Desalination and Water Treatment
www.deswater.com

doi: 10.5004/dwt.2020.25374

199 (2020) 13–23
September

Impact of a municipal waste landfill reclamation in Poland on groundwater quality based on field test

Elżbieta H. Grygorczuk-Petersons*, Józefa Wiater

Bialystok University of Technology, Department of Technology in Environmental Engineering, Wiejska 45E, 15-351 Bialystok, Poland, emails: e.petersons@pb.edu.pl (E.H. Grygorczuk-Petersons); j.wiater@pb.edu.pl (J. Wiater)

Received 27 September 2019; Accepted 17 December 2019

ABSTRACT

The aim of the study was to determine influence of municipal landfill before and after reclamation on groundwater quality. Piezometers were located near municipal landfill and were tested between 2009 and 2017, before reclamation in 2010 and for 7 years after reclamation. Conductivity, pH, heavy metal concentration, TOC, PAHs were determined in water. A negative impact of landfill site on water quality before reclamation and almost 6 years after reclamation was found. Stabilization of water quality occurred 7 years after reclamation. The decision to close disorganized landfills was justified and all of them should be recultivated in order to protect aquifers from their negative impact.

Keywords: Municipal waste landfill; Reclamation of landfill; Groundwater; Quality

1. Introduction

The sanitary landfill method is the oldest and the most common waste treatment technology in the world [1–4] because of its large handling capacity, low investment, and low operating cost [5]. However, it occupies a large land area and is not effective in reducing waste [1]. Some studies indicated that almost 95% of municipal solid waste (MSW) was disposed of by landfilling worldwide [6]. In the European Union (EU), the landfilling rate (landfilled waste as share of generated waste) compared with municipal waste generation dropped from 64% in 1995 to 23% in 2017 [7]. This reduction can partly be attributed to the implementation of European legislation, for instance Directive 62/1994 on packaging and packaging waste and Directive 31/1999 on landfill stipulated that Member States were obliged to reduce the amount of biodegradable municipal waste going to landfills. It is known that the impact of MSW landfills can cause pollution of all environmental components [8] including in particular the groundwater [9] as a consequence of its infiltration by the deposited waste [10].

Reduction of water pollution below the landfill should be ensured by limiting the amount of leachate [11], an appropriate leachate collection system [12], maintaining the leachate at the lowest possible level [1,11,13,14]. The amount of leachate, among others, depends on the permeability coefficient, and this indicator is closely related to the depth and unit weight MSW [1,11,13]. Also, the stability of the landfill depending, among others, on the physical composition, unit mass and durability factor of MSW [1,11,13] has an impact on water pollution around municipal waste landfills.

The negative impact of landfills on groundwater has been confirmed by many studies [12,15–27].

Shortly before Poland accession to the European Union, a vast majority of municipalities owned small municipal waste disposal sites. At that time, local authorities invested in the most popular waste disposal method, which was municipal waste deposit, while at the same time choosing the cheapest solutions for facilities. These landfills were most often built for convenience of inhabitants. They were often located in areas that were not geologically adapted at all (e.g., in depleted gravel excavations). For years the problem

* Corresponding author.

Presented at the 14th Conference on Microcontaminants in Human Environment, 4–6 September 2019, Czestochowa, Poland

1944-3994/1944-3986

of improperly built landfills was considered as non-existent. The approaches to landfills have changed since joining the European Union. The EU imposed an obligation on Poland to adapt its waste management system to its requirements. Polish state adapts its internal law to EU legislation. For this reason, landfills which did not comply with formal and legal requirements and had defective structures threatening environment were closed. In 2005, a total of 1,010 non-hazardous and neutral landfills were operated in Poland, and in 2017 their number decreased to 301 [28]. A requirement to monitor impact of landfills at various stages of their operation on environment has also been implemented. The monitoring studies, which were carried out, made possible to illustrate the problem of the impact of a small, not possessing drainage system and basal sealing of the bottom of the landfill on the environment at every stage of operation and after closure and reclamation.

The aim of the study was to determine influence of small landfill site situated in the territory of the European Union, on groundwater quality before and after reclamation.

2. Area and research methodology

The landfill site is located in the south-western part of Podlaskie Voivodeship, on Wysokomazowiecka Plateau in Poland (Fig. 1). It covers an area of 2.7 ha, in which the accommodation takes up to 1.30 ha and covers a total capacity of 16,200 m^3 (capacity used 12,068 m^3). The average thickness of the waste of the landfill site is 4.0 m.

It is located in the excavation of gravel mine, with natural sealing by impermeable soils, and its exploitation started in 2003. The landfill site was operated in 2003–2010 using the horizontal method consisting in layering the waste in layers 1-m thick. It is separated from roads and agricultural crops by a strip of protective greenery which constitutes a barrier eliminating the adverse impact of landfill on environment.

At landfill site, household and utility wastes from households and public utility facilities, agricultural, agri-food processing, large-size, used furniture, unsuitable for use, household electrical appliances, ash, slag, construction debris, excavation soil, stabilized sewage sludge and sludge from water treatment were deposited. The morphological composition of waste stored at landfill site is as follows: mineral and fine fraction for storage - 30%, biological - 22%, paper - 16%, textiles - 2%, plastics - 12%, glass - 12%, metals - 4%, hazardous waste in municipal waste - 2%. The inorganics had a big value in landfilled MSW, which according to Yang et al. [1] is good for the stability of a landfill site. The unit weight ranged from 5.1 to 5.5 kN/m^3 (laboratory tests) with a depth of 0–3 m. The landfill was characterized by a low degree of compaction [1]. All three accommodation units are surrounded by a ditch that collects leachate from landfill to drainage basins. Directions of water run-off at the landfill site are: N $\rightarrow$ S and E $\rightarrow$ W (Fig. 2). Groundwater occurs at a depth of 2.5–6 m below ground level (Fig. 3). In 2008, three piezometers were installed. Location of piezometer P1 was determined in direction of groundwater flow into landfill area and piezometers P2, P3 in groundwater outflow (Figs. 1 and 3).

In March 2010, reclamation of landfill was started and it was completed according to schedule on December 31, 2010. The waste was compacted and landfill was shaped. Mechanical reclamation was followed by uniform application of mineral soil layer and top soil-forming layer. The selection of upper reclamation layer was to shape an optimal 15 cm isolation layer of compact mineral soil and 25 cm soil-forming layer with a canopy shaped in a streamlined way outside the reclamation area.

In order to obtain a biological reclamation layer, a layer of humus was decomposed. In order to ensure optimal growth conditions for the grass mixture, this layer was fed with multicomponent fertilisers of the Azofoska type in the amount of 5 kg/m^2 of reclaimed land. The seedbed area was prepared at least 2 weeks before planned sowing of grass mixtures. Following composition of grass mixture was applied: red fescue - share in mixture 50%, boneless broom 20%, French pansy 20%, meadow grass 20%, white clover 10%. The mixture of grasses was applied in amount of 12 kg/100 m^2 on landfill site and 4 kg/100 m^2 on landfill slope. A phytosanitary zone was created for runoff water by planting trees of black alder, grey alder and bumblebee and elderberry bushes.

Water was taken from piezometers twice a year during periods before, during and after reclamation (from 2009 to 2017), in which pH, conductivity, concentration of total carbon, PAHs, heavy metals such as zinc (Zn), copper (Cu), lead (Pb), cadmium (Cd), chromium (Cr), mercury (Hg) were determined. The height of water column was also measured.

All analytical determinations were carried out, in accordance with the Polish Standard, with reference methods were specified in the Regulation of the Minister of Environment of 19 July 2016 on the forms and methods of monitoring surface water and groundwater bodies [29]. Electrolytic conductivity was measured using a conductivity meter.

Total organic carbon was determined using TOC (IR spectrometry), PAHs using GC gas chromatography. Heavy metals, such as chromium, copper and zinc, were determined by flame atomic absorption spectrometry (AAS), lead and cadmium by inductively excited atomic emission spectrometry (AES) and mercury by atomic absorption spectrometry (PN-EN 12338) (results for zinc, cadmium and mercury were given in this study, other results were below the limit of detection). Groundwater research was carried out between 2009 and 2017. Results of research were compared with limit values of indicators, water quality according to classes for groundwater contained in ordinances of Minister of Environment of 21 December 2015 on criteria and methods of assessing the condition of bodies of groundwater [30].

The minimum, maximum and arithmetic mean were determined in groundwater from piezometers - before and after reclamation of the municipal landfill site. The results of the physiochemical elements of the groundwater from the piezometers were used to conduct an analysis and to make conclusions based on the assessment of the influence of the landfill site on the quality of the groundwater in the immediate vicinity.

3. Research results and discussion

On the basis of the analysis of the quality of the groundwater in the piezometers P1–P3 situated in the area of the landfill site, most of the tested parameters did not meet the standards of very good water quality, that is, first class

(Table 1). At the same time, higher average pollutant values were found in water taken from the P2 piezometer, compared with the P3 and P1 piezometer.

Height of water column in piezometers was variable during studies (Fig. 4). The lowest level was recorded before landfill reclamation and the highest in the first year after landfill reclamation. In piezometers at outflow, it was similar to the year 2015, later differences between these piezometers were about 1 m. Height of water column in piezometer on water influence differed from piezometers located at outflow, and later it was similar. Water intake from quarter-strand formations occurred between 1 and 5 m under surface level, and such water occurred in case of analysed landfill site. Witkowski and Żurek [31] recorded a similar height of water column when investigating impact of the Tychy landfill on groundwater quality.

Conductivity of water from piezometers differed depending on piezometer location and research period (Fig. 5). During whole period of study, in the water of piezometers P1–P3 it assumed values characteristic for II–III class of purity [30]. In water of piezometers, P1 (tributary) conductivity ranged from 500 to 1,500 μS/cm, P2 from 550 to 2,343 μS/cm and P3 from 558 to 1,514 μS/cm. Definitely, the highest values and most dynamic changes of this indicator were observed in water of the second piezometer (P2) located in direct contact with the landfill. Mor et al. [32], Deshmukh and Aher [33], and Wiater [34] also report on impact of landfills on conductivity in piezometer water. The highest value occurred shortly before the start of reclamation, after 5 years of landfill operation. Since 2010, conductivity in piezometer water has decreased, and 2 years after reclamation, its value has increased exponentially. Between 2014 and 2015, there was a decrease in conductivity, followed by a slight increase. The lowest conductivity was found in

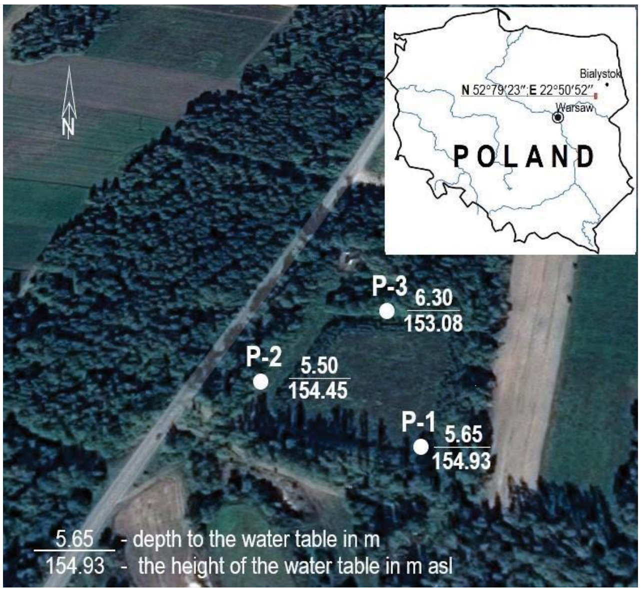

Fig. 1. Location and satellite image of the landfill.

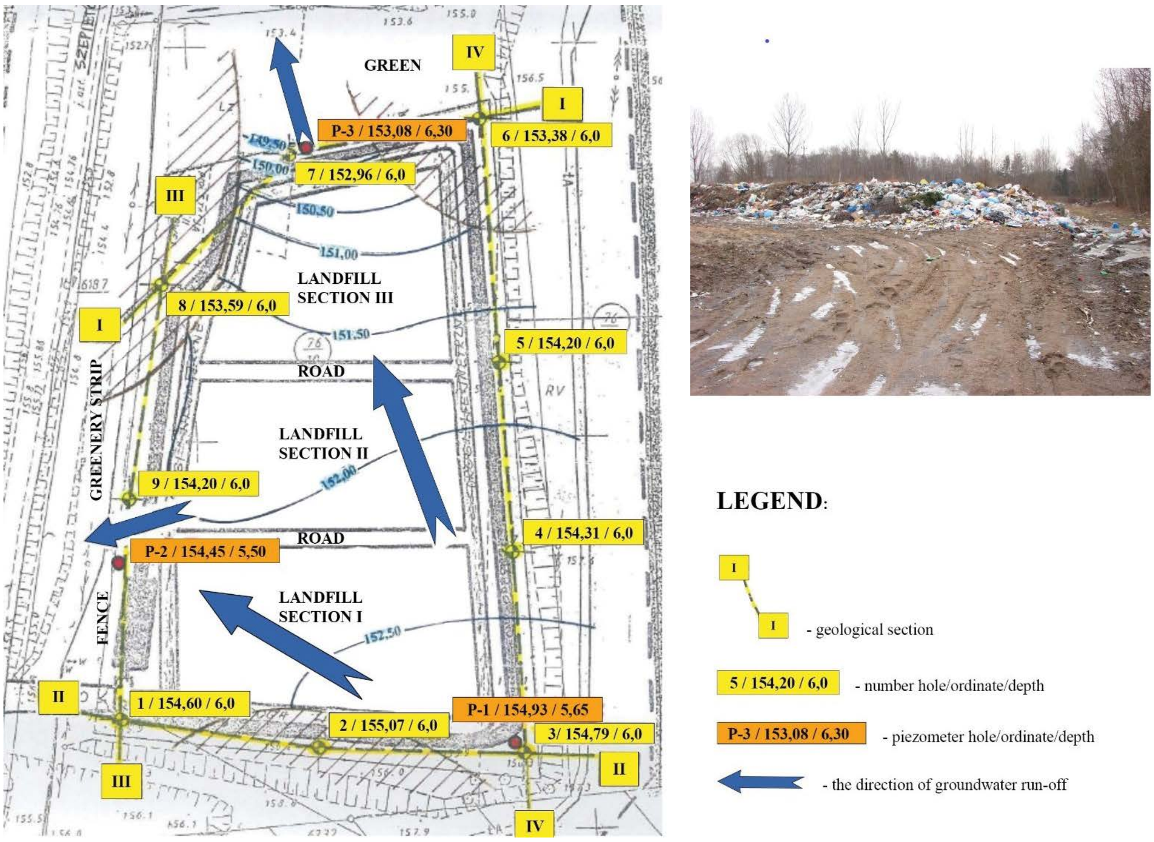

Fig. 2. Piezometers arrangement.

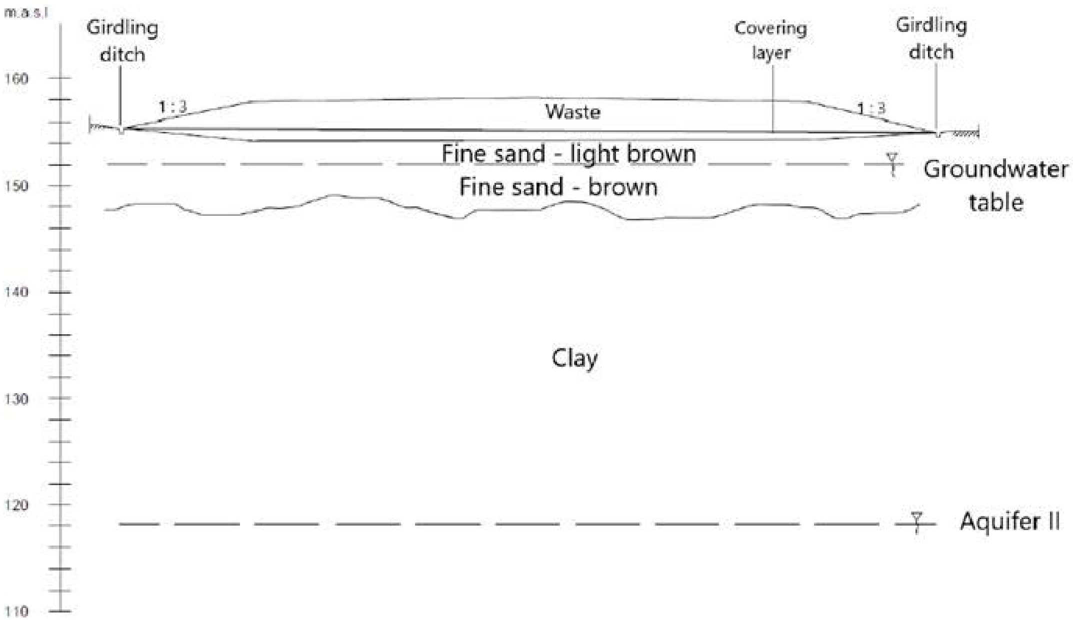

Fig. 3. Section of the landfill.

the water from third piezometer (P3), although in the first years it was higher than in water from the inflow, which indicates landfill impact on the value of this indicator. Water from first piezometer (P1) only in initial period of studies was characterized by low concentration of mineral salts, and later it was higher than in water from piezometers affected by landfill, which indicates that water was enriched with salts from surrounding soils. Reclamation process caused a slow stabilization of this indicator in the water from piezometers affected by landfill. Koc-Jurczyk and Rożak [35] examined leachate from the reclamation site and confirmed variability of conductivity during its exploitation and its stabilization after several years after reclamation. Similar observations were made by Ziyang et al. [36].

The pH of studied water ranged from nearly neutral to slightly alkaline in research period. In this study period, pH of analysed water from all piezometers varied from 6.7 to 7.4 (Fig. 6), which, in accordance with Regulation of the Minister of the Environment from 21 December 2015 on the criteria and method of assessing the status of bodies of groundwater [30], classified them as class I-III groundwater quality characterised by good chemical status. In 2008 and 2009, pH of inflow water (P1) was high, between 7.2 and 7.4 pH, then dropped to 6.8 pH and increased again by 0.4 units. Water from this piezometer was subjected to external influences. Water of piezometers subjected to landfill influences before and during reclamation was characterized by a pH close to neutral. After reclamation, pH of water from these piezometers increased slightly in order to be stabilized by 2014.

After 2014, it rapidly increased by 0.5 units in water from the second piezometer and fell from the third one. After 2016, pH of all piezometers in water was similar. Saarela [23] studied the pH of leachate from a closed landfill in Finland and found its high variability from 5.1 to 7.2. The reaction of examined water was shaped not only by the landfill impact but also by factors outside the landfill. Water pH in piezometers affected by landfill depends to a large extent on its age, changes in landfill conditions and type of waste deposited. At analysed landfill site, apart from municipal waste, sewage sludge and furnace waste were also deposited. After anaerobic acidic phase, which lasts from a few

Table 1
Scopes, average values of physicochemical and groundwater quality class

Piezometer			Physicochemical element					
			TOC (mg/L)	EC (μS/cm)	Zn (mg/L)	Cd (mg/L)	Hg (mg/L)	PAHs (ng L⁻¹)
Befor reclamation of the landfill	P1	Min.-Max.	3.28–18.3	548–606	0.013–0.017	0.0013–0.0020	0.0003–0.0007	1.46–24.2
		Mean	7.26	585	0.025	0.0017	0.00053	9.05
	P2	Min.-Max.	20.3–24.5	1,270–2,343	0.019–0.06	0.0018–0.0025	0.0008–0.0009	7.42–18.5
		Mean	21.65	1,983	0.030	0.0021	0.00087	13.77
	P3	Min.-Max.	6.55–11.9	124–1,270	0.010–0.061	0.0016–0.0026	0.0006–0.0008	6.1–22.7
		Mean	7.98	883	0.028	0.0020	0.0073	13.08
After reclamation of the landfill	P1	Min.-Max.	4.20–31.2	514–1,679	0.003–0.038	0.0012–0.0018	0.00012–0.0008	7.67–225.9
		Mean	8.65	1,000	0.025	0.0015	0.0043	78.9
	P2	Min.-Max.	9.88–83.8	550–2,258	0.003–0.058	0.0017–0.0026	0.00025–0.0009	10.0–93.9
		Mean	22.3	1,275	0.030	0.0021	0.00058	54.06
	P3	Min.-Max.	4.50–27.1	558–1,514	0.01–0.05	0.0010–0.0020	0.00022–0.0010	3.36–274.25
		Mean	13.0	1,025	0.028	0.0017	0.00069	94.97
					Limit value in classes [30]			
I			5	700	0.05	0.001	0.001	100
II			10	2,500	0.5	0.003	0.001	100
III			10	2,500	1	0.005	0.001	100
IV			20	3,000	2	0.01	0.005	500
V			>20	>3,000	>2	>0.01	>0.005	>500

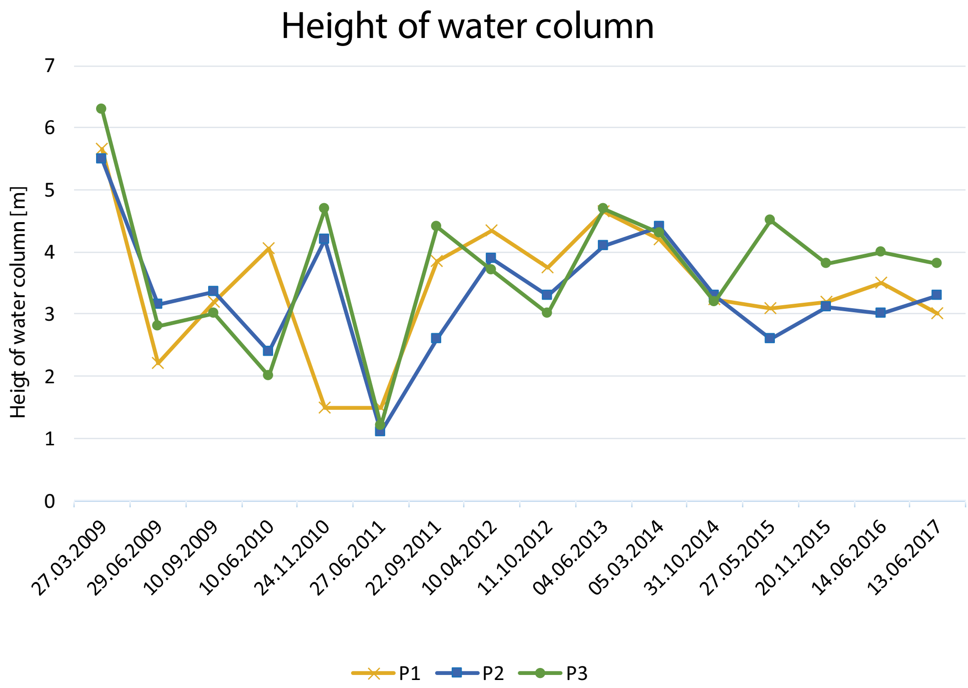

Fig. 4. Height of water column in piezometers.

to several years, concentration of ammonium nitrogen in leachate and then in water increases, which contributes to increase of their pH [37,38].

Concentration of all studied heavy metals (Cd, Zn, Hg) in water taken from piezometers P1–P3 (Figs. 7–9) - in research period was low and classified the water in I and II class of purity (good chemical status). Concentrations of cadmium were lower than those mentioned by El-Salam and Abu-Zuid [16] and higher lower than those mentioned by Przydatek and Kanownik [12], and Przydatek [19]. Low

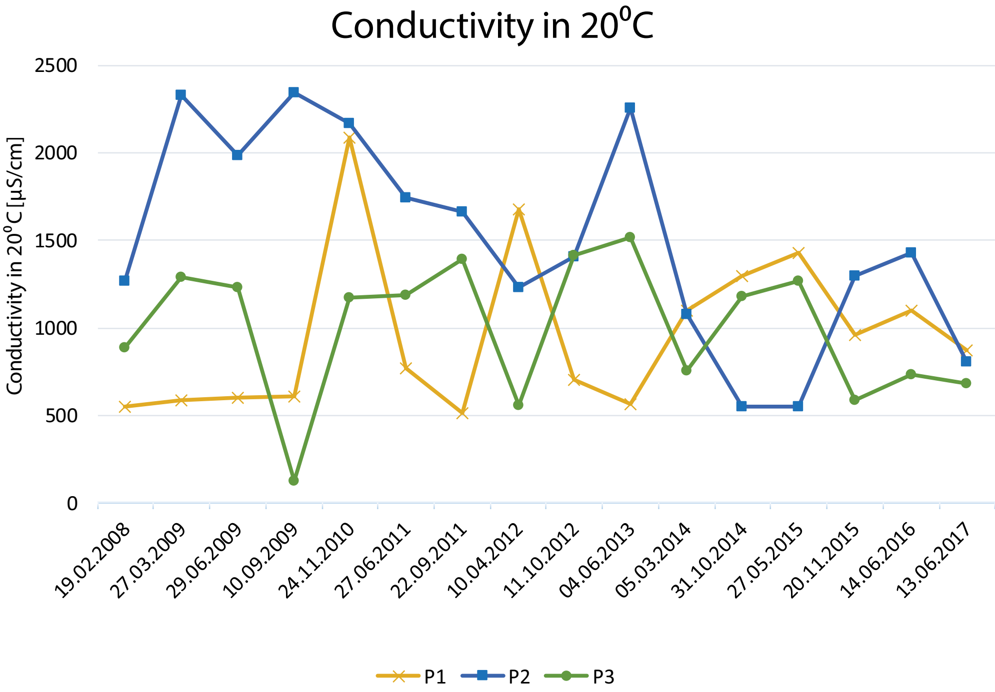

Fig. 5. Changes in conductivity of water in piezometers.

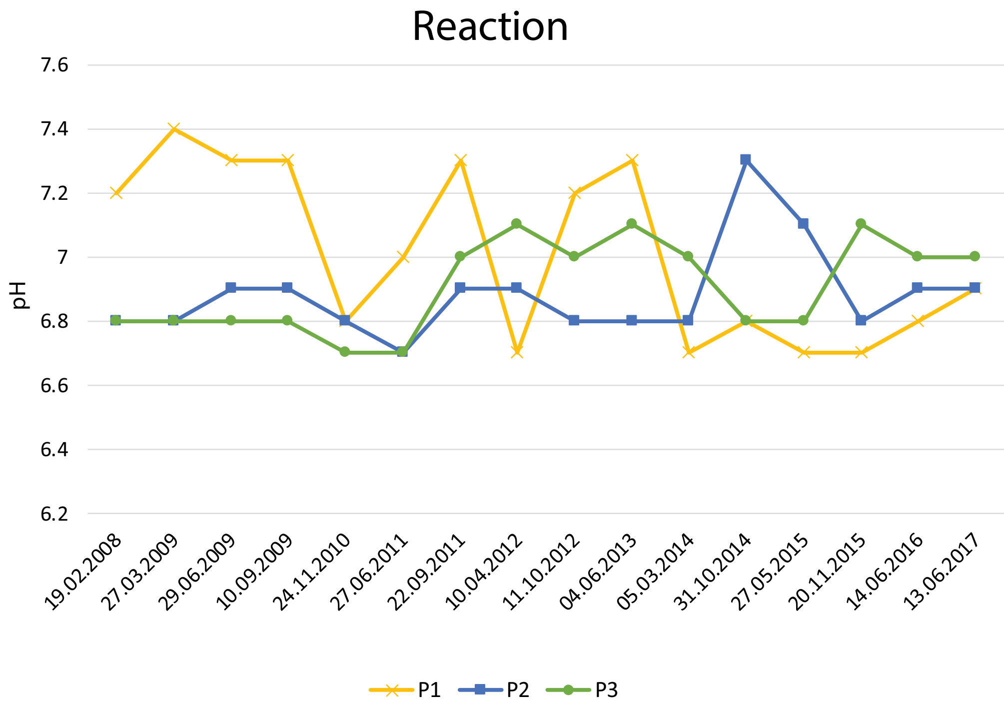

Fig. 6. Changes of water pH in piezometers.

concentration of metals in water may result from their pH, which was within neutral or slightly alkaline limits. Such values favor immobilization of metals and reduce their elutability. Zinc concentration in water from all piezometres underwent greatest changes during studies. A significant influence of landfill site was observed for water of second piezometer. Concentration of this element stabilized in the investigated water 7 years after completion of reclamation.

Mercury concentrations were low, but very variable, and were not subject to any trends and therefore it is difficult to say that they depended on influence of landfill site or surrounding environment (Fig. 9). During reclamation of

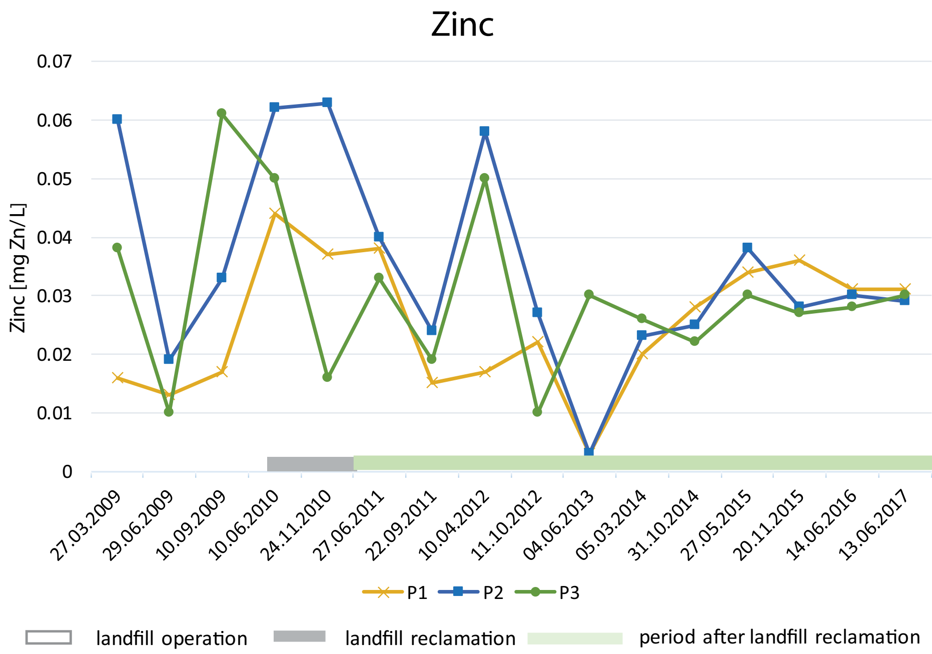

Fig. 7. Changes in zinc concentration in water from piezometers.

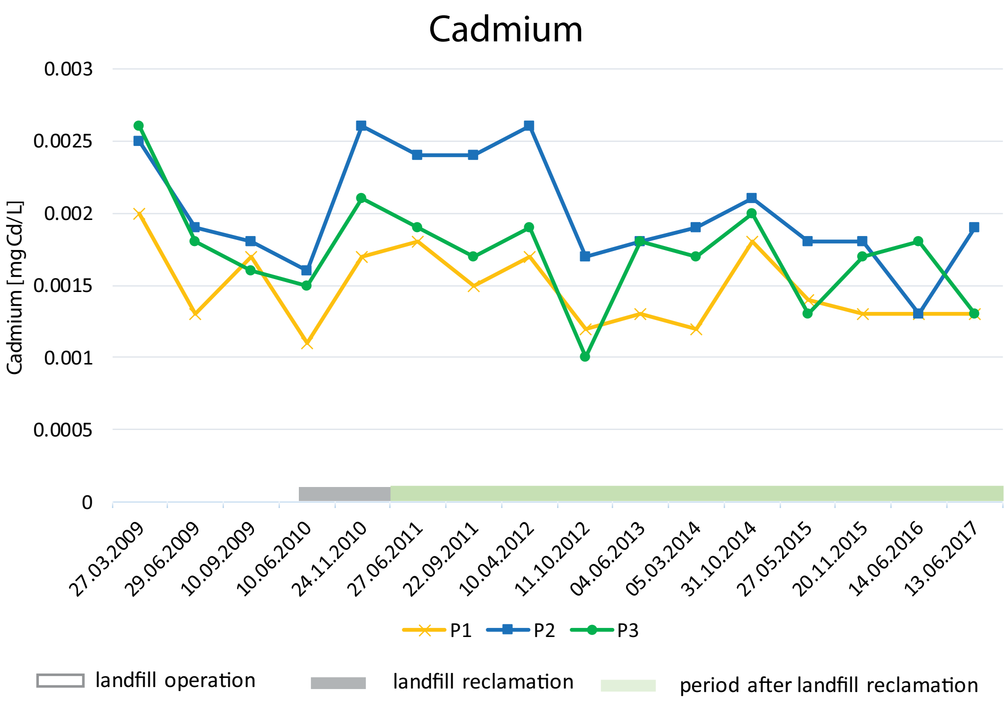

Fig. 8. Changes in cadmium concentration in water from piezometers.

all P1–P3 measurement points, water had an increased concentration (II purity class) of cadmium. Throughout entire research period, higher concentration of this metal occurred in water of two and three piezometers. After reclamation, the increase of cadmium occurred mainly in water of piezometer located closest to landfill (II class of purity). Two years after completion, a decrease in concentration of this metal in water of second piezometer was observed. Sources of cadmium may include sewage sludge as well as furnace waste [39], which was stored at landfills.

Concentration of total carbon in piezometer water indicates a clear influence of landfill site on its quality (Fig. 10). TOC in investigated water from piezometers changed in a wide range. Total organic carbon in water from all piezometers was, respectively, class I to class IV and even class V. This indicates an impact of pollutants contained in leachate generated at landfill sites on groundwater quality, which was proved by clearly increased carbon concentrations in water of piezometers P2 and P3 (Fig. 10). The highest values were found in P2 water after reclamation (max 83.79 mg/L) and the lowest in P1 water (Fig. 10). The water from the piezometer P1 from 2012 onwards was qualified to good quality water. The most visible changes occurred in water of a second piezometer located in direct impact zone of landfill. A value of TOC in the groundwater near the landfill, exceeding 10 mg C/L was reported by Koda et al. [15].

According to Huang et al. [40], the increase in the value of the concentration is a consequence of the natural processes occurring in the groundwater and an evident influence of the anthropogenic factor. Carbon concentration in landfill leachate alone varies widely and according to Christensen et al. [41] the range can be 30–29,000 mg/L. Leachings shape water quality in piezometer, especially in landfills without geomembranes. Prior to reclamation, the lowest concentration of TOC was found in water of this piezometer. After reclamation, it increased rapidly, which was associated with waste compaction and an increase in anaerobic processes in mass, which contributed to a nearly 80-fold increase in carbon concentration. Oleszkiewicz [42] states that the leachate composition is affected not only by the type of waste landfilled but also by its degree of compaction. Further years after reclamation, a decrease in concentration of carbon was observed, as well as in water of all piezometers after 7 years of its stabilization. Kulikowska and Sułek [43] reported the increase in pH and low concentration of TOC in leachate, and thus in water, indicate stabilization of landfill site.

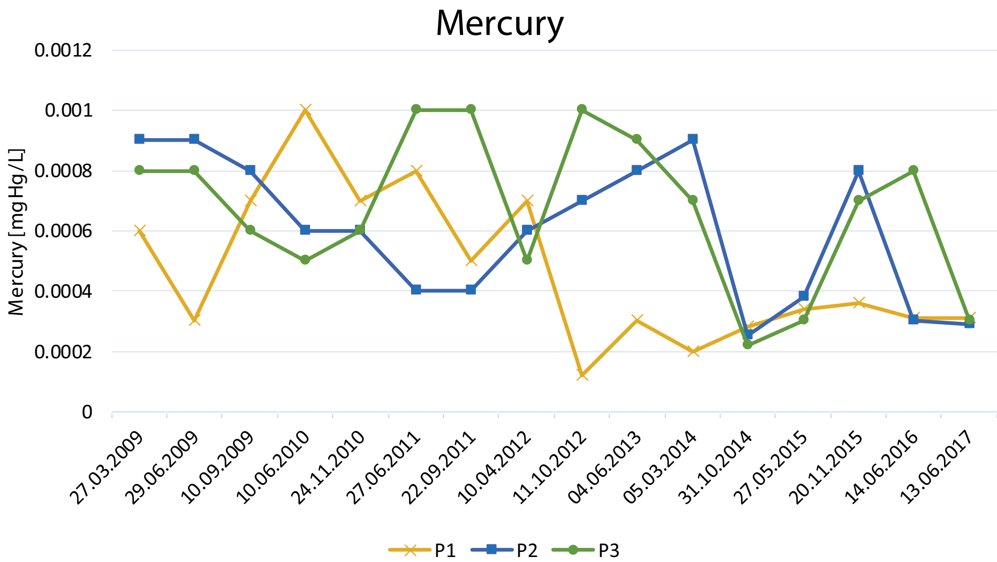

Fig. 9. Changes in mercury concentration in water from piezometers.

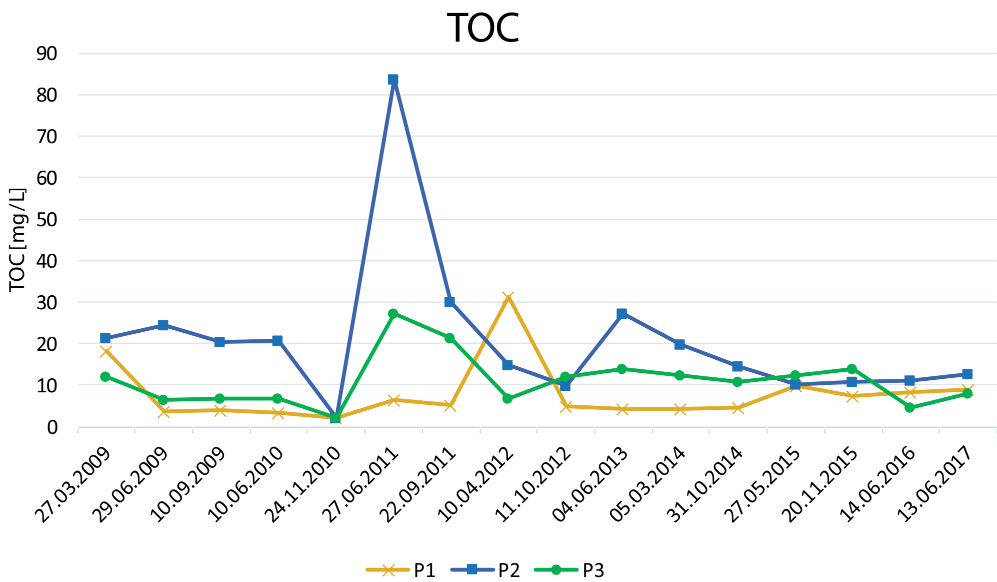

Fig. 10. Changes in TOC from water in piezometers.

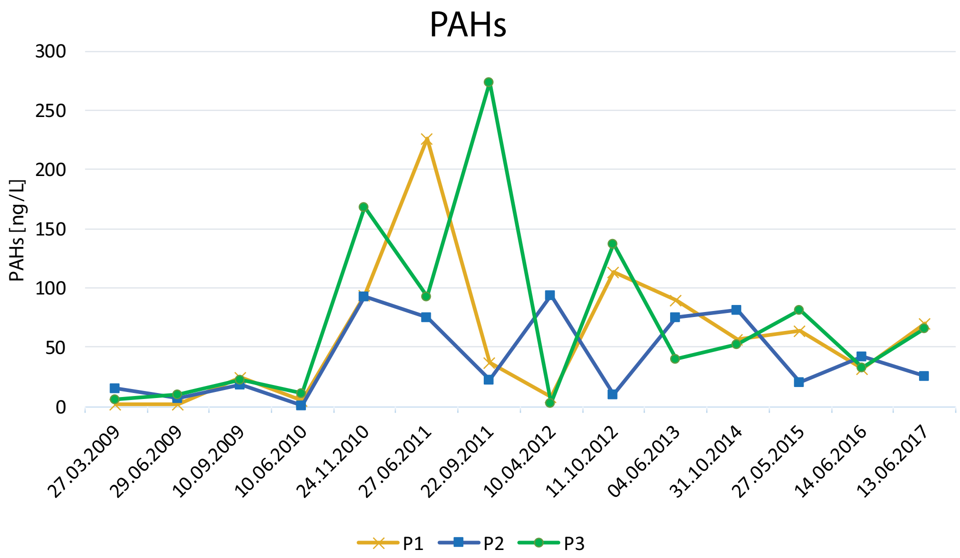

Fig. 11. Changes in PAHs in water from piezometers.

Water from third piezometer was the most stable due to concentration of TOC. Water at influence (piezometer I) was subjected to effects of carbon migration from surrounding soils. Concentration of organic carbon decreased quality of the examined water in all piezometers in whole research period in relation to other determined parameters. According to Srivastava and Ramanathan [44], groundwater flow helps in the dispersion and diffusion of leached pollutants in an aquifer system.

Concentration of PAHs in studied water depended on date of their collection, and to a lesser extent on location of piezometers. Until the beginning of reclamation of all piezometers in water, PAH concentrations were low and did not exceed 100 ng/L (1st class). Reclamation was followed by an increase in PAH concentration to about 275 ng/L (class III). Between 2010 and 2012, the concentrations changed dynamically in water from all piezometers and most of all in the third piezometer (Fig. 11). After 7 years, there was a decrease in concentration of these compounds and it ranged from 25 to 65 ng/L (I class). Changes in PAH concentrations in water near landfills are reported by Wysocka [45], who investigated the impact of unorganized landfills on PAH concentrations in piezometer water. It found an increase in their concentration after closure. Changes in PAHs concentration are the result of activities at landfill site, such as waste mass compacting in 2010 and subsequent biochemical transformations of waste mass. Slow reduction of PAHs concentration in piezometer water is the effect of plants covering the landfill. These compounds migrate faster than other organic compounds, hence their high concentration in water from third piezometer [46].

The sources of PAHs in investigated groundwater are leachate from landfill. In landfill, they are released mainly from paint and varnish packaging and from ashes and slag from individual furnaces [46]. PAHs in water also originates from sewage sludge deposited at old landfills, as reported by Grygorczuk-Petersons [47] based on research results of water affected by an unsealed landfill in Podlaskie Voivodeship.

The pollutants migrated from the landfill site into the groundwater in the immediate vicinity. According to Alslaibi et al. [48], and Baun and Berrin [49], the range of the impact was dependent on local geological and hydrogeological conditions, as well as dilution processes, redox reactions, and ion exchange occurring in the soil and water environment.

Han et al. [50] showed that the most intense groundwater pollution occurred in the area of landfills less than 20 years old. Normally, the concentration of leachate will be the highest during the first 3–8 years when biodegradation is occurring very rapidly [51].

Despite the negative impact of the examined landfill on the quality of groundwater, reclamation should be considered favorable, taking into account the time shift caused by migration of pollutants in the area. Erdogan et al. [52], Kostopoulou et al. [53], Nagendran et al. [54] said that reclamation on a landfill has an important role in removal of contaminants, besides imparting aesthetic value and erosion.

4. Conclusions

- Landfill sites without groundwater protection have a negative impact on groundwater quality, as shown by the results of TOC concentration and electrolytic conductivity and other pollutants in water of second piezometer.
- Reclamation of closed landfill contributed to stabilization of waste transformations in landfill, which is confirmed by obtained results of heavy metals, pH, TOC, PAHs after several years after treatment.
- Decisions of closure of disorganized landfills were right and all of them should be rehabilitated in order to protect aquifers from their negative impact.

References

[1] R. Yang, Z. Xu, J. Chai, A review of characteristics of landfilled municipal solid waste in several countries: physical composition, unit weight, and permeability coefficient, Pol. J. Environ. Stud., 27 (2018) 2425–2435.

[2] E. Koda, P. Osinski, A. Sieczka, D. Wychowaniak, Areal distribution of ammonium contamination of soil-water environment in the vicinity of old municipal landfill site with vertical barrier, Water, 7 (2015) 2656–2672.
[3] D. Laner, M. Crest, H. Scharff, J.W.F. Morris, M.A. Barlaz, A review of approaches for the long-term management of municipal solid waste landfills, *Waste Manage., 32* (2012) 498–512.
[4] K.R. Reddy, H. Hettiarachchi, J. Gangathulasi, J.E. Bogner, Geotechnical properties of municipal solid waste at different phases of biodegradation, Waste Manage., 31 (2011) 2275.
[5] J. Li, Ch. Wang, L. Du, Z. Lv, X. Li, X. Hu, Z. Niu, Y. Zhang, Did municipal solid waste landfill have obvious influence on polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans (PCDD/Fs) in ambient air: a case study in East China, Waste Manage., 62 (2017) 169–176.
[6] P. Ghosh, A. Gupta, I.S. Thakur Combined chemical and toxicological evaluation of leachate from municipal solid waste landfill sites of Delhi, India, Environ. Sci. Pollut. Res., 22 (2015) 9148–9158.
[7] Municipal Waste Statistics, (2018) https://ec.europa.eu/eurostat/statistics-explained/index.php/Main_Page
[8] N. Makarenko, O. Budak, Waste manage-ment in Ukraine: municipal solid waste landfills and their impact on rural areas, Annals Agrarian Sci., 15 (2017) 80–87.
[9] D. Fatta, A. Papadopoulos, M. Loizidou, A study on the landfill leachate and its impact on the groundwater quality of the greater area, Environ. Geochem. Health, 21 (1999) 175–190.
[10] R.B. Brennan, M.G. Healy, L. Morrison, S. Hynes, D. Norton, E. Clifford, Management of landfill leachate: the legacy of European Union Directives, *Waste Manage., 55* (2016) 355–363.
[11] R. Yang, Z. Xu, J. Chai, Y. Qin, Y. Li, Permeability test and slope stability analysis of municipal solid waste in advances in civil engineering Jiangcungou Landfill, Shaanxi, China, J. Air. Waste Manage., 66 (2016) 655–662.
[12] G. Przydatek, W. Kanownik, Impact of small municipal solid waste landfill on groundwater q*uality, Environ. Monit. Assess., 191* (2019) 169.
[13] R. Yang, Z. Xu, J. Chai, Seepage analysis of a multilayer waste slope considering the spatial and temporal domains of permeability, Adv. Civ. Eng., 5 (2019) 1–10.
[14] H. Singh, N.J. Raju, W. Gossel, P. Wycisk, Assessment of pollution potential of leachate from the municipal solid waste disposal site and its impact on groundwater quality, Varanasi environs, India, Arab. J. Geosci., 9 (2016) 131.
[15] E. Koda, A. Miszkowska, A. Sieczka, Levels of organic pollution indicators in groundwater at the old landfill and waste management site, Appl. Sci., 7 (2017) 638.
[16] 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.
[17] G. Pande, A. Sinha, S. Agrawal, Impacts of leachate percolation on ground water quality: a case study of Dhanbad city, Global NEST J., 17 (2015) 162–174.
[18] S. Xie, Y. Ma, P.J. Strong, W.P. Clarke, Fluctuation of dissolved heavy metal concentrations in the leachate from anaerobic digestion of municipal solid waste in commercial scale landfill bioreactors: the effect of pH and associated mechanisms, *J. Hazard. Mater., 299* (2015) 577–583.
[19] G. Przydatek, The quality of the groundwater in area of operation landfill site, Infrastruct. Ecol. Rural Areas, 1 (2013) 59–70.
[20] R. Nagarajan, S. Thirumalaisamy, E. Lakshumanan, Impact of leachate on groundwater pollution due to non-engineered municipal solid waste landfill sites of erode city, Tamil Nadu, India, Iran. J. Environ. Health Sci. Eng., 9 (2012) 35.
[21] A.M. Dhere, C.B. Pawar, P.B. Pardeshi, D.A. Patil, Municipal solid waste disposal in Pune city–an analysis of air and groundwater pollution, *Curr. Sci., 95* (2008) 773–777.
[22] O.N. Ağdağ, D.T. Sponza, Anaerobic/aerobic treatment of municipal landfill leachate in sequential two-stage up-flow anaerobic sludge blanket reactor (UASB)/completely stirred tank reactor (CSTR) systems, *Process Biochem., 40* (2005) 895–902.
[23] J. Saarela, Pilot investigations of surface parts of Tyree closed landdfills and factors affecting them, Environ. Monit. Assess., 84 (2003) 183–192.
[24] Y. Abu-Rukah, O. Al-Kofahi, The assessment of the effect of landfill leachate on ground-water quality-a case study. El-Akader landfill site-North Jordan, J. Arid. Environ., 49 (2001) 615–630.
[25] M.O. Looser, A. Parriaux and M. Bensimon, Landfill underground pollution detection and characterization using inorganic traces, Water Res., 33 (1999) 3609–3616.
[26] J.B. Christensen, D.L. Jensen, C. Gron, T.H. Christensen, Characterization of the dissolved organic carbon in landfill leachate-polluted groundwater, Water Res., 32 (1998) 125–135.
[27] E.D. De Rosa, E. Rubel, M. Tudino, A. Viale, R.J. Lombardo, The leachate composition of an old waste dump connected to groundwater: influence of the reclamation works, Environ. Monit. Assess., 40 (1996) 239–252.
[28] Central Statistical Office, Municipal Infrastructure in 2017, Warsaw, 2018 (in Polish).
[29] Regulation of the Minister of the Environment of 19 July 2016 on the Forms and Manner of Monitoring of Surface Water and Groundwater Bodies (in Polish).
[30] Regulation of the Minister of Environment of December 21, 2015 on the Criteria and Method of Assessing the Status of Homogeneous Bodies of Groundwater (OJ 2016, item 85), http://prawo.sejm.gov.pl/,(Accessed 6.02.2019).
[31] A.J. Witkowski, A.M. Żurek, Impact of old remediated sanitary landfills on groundwater, Contemp. Problems Hydrogeol., 13 (2007) 625–637.
[32] S. Mor, K. Ravindra, R.P. Dahiya, A. Chandra, Leachate characterization and assessment of groundwater pollution near municipal solid waste landfill site, Environ. Monit. Assess., 118 (2006) 435–456.
[33] K.K. Deshmukh, S.P. Aher, Assessment of the impact of municipal solid waste on groundwater quality near the Sangamner City using GIS approach, Water Resour. Manage. 30 (2016) 2425–2443.
[34] J. Wiater, Influence of waste disposal sites on the ground water quality and soil properties, Ecol. Eng., 26 (2011) 133–146 (in Polish).
[35] J. Koc-Jurczyk, J. Rożak, The composition of leachate from recultivated municipal landfill, Ecol. Eng., 27 (2011) 72–80 (in Polish).
[36] L. Ziyang, Z. Youcai, Y. Tao, S. Yu, C. Huili, Z. Nanwen, H. Renhua, Natural attenuation and characterization of contaminants composition in landfill leachate under different disposing ages, Sci. Total Environ., 407 (2009) 3385–3391.
[37] J. Surmacz-Górska, Degradation of organic compounds in landfill leachate, Lublin, 2011 (in Polish).
[38] E. Klimiuk, D. Kulikowska, J. Koc-Jurczyk, Biological Removal of Organics and Nitrogen from Landfill Leachates - A review, In: M. Pawłowska, L. Pawłowski, Eds., Management of Pollutant Emission from Landfills and Sludge, Taylor Francis Group, London, 2007, pp. 187–204.
[39] J. Antonkiewicz, Use incineration ash from binding heavy metals in soils, Environ. Prot. Nat. Res., 41 (2009) 398–405.
[40] G. Huang, J. Sun, Y. Zhang, Z. Chen, F. Liu, Impact of anthropogenic and natural processes on the evolution of groundwater chemistry in a rapidly urbanized coastal area, South China, Sci. Total Environ., 463 (2013) 209–221.
[41] T.H. Christensen, P. Kjeldsen, P.L. Bjerg, D.L. Jensen, J.B. Christensen, A. Baun, H.J. Albrechtsen, G. Heron, Biogeochemistry of landfill leachate plumes, Appl. Geochem., 16 (2001) 659–718.
[42] J. Oleszkiewicz, Operation of Landfill Sites, Decision-Maker Guide, LEM Project, Kraków, 1999.
[43] D. Kulikowska, P. Sułek, Removal of organic leachate pollutants by activated carbon adsorption, Part I:Process efficiency, Tech. Trans., 105 (2008) 89–98 (in Polish).
[44] S.K. Srivastava, A.L Ramanathan, Geochemical assessment of groundwater quality in vicinity of Bhalswa landfill, Delhi, India, using graphical and multivariate statistical methods, Environ. Geol., 53 (2008) 1509–1528.

[45] M.E. Wysocka, Influence of location of landfills on groundwater quality, Annual Set Environ. Prot., 17 (2015) 1074–1093.
[46] C. Rosik-Dulewska, Migration of PAHs from unsealed landfill to groundwaters, Annual Set Environ. Prot., 9 (2007) 335–344.
[47] E.H. Grygorczuk-Petersons, Effect of not-sealed municipal waste landfill on the underground water quality, Ecol. Eng., 48 (2016) 61–68 (in Polish).
[48] T.M. Alslaibi, Y.K. Mogheir, S. Afifi, Assessment of groundwater quality due to municipal solid waste landfill leachate, *J. Environ.* Sci. Technol., *4* (2011) 419–436.
[49] S. Baun, T. Berrin, Parametric fate and transport profiling for selective ground water monitoring at closed landfills: a case study, *Waste Manage., 38* (2015) 263–270.
[50] Z. Han, H. Ma, G. Shi, L. He, L. Wei, O. Shi, A review of groundwater contamination near municipal solid waste landfill sites in China, Sci. Total Environ., *569* (2016) 1255–1264.
[51] P. Agamuthu, Solid Waste: Principle and Management, University of Malaya Press, 2001, pp. 9–27.
[52] R. Erdogan, S. Mansuroglu, I. Uz, E. Oktay, A holistic management of waste landfills in Mediterranean city: the case study of Antalya Kizilli, J. Food Agric. Environ., 9 (2011) 636–640.
[53] P. Kostopoulou, A. Karagiannidis, P. Rakimbei, K. Tsiouvaras, Simulating the water balance in an old non-engineered landfill for optimizing plant cover establishment in an arid environment, Desalination, 250 (2010) 373–377.
[54] R. Nagendran, A. Selvam, K. Joseph, C. Chiemchaisri, Phytoremediation and rehabilitation of municipal solid waste landfills and dumpsites: a brief review, Waste Manage., 26 (2006) 1357–1369.