as well as low fertility.

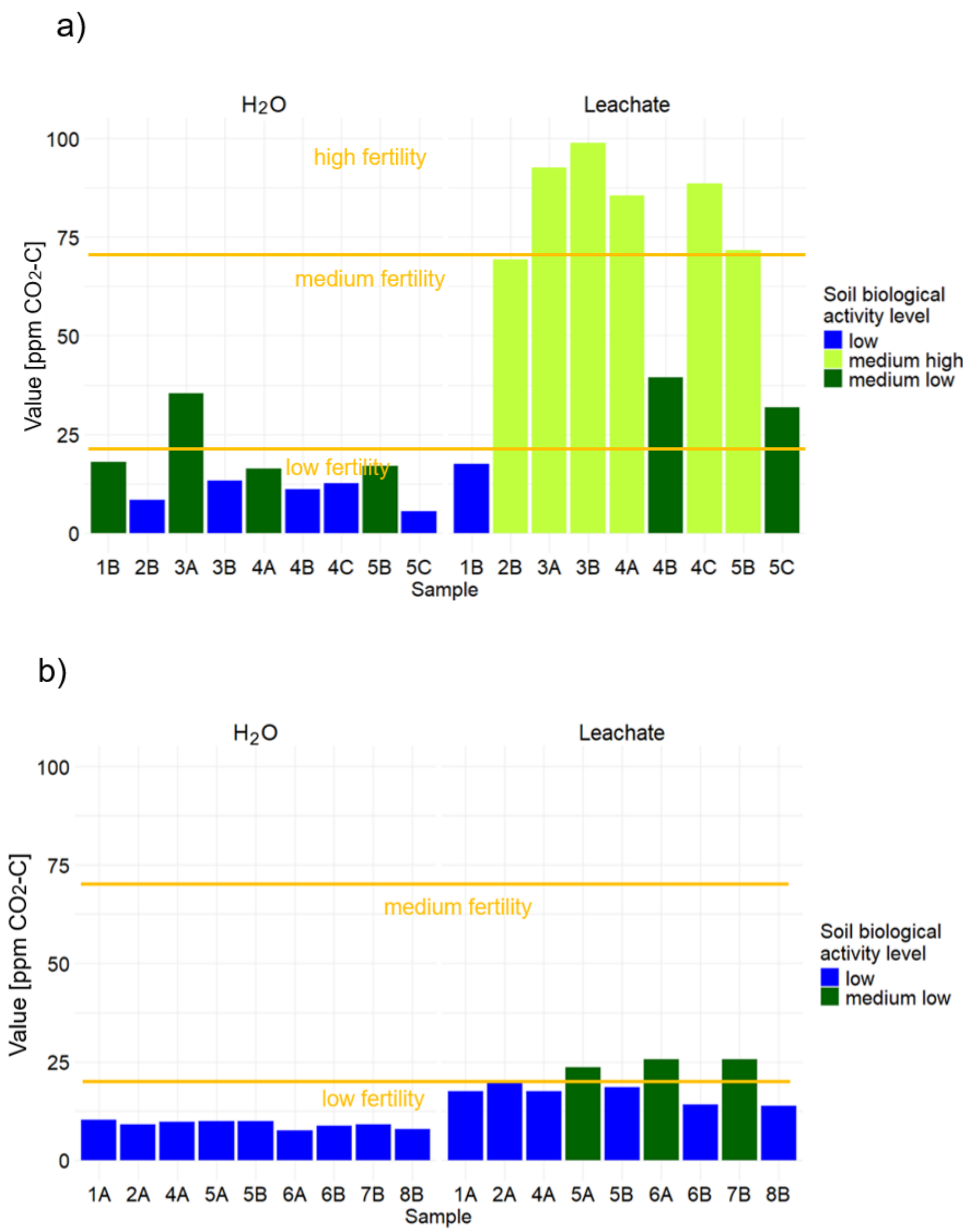


Figure 6.39. Respiration and fertility levels of soils from the investigated landfill sites: a) Zakroczym landfill, b) Zdounky landfill.

Based on the study, it is noted that regardless of soil type, watering the samples with H_2O from both Zakroczym and Zdounky resulted in a decrease in fertility level to "low" in 94% of cases, which is often associated with limited resources and poorer microbial activity. Djerdi et al. (2021) tested how CO_2 content changes depending on the addition of various concentrations (0.0795 μg/cm^2, 0.795 μg/cm^2, 2.384 μg/cm^2) of chlorpyrifos to the soil. They observed that with the increase in chemical concentration, CO_2 content in the soil also increased, exceeding the control sample with distilled water (12.25 μg

CO_2/g/h) already at the concentration of 0.795 µg/cm^2 of chlorpyrifos (23.85 µg CO_2/g/h). This is relevant in the context of the study's results, as it was observed that the amount of CO_2-C in soils flooded with landfill leachate from both Zakroczym and Zdounky was higher than that in soils with H_2O. Sciarappa et al. (2017) determined the average annual CO_2 values from 2013–2015 for all six replicates at 36.7 CO_2 ppm, indicating very good fertility and microbial activity. Rogers et al. (1983) found that plants (such as soybean, corn, pine, and sweetgum) achieve a greater biomass at higher CO_2 concentrations. According to the SOLVITA Manual Instruction (2019), proper development requires 40 CO_2-C for wheat, 24 CO_2-C for soybean, and 35 CO_2-C for alfalfa. Therefore, soybean could develop in soil watered with leachate in Zdounky, whereas all three plants could grow in soil from Zakroczym. Jo et al. (2022) recorded a significant increase in SPAD values, an indicator of chlorophyll content in leaves – the SPAD value in the 800ppm group was twice as high (45.87) as in the 400ppm group (25.85). Plant height also increased, the average fennel rose from 49.81 cm (at 400 ppm) to 57.64 cm (at 800 ppm).

Fungenzi (2015), found that CO_2 evolution was highly correlated with the clay content in the soil (r = 0.66) and strongly negatively correlated with the sand content (r = -0.31). On the other hand, McEachin (2022) found that soils with high clay content had a lower nitrogen mineralization rate than soils with sandy texture, which may hinder plant uptake. It was therefore decided to check the relationship between the clay content of the soils studied and CO_2 content, which confirmed the effect of soil type on CO_2 content, as can be seen in Fig. 6.40 below.

The Pearson's correlation coefficient (r) for the above study for Zdounky was equal r = -0.8593 (p-value = 0.0029, 95% confidence interval), which indicates a strong positive correlation, indicating that with an increase in clay content in the soil, the CO_2-C content also increases. In contrast, in the case of soils from Zakroczym, the coefficient r=-0.3896 (p-value = 0.3877, 95% confidence interval) indicates a negative relationship between the CO_2 content and clay content. However, since p = 0.3877, the result is not statistically significant.

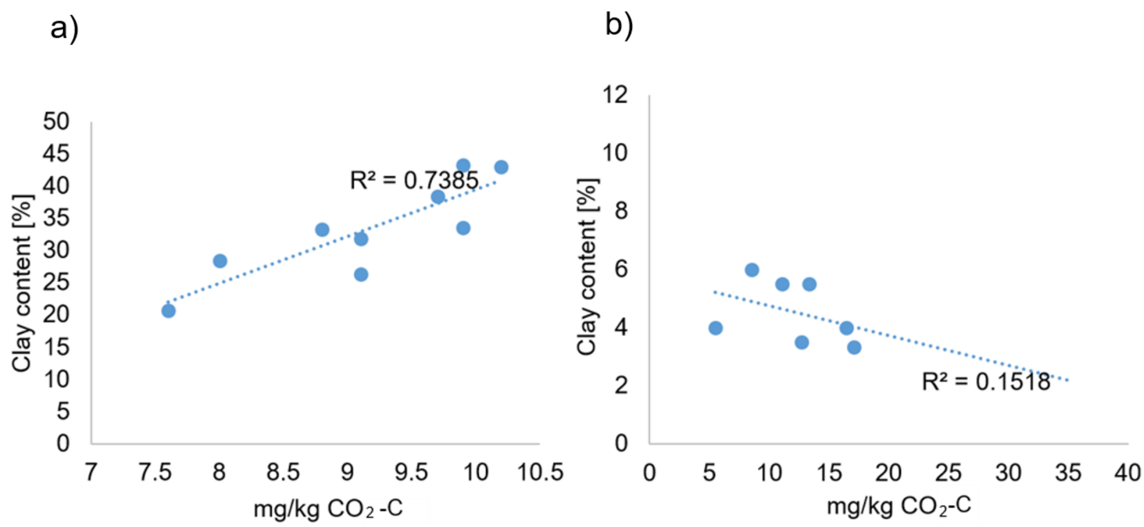


Figure 6.40. A study of the relationship between the content of clay fractions in the soil and the content of the CO_2-C: a) soils from Zdounky landfill, b) soils from Zakroczym landfill.

In the soils taken from Zakroczym, the average clay content was 5%, whereas in Zdounky it was 35.5%; therefore, higher CO_2-C values would have been expected for the soils taken from the Zdounky landfill. Nevertheless, the analysis conducted did not show higher CO_2-C values from the Zdounky landfill.

The CO_2 content may also depend on soil moisture, as shown in a study by Dilekoglu and Sakin (2017), who found a decrease in CO_2 emissions with an increase in moisture content. The soil CO_2 emission was measured as 54.47 g CO_2/m^2/week when the soil moisture was minimum (1.46%) and the soil moisture was maximum (18.77%) when the soil CO_2 emission was measured as 49.89 g CO_2-C/m^2/week. (Dilekoglu & Sakin (2017). The average CO_2 concentrations in the samples from Zdounky, according to the study in Section 6.2.1, had w_n = 24.85%, while those from Zakroczym had w_n = 11.77%, so the reduced CO_2-C emissions from the Zdounky landfill compared to the Zakroczym landfill may be affected by moisture content in the soils. To confirm this correlation, a Pearson correlation test was performed, which showed a statistically significant relationship between the natural moisture content of the soil and the CO_2-C content (r= -0.7529731). Therefore, it can also be concluded that the higher the amount of water in the soil, the lower is its microbiological activity.

In conclusion, the respiration study showed that soils watered with landfill leachate exhibited higher biological activity and fertility than soils with water, where the values were much lower, indicating the important role of leachate recirculation in landfills. At the same time, the analysis suggests that specific physicochemical conditions play a key role in respiration processes and require further research to optimize land use for landfill reclamation.

6.4.3. Summary of biomonitoring studies

This biomonitoring study included an analysis of leachate phytotoxicity and an assessment of soil respiration, which provided a comprehensive picture of the impact of landfill-derived pollutants on the environment. In the first part of the experiment, the effects of leachates on the germination of *Sinapis alba* L. seeds were assessed using the Phytoxkit and Phytotestkit assays. Root length measurements showed significant differences between the control and test samples, clearly indicating the toxic nature of the leachates, which was more pronounced in the tests conducted on solid test. Statistical analyses using Kruskal-Wallis and Wilcoxon tests confirmed these differences were significant, highlighting the sensitivity of the applied research methods. The second part of the study, which focused on soil respiration, revealed that soils irrigated with leachates exhibited significantly higher biological activity, as evidenced by the increased CO_2-C content. These results suggest that leachates not only negatively affect germination processes, but also stimulate soil microbial activity, which may lead to differences in soil fertility and reclamation potential. These findings underscore the importance of biomonitoring as a complement to traditional chemical and physical analyses. Due to their rapid response to stress factors, plants serve as early and highly valuable indicators of environmental change. Monitoring shifts in plant species composition allows the detection of even subtle signs of pollution, which may go unnoticed in purely chemical measurements. Integrating biomonitoring methods into standard procedures for assessing landfill impacts enables the early detection of degradation trends, identification of pollution migration pathways, and improved planning of reclamation efforts. As dynamic bioindicators, plants offer a holistic view of environmental conditions, making them indispensable components of comprehensive assessments of landfill impacts on ecosystems.

7. Discussion and Recomendations

A comparative analysis of the efficiency of technical and biological reclamation of MSW landfills in Zakroczym and Zdounky revealed significant differences resulting from the applied isolation methods. These differences were evident in terms of LFG emission, leachate production and quality, as well as the impact on the soil-water environment and the technical safety of the structures. Tab. 7.1 presents a comparison of the obtained research results with literature data, thanks to which greater transparency is noted compared to cover systems.

7.1. Comparison of the studied cover systems in the context of literature research

Table 7.1. Comparison of the studied cover systems in the context of literature research.

Own research		Literature approach
Mineral cover	**Synthetic cover**	
Landfill gas emission and its treatment		
• LFG emission 36.85 m^3 CH_4/1Mg MSW per year, 55.27 m^3 CO_2/1Mg MSW per year. • LFG composition monitoring showed a wide range of CH_4 levels (from nearly 0% to over 60%), with an average of 29.53% CH_4, suggesting diffuse emissions. • No statistically significant difference in CH_4 content (%) before and after reclamation, confirmed by Wilcoxon rank sum test.	• LFG emission 41.64 m^3 CH_4/1Mg MSW per year, 62.46 m^3 CO_2/1Mg MSW per year. • Minor variation in CH_4 levels (from 23.20% to over 38.70%), with an average of 31.60% CH_4; low CH_4 may be due to an on-site composting facility reducing organic content. • No statistically significant difference in CH_4 content (%) before and after reclamation, confirmed by ANOVA. • Higher gas production at the beginning of degradation until waste dries out, shortening the process.	• Gas emissions in mineral covers are diffuse and less effectively controlled (Cossu and Garbo, 2018). • Mineral covers result in partial CH_4 loss to the atmosphere, reducing biogas recovery efficiency (max production 80 m^3 CH_4/h vs. max collection 50 m^3 CH_4/h). GM systems retain more biogas (e.g., 140 m^3 CH_4/h produced, 130 m^3 CH_4/h recovered). Gas production under mineral covers lasts up to 30 years; under synthetics up to 22 years (Staub et al., 2011). • Jain et al. (2021) showed that with 508 mm annual rainfall unit gas

Own research		**Literature approach**
Mineral cover	**Synthetic cover**	
		production was 47 m³ CH_4 per 1 ton of MSW. • Kale and Gökçek (2020) estimated the emission for one of the analyzed landfills at 73.8 m³ CH_4 from 1 ton of MSW (landfill opened in 2012 and closed in 2031)
CH_4 oxidation potential		
• CH_4 oxidation: 7%.	• No oxidation due to the synthetic sealed landfill cover.	• According to Bian et al. (2021), CH_4 generation and oxidation vary with cover type: oxidation ranges from <10% to 100%. • Sadasivam and Reddy (2013) performed column tests in which the oxidation rate for clayey sand was 7%.
Leachate production and pollution level		
• Leachate volume: 626.4 m³/ha (18% of precipitation). • Average level of selected chemical properties of leachates: pH =8.32 [-] EC = 6 803 µS/cm Cr (VI) = 0.02 mg/l NH_{4+} = 114.4 mg/l P_{total} = 2.92 mg/l • LPI: min = 3.59, avg = 4.43, max = 7.70. • It can be assumed that rainwater has diluted the runoff, which is why it has lower pollutant loads.	• Leachate volume: 141.2 m³/ha (2.9% of precipitation). • Average level of selected chemical properties of leachates: pH = 8.38 [-] EC =10037µS/cm Cr_{total} = 0.77 mg/l Zn = 0.27 mg/l NH4+ = 408.6 mg/l P_{total} = 6.30 mg/l • LPI: min = 7.5, avg = 10.07, max = 13.5. • HDPE GM limited infiltration, increasing pollutant concentration.	• Podlasek (2023): 803-818 m³/ha/year. • Alslaibi et al. (2013): 453 m³/ha/year (Mediterranean, 322 mm rainfall). • Ng et al. (2019): synthetic cover percolation ~26-28 mm (2-3.1% of rainfall); CCL: 65 mm (7.5%). • Active landfill leachate discharge LPI should not exceed 5.696 (Hussein et al., 2019). • Abunama et al. (2021): LPI varies by leachate age:

Own research		Literature approach
Mineral cover	**Synthetic cover**	
		<5 yrs = 26.5; 5-10 yrs = 23.6; >10 yrs = 17.5. • Mineral covers enable to more oxygen and rainwater intake, increasing microbial diversity and pollutant mobility; synthetic covers lower redox potential, promoting anaerobic microbes like methanogens what enables immobilization of metals by precipitating them as sulfides (Morita et al., 2023).
Leachate recirculation potential		
• Recirculation is possible, which will have the effect of diluting the pollutant load carried by leachate by mixing it with precipitation, nevertheless for this purpose a tight gas capture system is needed.	• Recirculation only via injection under GM.	• Tight landfill covers require special leachate injection through a vertical borehole (Guérin et al., 2004). • Benson et al. (2012): GM-covered landfill recirculation can cause gas pressure buildup, risking slope instability. Synthetic cover may concentrate leachate pollutants due to a lack of rainwater infiltration.
Rainfall impact		
• Rainwater infiltrates the waste, generating leachate, especially during heavy rains.	• Rain causes surface runoff; even intense rain does not significantly increase leachate volume.	• Rain increases leachate and stimulates organic matter degradation (Luo et al., 2020; Miao et al., 2019).
Soil contamination in cover layers		
• Selected soil chemical properties: EC = 666.11 μS/cm pH = 7.81[-]	• Selected soil chemical properties: EC = 465.27 μS/cm pH = 7.32 [-] Zn = 29.5 mg/kg d.m. Pb =11.81 mg/kg d.m.	• Makuleke & Ngole-Jeme (2020), Wu et al. (2022): deeper landfill soils often show higher contamination and environmental risk.

Own research		Literature approach
Mineral cover	**Synthetic cover**	
Zn = 143 mg/kg d.m. (or 70 mg/kg d.m. after excluding outlying samples) Pb = 31.61 mg/kg d.m. Cu = 30.91 mg/kg d.m. Ni = 10.03 mg/kg d.m. Cd = 2.09 mg/kg d.m. • HMs order: Zn > Pb > Cu > Ni > Cd • Fluctuations and occasional exceedances (mainly Zn); deeper layers show higher concentrations, indicating insufficient isolation.	Cu = 29.50 mg/kg d.m. Ni = 36.92 mg/kg d.m. Cd = 0.71 mg/kg d.m. • HMs order: Zn > Ni > Cu > Pb > Cd; narrow data spread. • No exceedances in any sample.	
Groundwater impact		
• Monitoring indicates overall good groundwater quality in Zakroczym. • Zn and Cr^{6+} concentrations are well below WHO (2017) limits. • EC anomalies observed from 2008–2015 stabilized after 2011 (post-reclamation), indicating water quality improvement.	• In Zdounky, despite very low Zn levels and acceptable pH and EC (Class I standards), total Cr levels were elevated from 2008–2018, later dropping to levels acceptable for Class II waters by late 2018.	• Podlasek et al. (2021) found that the HMs Evaluation Index (HEI) before and after the landfill in Zdounky was 1.20 and 1.19 respectively, indicating the landfill had nearly the same impact on groundwater contamination as surrounding areas.
Material durability and percolation		
• Zakroczym cover mostly consists of clSa with hydraulic conductivity $k \leq 10^{-6}$ m/s and up to 82.5% sand content, with low Cl fraction (max 10%). • Soils classified as low plasticity (Ip ≤ 10%).	• Zdounky cover included cohesive soils such as Cl and sasiCl with Sa fraction up to 44%, Si 45%, and Cl up to 46%. • Soils were classified as medium and high plastic. • HDPE GM 1 mm used with $k \leq 10^{-15}$ m/s.	• CCL systems showed percolation after 2 years, increasing after 4 years to 0.15 mm/d due to stress cracking. Synthetic covers maintained stable percolation (~0.01 mm/d over 5 years) (Manassero et al., 1997). • Extreme temperatures can reduce mineral cover hydraulic conductivity, increasing percolation

Own research		Literature approach
Mineral cover	**Synthetic cover**	
		(Ojasanya & Dewoolkar, 2024). • GM aging is accelerated by UV exposure and insufficient protection; mechanical stress can cause damage (Sun et al., 2019). • HDPE GM lifespan at MSW landfills estimated at ~160 years (Rowe, 2005).
Stability limitations of analyzed landfill		
• Zakroczym landfill, despite steep slopes (2H:1V) and 18.5 m waste thickness, maintained safe stability conditions (FS ≈ 1.5); however, minimal margin exists—any strength parameter failure could reduce stability. • Main stability risks: steep slope and high waste thickness.	• Zdounky landfill showed no risk of GM-soil interface sliding (FS = 2.23), though slope was 3H:1V and waste thickness 13 m. • Slope change to 2H:1V yielded FS = 1.52—close to the stability threshold. • Bishop method showed FS = 2.455 for 13 m waste thickness. • GM has lower shear strength at interfaces, posing risks in wet conditions, potentially limiting final cover thickness and slope.	• Geosynthetic-covered slopes are designed at 3H:1V or flatter; mineral-covered slopes are typically steeper (e.g., 2H:1V) (Chetri, 2021). • Mineral covers offer favorable mechanical conditions along slopes (Cossu & Garbo, 2018). • GM use on 1V:4H slopes has caused slippage (Benson et al., 2012; Zhao & Karim, 2018); careful material selection is essential due to varying shear strength and fluid pressure effects (Liu et al., 1997).
Cover system construction difficulty		
• On-site soils meeting physical-mechanical criteria may be reused for reclamation. |	• GM requires long-distance transport and complex installation. • Higher investment needed (materials, labor: installation, sealing inspection).	• Mineral covers require heavy machinery for clay compaction (La Rocca, 2024). GM installation demands precise subgrade preparation (uniform, fine-grained, void-free) and careful welding.

Own research		Literature approach
Mineral cover	**Synthetic cover**	
	• Efficient biogas recovery with GM may offset higher cost via energy return.	• During installation, the number of seams is minimized with the control of power supply parameters. This process is correctly controlled by wire models, operating parameters with seam geometry and long-term switch, which complies with technical standards (Müller and Wöhlecke, 2019).
Material cost		
		• CCL system: 45 cm low permeability clay + 15 cm erosion layer = \$19,684, CCL-GM system: \$19,684 + 1.5 mm HDPE GM = \$39,383; HDPE GM alone = \$19,662. Possible to reduce mineral layer to cut costs (Janga & Reddy, 2024).
Life Cycle Assessment		
		• Mineral cover total carbon emissions (material, transport, construction): 19.0 kg CO_2/m^2; global warming potential (GWP): 100%. Synthetic cover: total carbon emissions = 14.0 kg CO_2/m^2; GWP = 92% (Ng et al., 2024). • Cumulative energy demand: 1061 GJ (mineral) vs. 235.7 GJ (synthetic) (La Rocca, 2024).
Leachate phytotoxicity		
• Highest *IG* in solid test: 22.2%; highest *IR*: 61.11%. • Strongest root growth stimulation in liquid *IR* = -	• Highest *IG* in solid test: 88.89%; highest *IR*: 79.44%.	• Wdowczyk & Szymańska-Pulikowska (2021): *IR* avg = 31.15% (100% concentration); max = 100%, min = -9.4%.

Own research		**Literature approach**
Mineral cover	**Synthetic cover**	
51.17%, but with *IG* = 12.5%, indicating root growth despite limited germination—suggesting possible leachate stimulation in some seeds. • Only 1 liquid test sample improved *IG* over control. • Mean *IR* (regardless of test type): 15.83%; mean IG: 7.22%. • 1 out of 6 samples classified as highly toxic (*IR*). • Mineral cover leachates showed lower *IG*/*IR* averages, fewer highly toxic samples, suggesting more moderate impact on *Sinapis alba* L. germination and growth.	• Strongest root growth stimulation in liquid IR = -95.7%, with *IG* = 12.5% suggesting a paradoxical positive effect on some seeds. • Only 1 liquid test sample improved IG over control. • Mean *IR*: 24.5%; mean *IG*: 28.47%. • 3 out of 6 samples classified as highly toxic (*IR*). • Synthetic cover leachates showed greater variability and extreme values, including possible positive effects on root growth, but overall higher toxicity (*IG* and *IR*).	• Palm et al. (2022): leachate at 50-70% dilution had a positive effect on *S. alba* L. and *T. aestivum* L. biomass and leaf growth. • Phytotoxicity tests showed that leachate at low concentrations can promote plant growth. At higher concentrations (50 and 100%), leachates caused inhibition of root and shoot growth (Wdowczyk and Szymańska-Pulikowska, 2021).
Soil respiration assessment		
• Soils irrigated with leachate (avg 63.80 ppm CO_2-C) showed medium-high microbial activity and better fertility than water-irrigated samples (avg 15.3 ppm CO_2-C). • Pearson correlation coefficient ($r = -0.389559$, $p = 0.3877$) suggests a negative relation between CO_2 content and clay fraction though not statistically significant ($p > 0.05$).	• In synthetic cover, leachate-irrigated soils had higher CO_2-C (avg 19.17 ppm) vs. water (avg 9.14 ppm), but overall low biological activity, 44% of samples exceeded the lower medium fertility threshold. • Correlation coefficient $r = 0.8593$ ($p = 0.002984$), indicating strong positive relation between clay content and CO_2-C.	• Research by Djerdi et al. (2021) showed that an increase in the concentration of the chemical causes an increase in CO_2 emissions. • Jo et al. (2022) observed that increasing CO_2 concentration to 800 ppm doubled the SPAD value (45.87 vs. 25.85) and increased the mean fennel length from 49.81 cm to 57.64 cm. • Findings align with literature (Rogers et al., 1983; Dilekoglu & Sakin, 2017; Djerdi et al., 2021;

Own research		Literature approach
Mineral cover	**Synthetic cover**	
		Jo et al., 2022) showing CO_2 emissions correlate with microbial activity, yield, fertility, moisture, and clay content, supporting the positive effect of leachate recirculation at landfills.

Due to the noticeable differences observed between the studied covers above, it was decided to divide the elements examined in the Tab. 7.1. into main categories: I – Technical aspects, II – Environmental resilience, III – Economic resilience, and IV – Social resilience, to assign to them elements that have either a negative or positive impact on the given function. Each of the studied elements was assigned a rank from -1 to 1, where: -1 – indicates a significant negative impact on the analyzed component, 0 – indicates a neutral impact, and +1 – indicates a noticeable positive impact. This ranking was applied to help answer the question of which of the studied covers is more beneficial. Appendix 3 contains the table with the assigned values grouped by category. The Fig. 7.1. presents a graphical summary of the analysis, which clearly indicates a more positive impact of the synthetic cover compared to the mineral cover. The synthetic landfill cover exhibits a more favorable overall performance compared to the mineral cover, as evidenced by its consistently higher positive rankings across multiple evaluation criteria. Notably, it demonstrates superior technical efficiency (e.g., permeability and mechanical resistance), enhanced environmental resilience (e.g., reduced leachate contamination and greater resistance to extreme weather), and improved economic and social outcomes, including lower operational costs and higher energy recovery potential.

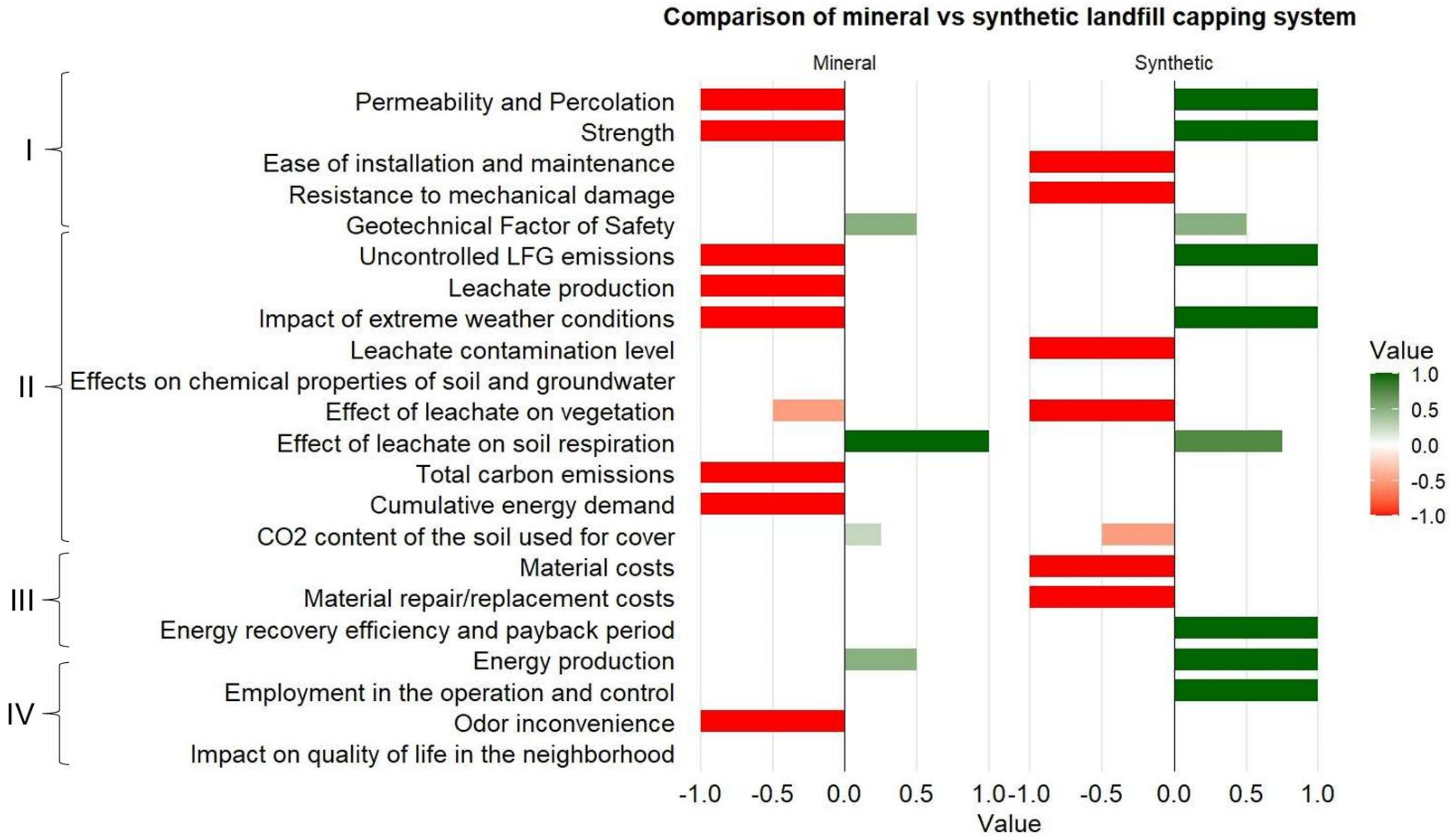


Figure 7.1. Final comparison of mineral and synthetic landfill cover system.

7.2. Proposal of recommendations on how to cover the MSW landfills

Based on the detailed analysis of the results of the own research conducted and data from the literature, the following recommendations can be made for the selection of the type of landfill cover:

Recommendations for the use of synthetic cover (GM HDPE):

1. **When the priority is to reduce leachate production**, synthetic covers significantly minimize rainwater infiltration (approximately 3% of precipitation), effectively reducing the volume of generated leachate compared to mineral covers (approximately 20% of precipitation).
2. **In cases where efficient LFG management and maximization of biogas recovery are essential**, HDPE GM provide a highly impermeable barrier, enabling CH_4 capture rates of up to 93% and significantly reducing uncontrolled LFG emissions.
3. **In areas with limited availability of suitable soils for reclamation in terms of both quality and quantity**, geosynthetics eliminate the need for transporting large volumes of mineral soils, although they require a higher initial investment for installation.
4. **On landfills with moderate slope gradients (gentler than 1V:3H)**, synthetic covers offer improved structural stability and enhance the overall safety of the landfill cover system.
5. **In locations exposed to intense rainfall or extreme temperatures**, geosynthetics provide superior resistance to percolation and more effectively limit leachate generation.
6. **On landfills planned for post-closure use with low vegetation (e.g., grassland or meadow ecosystems)** due to the limitations of root penetration through the synthetic GM.
7. **When a faster return on investment is desired through efficient energy recovery from biogas and climate impact mitigation**, synthetic cover systems demonstrate lower CO_2 emissions, reduced global warming potential, and significantly lower cumulative energy demand throughout their life cycle.

Recommendations for the use of mineral cover:

1. **In areas with high availability of local mineral materials**, the use of mineral cover can reduce transportation and material costs, as well as shorten installation time, despite the need for heavy construction equipment to compact the cover layers.
2. **On landfills with steeper slope gradients (up to 1V:1.5–2H)**, mineral covers offer higher shear strength at the interface with the subgrade and greater mechanical stability.
3. **In cases where LFG emissions are not a primary concern**, and the focus is on promoting microbiological activity and enhancing organic matter decomposition, mineral covers allow oxygen ingress, supporting aerobic microbial processes and partial CH_4 oxidation.
4. **When the priority is to reduce leachate toxicity**, mineral covers generate leachate with lower toxicity levels and less harmful effects on vegetation, making it possible to recirculate leachate without damaging the biologically active layers of the cover system.
5. **On landfills with post-closure plans involving the establishment of tall or deep-rooted vegetation**, mineral covers are more suitable due to their compatibility with such revegetation goals.
6. **In scenarios with less stringent requirements for LFG control and limited budgets**, mineral covers offer a more affordable initial investment. However, they are less effective and may involve higher maintenance costs over time.

The selection of a technical cover system should always consider local soil and hydrological conditions, the intended operational objectives of the landfill (such as energy recovery, emission control, and investment and operational costs), as well as the slope of the terrain. Synthetic covers are particularly recommended due to their more favorable environmental and energy-related performance and higher efficiency in biogas recovery. However, the final decision should be based on the planned direction of site reclamation and the availability of financial resources.

8. Conclusions and further research

8.1. General conclusions

The analysis carried out confirmed the impact of technical methods of reclamation on MSW landfill processes. The use of a synthetic layer in the cover effectively reduced the infiltration of rainwater, reducing the amount of leachate generated by four times compared to a mineral cover. Nevertheless, it was noted that the use of a GM was associated with an increase in the concentration of HMs and a higher LPI, indicating the need for more frequent monitoring of environmental pollutants for this type of isolation technology. The analysis of the physical and chemical properties of the soils at the studied landfills indicated significant differences that directly impacted the operation of the facilities. The dominant soils in Zakroczym are clayey sand soils characterized by higher CO_2 content, better aeration, and more intensive microbial activity, which positively affect the biodegradation rate of organic matter; however, an increase in HMs concentrations with depth was noted. In contrast, in Zdounky, there are clayey soils, in which no changes in HMs concentrations with depth were noted.

An analysis of LFG emissions and composition confirmed the significant effect of the type of cover used (mineral or synthetic) on the intensity and nature of CH_4 emissions. Despite lower specific CH_4 emissions per 1 Mg MSW with mineral cover, there was much greater variability and instability of CH_4 concentrations in LFG, suggesting the presence of diffuse emissions that are difficult to control. The synthetic cover, although it showed slightly higher specific emissions, guaranteed more stable and predictable gas conditions.

The use of synthetic covers generally shows more favorable environmental and operational effects than mineral covers. Synthetic covers are more effective in reducing uncontrolled LFG emissions and leachate generation, have better resistance to extreme weather conditions such as heavy rains or frost, and promote higher energy recovery efficiency. Mineral covers, on the other hand, are characterized by lower material costs, easier access to local raw materials, and the possibility of a higher thickness of landfilled waste, which may justify their use in specific local conditions.

This research makes a significant contribution to the development of the disciplines of civil engineering, geodesy, and transport, as it provides key information on the optimization of landfill design and reclamation, considering structural safety, the effectiveness of cover systems, and their impact on climate change. Simultaneously, it emphasizes the need for continuous monitoring of soil conditions and their strength parameters, gas emissions, leachate production, and the effectiveness of the applied protective methods. The obtained results can be used both in the design of new facilities and in the optimization of existing ones, with particular attention paid to the selection of appropriate geotechnical materials and structural parameters, which limit the negative environmental impact and ensure the long-term stability of landfills. The results of the predictive modelling of leachate generation, LFG emissions, and structural stability analyses represent a substantial contribution to landfill design, monitoring, and management. They enable the formulation of justified recommendations, using a holistic approach, encompassing several scientific disciplines, for selecting for the selection of isolation materials, considering economic, technical, and environmental aspects.

As a result, recommendations were developed indicating a preference for the use of geosynthetics in landfills, where the priorities are maximizing biogas recovery and reducing the amount of generated leachate. In contrast, mineral covers have been proposed as an alternative solution in cases that require low initial investment and integration with natural environmental conditions. However, it should be noted that such covers require enhanced supervision and effective management of diffuse emissions.

The results of the analyses made it possible to unequivocally confirm the research hypothesis, according to which the type of insulation material used affects the quantity and composition of landfill gas and generated leachate. The research and analysis carried out according to the adopted methodology allowed to fully realize the set objectives of the work and to prove the hypothesis.

8.2. Detailed conclusions

Monitoring studies:

1. Leachate analysis from 2008 to 2022 revealed that the Zdounky landfill, where an HDPE GM was used to limit rainwater infiltration, showed higher concentrations of HMs and a higher LPI (max. 13.5) compared to Zakroczym landfill, where the mineral reclamation layer contributed to the dilution of contaminants (max. LPI = 7.7).

2. Post-reclamation monitoring at the Zdounky landfill indicated an increase in CH_4 concentration in LFG (min–max: 23.2%–39.7%, median: 33.2%) compared to pre-reclamation values (min–max: 22.5%–34.8%, median: 31.5%), representing a 14% increase in CH_4 volume content. In contrast, the reclamation of the Zakroczym landfill resulted in only a 1.22% increase in CH_4 concentration (pre-reclamation min–max: 0.3%–65.7%, median: 48.5%; post-reclamation min–max: 0.3%–66.5%, median: 27%).

Laboratory studies:

1. Soils from the Zdounky landfill exhibited high cohesiveness, a substantial Cl fraction and higher plasticity, which may enhance the sorption of HMs. In contrast, Zakroczym soils were dominated by clayey sands with high sand content.

2. Soils from Zakroczym landfill showed greater variability and local exceedances of HMs limits (especially Zn), with a trend of increasing metal concentrations with depth—suggesting insufficient isolation and emphasizing the need for continuous monitoring of soil quality.

3. HMs concentration rankings in Zakroczym soils were as follows: Zn > Pb > Cu > Ni > Cd, however in Zdounky: Zn > Ni > Cu > Pb > Cd. It is noted that in both examined landfills the biggest problem is the concentration of Zn in the soil, while the most negligible impact is caused by Cd, the concentrations of which were below the detection level.

Modelling studies:

1. Leachate production modelling showed that mineral covers produced over 4-times more leachate per hectare (626.4 m^3) than synthetic covers (141.2 m^3). Annual percolation was estimated at 14.12 mm/year (2.9% of precipitation) for the GM cover and 90.58 mm/year (18% of precipitation) for the mineral cover.

2. Slope stability analysis indicated that, the FS > 1.5 in all scenarios. The highest slope stability was observed in Zdounky with the HDPE GM, a 1(V):3(H) slope, and a 13 m landfill height (FS = 2.23), with no slip risk. However, at a steeper slope of 1(V):2(H), FS dropped to 1.52, highlighting the limitations of GM on steeper slopes. Despite a steeper slope (1V:2H) and greater waste mass (18.5 m), Zakroczym achieved FS ≈ 1.5, indicating minimal safety margins when using mineral covers.

Biomonitoring studies:

1. Phytotoxicity tests showed that leachate from both landfills negatively affected seed germination and root growth of *Sinapis alba* L. This effect was particularly noticeable in solid-phase tests, which better simulate soil–leachate interactions. Leachates from Zdounky exhibited higher toxicity, confirmed by *IG*/*IR* indexes (mean *IR* in Zdounky: 24.12%, Zakroczym: 15.83%). However, statistical testing revealed no significant difference ($p = 0.061$) between solid-phase samples from both sites. The results also confirmed that different test formats (liquid vs. solid) reveal distinct toxicity profiles, with solid-phase tests being more sensitive to soil-like conditions.

2. Soil respiration studies found that soils irrigated with leachate had higher CO_2-C content, improved microbial activity, and greater fertility compared to those irrigated with water only. In the case of synthetic cover, although CO_2-C levels were higher in leachate-irrigated samples than in water-only ones, overall biological activity remained low. A strong positive correlation ($r = 0.8593$, $p = 0.002984$) between clay content and CO_2-C indicates that soil texture significantly affects CO_2 emissions. These findings support the conclusion that leachate recirculation enhances microbial activity and plant growth, suggesting its potential environmental benefits in landfill management.

Recommendations:

1. The choice of reclamation cover should be oriented to the landfill operational goals and local conditions, with synthetic covers - thanks to minimized infiltration, superior biogas recovery efficiency and more favorable environmental and energy performance - being the optimal option wherever emissions reduction and energy production are a priority, while mineral covers remain a better option when lower initial costs, availability of local land and the need for deep vegetation development are a key.

8.3. Suggested directions for further research

Issues related to landfill emissions represent an important area of research, particularly in the context of growing environmental challenges related to climate change and the development of new technologies. Existing solutions, such as the use of synthetic or mineral liners, while effective to some extent, still have certain limitations, such as percolation and leachate generation, or insufficient energy recovery from biogas, of which only approximately one-third is currently successfully converted.

Future research should focus on the development of advanced techniques using artificial intelligence (AI) to comprehensively analyze and optimize landfill cover systems. The complexity of this issue, which results from numerous interdependent environmental, technical, and economic factors, requires the use of AI tools to create accurate and sophisticated decision models. Further research is needed in the area of innovative cover materials, particularly those derived from recycled sources, as well as methods to support natural CH_4 oxidation and reduce stormwater infiltration. The use of alternative materials can significantly reduce the environmental impact and construction and maintenance costs of landfill covers.

The research directions I have chosen are driven by the need to find new, more efficient methods of managing emissions from landfills, an important step in the pursuit of sustainable development goals and taking steps to reduce climate change.

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- *Announcement of the Minister of Climate and Environment of 6 August 2022 on the publication of the consolidated text of the Regulation on landfills* (Obwieszczenie Ministra Klimatu i Środowiska z dnia 6 sierpnia 2022 r. w sprawie ogłoszenia jednolitego tekstu rozporządzenia Ministra Środowiska w sprawie składowisk odpadów, Dz.U. 2022 poz. 1902)
- *Announcement of the Minister of Infrastructure and Construction of 28 September 2016 on the publication of the consolidated text of the Regulation concerning the method of fulfilling obligations by industrial wastewater suppliers and the conditions for discharging wastewater into sewerage systems* (Obwieszczenie Ministra Infrastruktury i Budownictwa z dnia 28 września 2016 r. w sprawie ogłoszenia jednolitego tekstu rozporządzenia Ministra Budownictwa w sprawie sposobu realizacji obowiązków dostawców ścieków przemysłowych oraz warunków wprowadzania ścieków do urządzeń kanalizacyjnych, Dz.U. 2016 poz. 1757
- *Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste.*
- *Deponieverordnung*, Germany, 2009.
- Directive (EU) 2018/850 of the European Parliament and of the Council of 30 May 2018 amending Directive 1999/31/EC on waste.
- Ministry of Health and Welfare of Japan, 2001.

- *Regulation of the Minister of Maritime Economy and Inland Navigation of 11 October 2019 on the criteria and method for assessing the status of groundwater bodies* (Rozporządzenie Ministra Gospodarki Morskiej i Żeglugi Śródlądowej z dnia 11 października 2019 r. w sprawie kryteriów i sposobu oceny stanu jednolitych części wód podziemnych, Dz.U. 2019 poz. 2148)
- *Regulation of the Minister of the Environment of 21 December 2015 on the criteria and method for assessing the status of groundwater bodies* (Rozporządzenie Ministra Środowiska z dnia 21 grudnia 2015 r. w sprawie kryteriów i sposobu oceny stanu jednolitych części wód podziemnych, Dz.U. 2016 poz. 85)
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- *Resource Conservation and Recovery Act* (RCRA), U.S. Code of Federal Regulations (CFR) parts 239 through 282.
- The Landfill (England and Wales) Regulations, 1559/2002.

Norms:

- ČSN 83 8032 *Landfill sealing (Těsnění skládek)*.
- ČSN 83 8034 *Landfilling*.
- ČSN 83 8035 *Landfilling – Closure and Reclamation of Landfills* (*Skládkování odpadů – Uzavírání a rekultivace skládek*).
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- *Technical Code for Municipal Solid Waste Sanitary Landfill* (GB 50869-2013 and GB 51220-2017).

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Appendix

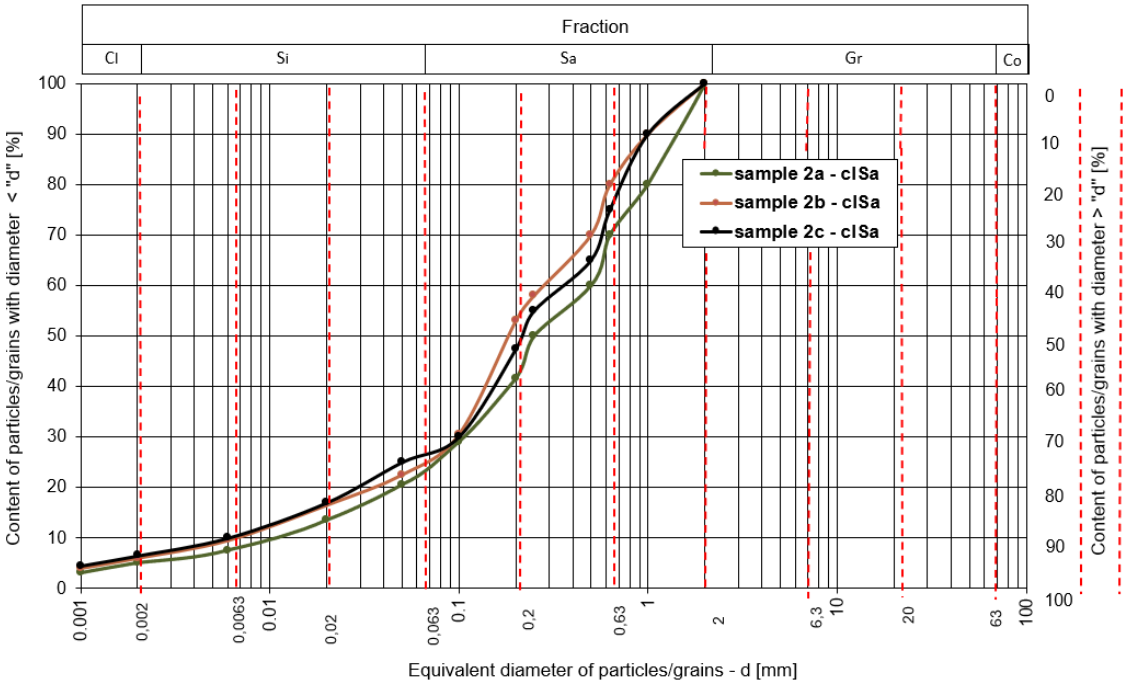
Appendix 1. Meteorological conditions used in the HELP model.

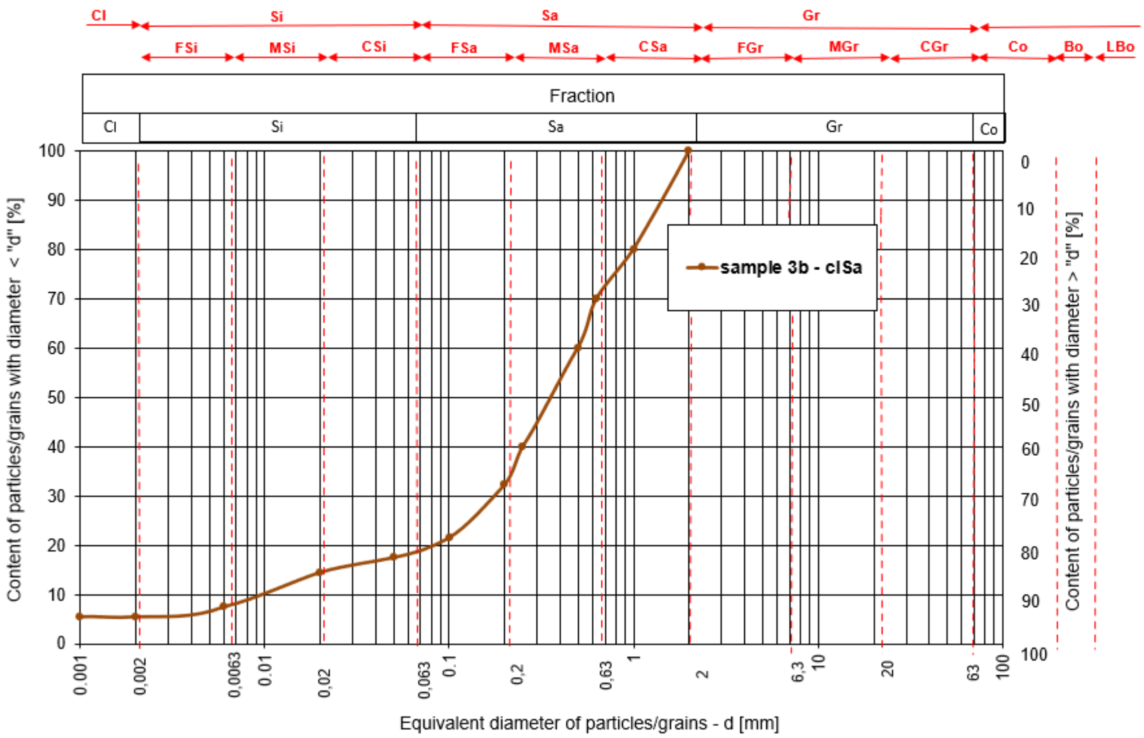
Warszawa Bielany Meterological Station				
Period 2016-2022	Precipitation [mm]	Temperature [°C]	Wind speed [m/s]	Humidity [%]
January	32.6	-1.6	3.0	85.1
February	35.0	1.4	2.9	81.5
March	28.5	4.5	2.7	62.5
April	30.5	9.8	2.8	62.3
May	56.1	14.6	2.5	64.1
June	60.3	20.2	2.3	53.5
July	79.3	20.2	1.8	68.8
August	65.3	20.0	2.1	70.4
September	44.8	14.7	2.2	65.5
October	59.0	10.1	2.6	83.2
November	25.3	5.1	2.7	74.4
December	40.4	1.7	3.0	87.7
Kromieryż Meterological Station				
Period 2016-2022	Precipitation [mm]	Temperature [°C]	Wind speed [m/s]	Humidity [%]
January	23.3	-0.6	2.0	81.7
February	31.5	2.1	2.3	76.3
March	18.7	5.3	2.1	66.1
April	37.8	10.1	2.2	61.6
May	60.2	14.5	1.8	67.3
June	70.6	19.9	1.7	67.0
July	82.8	20.5	1.4	64.7
August	69.1	20.6	1.5	66.6
September	67.7	15.4	1.5	74.0
October	46.7	10.6	2.0	78.4
November	30.3	5.5	2.1	83.7
December	33.1	1.5	2.1	83.5

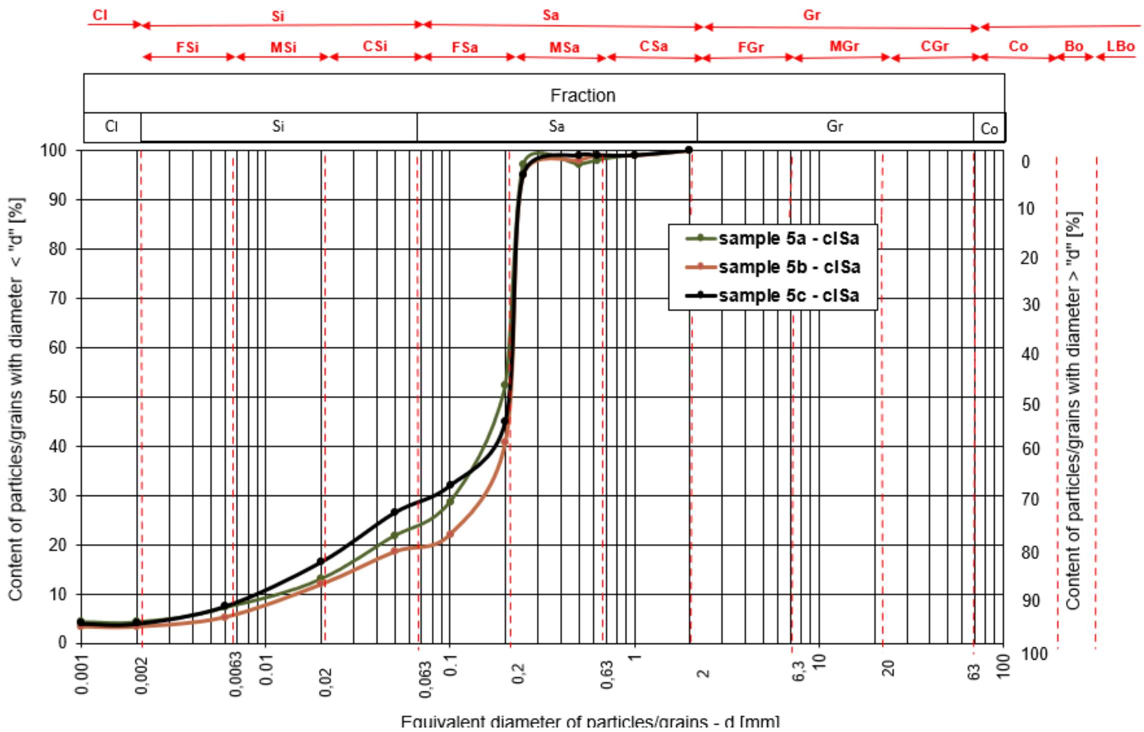
The annual insolation of the multi-year average for Zakroczym was 1943 h and for Zdounky was 1833 h.

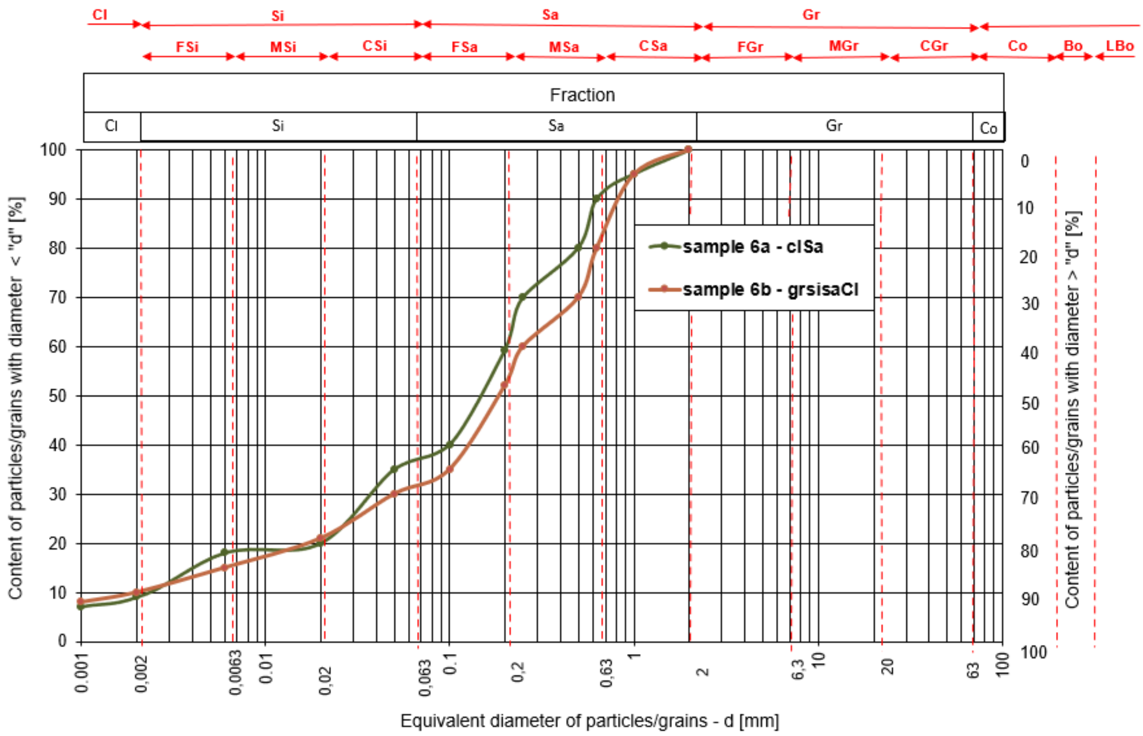
Appendix 2. Grain size curves of the tested soils.

a)

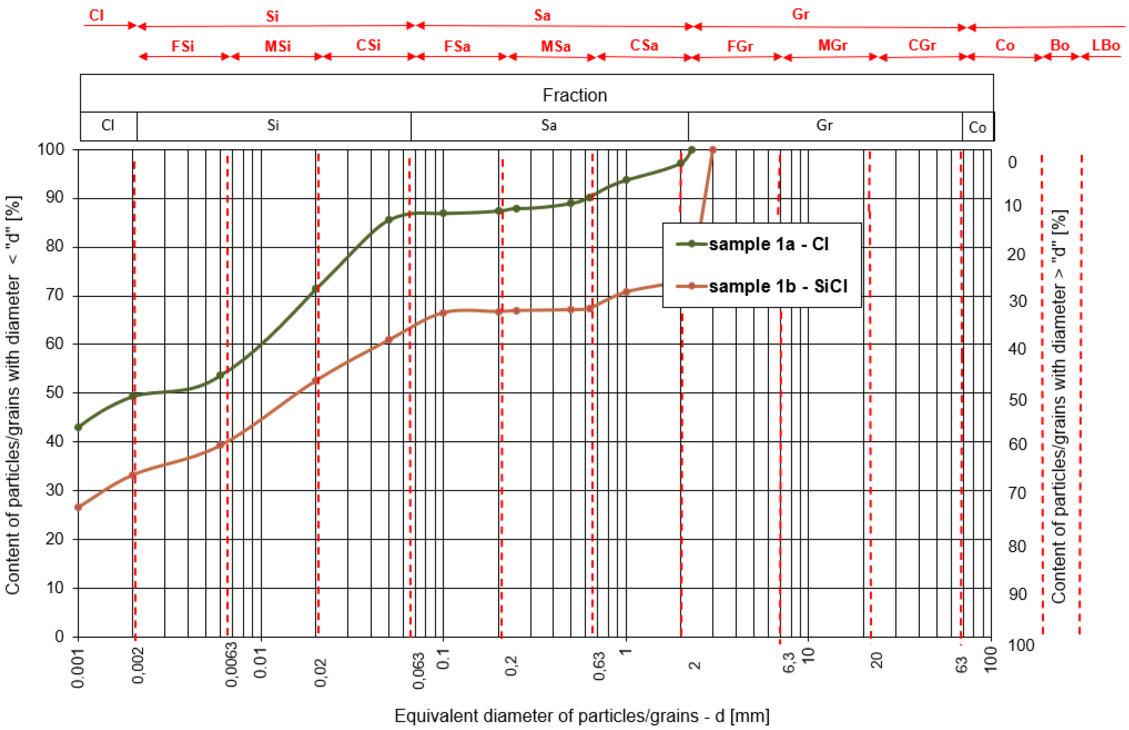




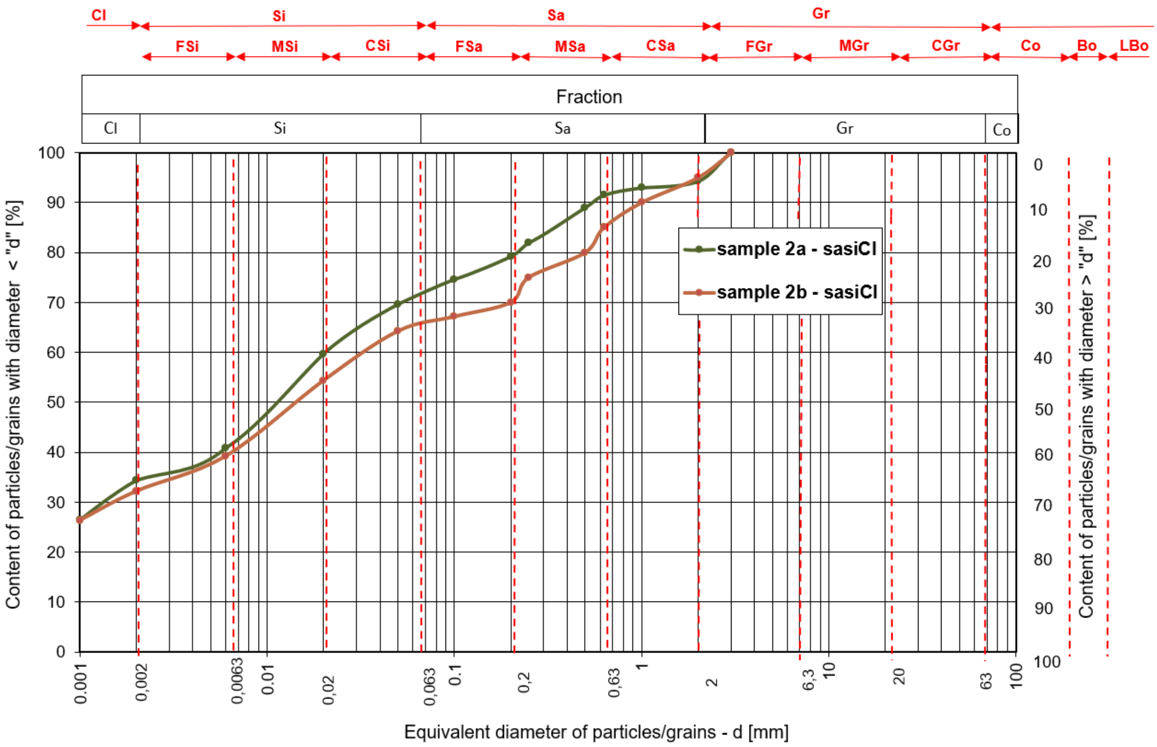

Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
sample 5a - clSa
sample 5b - clSa
sample 5c - clSa
Equivalent diameter of particles/grains - d [mm]


Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
sample 6a - clSa
sample 6b - grsisaCl
Equivalent diameter of particles/grains - d [mm]

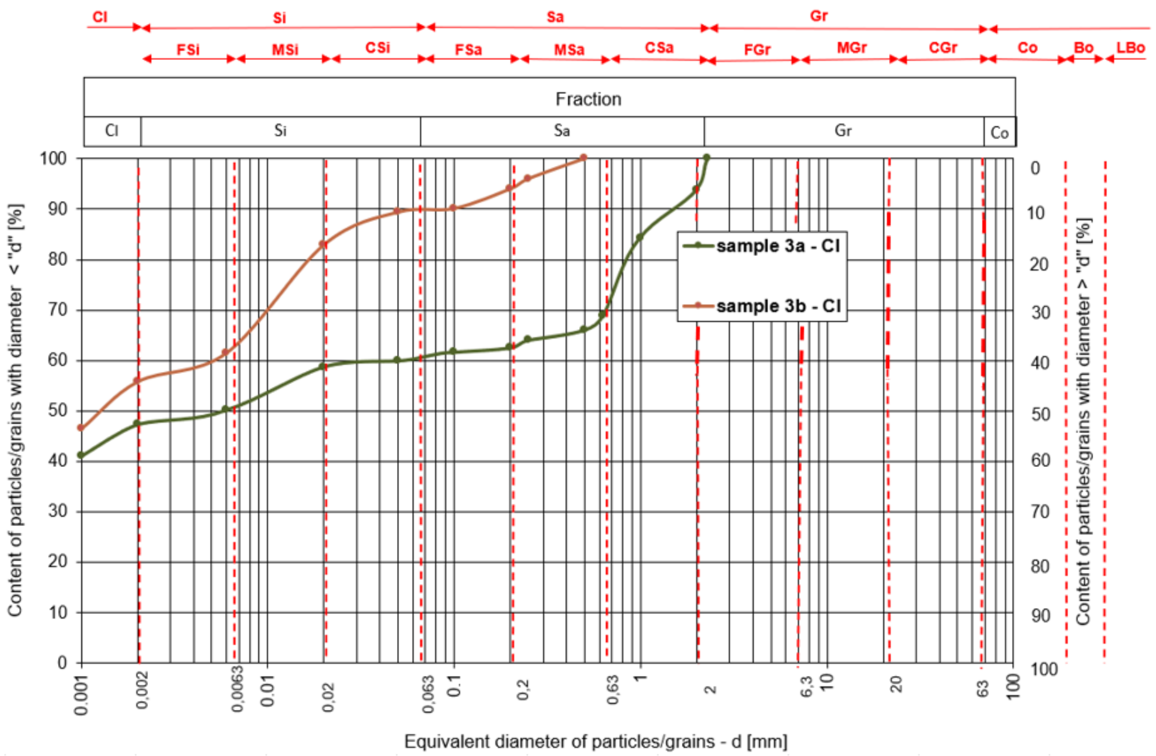
Zdounky:



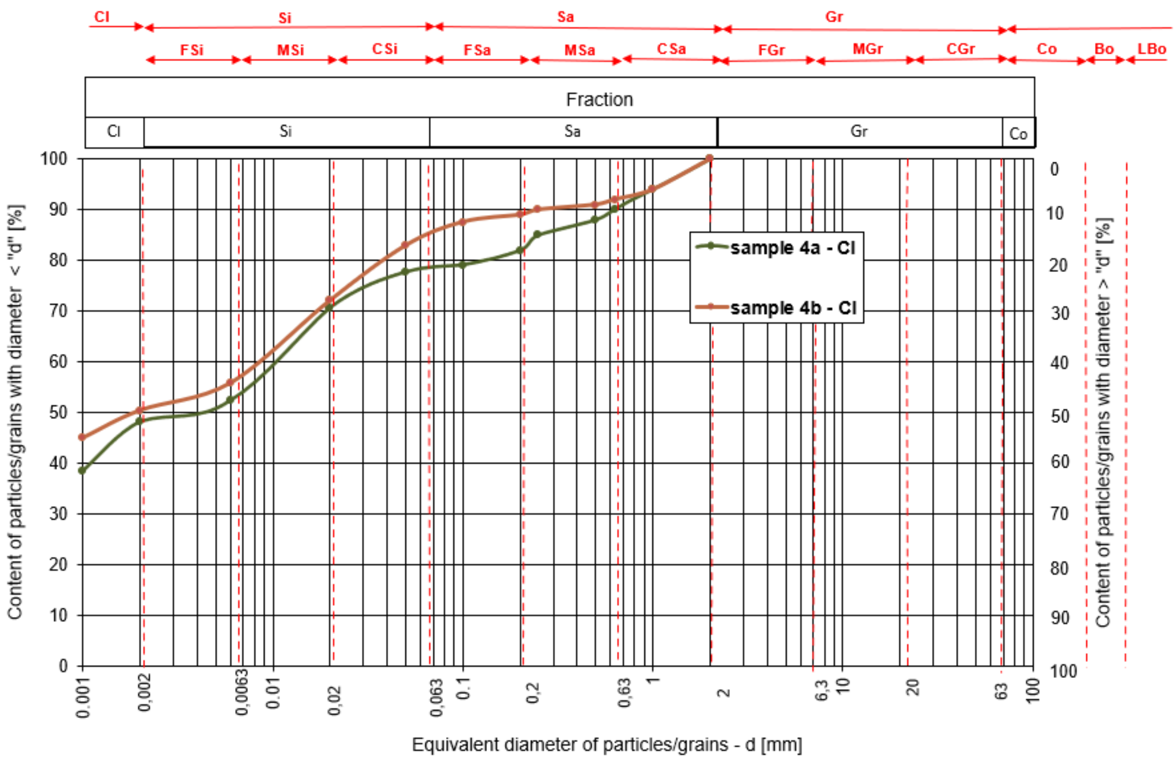
Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Cl
Si
Sa
Gr
Co
sample 1a - Cl
sample 1b - SiCl
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
Equivalent diameter of particles/grains - d [mm]



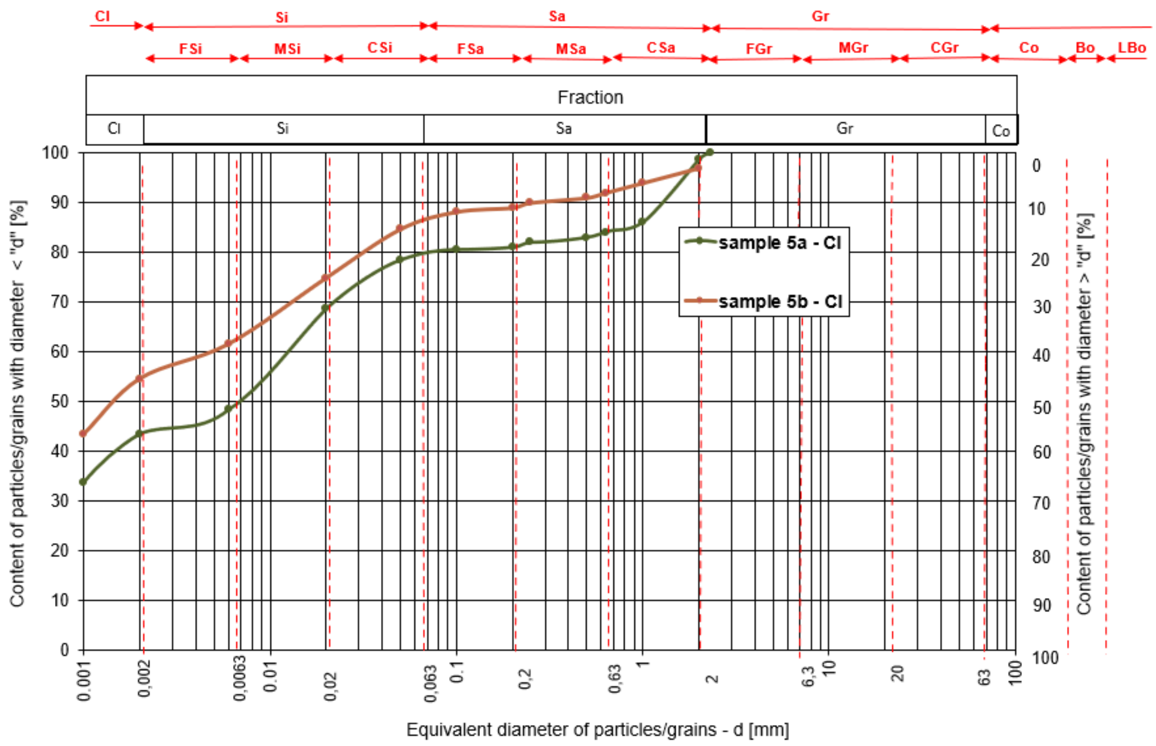
Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Cl
Si
Sa
Gr
Co
sample 2a - sasiCl
sample 2b - sasiCl
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
Equivalent diameter of particles/grains - d [mm]



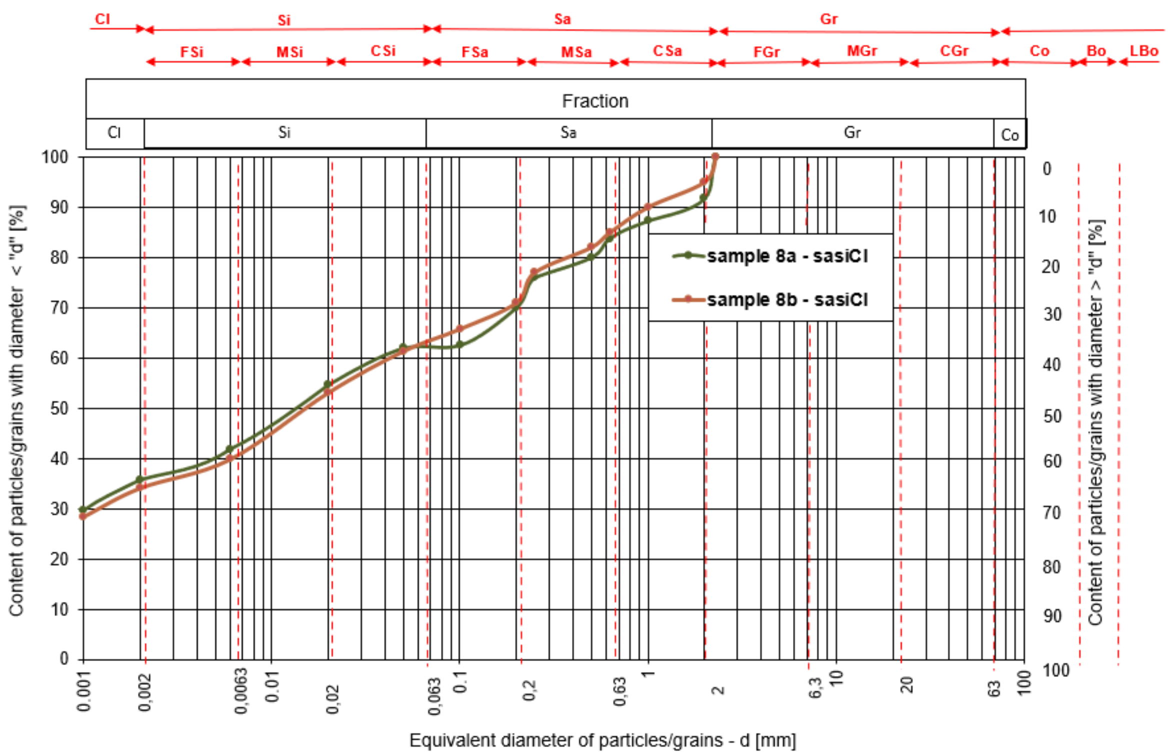
Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
sample 3a - Cl
sample 3b - Cl
0,001
0,002
0,0063
0.01
0,02
0,063
0.1
0,2
0,63
1
2
6,3
10
20
63
100
Equivalent diameter of particles/grains - d [mm]



Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
sample 4a - Cl
sample 4b - Cl
0,001
0,002
0,0063
0.01
0,02
0,063
0.1
0,2
0,63
1
2
6,3
10
20
63
100
Equivalent diameter of particles/grains - d [mm]



Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Cl
Si
Sa
Gr
Co
sample 5a - Cl
sample 5b - Cl
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
Equivalent diameter of particles/grains - d [mm]



Cl
Si
Sa
Gr
FSi
MSi
CSi
FSa
MSa
CSa
FGr
MGr
CGr
Co
Bo
LBo
Fraction
Cl
Si
Sa
Gr
Co
sample 8a - sasiCl
sample 8b - sasiCl
Content of particles/grains with diameter < "d" [%]
Content of particles/grains with diameter > "d" [%]
Equivalent diameter of particles/grains - d [mm]

Appendix 3. A table summarizing the parameters of the cover systems with evaluation.

Characteristics	Mineral cover system	Synthetic cover system
Technical aspects		
Permeability and Percolation	-1	1
Strength	-1	1
Ease of installation and maintenance	0	-1
Resistance to Mechanical damage	0	-1
Geotechnical Factor of Safety	0.5	1
Environmental resilience		
Uncontrolled LFG emissions	-1	1
Leachate production	-1	0
Impact of extreme weather conditions	-1	1
Leachate contamination level	0	-1
Effects on chemical properties of soil and groundwater	0	0
Effect of leachate on vegetation	-0.5	-1
Effect of leachate on soil respiration	1	0.75
Total carbon emissions (material, transportation, construction)	-1	0
Cumulative energy demand	-1	0
CO_2 content of the soil used for cover	0.25	-0.5
Economic resilience		
Material costs	0	-1

Characteristics	**Mineral cover system**	**Synthetic cover system**
Material repair/replacement costs	0	-1
Energy recovery efficiency and payback period	0	1
Social resilience		
Energy production	0.5	1
Employment in the operation and control	0	1
Odor inconvenience	-1	0
Impact on quality of life in the neighborhood	0	0
Total score	**-6.25**	**2.25**

Appendix 4. Graphical abstract of the research.