Desalination and Water Treatment
www.deswater.com
doi: 10.5004/dwt.2022.28949

275 (2022) 131–146
November

Temporal variations in groundwater chemical composition of landfill areas in the vicinity of agricultural lands: a case study of the Zdounky and Petrůvky landfills in the Czech Republic

Anna Podlasek[a,*], Magdalena Daria Vaverková[a,b], Eugeniusz Koda[a], Evan K. Paleologos[c], Dana Adamcová[b], Ayla Bilgin[d], Emily Rose Palm[e], Werther Guidi Nissim[f]

[a]*Institute of Civil Engineering, Warsaw University of Life Sciences – SGGW, Nowoursynowska 159, 02 776, Warsaw, Poland, emails: anna_podlasek@sggw.edu.pl (A. Podlasek), magdalena_vaverkova@sggw.edu.pl (M.D. Vaverková), eugeniusz_koda@sggw.edu.pl (E. Koda)*

[b]*Department of Applied and Landscape Ecology, Faculty of AgriSciences, Mendel University in Brno, Zemědělská 1, 613 00, Brno, Czech Republic, email: magda.vaverkova@uake.cz*

[c]*Department of Civil Engineering, Abu Dhabi University, Abu Dhabi, 59911, United Arab Emirates, email: evan.paleologos@adu.ac.ae*

[d]*Faculty of Engineering, Artvin Coruh University, Seyitler Campus 08000 Artvin, Turkey, email: ayla.bilgin@gmail.com*

[e]*Department of Agriculture, Food, Environment and Forestry (DAGRI), University of Florence, Viale delle idee, 30, 50019, Sesto Fiorentino, FI, Italy, email: emilyrose.palm@unifi.it*

[f]*Department of Biotechnology and Biosciences, University of Milano-Bicocca, Piazza della Scienza 2 – 20126-I Milano, Italy, email: werther.guidinissim@unimib.it*

Received 28 February 2022; Accepted 23 September 2022

ABSTRACT

The issue of groundwater contamination is one of several major global concerns, especially in areas exposed to the impact of waste disposal. The present study investigated the potential of combining both direct monitoring activities with computer-based model simulations to predict and describe contamination sources from two landfills and surrounding land in the Czech Republic (CR). The results have shown that groundwater quality at the two monitored landfills in the CR falls within the requirements set for pH values. Electrical conductivity (EC) values for the Petrůvky landfill showed homogeneity, however for the Zdounky landfill EC values were significantly higher. The concentrations of nitrate (NO_3^-) appear to have been affected both by the operation of the landfill and surrounding arable areas, although for both landfill sites, the average NO_3^- values were seen to meet the CR water quality requirements. The contents of most of the monitored indicators were found to be at the level of the natural background, therefore not exceeding the critical values set by regulatory agencies. It was demonstrated that random high concentrations of pollution indicators can potentially be explained by runoff of contaminants from the section of both landfills where tires and/or demolition wastes are stored. According to the Monte Carlo simulations it was also found that narrow contaminant plumes cannot be captured by a single downstream piezometer installed in the monitoring network. The outcomes presented in this work represent a novel aspect of environmental assessment as few studies have looked at groundwater quality in terms of temporal changes of contaminant indicators in locations where landfill facilities co-exist with agricultural lands.

Keywords: Groundwater; Monitoring; Contamination; Landfill facilities

* Corresponding author.

Presented at the 2nd International Conference on Strategies toward Green Deal Implementation – Water, Raw Materials and Energy (ICGreenDeal2021), 8–10 December 2021, held online by the Mineral and Energy Economy Research Institute, Polish Academy of Sciences

1944-3994/1944-3986

1. Introduction

Groundwater contamination by landfill leachate represents a major concern for the environment [1], especially in areas where few basic interventions are prescribed to minimize the leaching of contaminants [2]. For instance, disturbing results from the analysis of leachate samples from landfills in United States which shown the presence of pharmaceuticals, steroid hormones, animal and plant sterols, industrial and household chemicals, and other contaminants of emerging concern [3,4]. A number of scientific studies have demonstrated that the low quality of groundwater and surface water surrounding landfills is largely influenced by leachate percolation [5,6]. The pollution of surface water and groundwater is also considered a significant threat to human health [7]. Recently, Vaverková et al. [8] pointed out the increased danger from the release of viruses and other pathogens to the soil and groundwater resulting from the heavy pathogenic load of municipal solid waste (MSW) during epidemics, such as that observed during the COVID-19 pandemic.

To assess the risk of landfill operations on groundwater, regular monitoring of water quality is required [9]. The most common method is the use of groundwater monitoring wells to assess groundwater quality upstream and downstream of landfills [10]. Depending on the hydrogeological characteristics of a landfill's site, the composition, quantity, and location of release of the contaminants, and the monitoring system's spacing, number of wells, distance from the landfill cell that has leaked, and depth and frequency of sampling, the likelihood to detect contamination can be considerably low in some cases [11–13]. In the case of landfills, there are two routes identified as possible migration paths for of contaminants to groundwater groundwater: first, advective and dispersive transport and second, diffusive transport through geomembranes and clay liners [14]. Both of these pathways can affect the size of the contamination plume, and thus whether or not the position of the monitoring sites adequately captures the impact of the landfill on groundwater quality.

Alternative methods for monitoring groundwater quality have also been used when access to sampling points or instrumentation is limited, or when a greater understanding of potential contamination sources is needed. Groundwater monitoring results from control wells, located in the vicinity of the landfill and on the surrounding arable lands may be supplemented by non-invasive geophysical methods, for example, electrical conductivity measurements [15,16]. There have been several attempts made to link the direction of groundwater flow with the leachate contamination at landfill areas using numerical modelling techniques [5,17]. For precise evaluation of the range of contamination, advanced spatial interpretations using Geographic Information System (GIS) have also been used [5,18,19]. This approach is especially important to monitor sites where physicochemical parameters cannot be measured directly. Despite of typical hydrogeochemical measures taken to assess the water quality, also statistical techniques have been frequently used to distinguish sources of pollution and to indicate factors responsible for contaminant release at the landfill sites [6].

The state of groundwater quality at landfill sites may be affected by the operation of surrounding areas and facilities. It was found that the landfills and agricultural areas may have a mutual impact on groundwater quality and can interact with each other, and that a minimum distance of 300 m should be maintained between landfills and arable land [20]. This premise is also supported by the fact that the landfills may negatively affect crop yields, and even cause their degradation due to adverse soil conditions resulting from the impact of waste installations. Furthermore, due to the possible negative impacts of the landfill on arable lands, several studies insist that a minimum of 500 m should be set between these sites [21]. Simsek et al. [22] suggested that a buffer zone of 500 m is required between agricultural areas and landfill sites. Dolui and Sarkar [23] stress that prime agricultural land should be excluded when choosing the site for the landfill location especially because the percolation of contaminants from the waste body may degrade the fertility and quality of soils, reducing both productivity and overall soil health. While Jahan et al. [24] showed that properly managed landfill sites (segregation applied) may be used for agricultural production, Urme et al. [25] found that the landfills situated close to the agricultural lands expose them to various environmental hazards. The latter study revealed that a high-risk zone for agricultural lands can be delineated by 200–300 m radius from the dumping site. A landfill in the vicinity of agricultural areas may also be a possible source of food chain poisoning, leaving the negative impact of the economy [26].

In the present study, the quality of groundwater at the two landfills located in close proximity to agricultural lands was assessed using groundwater quality indicators supported by statistical analyses. The aim of the study was to evaluate the state of groundwater at the landfill sites and their neighbourhood, and to indicate possible factors which have an impact on groundwater contamination. Accurate identification of these factors can significantly contribute to the efficient management of both landfills and other surrounding facilities, and aid in the protection of water resources in regions susceptible to pollution. To our knowledge, the results presented here represent a novel aspect of this type of assessment as few studies have looked at groundwater quality in terms of temporal changes in major components in locations where landfill facilities co-exist with agricultural lands.

2. Study sites

2.1. Site description

Two landfill sites in the Czech Republic (CR) were the subject of this study (Fig. 1). The first is the Zdounky landfill (49°14′30.5″N 17°18′29.9″E), located in the Nětčice area in the Zlín Region and classified as S-OO3 in terms of technical security, intended to store municipal solid wastes from neighbouring sites [27]. The landfill began operating in 1998, and the current amount of area covered by the Zdounky landfill is 70,700 m^2. The total amount of wastes stored at the landfill is $35,000 \times 10^3$ $kg \cdot y^{-1}$. It serves an area with 75,000 inhabitants, and is surrounded by agricultural lands [27]. The facility was designed with five cells for waste storage, and currently, about 30% of the Zdounky landfill has been reclaimed, 55% of the landfill area is under operation

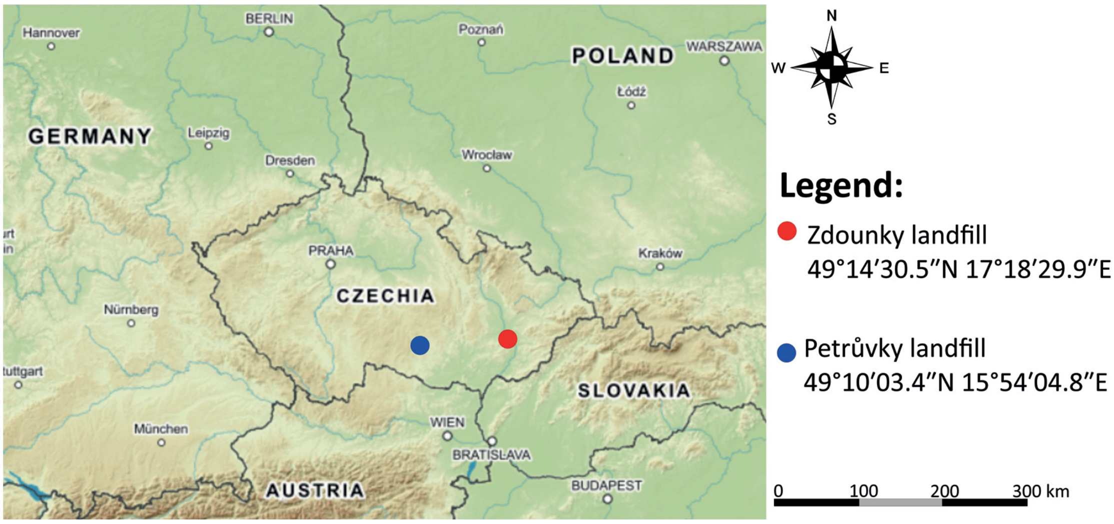

Fig. 1. Map indicating the geographic locations of the two sites of this study.

and 15% of the landfill is used as a composting plant [28]. It consists of a base sealing system and the top cover system of high-density polyethylene (HDPE) [27,29]. During the reclamation of the Zdounky landfill, the topsoil of the cover system was planted with the vegetation.

The second facility is the Petrůvky landfill (49°10′03.4″N 15°54′04.8″E), located 8 km from Třebíč. The landfill started operations in 1994, and, like the Zdounky landfill, is classified as S-OO3 which means that it receives "other wastes" with significant content of biodegradable compounds. The area of the Petrůvky landfill is 72,130 m^2. The total amount of wastes stored at that landfill is 31,000 × 10^3 kg·y^{-1}, and the total volume of the landfill is 600,000 m^3. The facility was designed with eight cells for waste storage, serving an area of 118,000 inhabitants [27]. Currently, ca. 50% of the Petrůvky landfill has been reclaimed. The landfill is surrounded by the forests and agricultural lands. The Petrůvky landfill is also designed as a sanitary landfill with base sealing system and top cover systems consisting of several protective layers, including HDPE geomembrane and soil layers.

2.2. Hydrogeological conditions

The surroundings of the landfill Zdounky area consist mainly of Paleogene rocks of marine origin, belonging to both the outer flysch and the Magura flysch. The Magura flysch is then slid onto these sub-units, which is represented in the SE near Zdounky by the Rača sub-unit. In terms of regional geological structure, the site is situated on the very edge of the Carpathian mountain-forming system. The Olšinka depression between 'Chřiby' and 'Litenčické Vrchy' is filled with rocks of the post-Silesian unit of the outer Carpathian flysch, while the marginal plug-in area of the entire mantle system is about 500 m from the edge of the area of interest [30]. The local translucent sediments are included in the Ždánice – Hustopeče formation in the facies of calcareous clays, saliva and sandstones of the mantle, that is, towards the Litenčické hills the frontal depth is already formed and is filled by the Carpathian to Upper Hellenic formation of the predominant saliva calcareous clays and clays with a thickness of approximately 500–700 m. Claystones at various degrees of weathering were verified in the landfill area by drilling. From a hydrogeological point of view, there are no significant differences between the sub-flysch units, and it is therefore possible to characterize this area as a whole with a common fractured permeability. The springs, with a few sporadic exceptions, are very irregularly distributed and have little yield. The hydrographic axis of the area is the surface stream Lipinka, 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^{-1} and which, after about 500 m, empties into the surface flow of Kotojedka. All these streams are significant for waste, leachate and water management.

The geological structure of the Petrůvky landfill area is characterized by thin overburden over a weathered rock base, which is built of Palaeozoic rocks of the Třebíč massif. In terms of hydrogeological zoning, the area of interest belongs to CR Zone No. 6550: "Crystalline in the Jihlava River basin". The most favourable conditions for groundwater cycling are in the fluvial sediments of more significant streams. The cycling depth is given by the depth of the local erosion base. Groundwater flow in pores and clefts is severely fluctuating and irregular, depending on local petrographic composition, tectonic predisposition, and character of Quaternary overburden. The zone of aeration is formed by clay-sandy formations to clay eluvium of syenites layers. The boundary between the aeration zone and the first aquifer is not well-defined and fluctuates depending on the seasonal distribution of precipitation. The general direction of stream groundwater conforms to the terrain slope, that is, towards SE. In terms of water management, the Petrůvky locality belongs to the Jihlava River basin. The nearest watercourse, the Zátoky stream, flows at a distance of approximately 900 m SE and S from the locality in the NE – SW direction. A nameless watercourse ends at the Zátoky stream below the landfill and drains surface runoff away from the landfill's perimeter troughs. On the Zátoky stream, two ponds have been built below the landfill catchment, designated as "Horní rybník" (Upper pond) and "Dolní rybník" (Lower pond).

2.3. Groundwater monitoring

The system to assess groundwater quality at the Zdounky landfill consists of the following monitoring piezometers: MV-1, MV-2, MV-4, MV-5, MV-6 and ST-1 (Fig. 2).

Two monitoring facilities (MV-1, ST-1), which are used to monitor the quality of groundwater flowing from the landfill area, are in the central part of the valley, below the landfill body. The piezometers MV-2B, MV-4, MV-5, and MV-6 are situated upstream of the landfill and they provide the reference monitoring with the background groundwater quality values of the landfill surroundings. These piezometers also reflect the impact of surrounding facilities on the groundwater quality within monitored area. Piezometer MV-2B (MV-2) is a replacement of the original monitoring piezometer MV-2A (cancelled during expansion landfill) The MV-2B was installed in September 2010 and was sampled for the first time during the autumn monitoring period of 2010. Monitoring points MV-4 and MV-5 indicate the flow and quality of groundwater at the east site of the landfill. The monitoring point MV-5 is also influenced by the runoff of contaminants from the sector related to the storage of demolition wastes. The latest monitoring point is the well MV-6, which was drilled in December 2014. This monitoring point is situated upstream of the landfill area and hence is not affected by the landfill operation, nevertheless it is impacted by the runoff of contaminants from arable lands surrounding the landfill (Fig. 2). Groundwater samples were first taken from MV-6 in the spring 2015. In the present study, the monitoring period of 11 April 2015 – 9 October 2018 was considered for the purpose of groundwater quality assessment at the Zdounky landfill.

Groundwater samples from the Petrůvky landfill were taken from the following monitoring points: HI-1, HI-6, HV-8 and HV-10 (Fig. 3). The monitoring piezometer HI-6 is situated at the northern edge of the area upstream of the landfill and represents the reference monitoring point. Monitoring piezometer HV-10 is situated downstream of the landfill and represents the furthest monitoring point from the landfill itself. Monitoring points HI-1 and HV-8 are also located downstream of the landfill, and together with monitoring point HV-10, they represent the impact of the landfill on groundwater, extending from a few to several hundred meters from the landfill's edge. These three piezometers may be considered as capturing the effect and dilution of a potential contamination plume that would emanate from both the closed and operational sectors of the landfill. The quality of groundwater was assessed with the use of the following groundwater quality indicators: pH, electrical conductivity (EC), ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), and total chromium (Cr_{total}). For the Petrůvky landfill, the monitoring period of 14 April 2015 – 18 September 2018 was considered for the evaluation of groundwater quality.

2.4. Collection of water samples

Sampling, transport and storage of groundwater samples were carried out in accordance with the CR legislative regulations and the decision on the integrated permit. Groundwater sampling was performed following the same procedure for each monitoring point (piezometer). Groundwater was pumped through each piezometer for a period of 15 min before the collection of a sample, and the amount pumped out was around 60 L. 6 L samples were then collected and taken to the laboratory for analysis. The measurement of the groundwater level was carried out immediately before and after pumping. Water temperature, pH and electrical

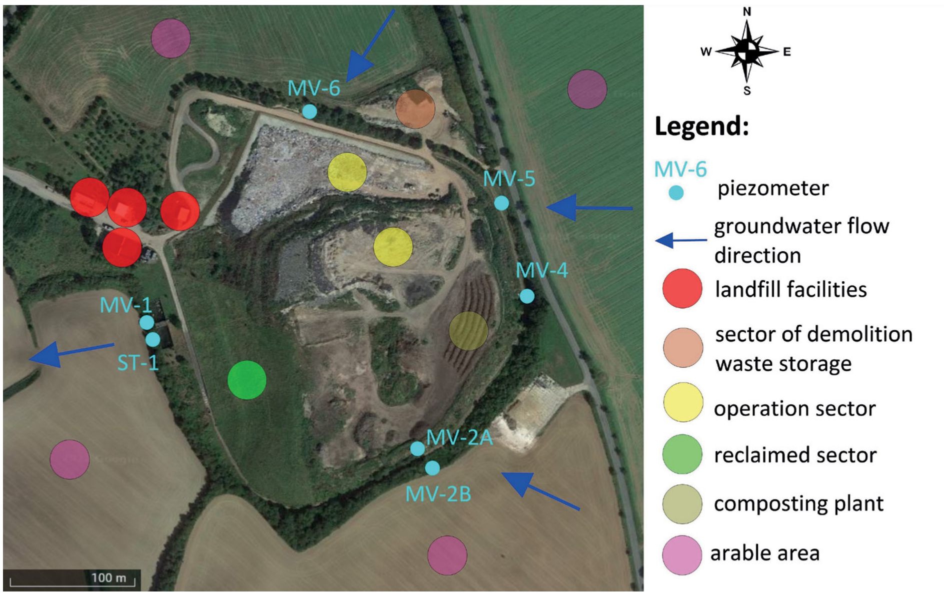

Fig. 2. Monitoring network of the groundwater quality at the Zdounky landfill, together with a view of the landfill's waste compartments.

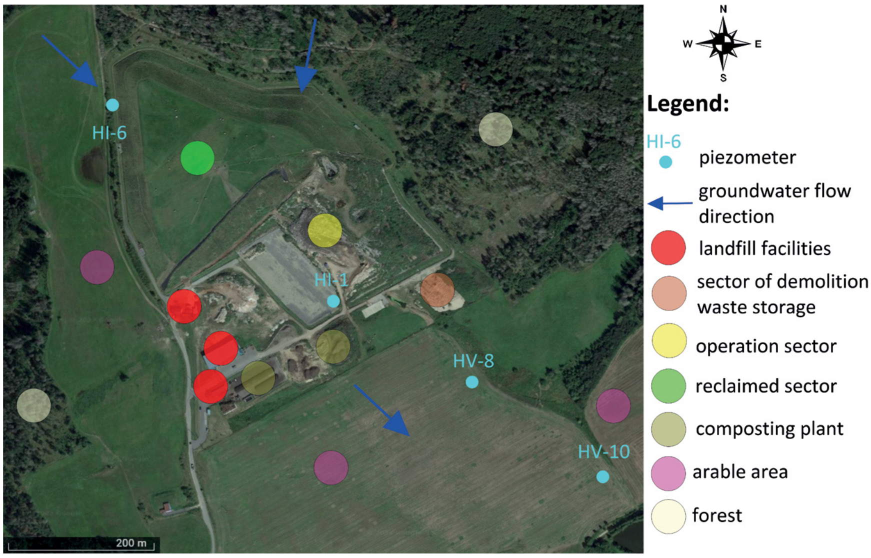

Fig. 3. Monitoring network of the groundwater quality at the Petrůvky landfill together with a view of the landfill's waste compartments.

conductivity (EC) were measured during the sampling. The parameters were considered stable when three consecutive readings, taken every 3 min, were within the following ranges of changes: water temperature ±0.2°C, pH ± 0.1, EC ± 5%. Sample measurements were compared with the critical values set by the Czech regulations for the landfill operation, which were obtained from a long-term monitoring of groundwater quality at the landfill and its surroundings. The sample concentrations were evaluated against the standards set in the CR [31]. The values obtained from the groundwater monitoring campaign were also compared against the criteria of ČSN (Czech State Standard) [32]. Finally, the detected concentrations were compared with the standards presented by the Environmental Protection Agency [33,34] and the World Health Organization (WHO) [35].

2.5. Monitoring data treatment and analysis

The parameters pH, EC, NH_4^+, NO_3^-, NO_2^-, and Cr_{total} were measured according to the analytical methods presented in Table 1.

The measured results from the piezometers monitoring groundwater quality at the two sites were evaluated against regulatory limits for analysed indicators. These limits are set in the CR [31,32], the environmental regulations of Ireland [33] and of the USA [34], and finally the recommendations of WHO [35], all of which are summarized in Table 2.

2.6. Statistical analysis

In this study, analysis of variance (ANOVA) and Principal Component Analysis (PCA) [42,43] were applied for data evaluation using SPSS-20 and XLSTAT software. ANOVA was used to assess which factors have an impact on the data and test whether the statistical discrepancy exists [44]. The PCA was to provide that the variation in the data set reduces the size of a large number of interrelated variables while at the same time protecting the data as much as possible [45].

Table 1
Methods for testing parameters of groundwater quality

Parameter	Method	References
pH	ČSN ISO 10523	[36]
EC, $mS\cdot m^{-1}$	ČSN EN 27 888	[37]
NH_4^+, $mg\cdot L^{-1}$	ČSN ISO 7150-1	[38]
NO_3^-, $mg\cdot L^{-1}$	ČSN ISO 7890-3	[39]
NO_2^-, $mg\cdot L^{-1}$	ČSN EN 26 777	[40]
Cr_{total}, $mg\cdot L^{-1}$	ČSN ISO 11083	[41]

2.7. Monte Carlo numerical simulations

Monte Carlo numerical simulations of groundwater flow and contaminant transport were performed for a heterogeneous aquifer, using the Spectral Turning Bands and the Particle Tracking methods [11–13]. The contaminant was assumed to be conservative and fully water soluble and a 2-dimension advection-dispersion equation was applied to analyze the migration [46]. A heterogeneous aquifer with a constant hydraulic gradient of 0.001 and mean ln$K$, where K is the hydraulic conductivity, equal to 1, and correlation length equal to 20 m for both x- and y-directions were considered. 3,000 Monte Carlo simulations and 8,000 particles were used to solve the groundwater and contaminant transport equations, and the results from these 3,000 numerical experiments were analysed. The total simulated region was 1,000 m long and 400 m wide, and it was discretized in cells of 2 by 2 m, creating a 500 × 200 grid. The landfill was depicted as a rectangular block with x-coordinates 10 and 60 m and y-coordinates 140 and 260 m. A system of six wells was placed downstream, 30 m from the edge of the

Table 2
Selected standards of water quality assessment

Standards and reference	Specific value	pH (–)	EC (mS·m⁻¹)	NH_4^+ (mg·L⁻¹)	NO_3^- (mg·L⁻¹)	NO_2^- (mg·L⁻¹)	Cr_{total} (mg·L⁻¹)
Critical value[a]	–	–	–	1.2	–	0.2	0.15[f]/0.30[g]
ČSN 75/143/1992 [32]	–	5–8.5	–	–	–	–	0.2
252/2004/Coll. [31]	–	6.5–9.5	–	0.5	50	0.5	0.05
Ireland's EPA, 2001 [33]	I/PV	6.5–9.5	250	0.5	50	0.5	0.05
	MCLG	–	–	–	10[d]	1[e]	0.1
US EPA, 2018 [34]	MCL	–	–	–	10[d]	1[e]	0.1
	SDWR	6.5–8.5	–	–	–	–	–
WHO, 2011 [35]	–	–	–	1.5[b]/35[c]	50	3	0.05

Notes: [a] – according to the monitoring recommendations of Petrůvky and Zdounky landfills, in accordance with the operating rules of selected landfills in the Czech Republic, [b] – value set as a threshold for odour concentration, [c] – value set as a threshold for taste concentration, [d] – concentration expressed as nitrate-N, [e] – concentration expressed as nitrite-N, [f] – value set for the Petrůvky landfill, [g] – value set for the Zdounky landfill, I – mandatory (imperative) value, PV – the parametric value which refers to the residual monomer concentration in the water as calculated according to specifications of the maximum release from the corresponding polymer in contact with the water, MCLG – Maximum Contaminant Level Goal, set at a level at which no known or anticipated adverse effect on the health of persons is expected to occur and which allows an adequate margin of safety, MCL – Maximum Contaminant Level set as the highest level of a contaminant that is allowed in drinking water, SDWR – Secondary Drinking Water Regulations, ČSN – Czech State Standard.

landfill, at x-coordinate 90 m, and the wells were distributed equally in the y-direction, over a distance of 100 m.

3. Results and discussion

3.1. Groundwater quality

The groundwater samples at the Zdounky and Petrůvky landfills were found to be in accordance with the requirements of the pH values (Table 3). At the Zdounky landfill, all pH values at the downstream piezometer MV-1 (6.8–7.5, min-max, respectively) were found to fall within the recommended range (6.5–9.5), and almost entirely stable during the monitoring period (Fig. 4). Similar values of pH detected in groundwater were presented by El-Salam and Abu-Zuid [47] and Abiriga et al. [9]. A pH of groundwater close to the value of 7 would indicate that groundwater in this area would be appropriate for agricultural or domestic purposes [48].

At the Petrůvky landfill, the mean, median, and maximum pH values (for the monitoring period 2015–2018) from the HV-10 piezometer were higher than from the rest of the samples taken from the other two piezometers positioned downstream from the landfill (HI-1 and HV-8). Instead, similarly slightly acidic conditions dominate HI-1 and HV-8, which is in accordance with the background conditions ascertained at HI-6; however, about 250 m further downstream from the HV-8 piezometer, the piezometer HV-10 registered alkaline conditions. Slightly alkaline conditions were found during the monitoring of a South African landfill and they were coincided with the neutral to alkaline pH that characterized the leachate [49].

The mean and median pH values at the HI-1 and HV-8 piezometers are lower than those allowed by the CR drinking water regulations at Decree No. 252/2004/Coll. [31], as well as the limits set by in the US EPA and Ireland's EPA [33] (Table 2). The mean and median of the pH at piezometer HV-10, despite the difficulty to explain this result from a plume's dispersive evolution perspective, falls within the recommended range. The pH revealed that those at the Petrůvky landfill exhibited more temporal variability than those at the Zdounky landfill (Fig. 4).

Table 3
Statistical summary of pH (–) in groundwater at the Zdounky and Petrůvky landfills

Landfill/Piezometer	Value				
	Min	Max	Mean	Median	Std. Dev.
Zdounky					
Downstream					
MV-1	6.8	7.5	7.1	7.1	0.2
Background					
MV-2B	6.7	7.7	7.1	7.2	0.3
MV-4	6.7	7.5	6.9	6.9	0.3
MV-5	6.7	7.5	7.0	7.0	0.3
MV-6	6.7	7.4	6.9	6.8	0.3
Petrůvky					
Downstream					
HI-1	5.4	6.4	5.8	5.7	0.4
HV-8	5.5	6.9	6.1	5.9	0.6
HV-10	5.6	9.0	7.7	8.1	1.2
Background					
HI-6	5.6	7.1	6.4	6.5	0.5

The most significant pH variations in time were observed at piezometer HV-10, which is located in a land depression, below the surrounding area. A potential explanation of the HV-10 pH anomaly may be that in early 2015 (Fig. 4) pollution from the landfill had reached the closest piezometers (with HI-1 exhibiting the lowest pH value of HI-1 and

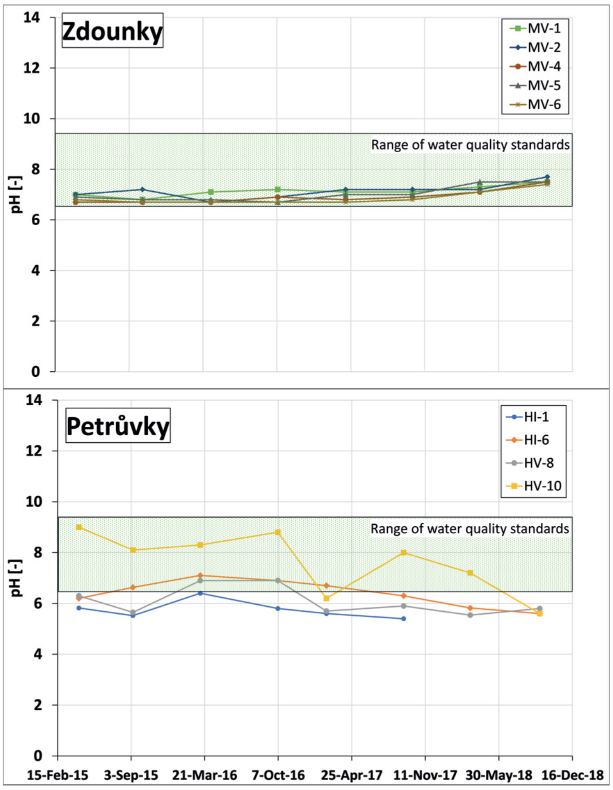

Fig. 4. Temporal changes of pH in groundwater within Zdounky and Petrůvky landfills.

HV-8, since it is nearer the landfill), but the plume had not travelled far enough to reach HV-10, which at high pH values reflected at that time local alkaline conditions. As time progressed the contamination plume reached HV-10, which, by the fall of 2018, exhibited a pH similar to the other monitoring piezometers.

In general, the Zdounky landfill EC values are significantly higher than those of the Petrůvky landfill (Table 4), with some values registering peaks that exceeded the regulatory limit (Fig. 5) of 250 mS·m^{-1}. In contrast, mean EC values from the monitoring period of 2015–2018 in groundwater samples collected at the Petrůvky landfill were consistently lower than the maximum values allowed by the regulations presented in Table 2. The measured values at the Petrůvky site fall within a very tight range with no significant change observed over time (Fig. 5). It was also shown, especially for the Petrůvky landfill, that the highest EC vales are in the sites closest to the landfill and diminishes farther away due to the dispersion and dilution. The differences in EC values and their range can be also a result of the construction of the piezometers, the lithology of the filtered aquifer, and the location of the piezometers in relation to the landfill site [50]. In particular, EC values at the monitoring point MV-6 at the Zdounky landfill consistently exceeded maximum allowable values. High EC values in the piezometer MV-6 may be due to high concentration of salts dissolved there [48], because of its proximity to the sector with demolition wastes, as well as the direct impact of the arable land. Overall variation between these two sites reflects how differences in the construction of individual landfills, the local lithology

Table 4
Statistical summary of EC (mS·m^{-1}) of the groundwater at Zdounky and Petrůvky landfills

Landfill/ Piezometer	Value				
	Min	Max	Mean	Median	Std. Dev.
Zdounky					
Downstream					
MV-1	203.0	231.0	214.3	211.5	12.7
Background					
MV-2	208.0	272.0	235.0	230.0	27.4
MV-4	173.0	183.0	179.0	180.0	4.9
MV-5	111.0	158.0	145.8	157.0	23.2
MV-6	325.0	336.0	330.8	331.0	4.6
Petrůvky					
Downstream					
HI-1	48.1	51.6	49.5	49.3	1.4
HV-8	25.9	56.0	45.3	47.9	9.5
HV-10	25.6	95.2	40.1	32.2	23.2
Background					
HI-6	20.9	27.8	25.4	25.8	2.2

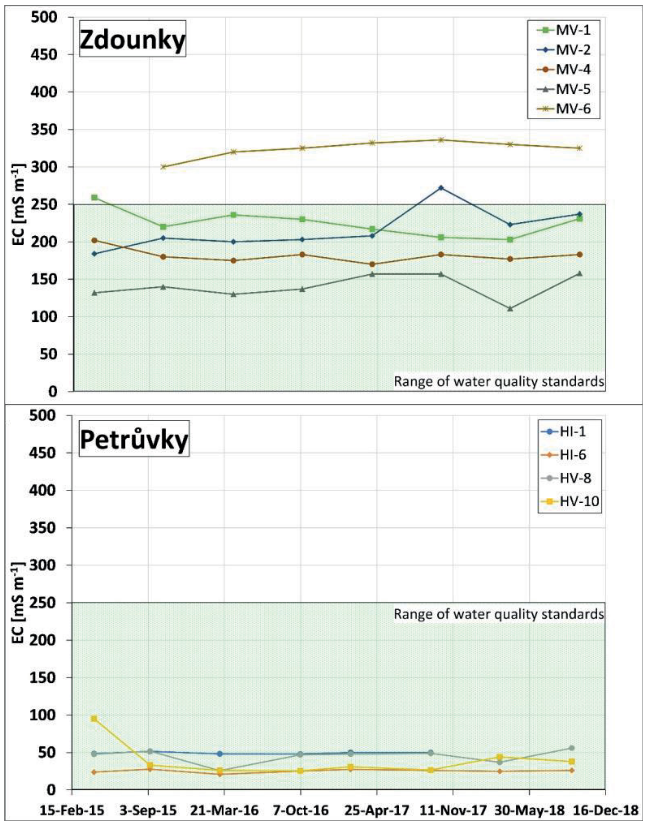

Fig. 5. Temporal changes of EC in groundwater within Zdounky and Petrůvky landfills.

and the way that surrounding land is used can impact the chemical properties of the groundwater.

Nitrogen (N) may be introduced to aquifers from many different sources. NH_4^+ in groundwater can originate naturally from the decay of organic matter, but concentrations greater than about 0.2 mg·L^{-1} are more often the result of contamination from agricultural fertilizers, or human and/or animal feces. NH_4^+ concentrations on the order of 1–10 mmol·L^{-1} have been observed in groundwater at locations contaminated by landfill leachate and wastewater disposal practices. Agricultural activities and septic systems have also resulted in locally elevated recharge rates of NH_4^+ [51]. In the present study, NH_4^+ concentrations (Table 5) were notably high in the piezometer HV-10 at the Petrůvky landfill. Piezometer HV-10 is the furthest away, downstream of the landfill, and the fact that higher NH_4^+ values are found there, relative to HI-1 and HV-8, which are also downstream but significantly closer to the landfill, is probably due to the existence of a land depression which favours runoff from surrounding areas.

The effect of arable lands on the piezometers' NH_4^+ values can be seen at HV-8 and HV-10, which are downstream of the Petrůvky landfill, and at MV-4 and MV-6, located upstream of the Zdounky site. All four of these piezometers are placed in cultivated lands and/or are receiving runoff from nearby arable lands.

At the Zdounky site, NH_4^+ concentrations measured at reference piezometers MV-4 and MV-6 exceed the allowable limits for NH_4^+ described in Table 2 more than eight and five times, respectively. The elevated levels of NH_4^+ concentrations may primarily reflect the impact of the surrounding agricultural areas. However, the much lower values at piezometer MV-5, which is also upstream and close to MV-4, and hence should have been equally affected by the fertilizing of agricultural lands, merits further investigation. The fact that the downstream piezometer MV-1 does not register high NH_4^+ values may be an indication that either the extent of the NH_4^+ plume is still limited, and has not reached this piezometer, or that the plume is narrow and has not been captured by this single downstream piezometer.

There are also some significant variations observed in time for NH_4^+ (Fig. 6). Temporal fluctuations in NH_4^+ concentrations are most readily observed in piezometer MV-4 at the Zdounky landfill, which is treated as a reference point reflecting the impact of surrounding agricultural areas on groundwater quality. Again, the difference between piezometers MV-5 and MV-4, which does not register such temporal variations, merits further investigation.

At the Petrůvky landfill, temporary increases in NH_4^+ concentrations in groundwater sample were also observed in piezometers located downstream of the landfill (HV-8, HV-10). Here, the fact that HV-8 appears to have lower NH_4^+ concentrations than HV-10, which is located further downstream and almost double the distance from the landfill than HV-8, makes it plausible that these NH_4^+ values mostly reflect the influence of surrounding agricultural areas and of runoff rather than pollution from the landfill. Although NH_4^+ is a typical groundwater pollution indicator of a landfill site that generally emanates from landfills at concentrations greater than other N forms, here the existence of arable

Table 5
Statistical summary of NH_4^+ (mg·L^{-1}) in groundwater at the Zdounky and Petrůvky landfills

Landfill/ Piezometer	Value				
	Min	Max	Mean	Median	Std. Dev.
Zdounky					
Downstream values					
MV-1	0.02	0.05	0.03	0.04	0.01
Background values					
MV-2	0.02	0.05	0.03	0.03	0.01
MV-4	0.03	9.00	3.15	1.63	3.62
MV-5	0.03	0.35	0.15	0.04	0.16
MV-6	1.57	3.96	2.65	2.49	0.93
Petrůvky					
Downstream values					
HI-1	0.05	0.05	0.05	0.05	0.00
HV-8	0.05	1.52	0.27	0.07	0.51
HV-10	0.05	3.62	0.73	0.05	1.26
Background values					
HI-6	0.05	0.75	0.14	0.05	0.25

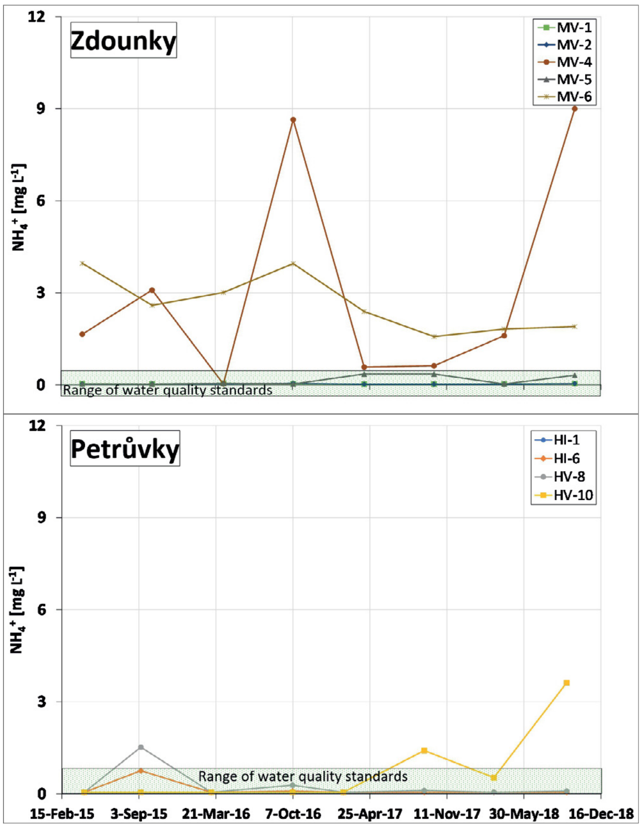

Fig. 6. Temporal changes of NH_4^+ in groundwater within Zdounky and Petrůvky landfills.

lands adjacent to the two landfills makes the situation complicated to distinguish the effect of landfills and agriculture with this small number of monitoring piezometers [52].

Following the monitoring results from the Zdounky landfill, it was noted that in the piezometer located downstream the landfill (MV-1), the concentration of NO_3^- is two times higher than the maximum allowable limit set by the environmental law (Table 6).

A similar situation was observed also for the monitoring point MV-2A, which is impacted by the surrounding arable lands (Fig. 7). A significant reduction of NO_3^- concentrations in piezometers MV-1 and MV-2 occurred starting in 2017. This is likely due to a change in the manner in which fertilizer was applied in surrounding arable fields during that same period. The intensive application of N fertilizers was reduced, resulting in lower leaching of unused N compounds to the environment in subsequent years during the monitoring period.

The data from our monitoring sites suggests that the concentrations of NO_3^- in groundwater are affected both by the landfills and surrounding agricultural areas. The NO_3^- concentrations in groundwater also show temporal variations. The most visible fluctuations were detected for the monitoring points MV-1 and MV-2 located at the Zdounky landfill. The concentrations measured in these monitoring points showed elevated values in comparison to the allowable limit of 50 mg·L^{-1}. Elevated concentrations of NO_3^- ions at the Petrůvky landfill were observed by piezometers HI-1 and HV-8 (Table 6). At both these monitoring points, the concentrations were measured at levels higher than the allowable 50 mg·L^{-1}. High NO_3^- concentrations close to the Petrůvky landfill may be also associated with the good oxidation conditions in groundwater. Because natural groundwater NO_3^- concentrations are generally low, concentrations greater than 1 mg·L^{-1} are likely due to anthropogenic activities [53,54]. The elevated concentrations of NO_3^- and NH_4^+ in groundwater typically indicate that groundwater quality was affected by the landfill leachate percolation [55] or agricultural activities. In the present study, given the position of the piezometers relative to the landfills (both up- and downstream) and the direction of the flow of the plume, it is likely that increased NO_3^- concentrations are the combined result of both the landfill and the long-term application of mineral and organic fertilizers, and also, as revealed by Han et al. [56], can result from the ploughing activities of agricultural fields, which promote nitrate infiltration through looser and aerated soils.

Table 6
Statistical summary of NO_3^- (mg·L^{-1}) in groundwater at Zdounky and Petrůvky landfills

Landfill/ Piezometer	Value				
	Min	Max	Mean	Median	Std. Dev.
Zdounky					
Downstream values					
MV-1	6.6	273.0	121.8	109.0	107.3
Background values					
MV-2	41.6	263.0	131.7	116.7	94.6
MV-4	0.1	17.0	2.3	0.2	6.0
MV-5	2.4	53.1	23.1	17.8	21.2
MV-6	0.1	2.5	0.8	0.4	0.8
Petrůvky					
Downstream values					
HI-1	46.5	68.1	56.7	56.7	7.70
HV-8	2.0	13.1	5.8	4.3	4.3
HV-10	2.0	3.3	2.3	2.3	0.5
Background values					
HI-6	2.0	15.0	4.4	2.5	4.4

Regarding NO_2^- in groundwater, for both analysed landfill sites it was observed that, in connection to the mean values measured, the requirements set by the water quality standards are met (Table 7). The concentrations of NO_2^- in groundwater were generally stable over time (Fig. 8), with some peaks observed, especially for the piezometer MV-5 that was likely impacted by the surrounding farmlands.

In addition to the possible impact of the agricultural lands on the content of N compounds in groundwater, it should be mentioned that the groundwater contamination by NO_3^- and NO_2^- can be observed particularly in the landfills without anti-seepage systems. For example, in several cases reported the landfills in Tibet, NO_3^- was found as the major contributor to groundwater contamination,

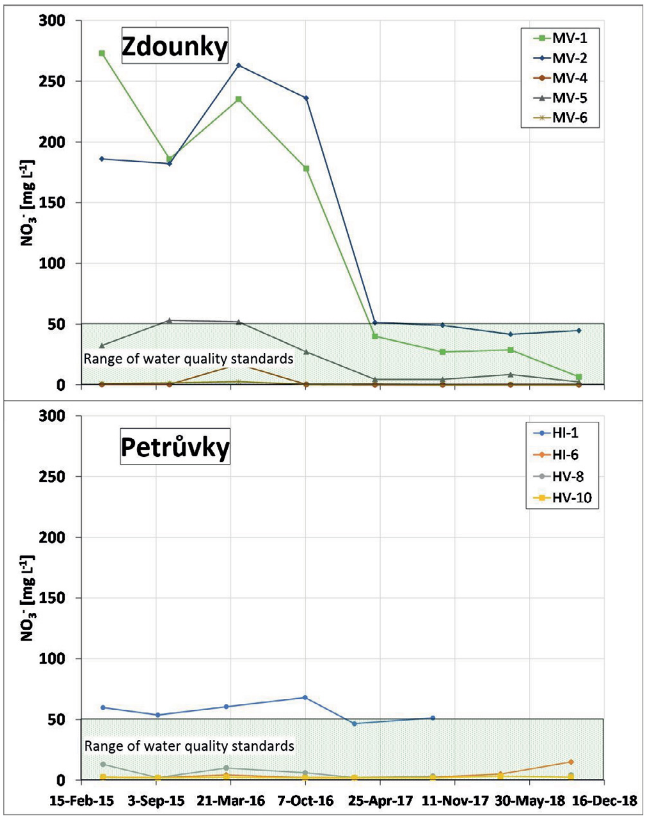

Fig. 7. Temporal changes of NO_3^- in groundwater within Zdounky and Petrůvky landfills.

Table 7
Statistical summary of NO_2^- ($mg \cdot L^{-1}$) in groundwater at Zdounky and Petrůvky landfills

Landfill/ Piezometer	Value				
	Min	Max	Mean	Median	Std. Dev.
Zdounky					
Downstream values					
MV-1	0.02	0.14	0.07	0.07	0.05
Background values					
MV-2	0.02	0.30	0.08	0.06	0.09
MV-4	0.02	0.13	0.05	0.05	0.04
MV-5	0.02	0.13	0.18	0.04	0.04
MV-6	0.02	0.07	0.05	0.06	0.02
Petrůvky					
Downstream values					
HI-1	0.05	0.09	0.06	0.05	0.02
HV-8	0.05	0.09	0.06	0.05	0.01
HV-10	0.05	0.08	0.06	0.05	0.01
Background values					
HI-6	0.05	0.09	0.06	0.05	0.01

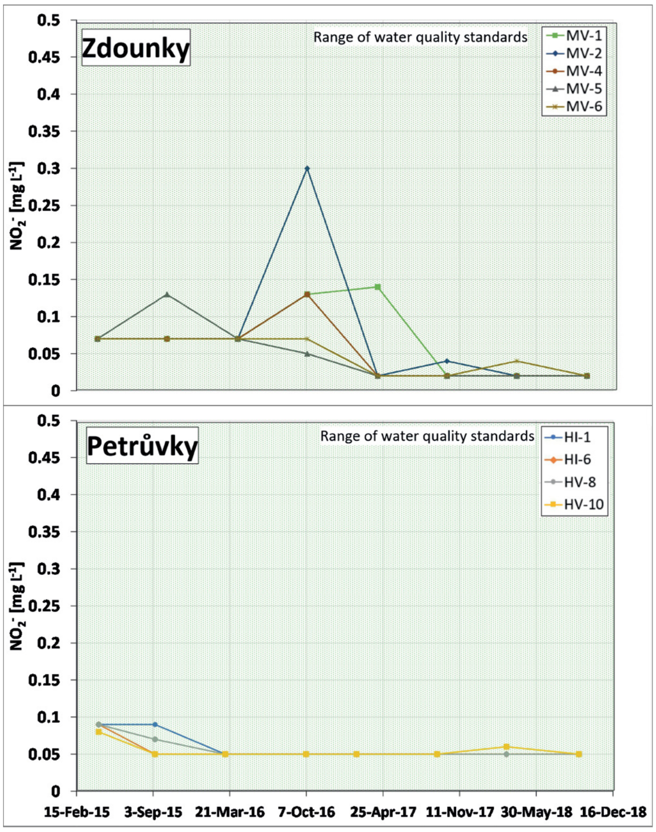

Fig. 8. Temporal changes of NO_2^- in groundwater at Zdounky and Petrůvky landfills.

followed by heavy metals [14,57]. Elevated concentrations of NO_3^- in groundwater due to the landfill operation were also revealed as a significant problem for Mekelle city in northern Ethiopia [58].

At both the Zdounky and Petrůvky landfills, the Cr concentrations did not exceed the critical values (0.15 and 0.30 $mg \cdot L^{-1}$ for the Petrůvky and Zdounky landfills, respectively), in accordance with the operating rules of selected landfills in the CR (Table 8). In terms of Cr_{total} concentrations in groundwater (Table 8), it was revealed that at Petrůvky site, detected values are lower than the allowable 0.05 $mg \cdot L^{-1}$ (50 ppb).

Variation in the concentrations was observed over the monitoring period for both landfills, but most noteably at Zdounky (Fig. 9). At that landfill, mean concentrations of Cr_{total} in groundwater, even at reference piezometers, are higher that the limit set by environmental laws. These high concentrations can potentially be explained by runoff of contaminants from the sector of tires or demolition wastes storage. Steel belts and bead wire in passenger car tires contain traces of Cr, as well as Mn, and these may be leaking from the tire sector of the landfill [59]. At the same time the geology of the site points to rock material of mantle origin, which may indicate a natural source of Cr [60], especially since piezometer MV-6, which also exceeds the regulatory limit, is found upstream of the tire section. A major impact on groundwater contamination by Cr compounds can be also observed in the vicinity of industrial landfills. Adamczyk and Haładaus [61] found that near industrial landfills, the range of measured concentrations can be as high as 1,500 $mg \cdot L^{-1}$. The presence of Cr close to the landfill sites can be also attributed to the operation of surrounding facilities. For instance, in the research performed by Akinbile [62], Cr in groundwater was observed at the level of 0.25 $mg \cdot L^{-1}$ in the distance of 100 m from the landfill and indicated the contamination originated from an adjacent abattoir and not from the landfill site.

It is worth noting that chromium (Cr) is widely distributed in the earth's crust and can be found in its trivalent form as Cr(III), or its hexavalent form Cr(VI). Cr(VI) is a known carcinogen upon inhalation, and in waters, apart from industrial pollution sources, under specific geochemical conditions it can be generated naturally in environments dominated by ophiolites and Mn [60,63]. Intense discussions over the last fifteen years about the potential of Cr(VI) to be carcinogenic upon ingestion led the state of California [64] to adapt a standard of 0.05 $mg \cdot L^{-1}$ (50 ppb), rather than the US EPA federal limit of 0.1 $mg \cdot L^{-1}$ (100 ppb) shown in Table 2. A recent draft document by WHO proposed again as guideline value for Cr_{total} the value of 0.05 $mg \cdot L^{-1}$ [65]. Cr_{total} is usually found in drinking water at an average of 0.001 $mg \cdot L^{-1}$ (1 ppb), well below the guideline value of 0.05 $mg \cdot L^{-1}$ [65].

According to the so-called "Dutch List" [66], groundwater with a content below 0.02 $mg \cdot L^{-1}$ of Cr_{total} is considered pure (class A). A Cr_{total} concentration over 0.05 $mg \cdot L^{-1}$ is a signal for detailed diagnosis (class B), and more than 0.2 $mg \cdot L^{-1}$ should be regarded as a pollutant requiring preventive action and remediation (class C).

The results of the Monte Carlo numerical simulations are especially relevant for a study of this type. Fig. 10 clearly

Table 8
Statistical summary of Cr_{total} (mg·L^{-1}) in groundwater within Zdounky and Petrůvky landfills

Landfill/ Piezometer	Value				
	Min	Max	Mean	Median	Std. Dev.
Zdounky					
Downstream values					
MV-1	0.02	0.08	0.05	0.06	0.02
Background values					
MV-2	0.02	0.08	0.05	0.06	0.02
MV-4	0.02	0.09	0.04	0.04	0.03
MV-5	0.02	0.13	0.06	0.06	0.04
MV-6	0.02	0.09	0.06	0.07	0.02
Petrůvky					
Downstream values					
HI-1	0.01	0.01	0.01	0.01	0.00
HV-8	0.01	0.02	0.01	0.01	0.00
HV-10	0.01	0.01	0.01	0.01	0.00
Background values					
HI-6	0.01	0.01	0.01	0.01	0.00

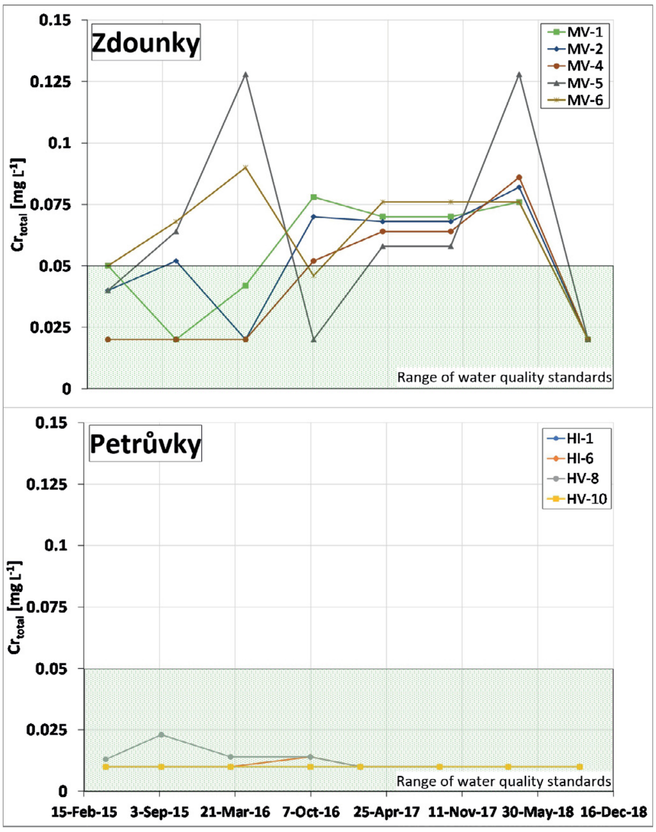

Fig. 9. Temporal changes of Cr_{total} in groundwater at the Zdounky and Petrůvky landfills.

demonstrates when monitoring systems that are set too close to a landfill, such as piezometer MV-1 at the Zdounky landfill, it is possible that the contaminant plume may pass in between the monitoring points, since dispersion may not yet have significantly expanded the plume laterally, causing it to be missed by the wells and not recorded in the collected samples.

Such a numerical approach is crucial for detecting the potential path of migration of contaminants, and also allows for the proper determination of the optimum number of groundwater monitoring points and their locations [67]. When setting the monitoring network in the research area it is also important to recognize the dispersive features of the environment. As reported by Paleologos et al. [46], in the case of a low dispersive environment, larger distances from the landfill are required to capture the contaminant plume by the observation points in the monitoring network. By contrast, in high dispersive environments, the contamination may be reliably detected in monitoring points close to the source.

3.2. Statistical analysis

According to the ANOVA analysis, carried out to distinguish differences between the data sets, NH_4^+ and NO_3^- were characterized by $p < 0.05$, indicating a statistically significant difference for the Zdounky landfill (Table 9). For the Petrůvky landfill, there was no statistical difference observed between the parameters ($p > 0.05$) (Table 10).

For the Zdounky landfill, a further analysis was carried out to identify the points where the pollution indicators vary (Appendix A). The ANOVA analysis indicates significant differences observed for NH_4^+ between MV-1 and MV-4; MV-2B and MV-4; MV-1 and MV-2B; MV-4 and MV-5. For NO_3^- significant differences were observed between piezometers MV-1 and MV-4; MV-1 and MV-5; MV-1 and MV-6; MV-2B and MV-4; MV-2B and MV-5; MV-2B and MV-6; MV-2B and MV-1.

Suitability for factor analysis of data was tested by Kaiser–Meyer–Olkin (KMO) and Bartlett tests. While KMO values range between 0 and 1, a value of KMO should be greater than 0.5 and is more suitable as an analysis factor as the KMO value gets closer to 1 [42,43]. As a result of the factor analysis, three factors were identified with eigenvalues > 1 ratio to total variance of which showed a gradually decreasing for the Zdounky and Petrůvky landfills (Tables 11 and 12).

For the Zdounky landfill, three factors explain 63.237% of the total variance (Table 11). The first factor explains 23.901% of the total variance, and NO_3^- and NO_2^- have weak load values. It shows the effect of the surrounding agricultural areas on the groundwater quality as the source of the first factor. The second factor explains 21.193% of total variance, and this factor shows that NH_4^+ and Cr_{total} have weak positive load values. The second factor source can be explained by the runoff of pollutants from agricultural areas and potentially the tire storage, or the storage of demolition waste. The third factor explains 18.143% of the total variance, and pH has moderate positive load values. The third factor being pH at the Zdounky site is supported by the almost completely stable pH values in the landfill during the monitoring period.

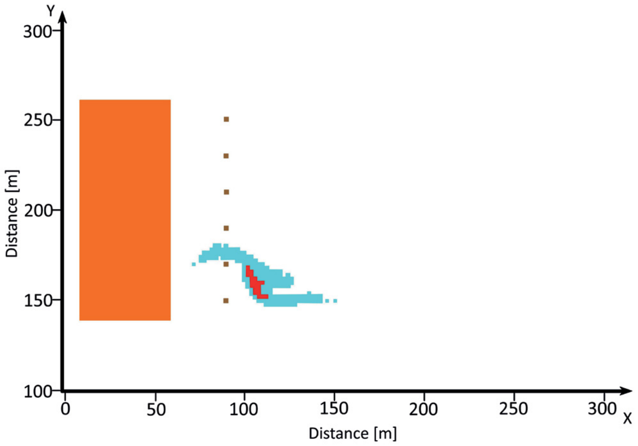

Fig. 10. System of six wells monitoring contamination from a landfill in a heterogeneous aquifer.

Table 9
ANOVA analysis for monitoring results from Zdounky landfill

Analysis		SS	df	MS	F	Sig.
pH	Between Groups	0.511	4	0.128	1.716	0.168
EC	Between Groups	41,610.350	4	10,402.588	0.689	0.604
NH_4^+	Between Groups	121.359	4	30.340	3.914	**0.010**
NO_3^-	Between Groups	136,474.518	4	34,118.630	8.155	**0.000**
NO_2^-	Between Groups	0.093	4	0.023	0.789	0.540
Cr_{total}	Between Groups	0.003	4	0.001	0.834	0.513

SS – sum of squares, df – degrees of freedom MS – mean sum of square, F – measure of statistic, Sig. – significance

Table 10
ANOVA analysis for monitoring results from Petrůvky landfill

Analysis		SS	df	Mean Square	F	Sig.
pH	Between Groups	2.689	4	0.672	0.674	0.614
EC	Between Groups	11720.299	4	2,930.075	0.421	0.793
NH_4^+	Between Groups	30.776	4	7.694	0.928	0.459
NO_3^-	Between Groups	1940.928	4	485.232	0.439	0.780
NO_2^-	Between Groups	0.027	4	0.007	1.018	0.412
Cr_{total}	Between Groups	9.820	4	2.455	0.246	0.910

SS – sum of squares, df – degrees of freedom MS – mean sum of square, F – measure of statistic, Sig. – significance

Table 11
Varimax rotated factor matrix (VF) for data sets for the Zdounky landfill

Parameter	VF1	VF2	VF3
pH	0.304	0.016	**0.426***
EC	0.051	0.142	0.014
NH_4^+	0.239	**0.478***	0.001
NO_3^-	**0.336***	0.095	0.324
NO_2^-	**0.439***	0.081	0.024
Cr_{total}	0.065	**0.459***	0.300
Eigenvalue	1.434	1.272	1.089
Variability (%)	23.901	21.193	18.143
Cumulative (%)	23.901	45.094	63.237

*Significant factor loading are bold faced (***strong > 0.75; **medium 0.50–0.75; *weak 0.50–0.30) [68].

Table 12
Varimax rotated factor matrix (VF) for data sets for the Petrůvky landfill

Parameter	VF1	VF2	VF3
pH	**0.459***	0.187	0.087
EC	0.030	0.170	**0.602****
NH_4^+	0.123	**0.388***	0.211
NO_3^-	**0.708****	0.002	0.010
NO_2^-	0.023	**0.653****	0.003
Cr_{total}	0.222	0.016	0.134
Eigenvalue	1.564	1.417	1.046
Variability (%)	26.068	23.617	17.439
Cumulative (%)	26.068	49.685	67.125

*Significant factor loading are bold faced (***strong > 0.75;**medium 0.50–0.75; *weak 0.50–0.30) (Liu et al., 2003).

For the Petrůvky landfill, three factors explain 67.125% of the total variance (Table 12). The first factor explains 26.068% of the total variance and pH and NO_3^- have weak and moderate load values, respectively.

The presence of alkaline conditions in agricultural areas as a first factor source may indicate that agricultural areas have alkaline conditions due to decomposition of rock minerals and pollution from landfills that create acidic conditions along the plume path. The second factor explains 23.617% of total variance, and NH_4^+ has weak positive load values, and NO_2^- has moderate positive load, respectively. The presence of high concentrations of NH_4^+ in the field as the source of the second factor indicates contamination with both agricultural fertilizers and landfill pollutants. Again, the impact of agricultural lands away from the landfill can be observed in the high ammonium values. The third factor explains 17.439% of the total variance, and EC has moderate positive load values (Fig. 11).

4. Conclusions

Groundwater quality at the landfill sites was monitored by sampling and analysing of selected indicators: pH, EC,

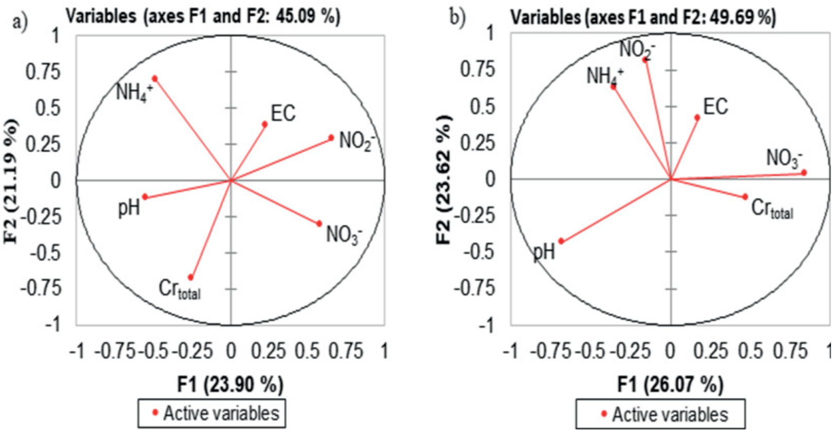

Fig. 11. Results of PCA analysis: (a) Zdounky and (b) Petrůvky.

NH_4^+, NO_3^-, NO_2^- and Cr_{total}. The pH anomaly was observed at the Petrůvky site, where a time-averaged acid reaction was found in close proximity to the landfill, while more alkaline conditions were measured in the agricultural area. In the initial period of monitoring, highly alkaline conditions in the agricultural areas were observed, but in the following years the groundwater gradually became acidic as well, as in the piezometers close to the landfill. This may indicate that initially the agricultural areas, furthest from the landfill, had alkaline conditions, because of the weathering of the rock minerals. Pollution emanating from the landfill created more acidic conditions over time. Relatively high values of EC in relation to background conditions may be the result of salt dissolution from the storage of construction waste in the Zdounky landfill and the impact of agricultural lands as well as runoff from roads and storage yards. The presence of high concentrations of NH_4^+ in the Petrůvky site indicates contamination with both agricultural fertilizers and pollutants from the landfill. Again, in the agricultural lands away from the landfill, the highest NH_4^+ values were found, which may be due to the existence of a lowering of the area conducive to the concentration of runoff from the surrounding agricultural lands. The impact of agricultural activity on the contamination of groundwater with N compounds can be observed in both of the monitored landfills, in which increased concentrations of these pollutants were found. Finally, the average concentrations of Cr_{total} in groundwater, even for reference background values, were higher than the limit set by environmental standards. This can be explained by pollutant spills from the tire disposal and demolition waste sectors, as well as the geology of the mantle rock material, which may indicate a natural source of Cr.

Funding

This research was supported by the National Science Centre, Poland, under a grant No. 2017/25/N/ST10/00909.

References

[1] G. Przydatek, W. Kanownik, Impact of small municipal solid waste landfill on groundwater quality, Environ. Monit. Assess., 191 (2019), doi: 10.1007/s10661-019-7279-5.
[2] P. Negi, S. Mor, K. Ravindra, Impact of landfill leachate on the groundwater quality in three cities of North India and health risk assessment, Environ. Dev. Sustainability, 22 (2020) 1455–1474.
[3] J.R. Masoner, D.W. Kolpin, E.T. Furlong, I.M. Cozzarelli, J.L. Gray, Landfill leachate as a mirror of today's disposable society: pharmaceuticals and other contaminants of emerging concern in final leachate from landfills in the conterminous United States, Environ. Toxicol. Chem., 35 (2015) 906–918.
[4] J.R. Masoner, D.W. Kolpin, E.T. Furlong, I.M. Cozzarelli, J.L. Gray, E.A. Schwab, Contaminants of emerging concern in fresh leachate from landfills in the conterminous United States, Environ. Sci. Processes Impacts, 16 (2014) 2335–2354.
[5] S. Mishra, D. Tiwary, A. Ohri, A.K. Agnihotri, Impact of municipal solid waste landfill leachate on groundwater quality in Varanasi, India, Groundwater Sustainable Dev., 9 (2019) 100230, doi: 10.1016/j.gsd.2019.100230.
[6] A.H. Baghanam, V. Nourani, H. Aslani, H. Taghipour, Spatiotemporal variation of water pollution near landfill site: application of clustering methods to assess the admissibility of LWPI, J. Hydrol., 591 (2020) 125581, doi: 10.1016/j.jhydrol.2020.125581.
[7] F. Parvin, S.M. Tareq, Impact of landfill leachate contamination on surface and groundwater of Bangladesh: a systematic review and possible public health risks assessment, Appl. Water Sci., 11 (2021) 1–17, doi: 10.1007/s13201-021-01431-3.
[8] M.D. Vaverková, E.K. Paleologos, A. Dominijanni, E. Koda, C.-S. Tang, M. Wdowska, Q. Li, N. Guarena, A.-M.O. Mohamed, C.S. Vieira, M. Manassero, B.C. O'Kelly, X. Qifeng, M.W. Bo, D. Adamcová, A. Podlasek, M. Uday, M. Anand, A. Mohammad, V.S.N.S. Goli, G. Kuntikana, E.M. Palmeira, S. Pathak, D.N. Singh, Municipal solid waste management under COVID-19: challenges and recommendations, Environ. Geotech., 8 (2021) 217–232.
[9] D. Abiriga, L.S. Vestgarden, H. Klempe, Groundwater contamination from a municipal landfill: effect of age, landfill closure, and season on groundwater chemistry, Sci. Total Environ., 737 (2020) 140307, doi: 10.1016/j.scitotenv.2020.140307.
[10] L.M.S. Pandey, S.K. Shukla, An insight into waste management in Australia with a focus on landfill technology and liner leak detection, J. Cleaner Prod., 225 (2019) 1147–1154.
[11] K. Papapetridis, E.K. Paleologos, Contaminant detection probability in heterogeneous aquifers and corrected risk analysis for remedial response delay, Water Resour. Manage., 47 (2011) 15, doi: 10.1029/2011WR010652.
[12] K. Papapetridis, E.K. Paleologos, Stochastic Modeling of Plume Evolution and Monitoring into Heterogeneous Aquifers, N. Lambrakis, G. Stournaras, K. Katsanou, Eds., Advances in the Research of Aquatic Environment. Environmental Earth Sciences, Springer, Berlin, Heidelberg, 2011, pp. 349–356.

[13] K. Papapetridis, E.K. Paleologos, Sampling frequency of groundwater monitoring and remediation delay at contaminated sites, Water Resour. Manage., 26 (2012) 2673–2688.
[14] D. Zeng, G. Chen, P. Zhou, H. Xu, A. Qiong, B. Duo, X. Lu, Z. Wang, Z. Han, Factors influencing groundwater contamination near municipal solid waste landfill sites in the Qinghai-Tibetan plateau, Ecotoxicol. Environ. Saf., 211 (2021) 111913, doi: 10.1016/j.ecoenv.2021.111913.
[15] M. Lech, J. Fronczyk, M. Radziemska, A. Sieczka, K. Garbulewski, E. Koda, Z. Lechowicz, Monitoring of total dissolved solids on agricultural lands using electrical conductivity measurements, Appl. Ecol. Environ. Res., 14 (2016) 285–295.
[16] E. Koda, A. Tkaczyk, M. Lech, P. Osiński, Application of electrical resistivity data sets for the evaluation of the pollution concentrations level within landfill subsoils, Appl. Sci., 7 (2017) 262, doi: 10.3390/app7030262.
[17] E. Koda, Influence of vertical barrier surrounding old sanitary landfill on eliminating transport of pollutants on the basis of numerical modeling and monitoring results, Pol. J. Environ. Stud., 21 (2012) 929–935.
[18] S. Thangaperumal, A study of groundwater quality and mapping using GIS techniques in Kodungaiyur, Chennai, Int. J. Adv. Res. Ideas Innovations Technol., 5 (2019) 618–624.
[19] K. Kalawapudi, O. Dube, R. Sharma, Use of neural networks and spatial interpolation to predict groundwater quality, Environ. Dev. Sustain., 22 (2020) 2801–2816.
[20] M.A. Nanda, A.K. Wijayanto, H. Imantho, L.O. Nelwan, I.W. Budiastra, K.B. Seminar, Factors determining suitable landfill sites for energy generation from municipal solid waste: a case study of Jabodetabek Area, Indonesia, Sci. World J., 2022 (2022) 9184786, doi: 10.1155/2022/9184786.
[21] M. Derakhshandeh, T. Taleb Beydokhti, Management of landfill locating of urban waste, Eur. Online J. Nat. Soc., 3 (2014) 32–39.
[22] C. Simsek, A. Elci, O. Gunduz, N. Taskin, An improved landfill site screening procedure under NIMBY syndrome constraints, Landscape Urban Plann., 132 (2014) 1–15, doi: 10.1016/j.landurbplan.2014.08.007.
[23] S. Dolui, S. Sarkar, Identifying potential landfill sites using multicriteria evaluation modeling and GIS techniques for Kharagpur city of West Bengal, India, Environ. Challenges, 5 (2021) 100243, doi: 10.1016/j.envc.2021.100243.
[24] E. Jahan, A. Nessa, M.F. Hossain, Z. Parveen, Characteristics of municipal landfill leachate and its impact on surrounding agricultural land, Bangladesh J. Sci. Res., 29 (2016) 31–39.
[25] S.A. Urme, M.A. Radia, R. Alam, M.U. Chowdhury, S. Hasan, S. Ahmed, H.H. Sara, M.S. Islam, D.T. Jerin, P.S. Hema, M. Rahman, A.K.H.M. Islam, M.T. Hasan, Z. Quayyum, Dhaka landfill waste practices: addressing urban pollution and health hazards, Build. Cities, 2 (2021) 700–716.
[26] N.S.I. Sharifah, A.M. Latifah, The challenge of future landfill: a case study of Malaysia, J. Toxicol. Environ., 5 (2013) 86–96.
[27] M.D. Vaverková, E.K. Paleologos, D. Adamcová, A. Podlasek, G. Pasternak, J. Červenková, Z. Skutnik, E. Koda, J. Winkler, Municipal solid waste landfill: evidence of the effect of applied landfill management on vegetation composition, Waste Manage. Res., 0734242X221079304 (2022) 1–10, doi: 10.1177/0734242X221079304.
[28] M.D. Vaverková, D. Adamcová, J. Winkler, E. Koda, P. Petrželová, A. Maxianová, Alternative method of composting on a reclaimed municipal waste landfill in accordance with the circular economy: benefits and risks, Sci. Total Environ., 723 (2020) 137971, doi: 10.1016/j.scitotenv.2020.137971.
[29] A. Podlasek, A. Jakimiuk, M.D. Vaverková, E. Koda, Monitoring and assessment of groundwater quality at landfill sites: selected case studies of Poland and the Czech Republic, Sustainability, 13 (2021) 7769, doi: 10.3390/su13147769.
[30] B. Balatka, T. Czudek, J. Demek, Regional division of the relief of Czechoslovakia, Proceedings of the CSSA, 78 (1973) 81–96 (in Czech).
[31] Collection of Laws, Decree 83/2014 Coll. Amending Decree 252/2004 Coll., Which Sets Out the Sanitary Requirements for Drinking and Hot Water and the Frequency and Scope of Drinking Water Control, as Amended, 2014, Praha, 33 p.
[32] ČSN 75 7143, Water Quality for Irrigation, Publishing of Standards, Prague, 1992 (in Czech).
[33] Ireland's EPA, Parameters of Groundwater Quality, Interpretation and Standards, Environmental Protection Agency, Ireland, 2001.
[34] US EPA, Edition of the Drinking Water Standards and Health Advisories Tables, U.S. Environmental Protection Agency, Washington, 2018.
[35] World Health Organization (WHO), Guidelines for Drinking-Water Quality (World Health Organization), 4th ed., Geneva, Switzerland, 2011, 542 p.
[36] ČSN ISO 10523, Water Quality, Determination of pH (in Czech).
[37] ČSN EN 27 888, Water Quality, Determination of Electrical Conductivity (in Czech).
[38] ČSN ISO 7150-1, Water Quality, Determination of Ammonium Ions. Part 1: Manual Spectrometric Method (in Czech).
[39] ČSN ISO 7890-3, Water Quality, Determination of Nitrate – Part 3: Spectrometric Method Using Sulfosalicylic Acid (in Czech).
[40] ČSN EN 26 777, Water Quality, Determination of Nitrite, Molecular Absorption Spectrometric Method (in Czech).
[41] ČSN ISO 11083, Water Quality, Determination of Chromium(VI), Spectrophotometric Method with 1,5-diphenylcarbazide (in Czech).
[42] J. Hair, R. Anderson, R. Tatham, W. Black, Multivariate Data Analysis, 5th ed., Prentice Hall, Upper Saddle River, NJ, 1998.
[43] S. Sharma, Applied Multivariate Techniques, Wiley, New York, 1996.
[44] M. Vega, R. Pardo, E. Barrado, L. Deban, Assessment of seasonal and polluting effects on the quality of river water by exploratory data analysis, Water Res., 32 (1998) 3581–3592.
[45] I. Jolliffe, Principal Component Analysis, 2nd ed., Springer, New York, 2002.
[46] E.K. Paleologos, M.T. Al Nahyan, S. Farouk, K. Papapetridis, Ensemble Contaminant Transport Modelling and Bayesian Decision-Making of Groundwater Monitoring, Proceedings of the Fifteenth International Conference on Civil, Structural and Environmental Engineering Computing, Civil-Comp Press, Stirlingshire, 2015, 10 pp.
[47] M.M.A. El-Salam, G.I. Abu-Zuid, Impact of landfill leachate on the groundwater quality: a case study in Egypt, J. Adv. Res., 6 (2015) 579–586.
[48] K. Karthik, R. Mayildurai, R. Mahalakshmi, S. Karthikeyan, Physicochemical analysis of groundwater quality of Velliangadu Area in Coimbatore District, Tamil Nadu, India, Rasayan J. Chem., 12 (2019) 409–414.
[49] O.O. Ololade, S. Mavimbela, S.A. Oke, R. Makhadi, Impact of leachate from northern landfill site in bloemfontein on water and soil quality: implications for water and food security, Sustainability, 11 (2019) 4238, doi: 10.3390/su11154238.
[50] D. Dąbrowska, A. Witkowski, M. Sołtysiak, Representativeness of the groundwater monitoring results in the context of its methodology: case study of a municipal landfill complex in Poland, Environ. Earth Sci., 77 (2018) 266, doi: 10.1007/s12665-018-7455-x.
[51] J.K. Böhlke, R.L. Smith, D.N. Miller, Ammonium transport and reaction in contaminated groundwater: application of isotope tracers and isotope fractionation studies, Water Resour. Res., 42 (2006), doi: 10.1029/2005WR004349.
[52] E. Koda, A. Sieczka, P. Osiński, Ammonium concentration and migration in groundwater in the vicinity of waste management site located in the neighborhood of protected areas of Warsaw, Poland, Sustainability, 8 (2016) 1253, doi: 10.3390/su8121253.
[53] N.M. Dubrovsky, K.R. Burow, G.M. Clark, J.M. Gronberg, P.A. Hamilton, K.J. Hitt, D.K. Mueller, M.D. Munn, B.T. Nolan, L.J. Puckett, M.G. Ruper, T.M. Short, N.E. Spahr, L.A. Sprague, W.G. Wilber, The Quality of Our Nation's Waters—Nutrients in the Nation's Streams and Groundwater, 1992–2004: (No. 1350), U.S. Geological Survey Circular, 2010.
[54] R. Cossu, L.E. Zuffianò, P.P. Limoni, G. De Giorgio, P. Pizzardin, T. Miano, D. Mondelli, R. Garavaglia, C. Carella, M. Polemio, M., How can the role of leachate on nitrate concentration and groundwater quality be clarified? An approach for landfills in operation (Southern Italy), Waste Manage., 77 (2018) 156–165.

[55] J. Wu, W.D. Zhao, J. Lu, S. Jin, J.Q. Wang, J.Z. Qian, Geographic information system based approach for the investigation of groundwater nitrogen pollution near a closed old landfill site in Beijing, China, Environ. Eng. Manage. J., 17 (2018) 1095–1101.
[56] Z. Han, S. Wang, J. Zhao, X. Hu, Y. Fei, M. Xu, Identification of nitrogen-sources in an aquifer beneath a municipal solid waste landfill in the vicinity of multiple pollutant sources, J. Environ. Manage., 268 (2020) 110661, doi: 10.1016/j.jenvman.2020.110661.
[57] E. Koda, A. Miszkowska, A. Sieczka, P. Osiński, Heavy metals contamination within restored landfill site in Poland, Environ. Geotech., 7 (2020) 512–521.
[58] T. Alemayehu, G. Mebrahtu, A. Hadera, D.N. Bekele, Assessment of the impact of landfill leachate on groundwater and surrounding surface water: a case study of Mekelle city, Northern Ethiopia, Sustain. Water Resour. Manage., 5 (2019) 1641–1649.
[59] Pinfohouse (Pollution Prevention Services, Iowa Department of Natural Resources) Anatomy of a Tire. Available at: https://p2infohouse.org/ref/11/10504/html/intro/tire.htm (Accessed, March 30, 2021).
[60] J.A. Izbicky, J.W. Ball, T.D. Bullen, S.J. Sutley, Chromium, chromium isotopes and selected trace elements, western Mojave Desert, California, Appl. Geochem., 23 (2008) 1325–1352.
[61] A.F. Adamczyk, A. Haładus, Zanieczyszczenie chromem wód podziemnych i powierzchniowych w rejonie składowisk Z.Ch. Alwernia, Mat. Konf. Geochemiczne. Hydrochemiczne i biochemiczne zmiany środowiska przyrodniczego na obszarach objętych antropopresją. Analiza stanu, prognoza, zapobieganie, Wyd. AGH Kraków, 1991 (in Polish).
[62] C.O. Akinbile, Environmental impact of landfill on groundwater quality and agricultural soils in Nigeria, Soil Water Res., 7 (2012) 18–26.
[63] J.W. Ball, J.A. Izbicky, Occurrence of hexavalent chromium in ground water in the western Mojave Desert, California, Appl. Geochem., 19 (2004) 1123–1135.
[64] California Water Boards 2018, Maximum Contaminant Levels and Regulatory Dates for Drinking Water, U.S. EPA vs California, Last Updated October 2018. Available at: https://www.waterboards.ca.gov/drinking_water/certlic/drinkingwater/documents/ccr/mcls_epa_vs_dwp.pdf (accessed March 10, 2021).
[65] World Health Organization (WHO), Chromium in Drinking-Water, Draft Background Document for Development of WHO Guidelines for Drinking-Water, 2019. Available at: https://www.who.int/water_sanitation_health/water-quality/guidelines/chemicals/draft-chromium-190924.pdf (Accessed March 10, 2021).
[66] B. Hölting, Hydrogeologie, 4 Aufl., F. Enke Edit. Stuttgart, 1992.
[67] E.K. Paleologos, Stochastic Flow and Particle Tracking Modeling for Contaminant Detection in Shallow, Heterogeneous Aquifers, Universal Journal of Mathematics & Mathematical Sciences, 5 (2014) 149.
[68] C.W. Liu, K.H. Lin, Y.M. Kuo, Application of factor analysis in the assessment of groundwater quality in a blackfoot disease area in Taiwan, Sci. Total Environ., 313 (2003) 77–89.

Appendix A

Table A1
ANOVA analysis of statistical difference between parameters monitored at the Zdounky landfill ($p < 0.05$)

Dependent variable	(I) VAR00001	(J) VAR00001	Mean difference (I–J)	Std. Error	Sig.
NH_4^+	MV-1	MV-2B	0.00375	1.39215	0.998
		MV-4	–4.23000	1.39215	**0.004**
		MV-5	–0.07500	1.39215	0.957
		MV-6	–2.61500	1.39215	0.069
	MV-2B	MV-1	–0.00375	1.39215	0.998
		MV-4	–4.23375	1.39215	**0.004**
		MV-5	–0.07875	1.39215	0.955
		MV-6	–2.61875	1.39215	0.068
	MV-4	MV-1	4.23000	1.39215	**0.004**
		MV-2B	4.23375	1.39215	**0.004**
		MV-5	4.15500	1.39215	**0.005**
		MV-6	1.61500	1.39215	0.254
	MV-5	MV-1	0.07500	1.39215	0.957
		MV-2B	0.07875	1.39215	0.955
		MV-4	–4.15500	1.39215	**0.005**
		MV-6	–2.54000	1.39215	0.077
	MV-6	MV-1	2.61500	1.39215	0.069
		MV-2B	2.61875	1.39215	0.068
		MV-4	–1.61500	1.39215	0.254
		MV-5	2.54000	1.39215	0.077
NO_3^-	MV-1	MV-2B	–9.88250	32.34121	0.762
		MV-4	119.54125	32.34121	**0.001**
		MV-5	98.43250	32.34121	**0.004**
		MV-6	121.01625	32.34121	**0.001**
	MV-2B	MV-1	9.88250	32.34121	0.762
		MV-4	129.42375	32.34121	**0.000**
		MV-5	108.31500	32.34121	**0.002**
		MV-6	130.89875	32.34121	**0.000**
	MV-4	MV-1	–119.54125	32.34121	**0.001**
		MV-2B	–129.42375	32.34121	**0.000**
		MV-5	–21.10875	32.34121	0.518
		MV-6	1.47500	32.34121	0.964
	MV-5	MV-1	–98.43250	32.34121	**0.004**
		MV-2B	–108.31500	32.34121	**0.002**
		MV-4	21.10875	32.34121	0.518
		MV-6	22.58375	32.34121	0.490
	MV-6	MV-1	–121.01625	32.34121	**0.001**
		MV-2B	–130.89875	32.34121	**0.000**
		MV-4	–1.47500	32.34121	0.964
		MV-5	–22.58375	32.34121	0.490