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Monitoring and Assessment of Groundwater Quality at Landfill Sites: Selected Case Studies of Poland and the Czech Republic

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Abstract: In order to protect the components of natural environment, each landfill must be properly secured and the monitoring program should be adopted. This study aims to present a comparative analysis of groundwater quality at selected landfill sites in Poland and the Czech Republic, with a special attention given to the levels and temporal changes of heavy metals (HMs) concentrations measured in collected groundwater samples. A secondary objective was to detect possible leakages of pollutants from the landfill body, into the groundwater, and further into the environment. The assessment of groundwater quality was based on a comparison of HMs concentrations with standards provided by the European environmental laws. On the basis of the long-term monitoring period, it was revealed, for the Polish landfill site, that the groundwater quality is improving over time, especially due to remedial works applied. For the Czech landfill, it was observed that the quality of groundwater is not negatively affected by the operation of the landfill, but in the immediate vicinity of the landfill, the groundwater quality is significantly affected by the agricultural use of neighbouring lands, as well as by the storage of construction and demolition wastes. The results showed that the leachate did not leak outside the landfills, especially due to minimal concentrations of HMs, measured in groundwater samples, taken from the piezometers located in the outflow direction from the landfills.

Keywords: contamination; groundwater; heavy metals; reclamation

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1. Introduction

Landfilling in the 21st century is still the most common method of waste management [1,2]. Nevertheless, proper Circular Economy (CE) policies have contributed to a recent decreasing trend in European countries [3]. Landfills are sources of greenhouse gases emissions [4] and possible threats to the aquatic environment, thus reduction and reclamation of landfill sites can be considered as a way to the implementation of Green Deal (GD) assumptions. The landfills have to be reclaimed in accordance with the GD strategies, including the application of environmentally-friendly technologies, and the solutions, then, can increase the protection of human health and the environment against hazardous and harmful substances arising from the landfill. The solutions (degassing systems, landfill gas development monitoring) should be applied to reduce methane emissions from the landfills. Engineering activities should be also perform to restore ecosystems and biodiversity during the biological reclamation of the landfills.

Urbanization and industrialization lead to the production of solid wastes that finally are disposed to landfills [5–7]. Improper location and operation of landfills may negatively affect soil, flora, air, surface water, and groundwater [7,8]. The most significant problem related to municipal solid waste (MSW) landfills are leachates [9]. The leachates are composed of a complex of chemicals, including organic substances, inorganic and organic salts, and heavy metals (HMs), which vary in concentration depending on a set of factors [10]. The composition of leachates is complemented by water, which constitutes 99% of their volume [11]. The leachates are formed as a result of rainwater seepage through the waste, biochemical processes occurring in the waste cells, and water in the waste itself [12]. The leachates can carry insoluble liquids (e.g., oils) and suspended solids composed of small particles [13]. In case of inadequate landfill sealing, these substances could infiltrate into the soil, contaminating groundwater for a long period of time [8]. The landfills, especially the old ones, without any protection systems applied, are most exposed to infiltration [14].

The landfills are also known as latent sources of HMs pollution in the environment. Due to a lack of drainage systems, landfills lead to the occurrence of toxic and threatening HMs in groundwater [15]. Therefore, landfill reclamation is a rescue for contaminated groundwater. Recently, it has been noted that the application of reclamation treatments positively affects groundwater quality [16]. The protection of water from contamination in adjacent areas can be applied by the construction of a vertical barrier, a component of which is also a suitable leachate drainage system [17]. In addition to their sealing function and role in preventing the migration of pollutants, the vertical barriers are also a tool for a CE. The materials of which the barriers are made are composed of mineral wastes and by-products, so they can be considered an excellent instrument of sustainable development (SD) [18]. By adapting legislation, the CE strategy is simpler to implement. GD and CE strategies for landfills are extremely important to reduce the negative interaction between the economy, the environment, and natural resources. The implementation of technologically innovative solutions contributes to SD and continuous improvement of the environment. Meeting the need for SD, it is desired to rehabilitate and close sanitary landfills. Moreover, where possible, it is also needed to use available resources from landfills.

Besides the introduction of additional technical seals in landfills, nowadays, there are also more complex leachate transfer systems. In the past, in old Polish landfills, the leachate was mainly captured by drainage or by band ditches and stored in retention tanks and collection wells. Nowadays, the systems are more complex and also have recirculation elements, pumping stations for leachate transfer, pumping pipelines, and systems for distribution of contaminated water on the surface of the landfill, with the possibility of dilution for irrigation purposes [17].

Apart from leachate capture systems, groundwater monitoring is also extremely important to ensure that leachate does not seep into the environment and cause harm to human health [13]. Groundwater monitoring should be performed at special measurement points, such as piezometers [11]. Each landfill should have at least three piezometers, with one located upstream and the others downstream from the landfill [11]. The Regulation of the Minister of the Environment on landfills [19] requires testing of groundwater parameters, such as pH, electrical conductivity (EC), total organic carbon (TOC), HM content, such as copper (Cu), zinc (Zn), lead (Pb), cadmium (Cd), chromium (Cr^{+6}), and mercury (Hg), and total polycyclic aromatic hydrocarbons (PAHs). Each country sets its limits that cannot be exceeded. However, in the European Union (EU) they are very similar to each other.

This article presents a comparative analysis of groundwater quality at selected landfills in two EU countries: Poland and the Czech Republic, which use different technical securing systems [20,21]. The difference in applied securing systems is related to the technical solutions used in the base and cover systems. It was hypothesized that properly reclaimed and technically secured landfills do not pose the threat to the groundwater. Hence, the aim of the study was to assess the groundwater contamination by HMs at MSW landfills and surrounding environment and to analyse the factors influencing contamination.

2. Materials and Methods

2.1. Characteristics of the Study Sites

2.1.1. Radiowo Landfill

The Radiowo landfill (52°16′37″ N, 20°52′45″ E) is located in the commune Stare Babice and partially in Warsaw, Poland. The landfill covers an area of approximately 16 ha, and its altitude is approximately 60 m. The landfill is surrounded by Forest Park Bemowo and Nature Reserves from the south and east sides. From the north side, the landfill is adjacent to a composting plant located in the Warsaw–Bielany District. On the west side, in the vicinity of the landfill slope, there are industrial facilities and a railway (Figure 1).

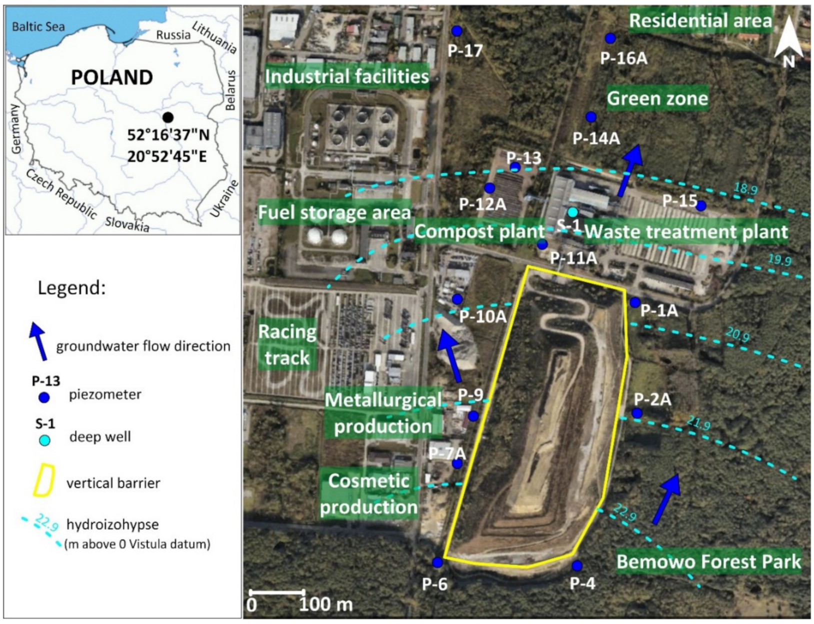

Figure 1. The view on the Radiowo landfill location together with its monitoring network and surroundings.

The landfill was constructed in the early 1960s when no legal restrictions, regarding location and environmental issues, were taken into consideration. The Radiowo landfill was used for disposal of the municipal waste from Warsaw in the period 1962–1991.

The landfill area is located within a denuded glacial plateau defined as the Warsaw-Błonie erosion terrace. It is the central part of the Warsaw basin, characterized as syncline, in the Mesozoic formations. The syncline is filled with tertiary formations covered with a complex of quaternary sediments. The oldest sediments, drilled at a depth of 270 m, are marls and limestones of the Upper Cretaceous. Above, there are sandy-dusty Oligocene deposits with glauconite, with a thickness of 60–90 m, then Miocene silts (with brown coal) up to several meters thickness, and the Pliocene layer formed as variegated clays (Tertiary roof). The surface of the Pliocene and its thickness are strongly diversified as a result of glacitectonic processes. In the area of the landfill, the depth to the Pliocene roof ranges from a few to several meters, and locally, it is exposed on the surface in the north-western part.

It was revealed in our previous study [22,23] that the landfill subsoil consists of sandy soils (thickness in range 2–15 m). In the south-eastern part of the area, the thickness of sandy sediments is significant (locally up to 20 m) due to the presence of erosion-denudation depressions in the top of the glacial tills and the existence of a small valley form (a small watercourse in the past). Towards the north, the thickness of the surface sandy sediments is small, and, in places, it fades into till clays with numerous sand pockets. Sandy layers with organic and mud interbedding at the northern border of the landfill are associated with aeolian-lake formations in the vicinity of the Lipkowska Woda stream.

The first groundwater level was measured at the depth 0.3–2 m below the surface level, with fluctuations dependent on the weather and the local drainage conditions. The deeper part of the subsoil consists of low permeable boulder and varved clays, with thickness 25–40 m, and average hydraulic conductivity at 10^{-9} m/s. Low hydraulic gradients in the south and east part of the area adjacent to the landfill result in very low velocity of the groundwater flow (inflow to the landfill). It was measured that the groundwater level increases from approximately 0.25 m in the south part to more than 0.75 m in the north-west part of the landfill site. Small increase of 0.2 m of the groundwater level was observed in the south part of the landfill. Along the downstream direction from the landfill, the velocity of groundwater flow in sandy soils is about 5.8×10^{-7} m/s (for a hydraulic gradient 10‰).

Initially, the protection system against the contamination was not applied at the landfill site. Since 1994, the reclamation works have been implemented. At that time, the solutions of stability improvement were introduced. Moreover, the mineral capping, planting of slopes, and installation of drainage and recirculation system were applied. One of the most important solutions introduced at the landfill, as an effect of reclamation works, was the installation of bentonite vertical barrier [24]. The function of the vertical barrier application was to minimize the migration of contaminants to the soil–water environment. The vertical barrier (width 0.6 m) was installed at a depth of 2 m below the top of the clayey soils, at the level 3.5–22 m below the surface. The permeability of the vertical barrier was below 10^{-9} m/s (subsoil practically impermeable) [23].

The monitoring network of groundwater quality at the landfill and its surroundings consists of eleven piezometers (Figure 1). Nine of them (P-2A, P-7A, P-9, P-10A, P-11A, P-12A, P-14A, P-15, P-17) are located downstream to the landfill and reflect the possible impact of the landfill on the quality of groundwater. Two piezometers (P-4 and P-6A) are located upstream to the landfill and reflect the background quality of groundwater.

2.1.2. Zdounky Landfill

The Zdounky landfill (49°14′29.2″ N 17°18′30.3″ E) is located in the area of Nětčice, part of the municipality of Zdounky in the Kroměříž district in the Zlín Region in the Czech Republic. The Zdounky landfill is classified to the S-OO3 group. The landfill was constructed at agricultural land of an area of approximately 10 ha. Currently, the landfill covers an area of approximately 7 ha. The object was designed to carry a volume of 907,000 m^3 of wastes, with the idea of serving for a population of 75,000 people. The total amount of deposited waste in the landfill is 1,280,750 Mg [25].

The surroundings of the Zdounky landfill are built of Paleogene rocks of marine origin, belonging to the Outer flysch and the Magura flysch. In terms of regional geological structure, the site is situated on the edge of the Carpathian mountain-forming system. 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. 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 $m^3\ s^{-1}$.

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.

For the Zdounky landfill, the base sealing system consists of several layers, including the mineral liner of clayey soil of a thickness 1 m, the geomembrane of high density polyethylene (HDPE) of 1.5 mm thickness and a drainage layer composed of sandy materials and used tires. Water that comes into contact with wastes and is therefore (or could potentially be) contaminated is treated separately from rainwater and never comes into contact with the soil environment.

The monitoring network of groundwater quality at the landfill and its vicinity consists of six piezometers (Figure 2).

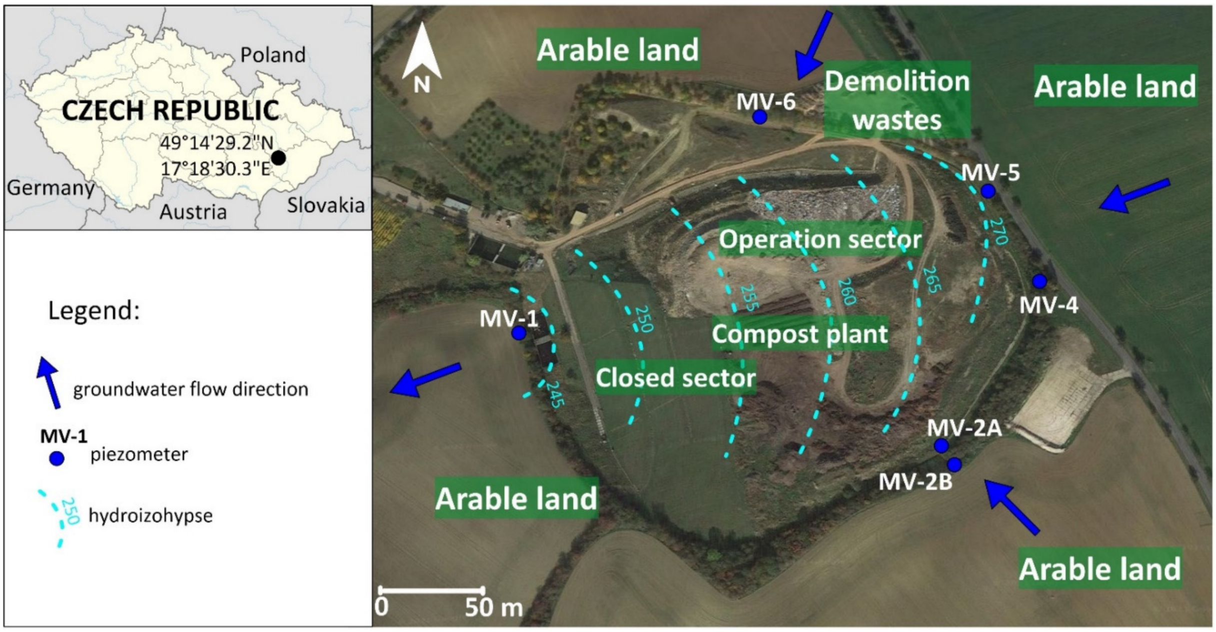

Figure 2. The view on the Zdounky landfill location together with its monitoring network and surroundings.

One piezometer (MV-1) is located downstream the landfill and therefore is treated as a monitoring point that reflects the impact of the landfill on groundwater quality. The rest of piezometers (MV-2A, MV-2B, MV-4, MV-5, and MV-6) are located upstream to the landfill and reflect the background and the possible impact of surrounding areas on groundwater. The piezometer MV-2B is a replaced monitoring of MV-2A. The monitoring points MV-4 and MV-5 reflect the groundwater quality at the east site of the landfill and may be influenced by the runoff of contaminants from the surrounding arable lands. The MV-5 monitoring point can be also directly affected by the runoff of contaminants from the sector intended to the storage of the construction and demolition wastes. The piezometer MV-6, due to its location, does not indicate the possible adverse effect of the landfill on the groundwater, nevertheless can reflect the impact of arable lands on groundwater quality.

2.2. Sampling and Experimental Analysis

Groundwater samples were collected twice a year, in spring (I) and in autumn (II), in accordance with the procedure outlined in PN-EN ISO 5667-3:2013-05 [26] and PN-ISO 5667-11:2004 [27] standards.

The measurements of HMs were started straight away after the collection of the samples. To assure the appropriateness of sampling, each sample was accompanied with the measurement of pH, temperature, and EC. All samples were collected into smoked glass bottles and stored at a temperature of 1–5 °C. The assessment of groundwater quality at the landfill site was based on a comparison of metal concentration measured in collected samples with standards provided by environmental laws (Table 1).

Moreover, for the Czech landfill, the monitoring results were compared with the so-called “critical values”, obtained from a long-term monitoring of groundwater quality at the landfill and its surroundings. The selected “critical” values were in accordance with the operating rules of landfills in the Czech Republic.

The analysed areas are low urbanized and the water from shallow water intakes is used for economic and municipal purposes. Due to that, the monitoring results of water quality were also compared with the limit values set for drinking water, established by the World Health Organization (WHO).

In the performed study, the monitoring period 2008–2018 was taken into consideration for the purpose of groundwater quality assessment at landfill sites.

Table 1. Standards of water quality according to the national and international law.

Standard	Specific Value	Cr_{total} [mg L^{-1}]	Zn [mg L^{-1}]	Pb [mg L^{-1}]	Cd [mg L^{-1}]	Hg [mg L^{-1}]
Ireland's EPA, 2001 [28]	I/PV	0.05	-	0.01	0.005	0.001
	MCLG	0.1	-	0	0.005	0.002
US EPA, 2018 [29]	MCL	0.1	-	-	0.005	0.002
	SDWR	-	5	-	-	-
WHO [30]	-	0.05	1	0.01	0.003	0.006
Polish Regulation [a] [31]	-	0.05		0.01	0.005	0.001
Polish Regulation [b] [32]	-	0.05	1	0.1	0.005	0.001
Geochemical bacground [32]	-	0.0001–0.010	0.005–0.050	0.001–0.010	0.0001–0.0005	0.00005–0.001

Notes: 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, [a]—limits set for drinking water, [b]—the limit values for "III" groundwater quality class (threshold values for good chemical status).

2.3. Assessment of the HM Pollution Level

In order to provide an overview of water quality with respect to the content of HMs, the Heavy Metal Evaluation Index (*HEI*) was calculated. The *HEI* index was determined using the following formula [33]:

$$HEI = \sum_{i=1}^{n} \frac{H_c}{H_{max}} \tag{1}$$

where:

H_c—monitored value of i-th heavy metal; H_{max}—maximum admissible concentration of i-th HM, according to WHO [30].

On the basis of *HEI* index the quality of water was assessed, concerning the following interpretations of the results: $HEI < 10$ indicates low level of contamination, $10 < HEI < 20$ indicates medium level of contamination, and $HEI > 20$ indicates high level of contamination [34].

The second measure applied for the evaluation of water quality, in terms of HMs, was Heavy Metal Pollution Index (*HPI*). This parameter allows for the determination of an overall quality of drinking water, using the following equations [35].

$$HPI = \frac{\sum_{i=1}^{n} W_i \cdot Q_i}{\sum_{i=1}^{n} W_i} \tag{2}$$

$$Q_i = \sum_{i=1}^{n} \frac{|M_i - I_i|}{S_i - I_i} \times 100 \tag{3}$$

where:

W_i—unit weight of the i-th HM, Q_i—sub-index of the i-th HM, n—number of considered HM, M_i—monitored value of the-th HM, I_i- ideal value of the i-th HM, S_i—standard value of the i-th HM.

Regarding the HPI index, the quality of water was classified assuming that: $HPI < 100$ indicates low contamination by HMs, $HPI = 100$ indicates the threshold value, and $HPI > 100$ indicates high contamination by HMs. It is also required that the water characterized by the $HPI > 100$ cannot be used for drinking purposes.

3. Results

3.1. Temporal Distribution of HMs in Groundwater

In the case of Radiowo landfill, it was revealed that the concentrations of total chromium (Cr_{total}) in groundwater were many times lower than acceptable for good groundwater chemical status (Figure 3).

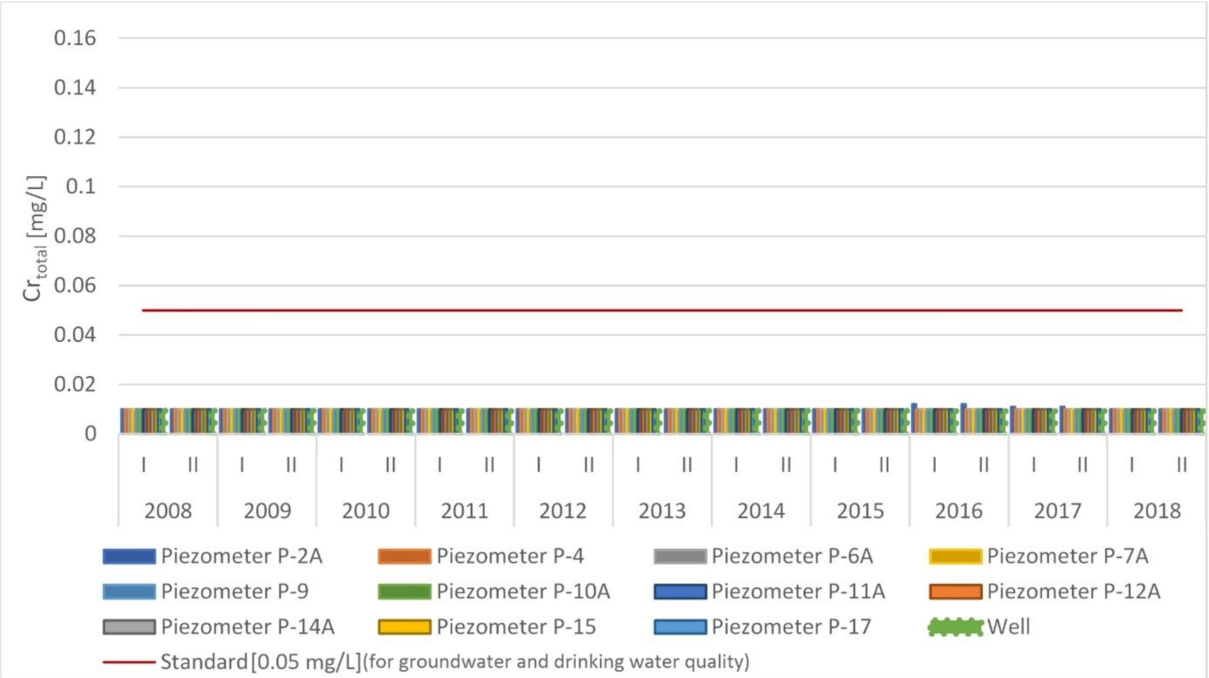

Figure 3. Temporal distribution of Cr_{total} in groundwater at the Radiowo landfill site.

The maximum concentrations of the Cr_{total} did not exceed 0.01 mg/L. The maximum values of this parameter were measured in piezometer P-9A (located downstream the landfill). Increased concentrations of Cr_{total}, however not exceeding the limit values, can be also the result of the location of this piezometer in the vicinity of the metallurgical production area (possible runoff of contaminants). The maximum concentrations of the Cr_{total} exceeding the range of values typical for the chemical background (0.0001–0.010 mg/L) can also indicate the possible impact of additional external sources of contamination.

The results of the monitoring studies performed at the Zdounky landfill indicate incidental peaks of Cr_{total} concentration in several piezometers (Figure 4). This situation can be clearly visible in piezometer MV-5, where the concentrations were even three times higher than the WHO standard [30] and the level required for the good chemical status of groundwater [30]. Incidentally, the concentrations were also elevated in piezometers MV-1, MV-4, MV-2A/B, and MV-6. These concentrations were not directly related to the possible impact of the landfill, as these piezometers are located at the upstream direction from the landfill, and therefore reflect only the possible influence of surroundings on groundwater quality. The increased Cr_{total} concentrations in that case can be the effect of the runoff of contaminants from the storage sector of tires or construction and demolition wastes [36,37].

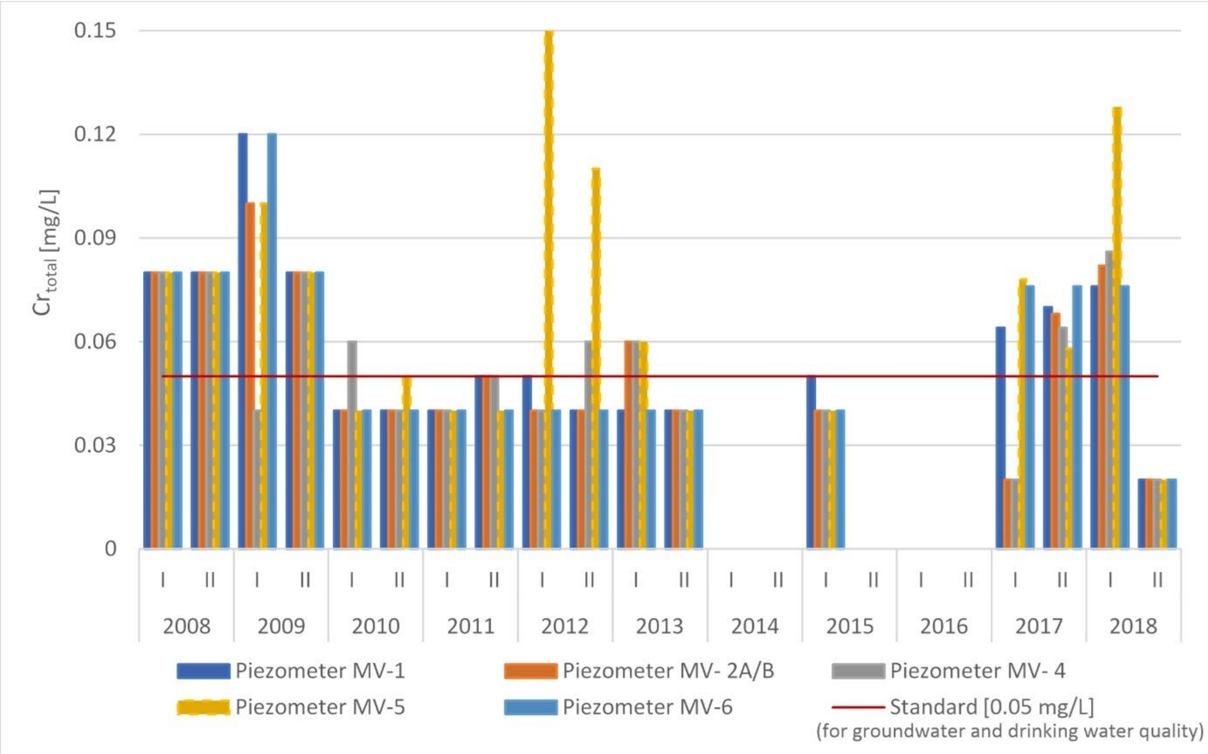

Figure 4. Temporal distribution of Cr_{total} in groundwater at the Zdounky landfill site.

For both monitored landfills, it was observed that concentrations of Zn are below the maximum limit set for good chemical status of groundwater quality. For the Radiowo landfill, some temporal changes in Zn concentration and its fluctuations were observed (Figure 5).

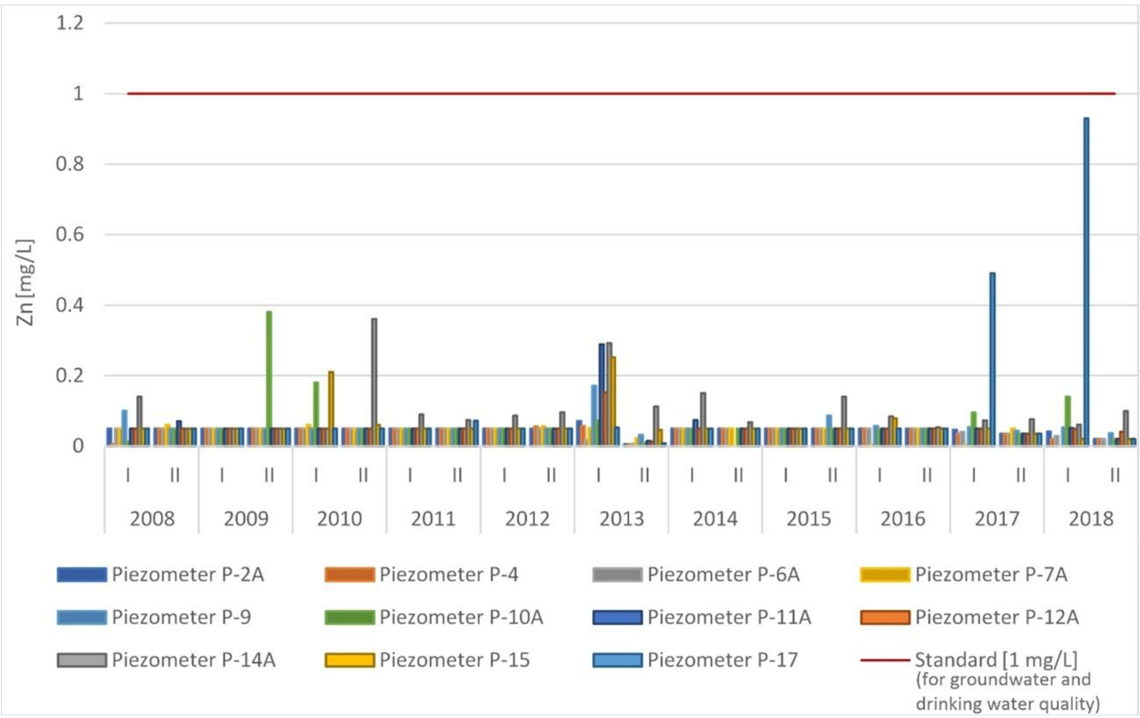

Figure 5. Temporal distribution of Zn in groundwater at the Radiowo landfill site.

The variability of Zn concentrations measured in piezometers can be attributed to the development of adjacent area and its possible impact on groundwater. Incidental increase in concentration of Zn may be due to the runoff from the adjacent street and the outflow of contaminants from the service area and production facilities.

At the Zdounky landfill site, for piezometers located downstream and upstream from the landfill, slight Zn concentrations in groundwater were observed, characteristic for the limits typical for the range of geochemical background (Figure 6). Therefore, the landfill and its surroundings do not contribute to groundwater contamination by Zn at the study site.

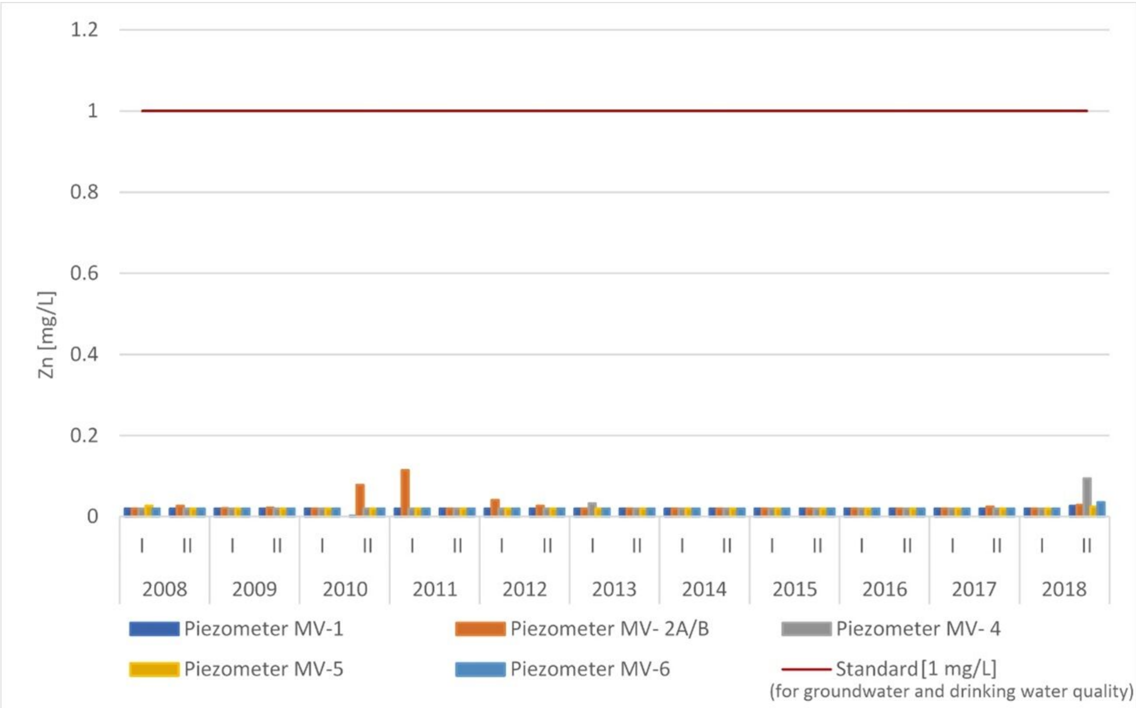

Figure 6. Temporal distribution of Zn in groundwater at the Zdounky landfill site.

The concentrations of Pb measured in groundwater in the area of the Radiowo landfill were below the limit set for the good chemical status (Figure 7).

In majority, the concentrations were in the range typical for the geochemical background. There were only two peaks for the entire monitoring period where the concentrations of Pb exceeded the value of 0.01 mg/L and therefore indicated that the water did not meet the requirements for water intended for human consumption. The similar tendency was observed at the Zdounky landfill site, for which the concentrations of Pb in groundwater indicated the good groundwater chemical status (Figure 8).

In 2008, the concentrations of Pb exceeded the limit set for the water intended for drinking purposes. Almost throughout the entire monitoring period, the values of Pb were in accordance with the limits of the concentrations characteristic for the geochemical background.

For the Radiowo landfill, the Cd concentrations in groundwater indicate good chemical status (Figure 9). The values were characteristic for the geochemical background and did not exceed the values subjected to the water intended for human consumption. This is very important from the point of view of the assessment of the landfill impact on groundwater. Taking into consideration that Cd is very strongly adsorbed on muds, humus, and organic matter, leading to the possibility of entry to the food chain, and subsequent accumulation in tissue, it is important to conduct thoughtful engineering activities that will not contribute to the release of this element to the environment [38].

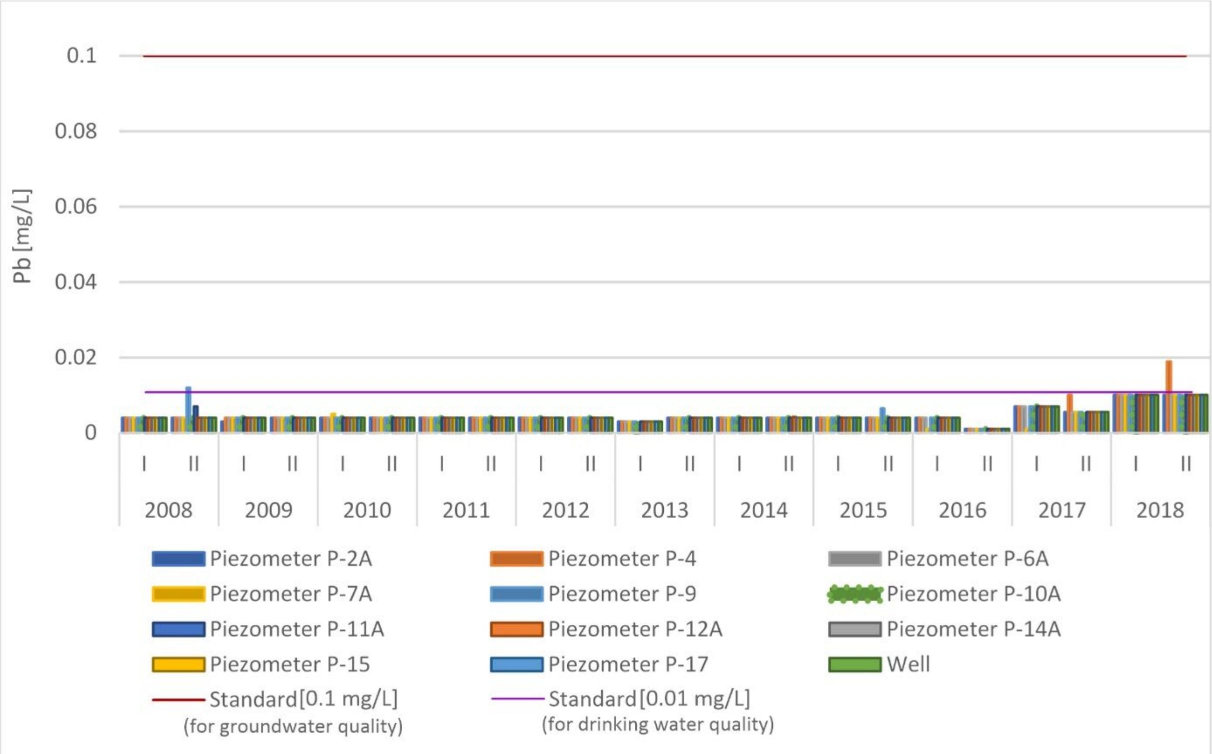

Figure 7. Temporal distribution of Pb in groundwater at the Radiowo landfill site.

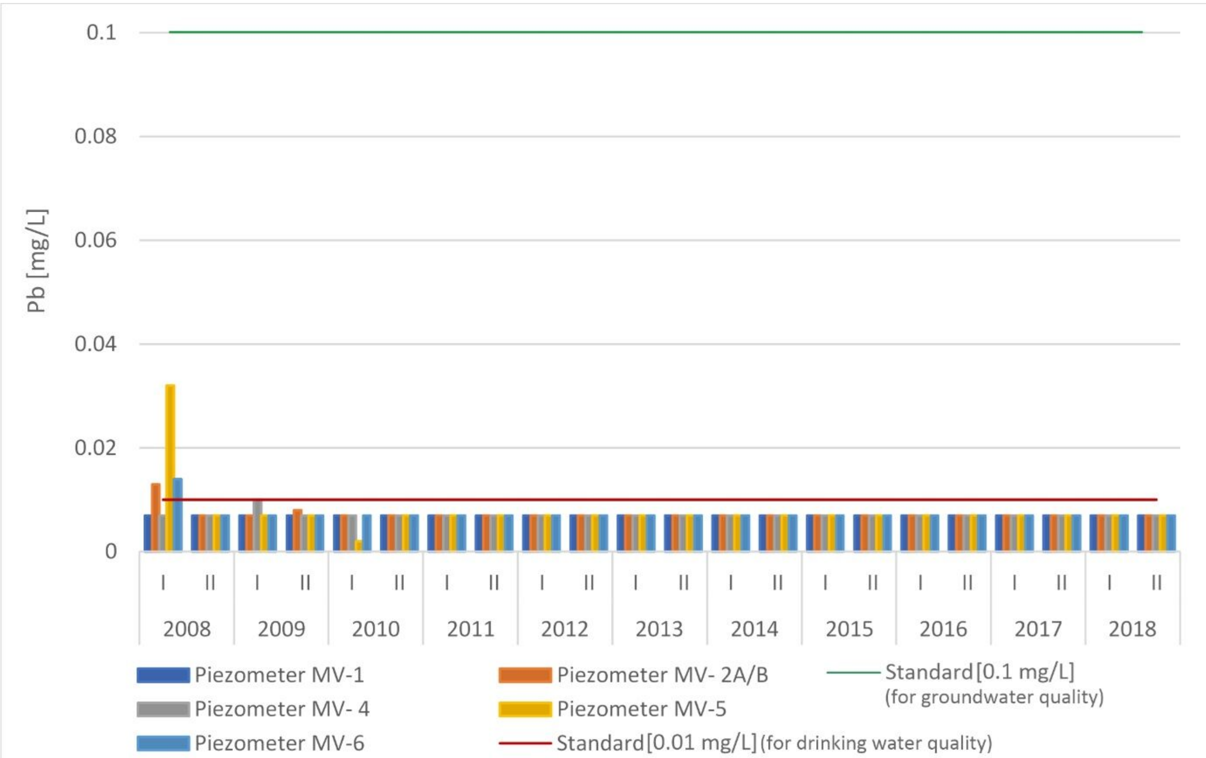

Figure 8. Temporal distribution of Pb in groundwater at the Zdounky landfill site.

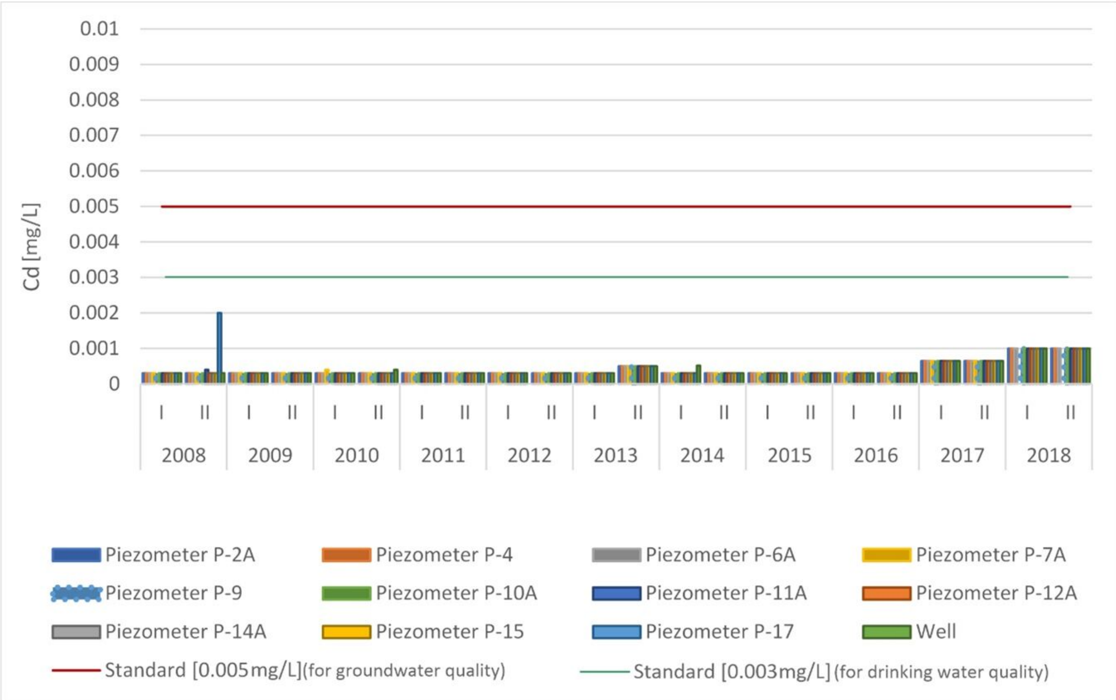

Figure 9. Temporal distribution of Cd in groundwater at the Radiowo landfill site.

The concentrations of Cd at the Zdounky landfill (Figure 10) were slightly higher than those recommended by WHO for drinking water. When it comes to other environmental requirements for drinking water quality, it can be stated that values of Cd observed at the Zdounky landfill were in accordance with the limits set by law [29]. In general, in comparison with the requirements for groundwater quality, the concentrations of Cd in groundwater indicated its good chemical status.

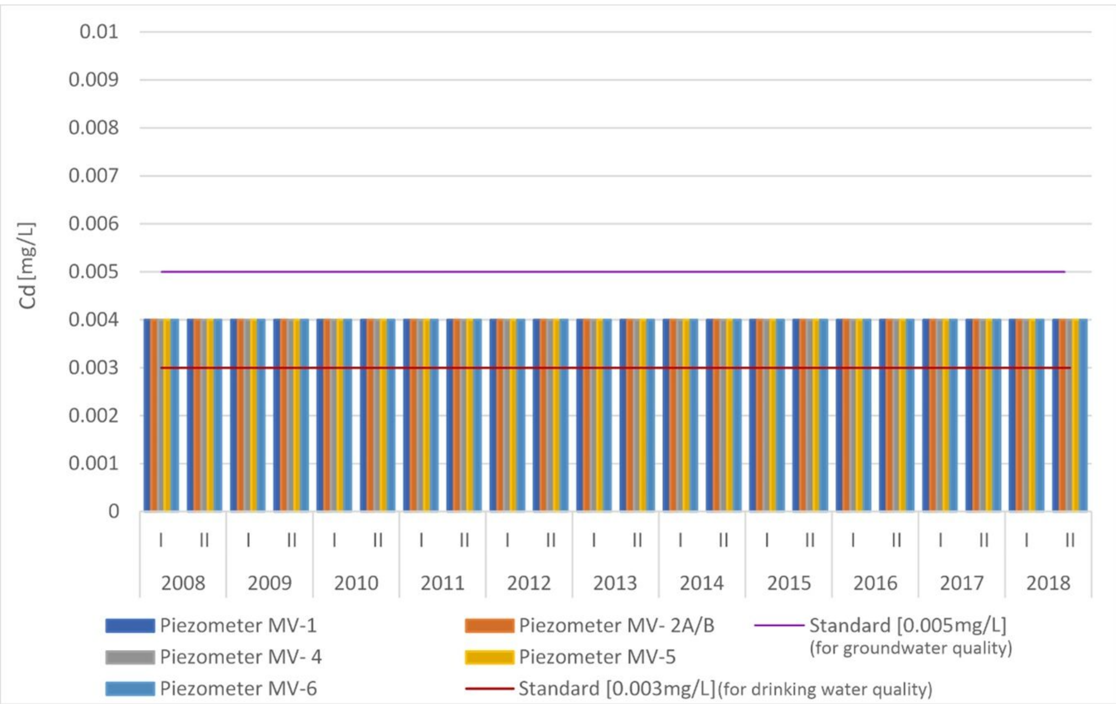

Figure 10. Temporal distribution of Cd in groundwater at the Zdounky landfill site.

For both monitored landfills, the concentrations of Hg were measured in ranges characteristic for the geochemical background (Figures 11 and 12).

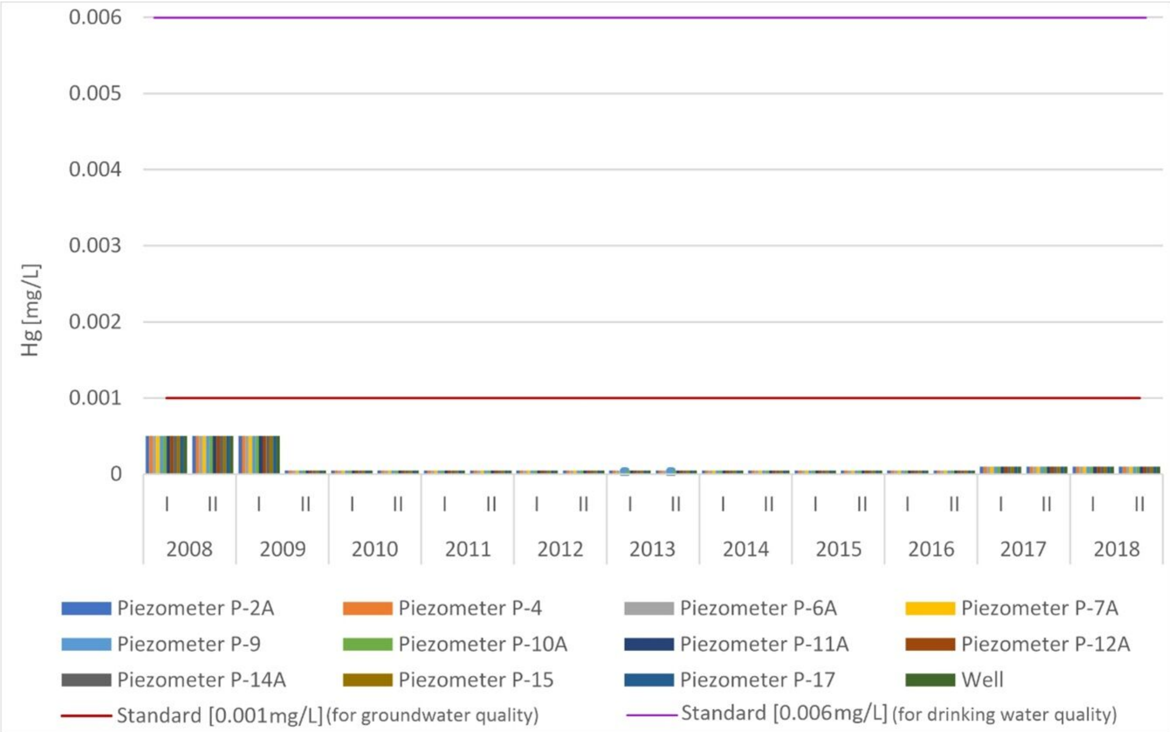

Figure 11. Temporal distribution of Hg in groundwater at the Radiowo landfill site.

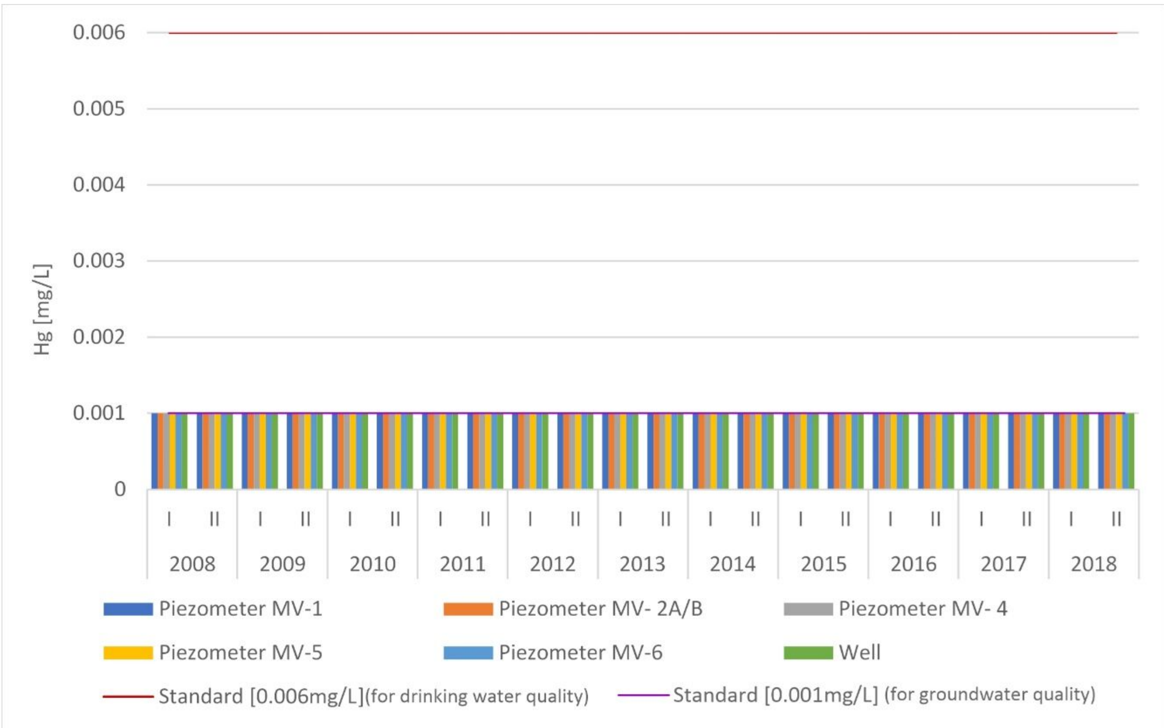

Figure 12. Temporal distribution of Hg in groundwater at the Zdounky landfill site.

There was no impact of the landfills on groundwater observed, in terms of Hg pollution. The requirements indicating good groundwater chemical status were met, as well as the limit set for drinking water was not exceeded. The results showed that the landfills and their surrounding facilities had no adverse influence on the Hg concentrations in groundwater.

Based on the results of statistical evaluation (Appendix A, Tables A1–A5), it was also checked what the form of HMs distribution is by using the measure of skewness [39] and kurtosis [40]. It was revealed that Cr_{total} concentrations in groundwater at Radiowo landfill site are highly skewed for each piezometer within the monitoring network. In contrast, for the Zdounky landfill, it was obtained that the distribution of Cr_{total} is approximately symmetric (Table A1). The calculated values of kurtosis indicated that the distribution of Cr_{total} in groundwater is leptokurtic for each piezometer at the Radiowo landfill site, whereas for the Zdounky landfill, in each of the piezometers, the distribution of this element was defined as platykurtic.

Regarding Zn concentrations in groundwater, it was found that the distribution is highly skewed for the Radiowo landfill. The same phenomenon was observed for almost each piezometer at the Zdounky landfill, with the exception of the piezometer MV-6 (distribution of Zn moderately skewed). For both landfill sites it was indicated that with regards to the values of kurtosis, the distribution of Zn are leptokurtic (Table A2).

As presented in Tables A3–A5, it was also revealed that the distribution of Pb, Cd, and Hg in groundwater at the Radiowo landfill is almost always highly skewed for each piezometer. The same tendency was also observed for the Zdounky landfill. The distribution of Pb in groundwater at the Radiowo landfill is characterized as leptokurtic, excluding piezometers P-9 and P-11A, for which the distribution can be defined as platykurtic. Some exceptions are also visible for the Zdounky landfill in which the distribution of Pb is leptokurtic for each piezometers, excluding MV-5.

3.2. Assessment of Groundwater Contamination by HMs Using Indices

By the *HEI* indices calculated for both analysed landfills, it was stated that the water can be classified as low contaminated by HMs (Figure 13).

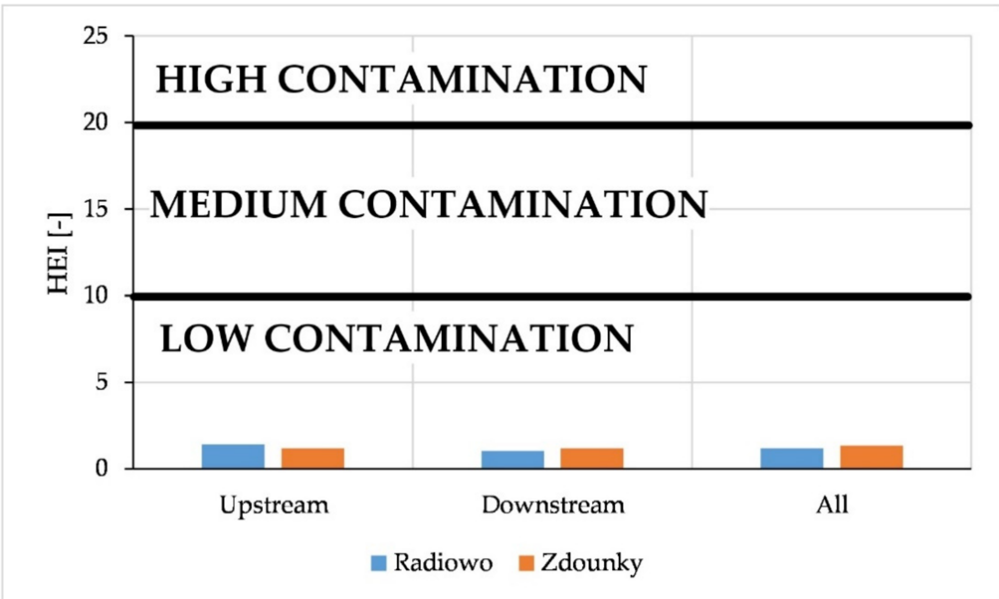

Figure 13. Comparison of *HEI* index between Radiowo and Zdounky landfill.

For the Radiowo landfill, the *HEI* index calculated based on piezometers located downstream the landfill was lower than for piezometers located at the upstream direction from the landfill. It may indicate that more significant impact on groundwater quality may have surrounding facilities than the landfill itself. For the Zdounky landfill, the *HEI* index calculated on the basis of data from piezometers located at the upstream and downstream the landfill were equal to 1.20 and 1.19, respectively. It may suggest that the Zdounky landfill had almost the same impact on groundwater contamination by HMs as surrounding areas. Because the possible impact of the landfill on groundwater can be evaluated by the analysis of observations from piezometers located downstream the landfill, it can be stated that the Zdounky landfill had greater potential to pollute the groundwater, in terms of HMs, than the Radiowo landfill. Regarding the *HPI* indices, it can be indicated that both landfills did not affect the groundwater in aspect of the deterioration of water quality for human consumption (Figure 14).

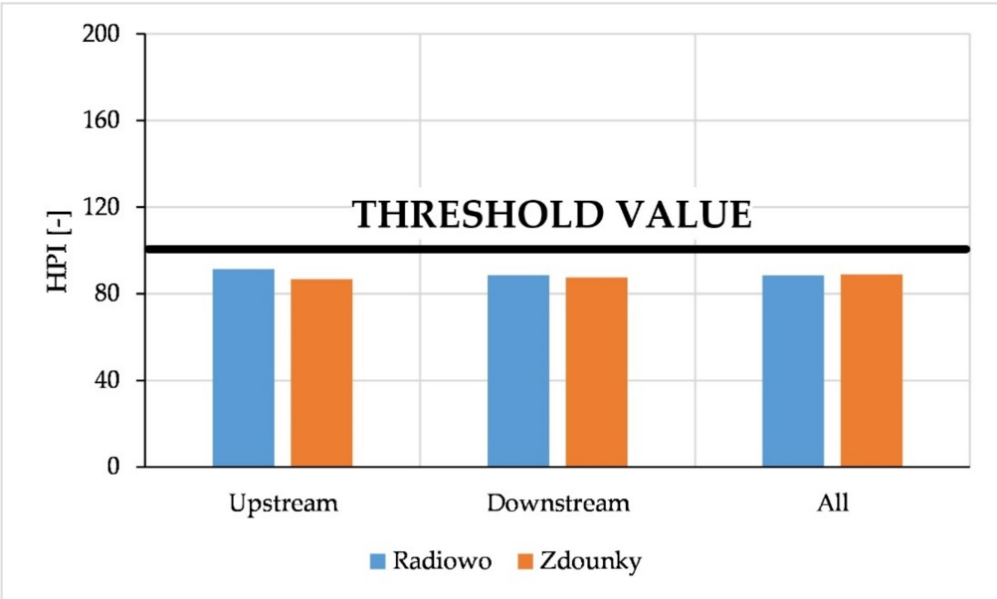

Figure 14. Comparison of *HPI* index between Radiowo and Zdounky landfill.

The threshold value of *HPI* = 100 was not exceeded. The values of *HPI* obtained at the lower level for piezometers located downstream the landfill than upstream the landfill pointed out that the surrounding areas may have greater impact on groundwater quality than the existence and operation of the landfill.

4. Discussion

Several studies have confirmed that the occurrence of HMs in groundwater can be treated as an indicator of the leakage of leachates from the landfill body [41,42]. The outcomes of presented study may suggest that properly applied sealing and technical protection of MSW landfills have significant effect on the reduction in groundwater pollution by HMs. This was evidenced by both the results from the 10-year monitoring period and the calculated HM pollution indices, which are one the most important methods of water quality evaluation. These measures facilitate the classification of water in terms of their utilization for drinking purposes [43]. However, a significant attention should be paid to incidental increases in concentrations of HMs in groundwater. Although HM pollution indices allow for a global assessment of water quality, including its suitability for human consumption, they do not highlight the alarming increases in the concentration of HMs (for example, Cr_{total} at the Zdounky landfill site for current study).

The groundwater contamination by HMs is a global problem, often related to the impact of landfills on the deterioration of water resources. In the case of the analysed landfills, no significant impact on groundwater quality was demonstrated, especially due to the application of an effective protection system against the contamination. The role of engineering solutions was highlighted also by Zeng et al. [44] who stated that Tibet waste landfills with effective liners caused no significant impact on groundwater contamination at MSW landfill sites. Nevertheless, even if the landfill is properly designed, some emergency cases can occur. The most probable reason of contamination near engineered landfill may result from the failure of the leachate collecting system (well, pipeline, pumping system) and the rapture of geomembrane [45]. The degradation of geomembrane associated with material aging and defect evolution, may increase leachate leakage significantly. Despite the fact that the impact of leachate leakage on surrounding groundwater may be small in the short term, it begins to increase in the medium term, whereas in the long term, the probability of groundwater pollution may reach 100% [46].

For the analysed sites, the greater impact on groundwater quality had neighbouring areas than the landfills. Low concentrations of HMs in groundwater, as revealed in study performed at the Radiowo landfill, may be also due to high adsorption of HMs in the unsaturated zone [47,48].

Nevertheless, in many regions of the world, landfills directly cause an increase in the concentration of HMs in groundwater. The study performed by Murtaza et al. [15] revealed that concentrations of Cd, Cu, Pb, and As in groundwater were higher than the permissible limit (60% of monitored samples exceeded the WHO limit). The increased levels of HMs in groundwater were attributed to the leaching of contaminants from the landfill body [49]. The leachate percolation and migration is also frequently evident from the quality of groundwater near uncontrolled landfills. For example, it was observed for the landfill in India (Kolkata) that groundwater quality may be affected by the leachate percolation and migration. In that case, it was found for not engineered landfill that major ions, including HMs, in groundwater were found to be high, indicating the influence of landfill leachate intrusion to groundwater [50]. The other sources of HMs contribution in the landfill area may be open wastewater drains, industrial, and intensive agricultural activities [47]. Generally, the problem of groundwater pollution exists in developing countries where the landfills are not properly engineered to prevent the release of contaminants [51]. For example, in the research performed by Amano et al. [52] in Ghana, it was found that the quality of water resources at the landfill site were above the limit for drinking water based on *HPI* index. They also revealed that the concentrations of Cd in surface water and groundwater, within the surrounding area of the waste landfill site, were extremely high. The literature findings also indicate the elevated concentrations of Pb in groundwater near landfills, as well as the pollution caused by Cu and Cr which may contribute to the toxicological risk at landfill sites [53–55]. The problem of environmental contamination is also visible, with respect to the occurrence of Al, As, Cd, Co, Cr, Cu, Fe, Mn, Mo, Ni, Pb, V, and Zn in groundwater, which may also affect the human health [56]. The occurrence of toxic HMs may be due to plastics, batteries, leather, metallic items, paint products, fluorescent lamps, metal scrap, and steel industrial waste from the surrounding areas [49]. In context of our findings, increased concentrations of HMs in groundwater were observed for Cr at Zdounky landfill, what was attributed to the direct impact of the runoff of contaminants from the area of tires and demolition waste storage.

The results of several studies described above indicate that the groundwater contamination at the landfill sites should be a concern. Groundwater quality assessment should be carried out with systematic measurements of HMs concentrations in groundwater for the purposes of determining the causes of alarming changes and taking corrective measures. It is also required to prevent other secondary pollutions, including air pollution and soil pollution [57].

5. Conclusions

Properly managed MSW landfills do not have to be regarded as the threat to the environment. The nuisances related to their existence can be reduced to minimum while the sealing system is used, or the reclamation works are successively applied at the landfill site. Applied engineering activities may contribute to SD in the region of the landfill and also may improve the quality of human life. Thanks to applied protection systems, the people are not exposed to the direct impact of toxic substances arising from the landfills. The establishing of a monitoring network of groundwater quality is also helpful when assessing the impact of the landfill on groundwater as well as when checking the impact of surrounding areas on the groundwater deterioration.

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Appendix A

Table A1. Summary of statistical evaluation of Cr_{total} [mg/L] content in groundwater at the Radiowo and Zdounky landfill sites.

Piezometer	Mean	Median	Minimum	Maximum	Variance	Standard Deviation	Skewness	Kurtosis
				Radiowo landfill				
P2-A	0.010	0.010	0.010	0.012	0.000	0.001	2.232	3.898
P-4	0.010	0.010	0.010	0.010	0.000	0.000	n.d.	n.d.
P-6A	0.010	0.010	0.010	0.010	0.000	0.000	n.d.	n.d.
P-7A	0.010	0.010	0.010	0.011	0.000	0.000	4.690	22.000
P-9	0.012	0.010	0.010	0.030	0.000	0.005	3.352	12.267
P-10A	0.011	0.010	0.010	0.018	0.000	0.002	3.877	15.730
P-11A	0.011	0.010	0.010	0.018	0.000	0.002	3.954	16.185
P-12A	0.010	0.010	0.010	0.010	0.000	0.000	n.d.	n.d.
P-14A	0.010	0.010	0.010	0.010	0.000	0.000	n.d.	n.d.
P-15	0.010	0.010	0.000	0.018	0.000	0.003	−1.043	11.007
P-17	0.010	0.010	0.010	0.010	0.000	0.000	n.d.	n.d.
				Zdounky landfill				
MV-1	0.04	0.04	0.00	0.12	0.00	0.03	0.25	−0.15
MV-2 A/B	0.04	0.04	0.00	0.10	0.00	0.03	0.12	−0.96
MV-4	0.04	0.04	0.00	0.09	0.00	0.03	−0.12	−1.11
MV-5	0.05	0.05	0.00	0.15	0.00	0.04	0.49	−0.33
MV-6	0.04	0.04	0.00	0.12	0.00	0.03	0.34	−0.31
P2-A	0.046	0.050	0.005	0.071	0.000	0.013	−1.898	5.738

n.d.—not defined.

Table A2. Summary of statistical evaluation of Zn [mg/L] content in groundwater at the Radiowo and Zdounky landfill sites.

Piezometer	Mean	Median	Minimum	Maximum	Variance	Standard Deviation	Skewness	Kurtosis
				Radiowo landfill				
P-4	0.042	0.050	0.005	0.057	0.000	0.016	−1.601	1.380
P-6A	0.043	0.050	0.007	0.050	0.000	0.013	−1.788	2.108
P-7A	0.045	0.050	0.004	0.060	0.000	0.016	−2.100	3.482
P-9	0.059	0.050	0.032	0.171	0.001	0.030	3.143	10.982
P-10A	0.072	0.050	0.009	0.380	0.006	0.079	3.223	11.738
P-11A	0.059	0.050	0.015	0.289	0.003	0.053	4.253	19.287
P-12A	0.052	0.050	0.012	0.152	0.001	0.024	3.553	16.294

Table A2. *Cont*.

Piezometer	Mean	Median	Minimum	Maximum	Variance	Standard Deviation	Skewness	Kurtosis
P-14A	0.105	0.080	0.050	0.360	0.006	0.079	2.397	5.745
P-15	0.065	0.050	0.020	0.252	0.003	0.056	2.881	7.807
P-17	0.107	0.050	0.008	0.930	0.043	0.207	3.594	13.116
Zdounky landfill								
MV-1	0.016	0.020	0.000	0.027	0.000	0.009	−1.271	−0.111
MV-2 A/B	0.025	0.020	0.000	0.114	0.001	0.026	2.316	6.489
MV-4	0.019	0.020	0.000	0.094	0.000	0.019	2.877	11.715
MV-5	0.016	0.020	0.000	0.027	0.000	0.009	−1.232	−0.155
MV-6	0.016	0.020	0.000	0.036	0.000	0.010	−0.741	0.255

Table A3. Summary of statistical evaluation of Pb [mg/L] content in groundwater at the Radiowo and Zdounky landfill sites.

Piezometer	Mean	Median	Minimum	Maximum	Variance	Standard Deviation	Skewness	Kurtosis
Radiowo landfill								
P2-A	0.005	0.004	0.001	0.010	0.000	0.002	1.775	3.717
P-4	0.005	0.004	0.001	0.019	0.000	0.004	2.886	9.544
P-6A	0.005	0.004	0.001	0.010	0.000	0.002	1.787	3.853
P-7A	0.004	0.004	0.001	0.010	0.000	0.002	1.419	3.403
P-9	0.005	0.004	0.001	0.012	0.000	0.003	1.529	2.077
P-10A	0.005	0.004	0.001	0.010	0.000	0.002	1.787	3.853
P-11A	0.005	0.004	0.001	0.010	0.000	0.002	1.505	2.650
P-12A	0.005	0.004	0.001	0.010	0.000	0.002	1.779	3.846
P-14A	0.005	0.004	0.001	0.010	0.000	0.002	1.787	3.853
P-15	0.005	0.004	0.001	0.010	0.000	0.002	1.787	3.853
P-17	0.005	0.004	0.001	0.010	0.000	0.002	1.787	3.853
Zdounky landfill								
MV-1	0.008	0.007	0.000	0.070	0.000	0.014	4.336	19.809
MV-2 A/B	0.009	0.007	0.000	0.070	0.000	0.014	4.238	19.156
MV-4	0.008	0.007	0.000	0.070	0.000	0.014	4.303	19.596
MV-5	0.054	0.045	0.000	0.150	0.002	0.043	0.489	−0.334
MV-6	0.009	0.007	0.000	0.070	0.000	0.015	3.569	13.791

Table A4. Summary of statistical evaluation of Cd [mg/L] content in groundwater at the Radiowo and Zdounky landfill sites.

Piezometer	Mean	Median	Minimum	Maximum	Variance	Standard Deviation	Skewness	Kurtosis
Radiowo landfill								
P2-A	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-4	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-6A	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-7A	0.0004	0.0003	0.0003	0.0007	0.0000	0.0001	2.2065	3.7459
P-9	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-10A	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-11A	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.0789	3.4079
P-12A	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-14A	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-15	0.0004	0.0003	0.0003	0.0010	0.0000	0.0002	2.1061	3.4662
P-17	0.0005	0.0003	0.0003	0.0020	0.0000	0.0004	2.9317	9.5506
Zdounky landfill								
MV-1	0.0031	0.0040	0.0000	0.0040	0.0000	0.0017	−1.3988	−0.0572
MV-2 A/B	0.0031	0.0040	0.0000	0.0040	0.0000	0.0017	−1.3988	−0.0572
MV-4	0.0031	0.0040	0.0000	0.0040	0.0000	0.0017	−1.3988	−0.0572
MV-5	0.0031	0.0040	0.0000	0.0040	0.0000	0.0017	−1.3988	−0.0572
MV-6	0.0031	0.0040	0.0000	0.0040	0.0000	0.0017	−1.3988	−0.0572

Table A5. Summary of statistical evaluation of Hg [mg/L] content in groundwater at the Radiowo and Zdounky landfill sites.

Piezometer	Mean	Median	Minimum	Maximum	Variance	Standard Deviation	Skewness	Kurtosis
				Radiowo landfill				
P2-A	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-4	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-6A	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-7A	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-9	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-10A	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-11A	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-12A	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-14A	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-15	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
P-17	0.0001	0.0001	0.0001	0.0005	0.0000	0.0002	2.2082	3.2965
				Zdounky landfill				
MV-1	0.0008	0.0010	0.0000	0.0010	0.0000	0.0004	−1.3988	−0.0572
MV-2 A/B	0.0008	0.0010	0.0000	0.0010	0.0000	0.0004	−1.3988	−0.0572
MV-4	0.0008	0.0010	0.0000	0.0010	0.0000	0.0004	−1.3988	−0.0572
MV-5	0.0010	0.0010	0.0000	0.0050	0.0000	0.0010	3.2508	13.8396
MV-6	0.0008	0.0010	0.0000	0.0010	0.0000	0.0004	−1.3988	−0.0572

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