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Assessment of Shallow Groundwater Purification Processes after the Construction of a Municipal Sewerage Network

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Abstract: In the present study, the impact of the construction of the sewerage network (2014) on groundwater quality was studied on the example of a middle-sized settlement in the Great Hungarian Plain and changes in water quality were assessed using GIS and multivariate statistical analysis. As a result of the pollution that has been going on for decades, the groundwater supply of the municipality has become heavily polluted. In the period before sewerage (2011–2013), clear spatial differences were shown in the degree of contamination in the area of the settlement: the degree of contamination increased in the direction of local groundwater flow. Based on our results, the construction of the sewerage network resulted in marked changes in the quality of groundwater wells. Our studies showed a clear decrease in the concentrations of inorganic pollutants (NH_4^+, NO_2^-, NO_3^-, PO_4^{3-}). In the fifth year of the post-sewerage period (2019), the clear spatial difference between wells was eliminated. We have observed a strong descent (1.5 to 2 m) in groundwater levels in the post-sewerage period, the reason for that is clearly due to that wastewater outflow was eliminated. On this basis, it can be declared that the purification processes in the area have started, but at the same time, our results show that pollutants entering environment undergo a number of transformations and remain in the system for a long time.

Keywords: groundwater quality; environmental pollution; cleaning process; sewerage network; GIS; multivariate statistical analysis

1. Introduction

In parallel with the construction of the drinking water network, the sewerage network and wastewater treatment plants were not established in many countries around the world, therefore municipal wastewater from households is stored in various installations without insulation in the yard of households (cesspits, soakaways, septic tanks) became a widespread practice [1–5]. Wastewater flowing out of these point sources resulted in a sharp degradation of the groundwater supply in the municipalities [6–9].

However, the deterioration of groundwater status occurred not only in Hungary, it is a global problem, which has thus become the focal point of environmental research [10]. Numerous studies show that, in addition to industrial and agricultural activity, one of the most significant sources of pollution is municipal wastewater, the treatment and cleaning of which is often unresolved in developing and developed countries alike [11–13]. In the more developed urban environment, wastewater exfiltration from the sewer system poses a threat to the subsurface water resources [14].

Nitrogen loading to groundwater in rural and urban areas, mainly from domestic sewage leaching, has become an increasingly important environmental problem in recent years [15,16]. Nitrogen can enter the groundwater system in various organic and inorganic forms and, due to its solubility, it leaches easily from the point of infiltration into deeper aquifers, causing contamination [17,18]. Lack of wastewater treatment has led to widespread nitrogen contamination of groundwater systems not only in less developed

regions of the world, but also in developed regions [19]. In the municipal environment, organic phosphorus from uninsulated sewage tanks and pit latrines leaks into the soil and groundwater in large quantities, causing very serious pollution [20,21]. Human metabolism processes contribute ~2 g/day, and detergents used contribute a further ~2 g/day of phosphorus load [22].

Preserving and improving groundwater quality in the urban environment is a complex task due to the many sources of pollution [23–25]. Although groundwater monitoring is of great interest to water decision-makers for establishing strategies to protect water resources, due to the high costs there is not enough data available at a sufficient level of detail, especially for long-term monitoring [26,27]. For this reason, only few studies have been conducted to assess the impact of the sewerage network on groundwater [28,29]. In addition, knowledge of the baseline condition is necessary to assess the changes, but most municipalities do not have such a settlement-wide data making comparative analysis impossible. One of the aims of our study is to fill this scientific gap.

Assessment of groundwater quality is crucial for environmental management and land development [30,31]. A complex analysis not only presents the water chemistry parameters, but also combines the results of water quality index (WQI), geographic information system (GIS) and statistical analysis, with a strong emphasis on the mapping and interpretation of environmental data, which is of great importance in environmental science and environmental policy as well [32–35].

Multivariate statistical methods are a valuable tool for assessing spatial and temporal differences, interpreting complex water quality datasets, identifying the source of pollutants (natural or anthropogenic), and assisting in the design of monitoring networks, effective water management and possible practical solutions for pollution remediation [36]. Comprehensive groundwater quality studies using multivariate statistical analyses have been carried out in a number of studies. Demirel and Güler (2006) identified anthropogenic factors affecting groundwater body chemistry in the coastal zone of the Mersin-Erdemli basin using hierarchical cluster analysis (HCA), principal component analysis (PCA) and geochemical modelling techniques [37]. Results showed that seasonal variations are caused by fertilizer application in the early summer period. Cloutier and colleagues (2008) used PCA and HCA to make the large dataset available transparent and to identify processes that determine the geochemical properties of groundwater [38]. Yidana and colleagues (2010) used hierarchical cluster analysis in addition to traditional graphical representation to isolate groundwater bodies with different salinity in southwestern Ghana [39]. Cluster analysis proved to be an effective statistical tool, allowing the grouping of elements with similar properties into correlation pairs. Lin and colleagues (2012) investigated the factors influencing the temporal variability and chemical composition of near-surface groundwater in Manukan Island, Malaysia. By using analysis of variance, PCA, HCA and other geostatistical techniques, they found that near-surface groundwater is unsuitable for use as drinking water [40]. Farsang and colleagues (2017) investigated 12 inorganic contaminants in the groundwater of the city of Szeged. Discriminant analysis was used to evaluate spatial variations in groundwater quality, and R/S analysis was used to investigate water level fluctuations in the groundwater system and the Tisza River [41].

Hungary ratified the 2000/60/EC Water Framework Directive and Directive 91/271/EEC on urban wastewater treatment by joining the European Union, in accordance with which the establishment of the sewerage network of municipalities with pollutant loads above 2000 inhabitant equivalents (IE) is currently ongoing. Very significant progress has been made in the collection and purification of municipal wastewater in Hungary in the last decade. The difference in the proportion of homes connected to the water and sewerage decreased from 41.1% in 2000 to 12.3% in 2019 [42].

As a result of the investments realised, the amount of municipal wastewater entering the groundwater system has been significantly reduced, the effect of which extends to the entire urban environment [43,44]. Since the construction of a sewerage network is a very heavy financial burden for municipalities, the question may arise as to the benefits of

these investments. Does it really solve the problems and, if so, how quickly do the desired positive changes occur? In order to accurately determine the extent, intensity and duration of the expected purification processes, long-term groundwater monitoring is essential, during which a pre-investment baseline assessment is required.

In the present study, the impact of the construction of the sewerage network on groundwater quality was studied on the example of Báránd a middle-sized settlement in the Great Hungarian Plain and changes in water quality were assessed. Regular water quality tests have been carried out in the village since 2011, which have continued after the construction of the sewerage network municipality in 2014 and are still ongoing. In addition, groundwater monitoring, which has been going on for a decade, provides an opportunity to carry out a complex, comparative research that can not only bring important value added for both domestic and international literature, but also has economic importance.

2. Materials and Methods

2.1. Site Location and Characteristics

The investigated settlement Báránd is located in the eastern part of the Great Hungarian Plain, in the Nagy-Sárrét region on the western part of the alluvial deposit of the Sebes-Körös River (Figure 1), and has a population of 2567 [45].
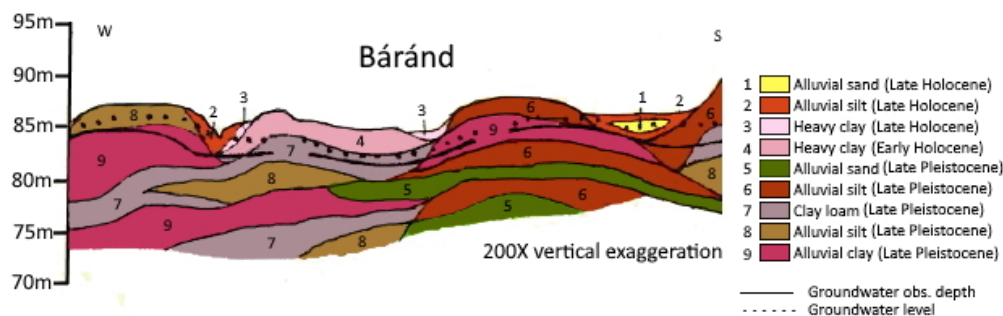

Figure 1. Geological cross section of the study area (Modified based on Rónai 1980 [46]).

The altitude of the Nagy-Sárrét is typically 85–89 m, and the region is classified as a flat plain (relative relief 0–3 m/km^2). The groundwater level can be found close to the surface, at a depth of 1–2 m; consequently, all the soil types have been formed under the influence of water [47]. In the study area, the most frequent soil types are Solonetz, Vertisol, Kastenozem, and Chernozem, and in the built-up area—as a result of anthropogenic effects—Technosol [48].

The annual water production of the settlement has been between 90,000–120,000 m^3 in the last decade, the amount of water supplied to households was 70,000–90,000 m^3. According to our calculations, 40–60% of the municipal wastewater stored in uninsulated sewage tanks could have leaked into the soil, which meant 30,000–55,000 m^3 of leaking wastewater per year at the municipal level in the pre-sewerage period. Since the municipality has sewage discharges of more than 2000 inhabitants equivalent, a sewerage system was constructed in 2014 in accordance with the relevant legislation.

2.2. Field Sampling and Laboratory Analysis

In the present study an analysis of the water samples collected before (2011, 2012, 2013) and after (2017, 2018, 2019) the construction of the sewerage network (2014) was performed. The number of wells has been extended to 40 from 2013 during summer sampling in order to determine the contamination as accurately as possible in space (Figure 2). The base depth of the monitoring wells ranged from 6 to 10 m.
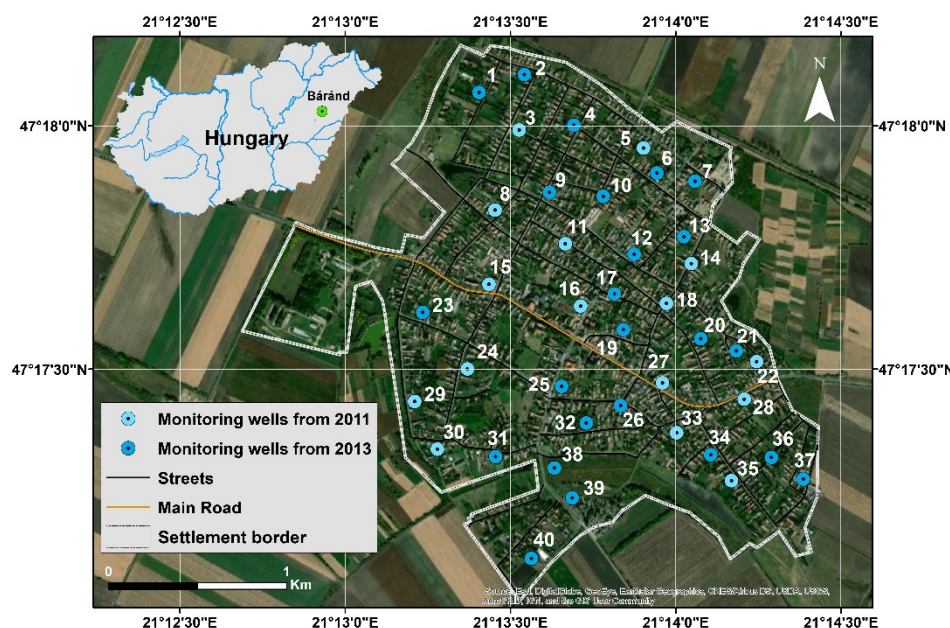
Figure 2. Location of the monitoring wells in the investigated settlement in Hungary.

The most characteristic pollutants of municipal wastewater were included in the analysis. The ammonium ion (NH_4^+), nitrite ion (NO_2^-), nitrate ion (NO_3^-), orthophosphate ion (PO_4^{3-}) and organic matter (COD) content of the water samples taken were determined on the basis of the Hungarian Standards (HS ISO 7150-1:1992, HS 448-18:2009, HS 1484-13:2009, HS 12750-17:1974) in the Complex Laboratory of the Institute of Earth Sciences, University of Debrecen. Chloride ion (Cl^-) sulphate-ion (SO_4^{2-}), p- and m-alkalinity (HCO_3^-/CO_3^-) have been defined in the post-sewerage period on the basis of the Hungarian standards (HS 1484-15:2009, HS 260–7:1987, HS 448-11:1986). The sodium ion (Na^+) content of the water samples was measured regularly, magnesium (Mg^{2+}) calcium (Ca^{2+}) and potassium (K^+) ion was measured in the post-sewerage period using a PERKIN ELMER 300 atomic absorption device. Data quality for major components was estimated by charge balance [49]. Deviation between the sum of concentrations of major cations and the sum of concentrations of major anions, in equivalent units, varies between −5% and +5%. The pH and EC values were determined using a WTW 315i and CONSORT C3010 measuring instruments. The results were assessed on the basis of the relevant limits of the Joint Regulation KvVM-EüM-FVM No 6/2009 (IV. 14). The cation and anion composition of the water samples from the post-sewerage period was evaluated using Piper and Durov diagrams generated with Grapher 17 software.

The results of field measurements with a Trimble S9 GPS device to create digital elevation models for the surroundings of the municipality to determine the absolute height of groundwater levels (Figure 3).

2.3. Statistical Analysis and GIS

Geographical information system (GIS) applications are widely used to determine the spatial distribution of groundwater quality parameters by integrating spatial data with other geographical information [50–53]. The geovisualization of the results was performed with ArcGIS 10.4.1 software, the spatial distribution of the contaminants with Kriging interpolation using the software Surfer 19. In environmental sciences, interpolation can be considered as primary form of geostatistical analysis, which is often used to predict values in unobserved locations based on the values in observed locations. Ordinary kriging, a type of interpolation has become a widely used interpolation method for assessment of

spatial variation of groundwater quality [54]. A semivariogram can be used to quantify spatial dependence:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [Z(Xi + h) - Z(Xi)] \tag{1}$$

where $\gamma(h)$ illustrates the semivariogram as a function of the lag distance or separation vector h between two points. $N(h)$ represents the number of observation pairs divided by distance h, and $Z(Xi)$ represents the random variable at position Xi [55].
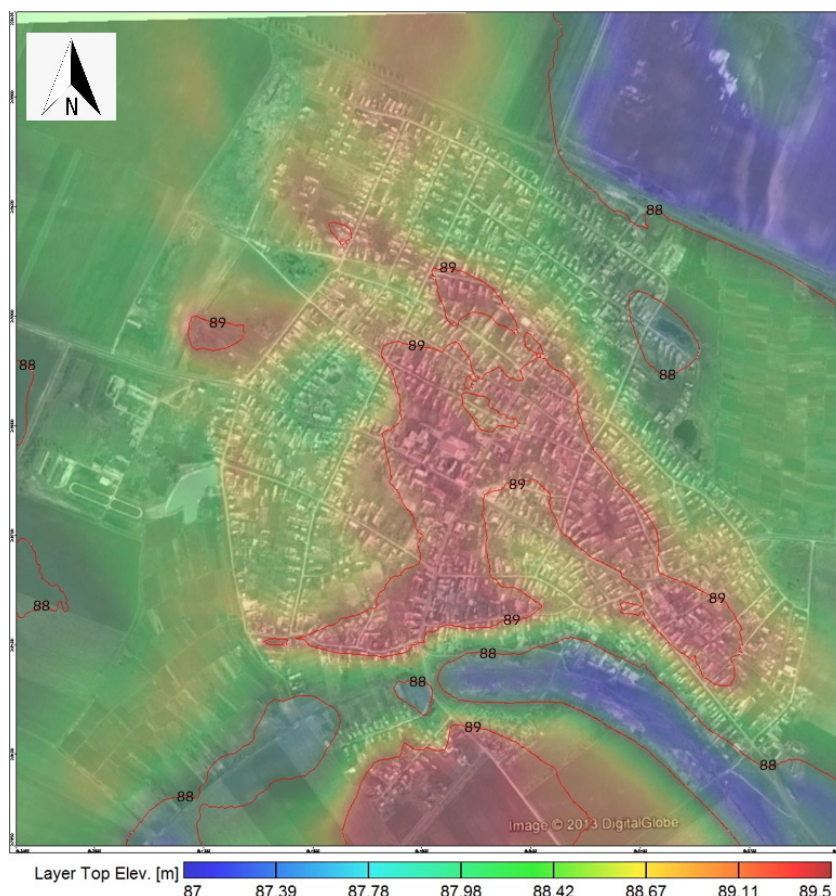

Figure 3. Digital relief map of the investigated settlement.

Ordinary Kriging is a linear appropriate interpolation technique with which interpolated (predictive) maps can be created. The spatial distribution of different parameters can be determined according to the equation below [56]:

$$\hat{Z}(X0) = \sum_{i=1}^{n} n \, \lambda i \, Z(Xi) \tag{2}$$

where $\hat{Z}(X0)$ is the predictable value at $X0$ points, while n is the number of the sampled point $Z(Xi)$ is the recognized value at sampled Xi points, and λ is the weight assigned to the sampled point.

The software IBM SPSS 22 was used to perform statistical processing of the data series and to represent the results. In addition to calculating the base statistical values, the normality of the series was examined as well. In addition to the calculation of the basic statistical values, mean, lower, upper quartile, mode, median and standard deviation, the results were plotted on different scatterplot and boxplot diagrams for better illustration. This is because boxplot diagrams are suitable for plotting interquartile range, median, highest and lowest values. In order to determine the strength of the relationship between the variables,

Spearman's rank correlation test was performed. Principal component analyses were been performed to organize the variables into components and to better explore geochemical background processes. In addition, the results of the principal component analysis were used to investigate the spatial separability of monitoring wells. The number of principal components was determined based on the Kaiser criterion [57]. The suitability of the data for analysis was reported by Kaiser–Meyer–Olkin (KMO) and Bartlett probes. If the KMO value is higher than 0.5 and in the case of the Bartlett test $p < 0.05$, then the database is suitable for the study. A hierarchic (Ward method) and two-stage cluster analysis were carried out to separate groundwater wells with similar characteristics from each other. Discriminant analyses were carried out to determine whether conditions before and after the construction of the sewerage network can be separated or not. The test was performed using the Wilks 'Lambda method. Fisher's (1936) two-group discriminant analysis is suitable for distinguishing between different groups [58].

3. Results and Discussion

3.1. Water Quality before the Construction of the Sewerage Network

According to our calculations, 40–50% of the municipal wastewater leaked from uninsulated septic tanks into the groundwater, which meant an annual raw sewage load of 50,000–60,000 m^3. As a result of the pollution that has been going on for decades, the groundwater supply of the municipality has become heavily polluted. Anthropogenic effects were detected for all groundwater wells during sampling before the construction of the sewerage network. The statistical summary of the parameters investigated between 2011–2019 are shown in Table 1. To obtain a more detailed picture of the changes in the degree of contamination, we plotted the EC, NH$_4^+$, NO$_3^-$, PO$_4^{3-}$ values of each well on a boxplot diagram (Figure 4).
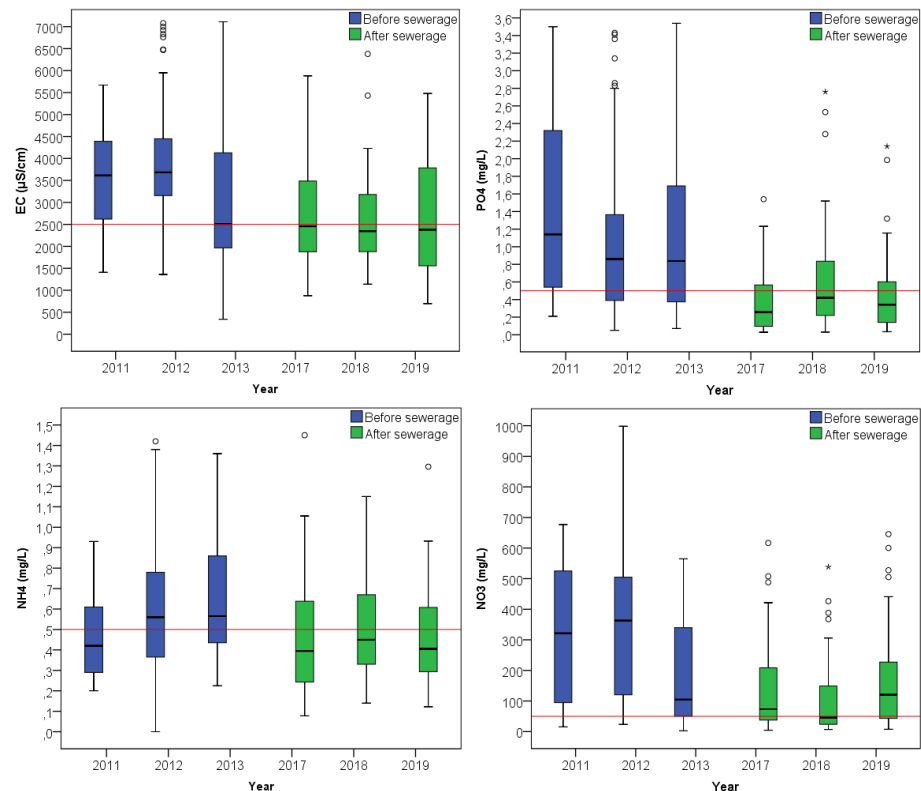

Figure 4. Box plots of EC, PO$_4^{3-}$, NH$_4^+$ and NO$_3^-$ before and after the establishment of the sewerage network. The bottom and top of each box represent the lower and upper quartiles, and the line inside each box represents the median. The red line indicates the contamination limit value.

Table 1. Descriptive statistics of the examined hydrochemical parameters. (+) indicates positive changes compared to the 2013 reference year, (−) indicates negative changes based on the contamination limit for the given parameter.

Parameter	Mean						St. Deviation					
	2011	2012	2013	2017	2018	2019	2011	2012	2013	2017	2018	2019
pH	7.83	7.76	8.25	7.51	7.99	7.26	0.257	0.275	0.457	0.301	0.417	0.351
EC (μS/cm)	3524	3877	3032	2845 (+)	2637 (+)	2726 (+)	1194	1280	1701	1785	1136	1567
NH_4^+ (mg/L)	0.539	0.609	0.691	0.529 (+)	0.653 (+)	0.512 (+)	0.288	0.337	0.373	0.545	0.773	0.505
NO_2^- (mg/L)	0.269	0.265	0.306	0.204 (+)	0.211 (+)	0.267 (+)	0.24	0.36	0.33	0.40	0.29	0.42
NO_3^- (mg/L)	327.1	361.9	187.8	142.7 (+)	109.8 (+)	163.5 (+)	217.4	266.5	164.4	159.3	129.5	170.7
PO_4^{3-} (mg/L)	1.36	1.12	1.22	0.39 (+)	0.65 (+)	0.49 (+)	0.93	0.78	1.09	0.39	0.65	0.49
COD (mg/L)	5.14	5.49	6.85	7.65 (−)	7.16 (−)	7.62 (−)	1.94	1.85	3.94	3.23	5.78	3.76
Na (mg/L)	424.6	361.8	237.9	377.9 (−)	352.4 (−)	379.3 (−)	285.9	213.5	141.7	389.6	298.6	330.1

Parameter	Minimum						Maximum					
	2011	2012	2013	2017	2018	2019	2011	2012	2013	2017	2018	2019
pH	7.24	7.01	7.23	7.02	7.21	6.81	8.39	8.44	9.42	8.3	8.87	8.25
EC (μS/cm)	1410	1360	340	876 (−)	1140 (−)	695 (−)	5670	7080	7670	9290 (−)	6380 (+)	8910 (−)
NH_4^+ (mg/L)	0.2	0.211	0.225	0.078 (+)	0.140 (+)	0.122 (+)	1.81	2.13	1.89	3.42 (−)	3.97 (−)	3.36 (−)
NO_2^- (mg/L)	0.015	0.013	0.017	0.006 (+)	0.000 (+)	0.001 (+)	1.36	1.61	1.28	1.86 (−)	1.37 (−)	1.94 (−)
NO_3^- (mg/L)	15.38	23.35	8.36	4.46 (+)	6.95 (+)	7.61 (+)	676.8	1140	564.8	616.6 (−)	538.3 (−)	645.5 (−)
PO_4^{3-} (mg/L)	0.21	0.05	0.07	0.03 (+)	0.03 (+)	0.04 (+)	3.52	3.43	4.07	1.54 (+)	2.76 (+)	2.14 (+)
COD (mg/L)	1.84	1.68	2.40	2.90 (−)	1.1 (+)	1.66 (+)	9.36	10.29	18.20	17.68 (+)	36.8 (−)	16.65 (+)
Na (mg/L)	14.35	14.1	8.90	75.8 (−)	97.4 (−)	52.1 (−)	1120	1485	653.2	2254 (−)	1828 (−)	2019 (−)

Parameter	Lower Quartile						Upper Quartile					
	2011	2012	2013	2017	2018	2019	2011	2012	2013	2017	2018	2019
pH	7.65	7.56	7.92	7.29	7.68	6.96	8.01	7.91	8.56	7.74	8.29	7.41
EC (μS/cm)	2585	3152	1950	1871 (+)	1874 (+)	1511 (+)	4395	4515	4310	3507 (+)	3190 (+)	3792 (+)
NH_4^+ (mg/L)	0.29	0.36	0.43	0.243 (+)	0.325 (+)	0.289 (+)	0.61	0.785	0.865	0.643 (+)	0.680 (+)	0.617 (+)
NO_2^- (mg/L)	0.03	0.04	0.052	0.021 (+)	0.009 (+)	0.024 (+)	0.32	0.297	0.416	0.167 (+)	0.300 (+)	0.308 (+)
NO_3^- (mg/L)	94.41	119.3	50.16	37.65 (+)	23.72 (+)	43.09 (+)	527.6	507.1	341.7	221.1 (+)	153.1 (+)	244.8 (+)
PO_4^{3-} (mg/L)	0.54	0.385	0.374	0.095 (+)	0.215 (+)	0.137 (+)	2.35	1.67	1.75	0.577 (+)	0.863 (+)	0.604 (+)
COD (mg/L)	3.58	3.54	4.02	5.24 (−)	3.59 (+)	4.57 (−)	6.57	6.29	8.05	9.89 (−)	8.48 (−)	10.1 (−)
Na (mg/L)	209.1	215.1	132.2	154.1 (−)	182.4 (−)	185.4 (−)	645.2	395.6	312.8	422.5 (−)	447.8 (−)	477.9 (−)

The presence of NH_4^+ indicates the decomposition of the organic matter content of the waters, which indirectly refers to wastewater load and fresh contamination. In the period before sewerage the mean of NH_4^+ concentrations (0.61 mg/L) and the upper quartile (0.78 mg/L) exceeded the contamination limit (0.5 mg/L). In the vast majority of the wells, NO_3^- concentrations several times above the limit value (50 mg/L) were measured. The lower quartile value (94.6 mg/L) was more than double that of the limit while the upper quartile (493.2 mg/L) was nearly 10 times higher than the limit. PO_4^{3-} concentrations also show the strong contamination of groundwater. In 72.2% of the measurements before the construction of the sewerage network concentrations above the relevant contamination limit of 0.5 mg/L were found. The total ion content in the studied groundwater wells exceeded or significantly exceeded the limit of 2500 μS/cm in the majority of wells. The mean value of the data series was 3637 μS/cm and the upper quartile value was 4395 μS/cm. The mean Na^+ concentration was nearly twice the limit value (200 mg/L). Significant differences in spatial concentration trends suggest that, in addition to soil factors, concentrations were largely influenced by the typically high sodium content of communal wastewaters. Concentrations exceeding the limit value for COD contamination (4.5 mg/L) were measured in 71% of the wells.

These results are consistent with the results of other studies carried out in a village environment which showed strong pollution of the groundwater resources of the settlements [59–62].

3.2. Water Quality Changes after the Construction of the Sewerage Network

Examining the data series for the period before (2011–2013) and after (2017–2019) the construction of the sewerage network significant positive changes in groundwater quality were shown using various geoinformatic and statistical methods. Our studies showed a clear decrease in the concentrations of inorganic pollutants (NH_4^+, NO_2^-, NO_3^-, PO_4^{3-}) (Table 1) (Figure 4).

Although the average values of NH_4^+ concentrations continued to be above the limit value in each studied year, a steady decreasing trend can be observed. While concentration over 1 mg/L was measured in 17.5% of the wells in large sample 2013, only such concentrations were measured in only 5% of the wells in 2019. NO_2^- concentrations also decreased to a detectable extent, from an average value of 0.416 mg/L in 2013 to 0.308 mg/L in 2019. The decrease in the concentration of NO_3^- also indicates the loss of a significant part of the contaminant supply, as the average value decreased from 187.8 mg/L (2013) to 109.8 mg/L in 2018. The lower quartile value was below the limit value during each large sampling of the period after the construction of the sewerage network. The average PO_4^{3-} concentration in 2019 decreased by 60% compared to the reference year 2013, reducing it to the level of the contamination limit (0.5 mg/L). In 32.5% of the wells concentrations above the limit were measured in 2019 compared to 65% in 2013. In the fifth year following construction of the sewerage network, the proportion of the areas of the municipality where the EC value was below the limit value (2500 µS/cm) increased compared to the reference year.

3.3. Results of the Discriminant Analysis

A discriminant analysis was performed to determine whether the water chemistry parameters indicate whether the sample was taken before or after sewerage. In order to determine this, the data series were categorised into pre-sewerage (2011–2013) and post-sewerage (2017–2019) and discriminant analysis was performed (Table 2). The Wilks–Lambda test showed a significant result ($p < 0.001$). A total of 85.6% of the cross-validated values were successfully classified into their original category. Applying discriminant analysis, it can be determined with 82.4% probability whether the sample is from the period prior to or after the establishment of the sewerage network (Table 2). On this basis, the construction of the sewerage network resulted in marked changes in the quality of groundwater wells.

Table 2. Classification results of the discriminant analysis.

Classification Results [a,c]			Predicted Group Membership		Total
		Sewerage Status	1	2	
Original	Count	Before sewerage	175	23	198
		After sewerage	29	91	120
	%	Before sewerage	88.4	11.6	100.0
		After sewerage	24.2	75.8	100.0
Cross-validated [b]	Count	Before sewerage	173	25	198
		After sewerage	31	89	120
	%	Before sewerage	87.4	12.6	100.0
		After sewerage	25.8	74.2	100.0

[a] In total, 83.6% of original grouped cases correctly classified. [b] Cross validation was done only for those cases in the analysis. In cross validation, each case was classified by the functions derived from all cases other than that case. [c] In total, 82.4% of cross-validated grouped cases were correctly classified.

3.4. Spatio-Temporal Changes in Groundwater Quality

In order to determine the spatial distribution of the pollutants, interpolated maps were produced based on the concentrations at the sampling points (Figure 5).

The concentration of PO_4^{3-} in the year before the sewerage showed the highest values in the central areas of the municipality (2 mg/L isoline) and a decreasing trend towards the north and south of the municipality (Figure 5). This pattern showed a high similarity with the spatial distribution observed in 2011–2012. However, concentrations were above 0.5 mg/L in most parts of the municipality. Further, five years after the construction of the sewerage, in 2019, significant changes were observed. The central area of the agglomeration remained the most polluted, but there were no areas with concentrations above 2 or 3 mg/L, with only a small area above 1.5 mg/L. The other significant change was that the area with concentrations below the limit value had increased significantly (Figure 5). While in 2013,

concentrations below the limit were only measured in the north and south peripheral areas of the municipality, in 2019 concentrations below the limit were found in all but the central areas of the municipality and a few small areas.
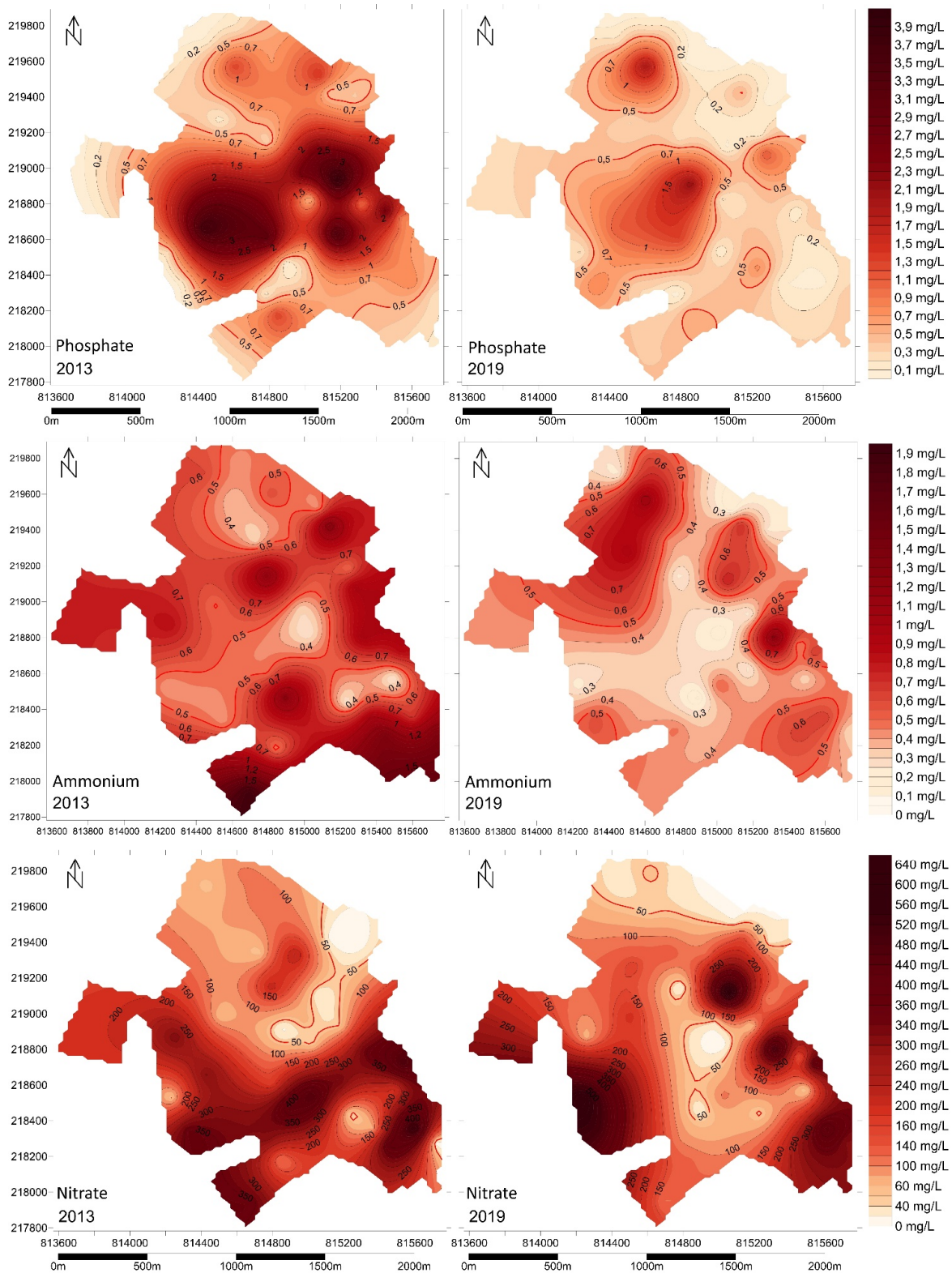

Figure 5. Spatial distribution of PO_4^{3-}, NH_4^+ and NO_3^- concentration before (2013) and five years after the establishment of the sewerage network (2019).

In 2013, the highest NH_4^+ concentrations were measured in the South and East areas of the municipality (>0.7 mg/L); concentrations below 0.5 mg/L were detected in the North and Central areas (Figure 5). By 2019, the proportion of areas of the municipality where the map shows concentrations below 0.5 mg/L had increased significantly. However, concentrations above 0.2 mg/L still indicate high levels of pollution.

For NO_3^-, a very characteristic spatial distribution was observed in the reference year prior to sewerage (Figure 5). Similar to the data for 2011 and 2012, the southern areas of the municipality showed very high levels of pollution. Almost the entire area south of the centre of the municipality showed concentrations above 200 mg/L, with a continuous band of concentrations above 300 mg/L. In contrast, the northern part of the municipality was less polluted, but concentrations above the limit value of 50 mg/L were also measured in the majority of the area. By 2019, the north–south difference had disappeared, with a significant positive change in the central part of the area. The most polluted areas are still the South-West and South-East parts of the municipality, but their area coverage has decreased significantly (Figure 5). In parallel, the proportion of areas with concentrations below 100 mg/L has increased significantly.

In the period before the construction of the sewerage network, clear spatial differences were shown in the degree of contamination in the area of the settlement. The degree of contamination increases in the direction of local groundwater flow, which was detected for most of the parameters studied (NH_4^+, NO_2^-, NO_3^-, EC). The horizontal hydraulic gradient slopes toward the Eszteró-Canal in the southern vicinity of the settlement, which drains the area.

NO_3^- concentrations before the construction of the sewerage network showed a very significant degree of contamination in the S areas of the municipality. The N part of the settlement was less polluted, but in the vast majority of the area concentrations above the limit were measured. Wells located south had an average NO_3^- value of 429 mg/L, while wells located north had an average value of 147 mg/L. There were also significant differences in the EC values of groundwater wells. While the average EC value of south-located wells was 4396 μS/cm, the average value of north-located wells was 2476 μS/cm. A total of 87% of the wells with NH_4^+ concentrations higher than 1 mg/L were located in southern areas. Further, 62% of wells with NO_2^- concentrations lower than 0.2 mg/L were in the northern part of the municipality.

A two-step cluster analysis was performed to determine the percentage of wells we classified as "northern" and "southern" (20-20 wells) in the lower and higher contaminated cluster before and after sewering. We also examined the weight of each water chemistry parameter in the formation of the lower and higher contaminated cluster (Figure 6).

In the period 2011–2013, before sewerage, EC had the greatest weight in the formation of clusters. The second and third dominant parameters were NO_3^- and NO_2^-. Nearly 60% of all samples were in the higher contamination cluster (1), with 65.3% of these samples coming from wells located in the south. In the lower contamination group (2), 61.3% of the samples were from wells located in the north. Therefore, it can be seen that there is of course no clear line between the boundaries of the two areas, but that a transition zone is also emerging. At the same time, the analysis shows that a marked north-south difference appears. This is confirmed by the markedly different average values of the parameters of the two clusters (Figure 6). The higher contaminated cluster had an average EC value of 4396 μS/cm and an average NO_3^- value of 430 mg/L, compared to 2476 μS/cm and 147 mg/L for the lower contaminated cluster.

For the 2017/2018/2019 post-sewerage database, NO_3^- and EC were the main clustering factors. The third most important parameter was COD. The purification processes are illustrated by the fact that, in contrast to the pre-sewerage period, more than 70% of the water samples were placed in the lower pollution cluster. Furthermore, 52.3% of the water samples in the cluster came from wells located in the north. This shows that the differences between northern and southern wells are decreasing. The average EC value decreased for both higher and lower contaminated clusters in the post-sewerage

period (from 4396 μS/cm to 4168 μS/cm; from 2476 μS/cm to 2193 μS/cm). An even more significant decrease was observed for NO_3^-. From 429 mg/L to 286 mg/L for the higher contaminated cluster and from 147 mg/L to 84 mg/L for the lower contaminated cluster.
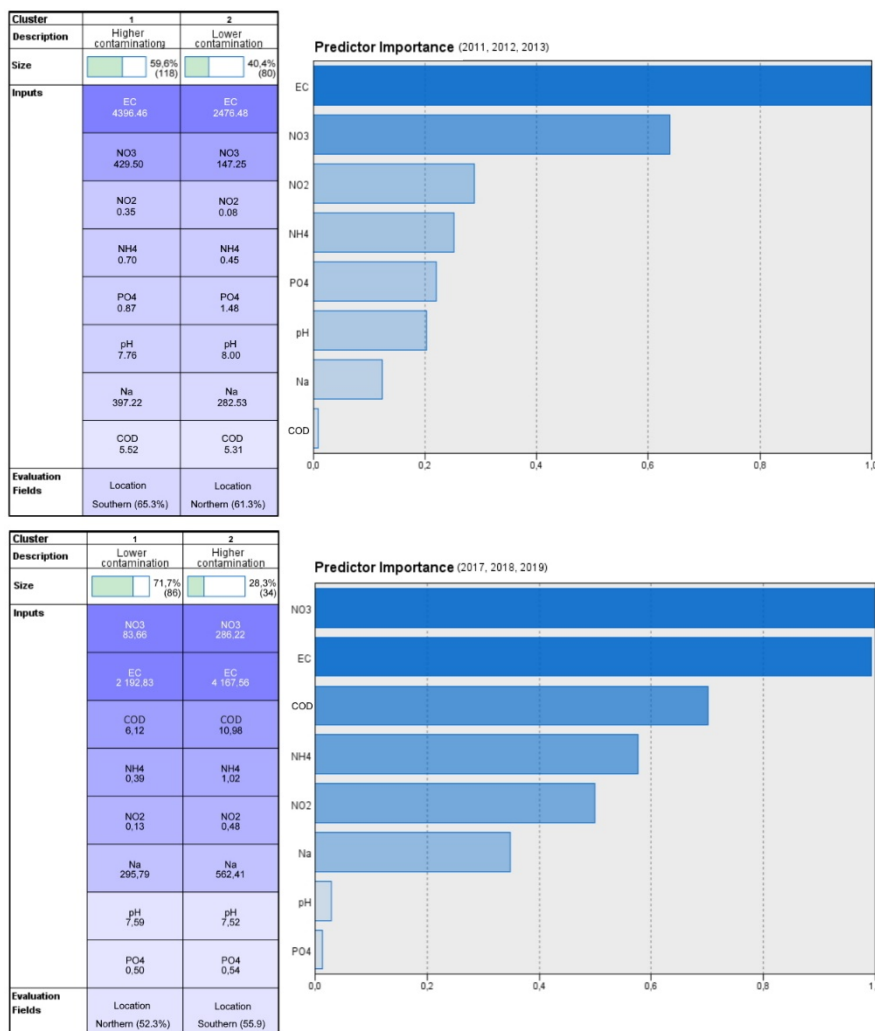

Figure 6. Results of two-step cluster analysis for data before and after the establishment of the sewerage network.

Since hierarchical cluster analysis is not suitable for large sample sizes, the results of the summer sampling before sewerage (2013) and five years after sewerage (2019) were included in the analysis. However, an important advantage of the analysis is that it does not work with a predefined number of clusters, so that grouping wells by common characteristics gives a more detailed picture. The dendograms produced by the Ward method are shown in Figure 7. Based on the results of 2013, a south-only cluster of 12 wells based on hierarchical cluster analysis was clearly defined (Figure 7). The other dominant cluster contained northern wells in 74%. In addition, a third cluster was also formed the five wells of which can be found in the central part of the settlement (17, 18, 24, 25, 27). In the fifth year following the establishment of the sewerage network, this clear difference between wells was eliminated with a decrease in the level of contamination, with approximately the same proportion (52–48%) of north and south wells in the two established clusters. The decrease in the differences between northern and southern wells is associated with a decrease in pollution.
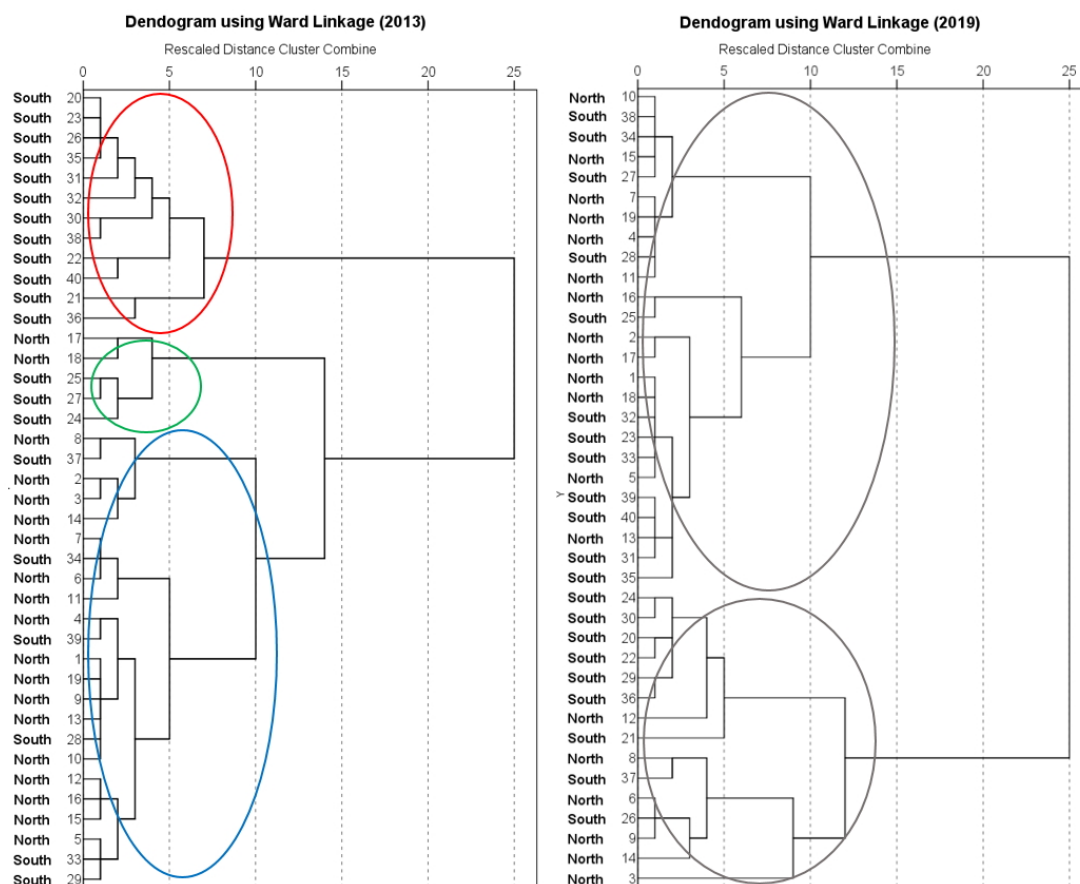
Figure 7. Ward's dendogram based on the location of sampling sites before (2013) and after (2019) the construction of the sewerage network.

3.5. Results of Principal Component Analysis

In order to explore the degree of monotonic relationship between the examined parameters, correlation calculations were performed with the help of SPSS 22 software (Table 3). Among the examined parameters, a significant ($p < 0.001$), strongest positive relationship was found between NO_3-EC ($r = 0.779$) and a moderately strong correlation between Na-EC ($r = 0.445$) and NO_2–EC ($r = 0.408$). NO_3–EC and Na-EC values were also plotted on an ordination diagram (Figure 8).

Table 3. Correlation matrix of the investigated parameters.

Parameter	pH	EC	NH_4^+	NO_2^-	NO_3^-	PO_4^{3-}	COD	Na^+
pH	1.000							
EC	−0.115 *	1.000						
NH_4^+	0.044	0.408 **	1.000					
NO_2^-	−0.044	0.357 **	0.335 **	1.000				
NO_3^-	−0.112 *	0.771 **	0.297 **	0.205 **	1.000			
PO_4^{3-}	0.329 **	0.112 *	0.029	−0.095	0.192 **	1.000		
COD	−0.117 *	0.165 **	0.188 **	0.221 **	−0.106	0.080	1.000	
Na^+	−0.016	0.445 **	0.066	0.173 **	0.329 **	−0.002	0.124 *	1.000

* Correlation is significant at the 0.05 level (2−tailed); ** correlation is significant at the 0.01 level (two−tailed).

To explore the background geochemical processes, the database was subjected to principal component analysis. Since the data series did not show a normal distribution, they were normalised. The Kaiser–Meyer–Olkin test result showed a value above 0.6 and the Bartlett test result showed $p < 0.001$, so the conditions for principal component analysis were met. Three principal components were identified according to the Kaiser

criterion, explaining 65.81% of the total variance. The first principal component was formed by NO_3^- EC-Na. The second principal component was formed by $COD-NH_4^+-NO_2^-$, indicating a close relationship between organic matter degradation, ammonification and nitrification (Table 4). The third principal component was formed by PO_4^{3-}- pH, which can be explained by the different solubility of phosphate at different pH values.
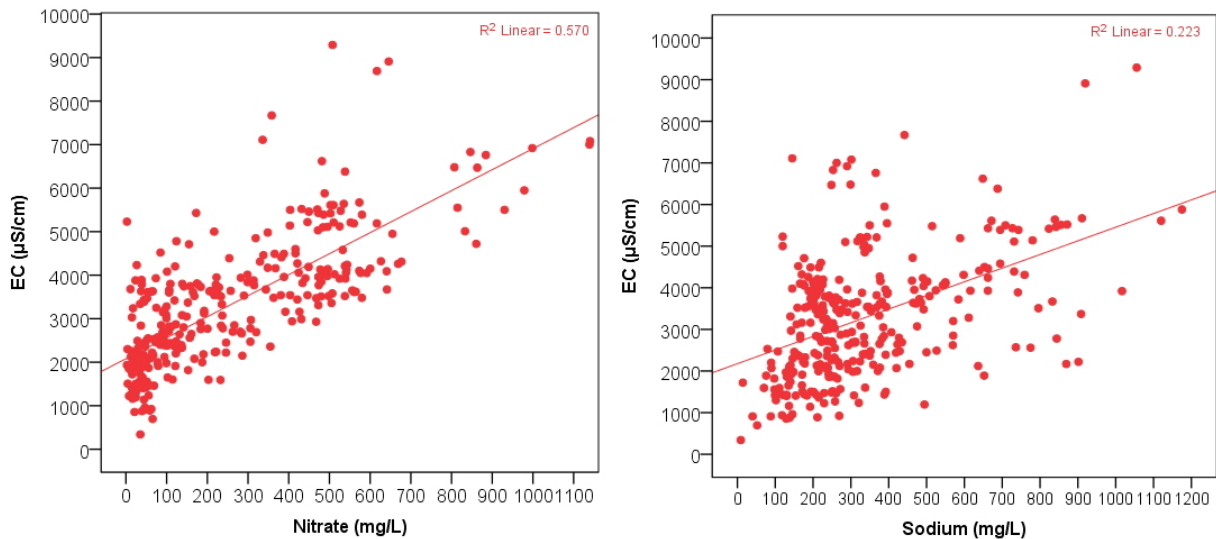

Figure 8. Scatter plot diagram for $EC-NO_3^-$ and $EC-Na^+$.

Table 4. Loadings of the variables on three significant principal components.

	Rotated Component Matrix [a]		
Parameters	**Component**		
	1	**2**	**3**
NO_3^-	**0.888**	−0.063	0.074
EC	**0.877**	0.307	0.021
Na^+	**0.569**	0.110	−0.083
COD	−0.108	**0.743**	−0.015
NH_4^+	0.212	**0.717**	0.147
NO_2^-	0.255	**0.657**	−0.109
PO_4^{3-}	0.148	−0.052	**0.810**
pH	−0.173	0.068	**0.790**

Extraction method: principal component analysis. Rotation method: varimax with Kaiser normalization.
[a] Rotation converged in four iterations.

Data from the wells located north and south were subjected to principal component analysis, which clearly showed the difference between the north and south wells. After the construction of the sewerage network, the size of the transition zone between groups increased (Figure 9).

3.6. Anion and Cation Content of Groundwater

Piper and Durov diagrams were created to understand the hydrochemical facies (Figure 10). The dominant cations and anions were Ca^{2+} and Cl^- since the majority of the samples were classified as Cl-Ca type and SO_4*Cl-Ca*Mg type. The hydrochemical facies of the shallow groundwater were determined by rock dominance, evaporation and anthropogenic activities. The main source of chloride was municipal wastewater beside of geogene origin.
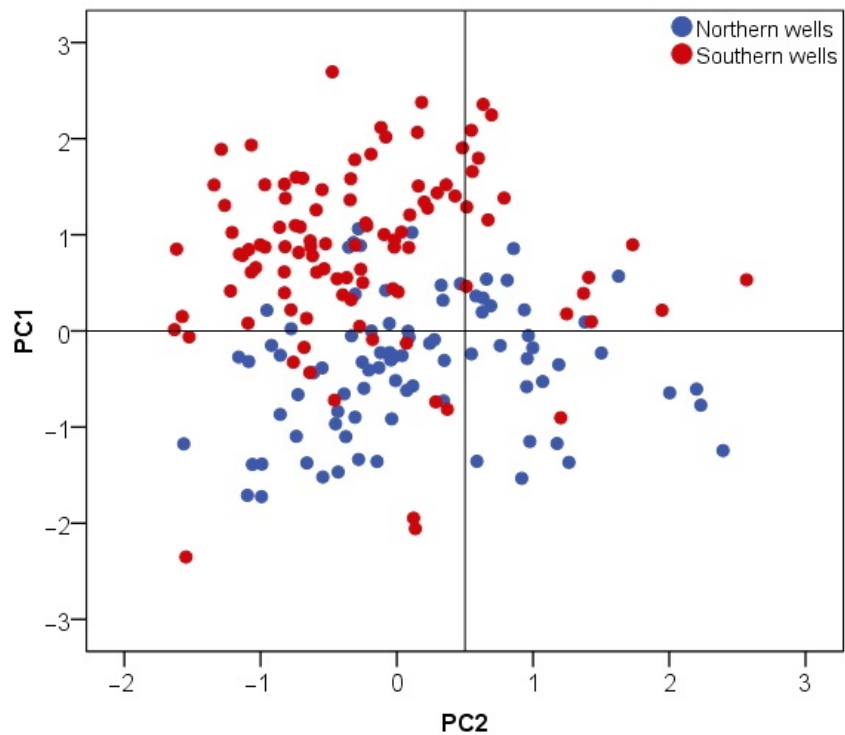
Figure 9. Bivariate plot of the scores of PCs 1 and 2 according to zone of location of the monitoring wells before sewerage.
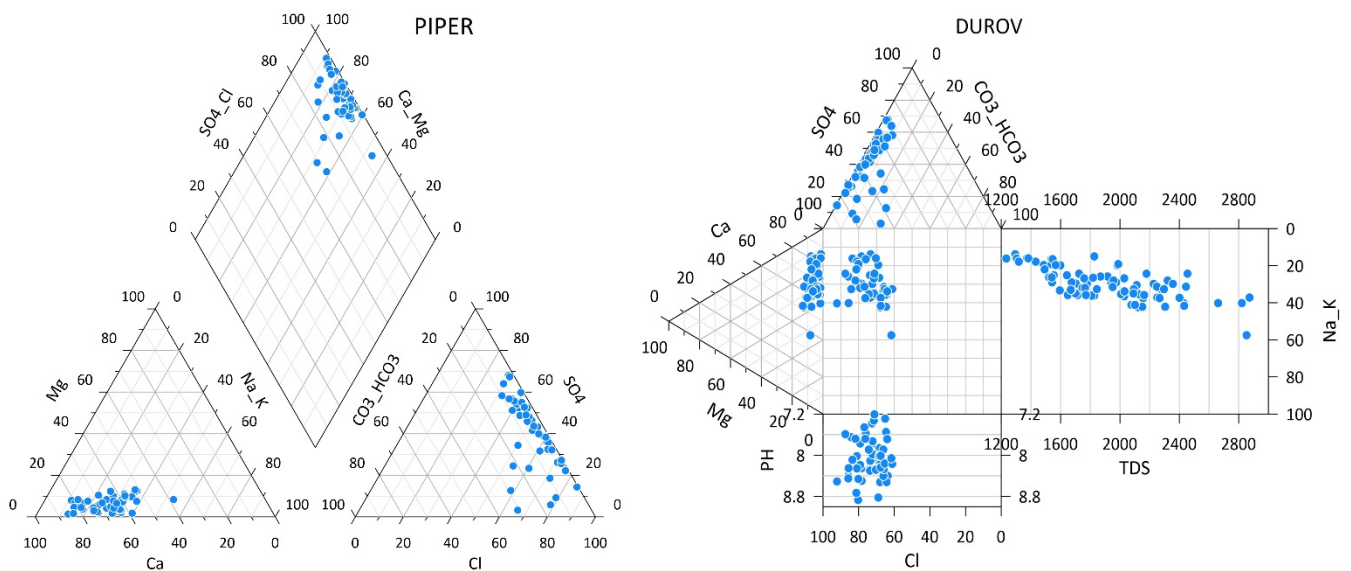

Figure 10. Piper and Durov diagrams for groundwater samples of the study area after sewerage.

3.7. Changes in Groundwater Level after the Construction of the Sewerage Network

Changes in the groundwater level after the construction of the sewerage network were assessed on the basis of water levels measured during the summer sampling (Figure 11). In the year preceding the establishment of the sewerage network in 2013, the groundwater level was between 88.51 m a.s.l. and 86.98 m a.s.l. The groundwater level was highest in the inner parts of the settlement, while the deepest water level could be measured in the southern areas of the settlement.
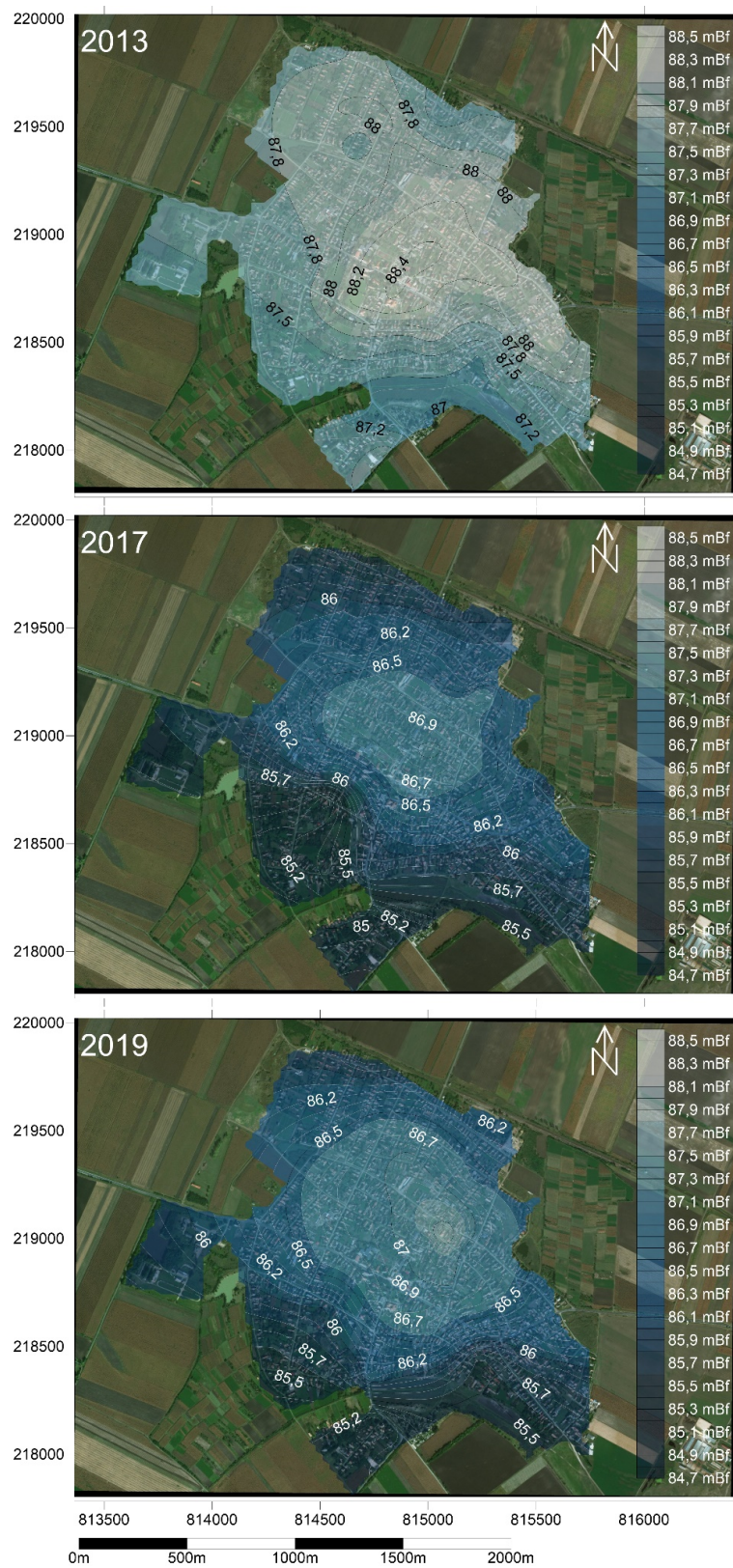
Figure 11. Groundwater table (a.s.l.) before (2013) and after (2017, 2019) the construction of the sewerage network.

In the period after the establishment of the sewerage network, significant changes can be observed as a result of that wastewater outflow was stopped. There was a strong

descent in groundwater levels. The average decrease in water levels in the period after the establishment of the sewerage network was 1.5 to 2 m, which cannot be explained by changes in precipitation conditions, the reason for the decrease in water levels is clearly due to that wastewater outflow was eliminated.

We found that differences between low- and high-water levels increased after the construction of the sewerage network. While the difference between the minimum and maximum water levels was 1.53 m in 2013, this increased to 2.29 m by 2017 and was 2.15 m in 2019. The phenomenon can be explained by that wastewater from households made the water level in the municipality more balanced. In southern areas with low groundwater levels, the damming effect due to wastewater outflow was even stronger than in areas with higher water levels.

A significant, strong negative correlation between the groundwater data series of the period before and after the establishment of the sewerage network (r = −0.812) has been demonstrated. Applying the Wilcoxon test, we showed that there was a clear background process (sewerage) behind the differences in the three-year data series. We found that by discriminant analysis water level data can be used to determine with an accuracy of 94.2% whether the water level data is derived from the period before or after the establishment of the sewerage network (Table 5).

Table 5. Classification results of the discriminant analysis for groundwater levels.

Classification Results [a,c]					
		Sewerage Status	Predicted Group Membership		Total
			Before Sewerage	After Sewerage	
Original	Count	Before sewerage	43	4	47
		After sewerage	3	70	73
	%	Before sewerage	91.5	8.5	100.0
		After sewerage	4.1	95.9	100.0
Cross-validated [b]	Count	Before sewerage	43	4	47
		After sewerage	3	70	73
	%	Before sewerage	91.5	8.5	100.0
		After sewerage	4.1	95.9	100.0

[a] In total, 94.2% of original grouped cases were correctly classified. [b] Cross validation was done only for those cases in the analysis. In cross validation, each case was classified by the functions derived from all cases other than that case. [c] A total of 94.2% of cross-validated grouped cases were correctly classified.

The negative effect of falling groundwater levels as a result of the construction of the sewerage network can be seen in trees drying out, subsiding houses and fracturing house walls. These processes are currently based only on observations of residents and have not yet been followed up by scientific studies, but we also consider it of paramount importance to monitor the potential negative impacts of wastewater investments.

4. Conclusions

Based on the results of water sampling carried out in the pre-sewerage period (2011–2013), a strong contamination of the groundwater supply of the settlement was found. Significant positive changes in the quality of groundwater were detected when the data series of the time periods prior to (2011–2013) and after (2017–2019) the establishment of the sewerage network using various geoinformatic and statistical methods. The results of the analysis of the large sample prior to (2013) and after (2017–2019) the establishment of the sewerage network showed the clear decrease of the concentration of inorganic contaminants (NH_4^+, NO_2^-, NO_3^-, PO_4^{3-}).

Applying discriminant analysis, it can be determined with 82.4% probability whether the sample is from the period prior to or after the establishment of the sewerage network.

On this basis, the construction of the sewerage network resulted in marked changes in the quality of groundwater wells.

In the period before sewerage, clear spatial differences were shown in the degree of contamination in the area of the settlement. We found that the degree of contamination increases in the direction of local groundwater flow, which was detected for most of the parameters studied (NH_4^+, NO_2^-, NO_3^-, EC). The results of the principal component and two-stage cluster analyses also supported the differences between the N and S areas. At the same time, of course, there is no clear line between the boundaries of the two areas, but a transitional zone is also formed. In the fifth year following the establishment of the sewerage network, this clear difference between wells was eliminated with a decrease in the level of contamination, with approximately the same proportion of north and south wells in the two established clusters.

It has also been shown that groundwater levels have decreased significantly in the post sewerage period. The average decrease of 1.5–2 m can be explained by the cessation of sewage outflow in the settlement. Any further decrease could increase the negative effects of the investment.

On this basis, it can be declared that the purification processes in the area have started, but at the same time, our results show that pollutants entering environment undergo a number of transformations and remain in the system for a long time. Therefore, further monitoring and environmental measures may be necessary.

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