Evaluation of the soil quality in the vicinity of Inowrocław soda plants

Katarzyna Maria Wiatrowska*, Jolanta Komisarek, Jerzy Marcinek,
Michał Kozłowski

Poznań University of Life Sciences, Poland

*Corresponding author's e-mail: kawiatr@up.poznan.pl

Abstract: The aims of this study were to assess the quality of soils affected by strongly saline waste products and to analyze the spatial distribution pattern of soil properties influenced by a soda plant. Soil samples were analyzed for electrical conductivity (EC), sodium adsorption ratio (SAR), pH, trace element content and dehydrogenases activity. The research was conducted in the vicinity of the waste ponds of the Ciech Soda Poland S.A. company (Central Poland), where 35 groundwaters, 63 surface (0–20 cm) and 60 subsurface (80–100 cm) samples were collected. Although the waste ponds are currently not used for regular disposal of waste products, a high level of salinity of groundwater and soil, especially in the subsurface layer, is still being observed. The electrical conductivity of soil saturation extract (ECse) values varied between 0.9–15.4 and 1.0–87.2 dS·m^{-1} for surface and subsurface layer, respectively. Analysis of the spatial distribution of soil salinity reveals a correlation with microrelief and groundwater levels. Maps of the spatial distribution of heavy metals showed that their content in the subsurface layer was a result of chemical migration from the waste ponds, and in the surface layer by other sources (atmospheric deposition). The soil salinity did not influence dehydrogenases activity.

Introduction

As soil plays major role in the maintenance of both local and global environmental quality, the need to evaluate the quality of soil resources has increased. Despite the fact that soil is the basis of food and fiber production, human-induced degradation of nearly 15% of the world's total land area is being observed. Currently, soil degradation as a result of soil erosion, atmospheric pollution, extensive soil cultivation, over-grazing, salinization and desertification, is one of the most urgent global concerns extenuated by demographic pressure (Oldeman 1994, Lal 1998).

A widely accepted definition of soil quality was given by Karlen et al. (1997) as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation." The definition proposed by these authors emphasises the role of soil both for food and fiber production and as well to maintain the quality of the environment. However, there is a lack of agreement what kind of indices should be used to evaluate soil quality. Some authors (Doran 1996, Doran and Perkin 1996, Li et al. 2013) used physical (texture, infiltration rate, bulk density, water retention capacity), chemical (pH, total C, EC, nutrient level), biological and biochemical (C, N, microbiological biomass, potentially mineralizable N, soil respiration, dehydrogenases activity or ATP content) indices for quantifying soil quality.

Soil enzymes activities are widely used as important indicators of soil quality and soil biological activity as they are closely related to soil properties, soil type and environmental conditions (Singh 2015, Guangmining et al. 2017, Zheng et al. 2017). Salinity affects plants and microbes via two mechanisms: osmotic effect and specific ion effects (Yan et al. 2015, Oren 1999, Zheng et al. 2017). Soluble salts increase the osmotic potential of soil water not only by impeding water absorption by roots and microbes, but also by drawing water out of cells. Additionally, high concentrations of certain ions like Na$^+$ and Cl$^-$ are toxic to many plants and microbes. Moreover, salinity affects enzyme activities by denaturing proteins by disturbing their tertiary structure and reducing their solubility (Zehng et al. 2017). In consequence, soil salinity changes microbial community structure, reduces microbial biomass and their activity and disturbs enzyme activities.

Guangmining et al. (2017) studied the influence of soil salinity on the activity of three enzymes: urease, alkaline phosphatase and catalase. These results showed a significant reduction in activity of all enzymes with increasing soil salinity. Furthermore, path analysis revealed that salinity had little direct effect, but had an important indirect effect via soil properties on urease and alkaline phosphatase activity. Also other authors (Telesiński 2012, Zheng et al. 2017) found

that NaCl had a negative correlation with activities of some soil enzymes such as dehydrogenases, catalase, invertase and urease. Zheng et al. (2017) thought that soil salinity affects the enzymatic activity through a variety of abiotic and biotic factors. Soil salinity changes soil physico-chemical properties, reduces the number of microorganisms and the microbial respiration and finally lowers osmotic potential, which leads to accelerated microbial cell lysis.

Nonetheless, the proposed indices did not satisfy all the requirements for a universal soil quality index, such as: sensitivity to the presence of the greatest possible number of degrading agents, consistency in the direction of response to a particular stress factor and reflection of the different levels of degradation (Doran and Zeiss 2000, Gil-Sotres et al. 2005). Therefore, the soil quality indices should be locally selected according to soil function and nature of stress.

This paper is concerned with one of the largest industrial plants in Poland affecting soil salinity known as Ciech Soda Poland S.A. in Mątwy (previously Inowrocław Soda Plants). Soil salinity in Poland is a problem caused by a constant supply of water rich in soluble salts from natural or anthropogenic sources. Industrial and mining activities are the major sources responsible for the emergence of salt-affected soils (Hulisz 2008). The aim of this study was to analyse the quality of soils affected by strongly saline waste products. Additionally the spatial variability of some soil properties that may be altered by industry was investigated. The assessment of the soil quality was done on the basis of selected parameters; some typically used for describing saline soil like: EC, SAR, pH, Cl⁻ and additionally trace element content as well as dehydrogenases activity (DHA).

Study area

The Ciech Soda Poland S.A. plant is located in the Noteć river valley at the Kujawy region (central Poland), within the western part of the Kujawsko-Pomorskie anticline. A characteristic form of this area is a salt dome, which occurs in the centre of Inowroclaw. The salt dome determines the land relief, layout of geological layers and water regime of the city and its neighborhood. The sample area lies 2.5 km beyond the southern border of the salt dome (Marek and Znosko 1972). The soda plant has been in operation since 1882 and waste products have been collected in open sediment ponds next to it. The semi-fluid waste products carried away to settling ponds contained mainly $CaCO_3$, $CaSO_4$, $Ca(OH)_2$, $Fe(OH)_3$, $NaOH$, NH_4OH, KCl, $NaCl$, $MgCl_2$, $CaCl_2$, Na_2SO_4, silicates and aluminosilicates (Cieśla et al. 1981, Hulisz and Piernik 2011). This long-term deposition of extremely saline sludges has caused significant groundwater pollution, especially because only new settling ponds have appropriate sealed bottoms, but older ones, situated directly on permeable ground, are without any sealing (Strzelecka et al. 2011). Even though, after modernization of the technological process, settling ponds are currently not used and diluted fluid waste products are carried away to the Vistula river, their adverse influence on the soil environment is still being observed (Hulisz and Piernik 2011). The use of the settling ponds has led to a rise of groundwater level and a change in flow pattern. A shallow level of saline-affected groundwater has a destructive effect on the soil

structure and results in the accumulation of alkaline cations, mainly sodium, in soils. Consequently, the area not suitable for agriculture is extending and a succession of natural salt-tolerant plants is occurring (Cieśla et al. 1981, Piernik et al. 1996).

Materials and methods

Sampling area

An area of about 25.6 ha, bounded by the settling ponds on the West, Poznańska street on the East and Bagienna street on the North, was sampled in December 2013 (Fig. 1). Research was conducted on a flat area slightly sloping in the easterly direction. For the purpose of the study, 35 water, 63 soil surface (0–20 cm) and 60 subsurface (80–100 cm) samples were collected. At the same time the groundwater table was measured. Soil samples collected for enzymatic activity were stored in polyethylene containers at 4°C and immediately analysed at the laboratory. The position of each sampling point was measured by using a GRX-1 device by SOKKIA. The measurement accuracy of this device for X and Y coordinates was 10 mm + 1 ppm, while for the Z coordinate, 20 mm + 1 ppm, respectively. The applied communication and control software was the Sokkia Spectrum Field ver. 8.1 installed at the SHC-250 controller, while post-processing was performed using the Topcon Link ver. 8 and Spectrum Survey Office ver. 8.2 programmes. A digital elevation model of the study area, with 2 m spatial resolution was obtained from LiDAR data.

Soil and water analyses

The soil samples collected were air-dried and sieved throughout a 2 mm sieve. The soil reaction (pH) was measured potentiometrically in a soil:water suspension 1:5 (v/v) according to the Polish Standard PN-EN ISO 10390. The electrical conductivity ($EC_{1:5}$) was determinated at a soil:water ratio 1:5 (m/v) as reported in PN-EN ISO 11265+AC1. Taking into consideration that salinity classes are given for electrical conductivity of soil saturation extract (EC_{se}), then the latter was calculated using the relationships (DERM 2011):

$$EC_{se} = EC_{1:5} \cdot \frac{Q_{1:5}}{Q_{se}} \qquad Q_{1:5} = 500 + 6ADMC$$

where:

EC_{SE} – electrical conductivity of soil saturation extract ($dS \cdot m^{-1}$)

$Q_{1:5}$ – water content at equivalent of 1: 5 soil:water suspension

Q_{se} – water content equivalent to soil saturation (%)

500 – constant value in the equation

ADMC – air dry water content expressed as $kg \cdot 100kg^{-1}$

The soluble chloride content was determined by Mohr's method by titrating the soil extract with silver nitrate in the presence of potassium chromate as an indicator (van Reeuwijk 2002, PN-ISO 9297). In the soil:water (1:5) extracts, the concentration of alkaline cations was measured in order to calculate sodium adsorption ratio (SAR), which was estimated accordingly:
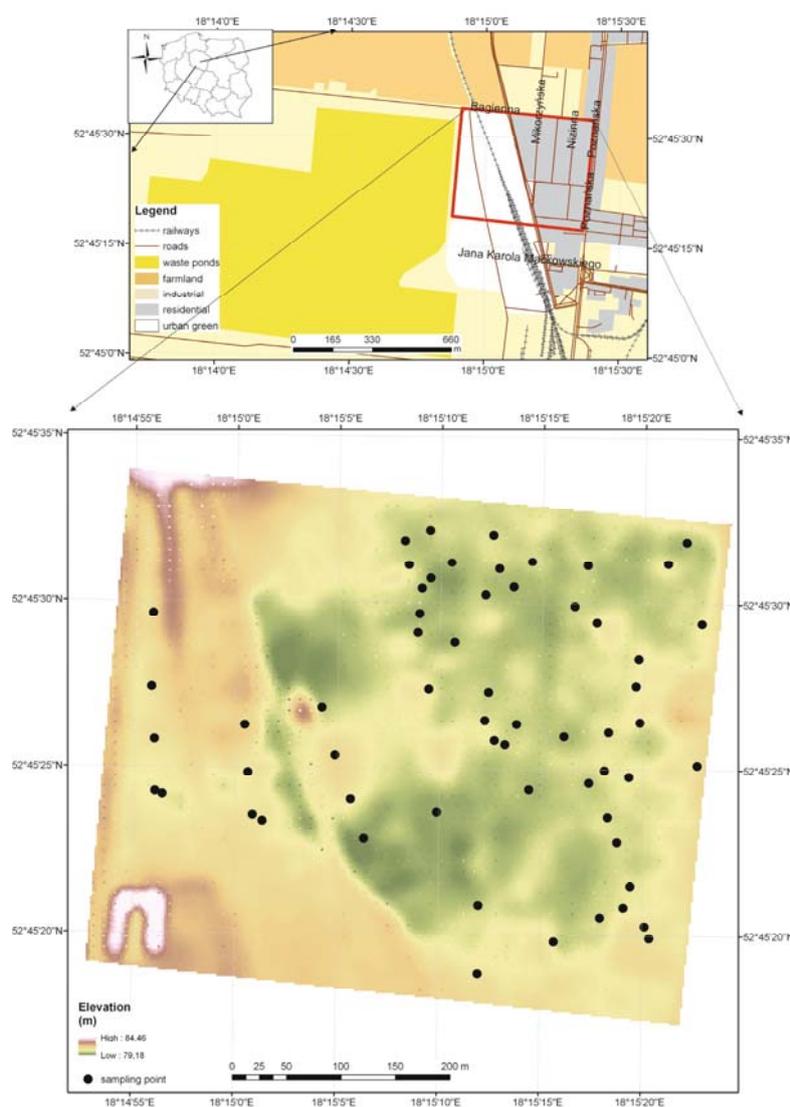
Fig. 1. Localization of study site and digital elevation model of an area studied with sampling points

$$SAR = \frac{Na^+}{\sqrt{(Ca^{2+} + Mg^{2+})}}$$

where:
Na^+, Ca^{2+}, Mg^{2+} – cation concentrations in meq·dm^{-3}

The total content of heavy metals (Cd, Cu, Pb, Zn) was determined in a soil:aqua regia mixture 1:10 (m/v), in a closed system. The concentrations of water soluble cations (Ca^{2+}, Mg^{2+}, K^+, Na^+) were determinated by using a 249 FSAA Agillent Technologies atomic absorption spectrometer. Additionally, in the surface soil samples organic carbon content and dehydrogenases activity were measured. All analyses were duplicated except for dehydrogenases activity which was done in triplicate. Total organic carbon was determined by dry combustion using Multi N/C 3100 (Jena Analytics). Prior to the analysis carbonates were removed by HCl treatment. An internal laboratory reference material and reagent blank were used in order to control the accuracy of soil analyses. The

values obtained for all parameters (heavy metals, basic cations and organic carbon content) fall in the required range of reference content. The dehydrogenases activity was assessed as described by Tabatabai (1982) with 2,3,5-triphenyltetrazolium chloride. To present the concentration of heavy metal and evaluate potential contamination of the soil studied, the geoaccumulation index (I_{geo}) proposed by Müller (1969) was applied. This index was calculated by using the following formula:

$$I_{geo} = \log_2\left(\frac{C_n}{1.5 \cdot B_n}\right)$$

where:
C_n – the measured content of metal (mg·kg^{-1})
B_n – the geochemical background value of the element (mg·kg^{-1})
1.5 – factor introduced to mitigate impact of fluctuation in the element content which may be attributed to lithogenic effect

Water samples were analysed for physico-chemical and chemical parameters such as pH, electrical conductivity, Cl, Ca, Mg, K and Na according to Polish standard methods (PN-EN ISO 10523, PN-EN 27888, PN-ISO 9297).

Statistical and spatial analysis

Statistical treatment (summary statistic, correlation analysis and principal component analyses – PCA) was performed with Statistica v. 7. The maps of the spatial distribution were prepared using Surfer v. 8 software.

Results and discussion

Groundwater characteristics

The groundwater was the main source of soil salinity in the study area since it occurred close to the ground surface. During sampling time, the groundwater table was noted between 0.45 and 0.90 m bgl (below ground level). Additionally, some areas showed signs of permanent wet condition (redoxymorphic colours). The measurements of groundwater level revealed that saline waters move from settling ponds in a E/SE direction, towards Maćkowskiego street (Fig. 2). The groundwater samples collected showed neutral to slightly alkaline pH,

as a consequence of a high concentration of alkaline cations (Table 1). The salinity of groundwaters, measured as electrical conductivity (EC), varied between EC 0.69 to 39.3 dS·m^{-1} (25°C). Significantly higher results of EC, in areas adjacent to the settling ponds, were reported by Hulisz (2003) and Strzelecka et. al (2011). However, these authors collected samples during the summer period. In Poland, the salinity is strongly dependent on the amount and distribution of precipitation and changes significantly during the year. Annual evapotranspiration in Poland is lower than annual precipitation, leading to a natural leaching of soluble salts in the soil profile and decreasing the salinity level at the surface soil horizon. However, during summertime, the evapotranspiration usually exceeds precipitation causing an increase in soil salinity. As in this study groundwater samples were collected after wet period, lower values of EC were expected (Fig. 3). The spatial variability of groundwater EC was very high and correlated with microrelief and the direction of water flow. The highest salinity of shallow groundwater was noted in local depressions, in the close vicinity to waste ponds and at Marcinkowskiego street. A basic statistical analysis reveals a strong and positive correlation between EC values and the concentrations of Cl$^-$ (0.98), Na$^+$ (0.93), Ca^{2+} (0.81) and K$^+$ (0.42) ions (at $p<0.05$).
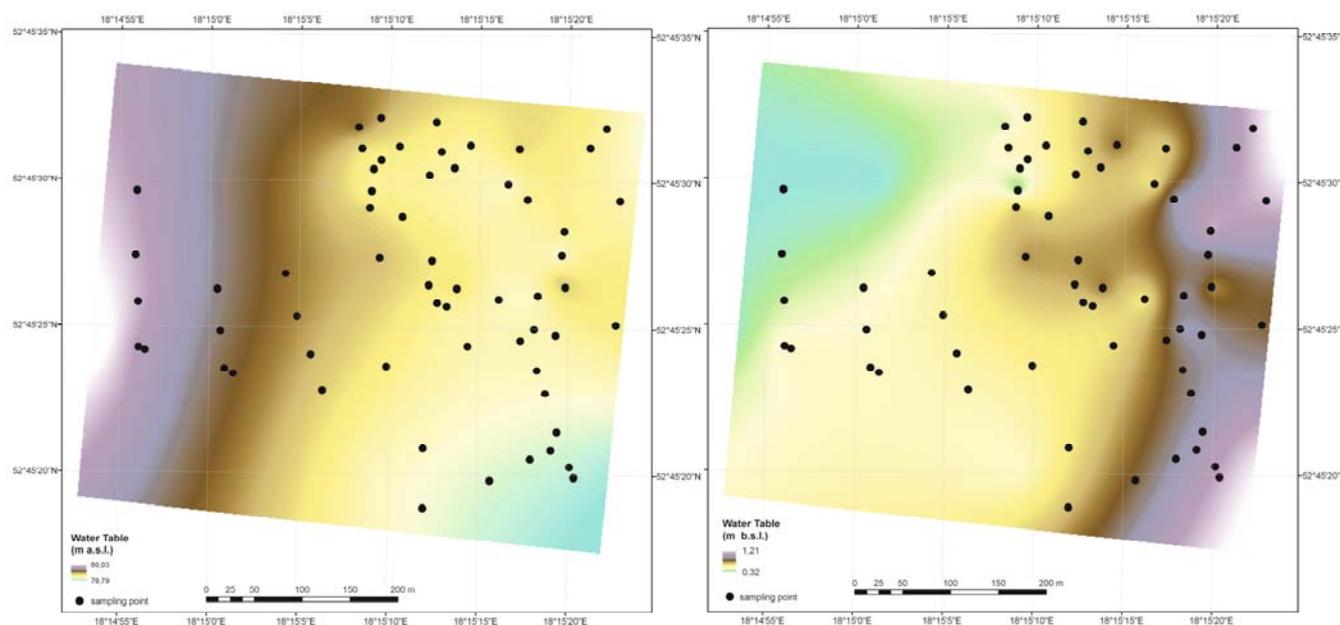

Fig. 2. Map of a hydroisohypse and hydroisobath

Table 1. Selected shallow groundwater properties

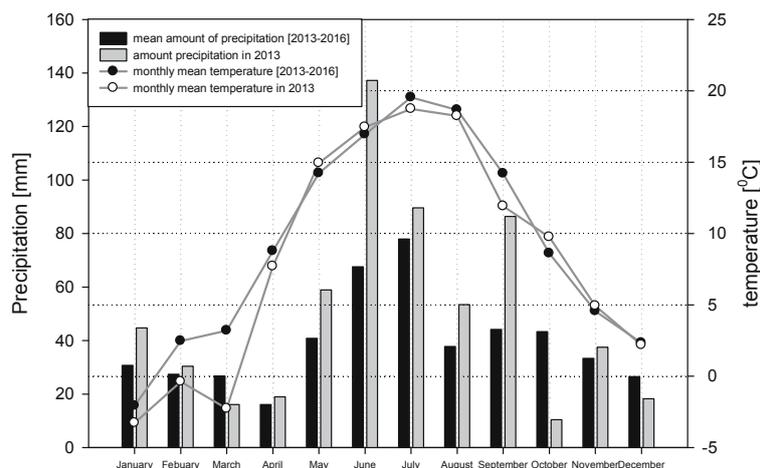| Value	EC [dS·m^{-1}]	SAR	pH	Cl$^-$	Ca^{2+}	Mg^{2+}	K$^+$	Na$^+$
				[mg·dm^{-3}]				
Minimum	0.69	0.002	7.08	27.65	51.49	13.32	0.33	0.08
Maximum	39.3	32.68	8.62	16522	993	238	209	4271
Mean	4.94	4.37	7.77	1364.36	271	78.21	43.93	496
Median	2.73	1.86	7.76	386.87	155	58.08	27.63	152
SD	7.07	6.72	0.44	2982	246	55	48	865

Fig. 3. Distribution of precipitation and monthly mean temperature in 2013 and 2013–2016 (www.wunderground.com)

Soil salinity

The soils of the study area were classified as Mollic Gleysols Amphisalic (WRB 2014). The thickness of the mollic horizon (A) varies between 30–60 cm and organic carbon content was 9.79–116.2 g·kg^{-1}. The surface layer was characterized by texture of sand, loamy sand and sandy loam. The subsurface layer had slightly coarser texture, which was categorized as sand and loamy sand. Soil samples were classified as alkaline to strongly alkaline as a consequence of a high content of basic cations (Table 2). The influence of shallow saline groundwater led to a disturbance in the chemical composition of the soil adsorbing complex. Particular attention should be paid to the high content of water-soluble sodium, which amounted to 361.8 mg·100 g^{-1} in the surface layer and up to 882 mg·100 g^{-1} at 80–100 cm bgl. Meanwhile, other cations (Ca, Mg, K) occurred in relatively small amounts (Table 2). A similar pattern was obtained by Cieśla et al. (1981). However, in comparison to their findings a higher content of water-soluble cations was observed, especially for Ca^{2+}, K$^+$ and Na$^+$. Among water-soluble anions, only chloride concentration was measured. The study area showed high variability in Cl$^-$ content, in the range 1.42–197.1 and 0.71–627.1 mg·100g^{-1} for surface and subsurface layer, respectively. The spatial variability of Cl$^-$ content was very high and it corresponded to the microrelief (Fig. 4). The highest amount of chloride ions was observed in local depressions. In all sampling points, the content of this ion increased with depth. In contradiction to cation concentrations, chloride content was a few times lower than reported by Cieśla et al. (1981), Strzelecka et al. (2011) and Hulisz and Piernik (2013). According to Smith (1972), chloride ions are not sorbed by soil colloids. Moreover, they are easily transported through the soil by the movement of water, as a repellent interaction between anions and negative surface charge of soil colloids occurs (Smith 1972, Öberg 2003). As soil samples were taken in winter, during the period when precipitation over evapotranspiration and downward movement of water is observed, chloride ions were leached down the soil profile.

The ECse (25°C) of surface soil samples, except for the areas adjacent to waste ponds, showed values below 2 mS·dm^{-1} (tab. 2). Over 70% of surface samples were classified as non saline, 23% as slightly, 5% as moderately and only 2% as strongly saline (DERM 2011). The highest ECse values were found, like Cl ions, in local depressions. Relatively greater salinity was found in the deeper layer (80–100 cm), over 13% of subsurface soil samples were classified as very strongly saline and only 22% as non saline. The spatial variability of salinity in this layer was very high and corresponded to the microrelief (Fig. 5). A significant (at p<0.05) and negative correlation was found between soil ECse and shallow groundwater level. The coefficients of determination amount -0.43 and -0.52 for surface and deeper samples, respectively. Lower values of the coefficient of determination than expected could be a result of soil management: manuring, irrigation and soil tipping. Excessive salinity of the deeper layer was caused by both slurry percolating through the banks of the waste ponds and the high level of the groundwater table. Additionally, the study area has low annual precipitation that results in seasonal upward water movement and secondary salinisation of the lower part of the soil profile (Fig. 3). Moreover, as the study area is located near a salt extrusion, extensively mined in the past, a natural source of soil salinity can not be excluded. In comparison to Cieśla et al. (1981) and Strzelecka et al. (2011) studies, ECse values noted in this research for surface layer were significantly lower. Hulisz and Piernik (2013) also reported higher values of soil salinity in the Inowrocław-Mątwy region with the ECse values varying between 52–99 dS·m^{-1}. Nonetheless, in the humid climate of Poland salinity level is very variable – it decreases after rains and increases during the summer. As soil samples were collected after a wet autumn, ions had been washed out into the deeper part of the soil profile

Salts in the soil can influence soil processes by: (a) the salt concentration in the soil solution (salinity) which determines the osmotic potential, and (b) the concentration of exchangeable sodium (sodicity) which affects soil structure stability. Four surface and 28 subsurface soil samples were classified as saline (pH < 8.5; ECse > 4.0 dS·m^{-1}; SAR < 13) according to the USDA classification (Yan et al. 2015). These samples were collected in local depressions, where the shallowest groundwater table was observed. Only one subsurface sample fulfilled the criteria for saline-sodic soil (pH < 8.5; ECse > 4.0 dS·m^{-1}; SAR > 13). It was collected close to the eastern bank of the waste ponds.
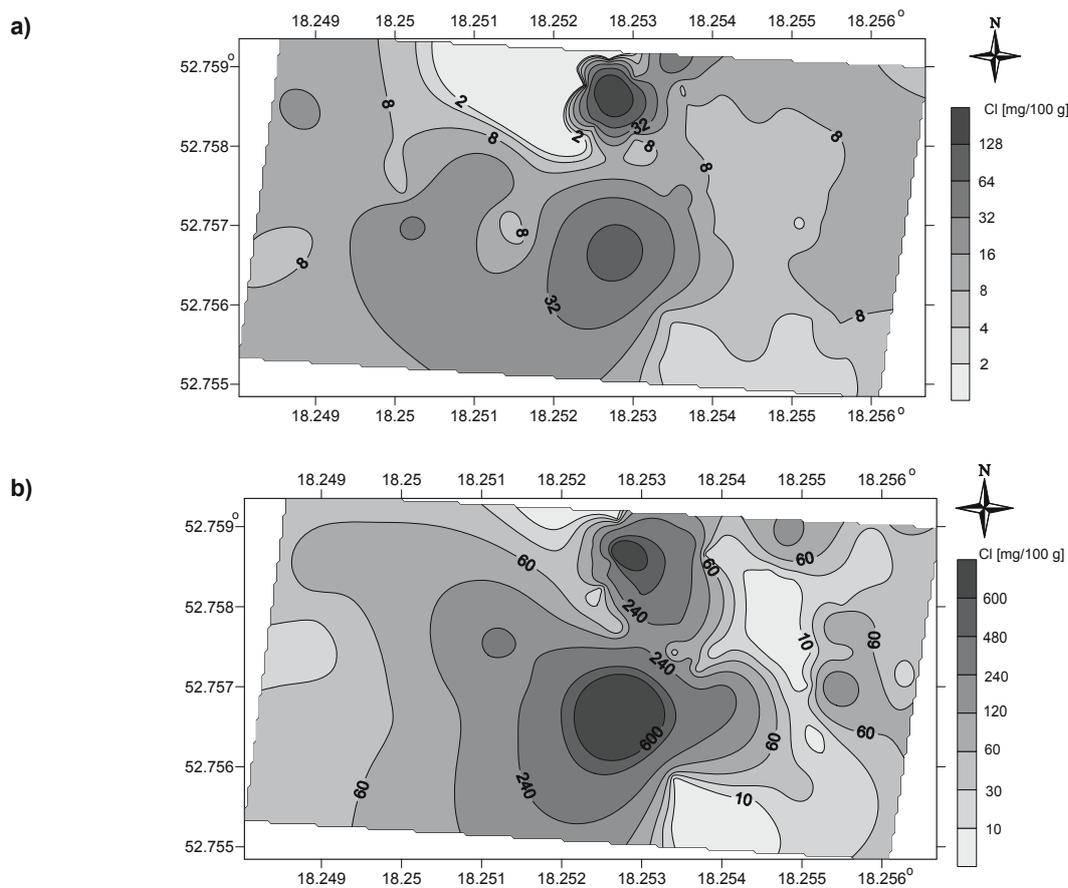
Fig. 4. Maps of the spatial variability of chloride ions content in a). 0–20 cm layer and b). 80–100 cm layer

Table 2. The electrical conductivity, content of water-soluble components and basic soils properties

Properties	Range	Mean	Median	Range	Mean	Median
	0–20 cm			80–100 cm		
C [g·kg^{-1}]	9.79–116.2	28.96	24.56	x	x	x
pH$_{H2O}$	7.89–10.16	8.48	8.46	7.31–12.2	8.46	8.41
EC$_{se}$ (25°C)[dS·m^{-1}]	0.91–15.43	2.05	1.63	1.06–87.2	9.87	3.56
Sand [%]	62–91	80	83	75–97	88	90
Silt [%]	7–33	16	14	2–23	10	8
Clay [%]	1–15	4	3	1–7	3	2
Ca^{2+} [mg·100g^{-1}]	0.03–268.6	38.29	29.96	0.003–856.4	108.69	49.68
Mg^{2+} [mg·100g^{-1}]	0.10–13.59	4.54	4.25	0.003–108.0	12.00	7.09
K$^+$ [mg·100g^{-1}]	3.7–64.63	17.64	13.10	0.44–95.53	17.85	12.13
Na$^+$ [mg·100g^{-1}]	0.9–361.8	27.84	15.38	0.90–882.9	104.86	19.94
SAR	0.14–10.11	1.56	0.94	0.10–18.54	3.18	1.28
Cl$^-$ [mg·100g^{-1}]	1.42–197.1	10.93	4.74	0.71–627.1	60.56	14.53
Dehydrogenases activity [mg TFFg^{-1}d.m.·h^{-1}]	0.05–2.93	0.48	0.34	x	x	x

Contamination levels with heavy metals

The level of soil contamination was assessed using geochemical index (I$_{geo}$). This index enables the assessment of heavy metal contamination by comparing current and pre-industrial metal content (Grzebisz et al. 2002, Baran and Wieczorek 2015, Hanif et al. 2016). The background values of Cd, Cu, Pb and Zn were taken from Kabata-Pendias and Pendias (2001) and amounted to 0.22, 6.5, 18 and 40 mg·kg^{-1}, respectively. According to Müller (1969) the I$_{geo}$ values classified soils as: uncontaminated (I$_{geo} \leq 0$ – class 0); uncontaminated to moderately contaminated (0<I$_{geo} \leq 1$ – class 1); moderately contaminated (1<I$_{geo} \leq 2$ – class 2); moderately to heavily contaminated (2<I$_{geo} \leq 3$ – class 3); heavily contaminated (3<I$_{geo} \leq 4$ – class 4); heavily to extremely contaminated (4<I$_{geo} \leq 5$

– class 5); extremely contaminated ($I_{geo} \geq 5$ – class 6). The values of the geoaccumulation index varied among elements, but for all of them higher I_{geo} values were found in surface samples (Tab. 3), indicating that the main source of heavy metals in this area is atmospheric deposition. The highest values of I_{geo} were found for cadmium: 1.31 and 0.78 in surface and subsurface samples respectively. This may provide evidence that soils in the surface layer were moderately contaminated (class 2) and in the subsurface layer were uncontaminated to moderately contaminated with this metal. Based on the geoaccumulation indices calculated for Cu, Pb and Zn, it was established that soils in surface and deeper layer were uncontaminated to moderately contaminated with these elements. Baran and

Wieczorek (2015) also reported considerably higher values of I_{geo} for Cd than for other elements (Cu, Ni, Pb and Zn). Comparing spatial distribution maps of heavy metals and ECse no similarity was found for surface samples. However, for subsurface samples a similar spatial distribution pattern was found for Cd, Cu and Zn (Fig. 6). This may suggest that waste products deposited in waste ponds do not have any influence on heavy metal content in the surface layer, but they may affect the deeper parts of the soil profiles as a result of capillary rise. In saline soils, the solubility of heavy metals, especially Cd and Cu, increases due to the formation of chloride complexes and the decrease of surface charge density of soil colloids (Du Laing et al. 2008, Sherene 2010, Kadkhodaie et al. 2012).
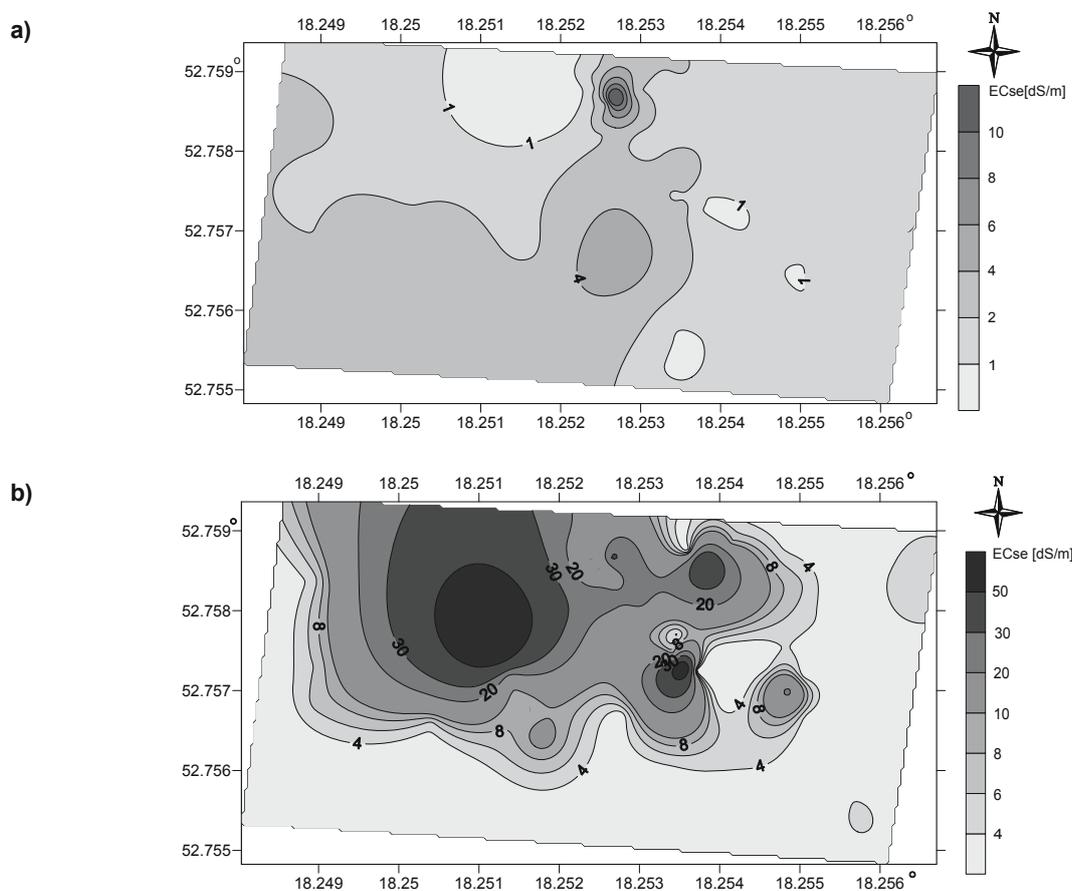

Fig. 5. Maps of the spatial variability of ECse in a). 0–20 cm layer and b). 80–100 cm layer

Table 3. The share of soil samples in particular Müller's classes

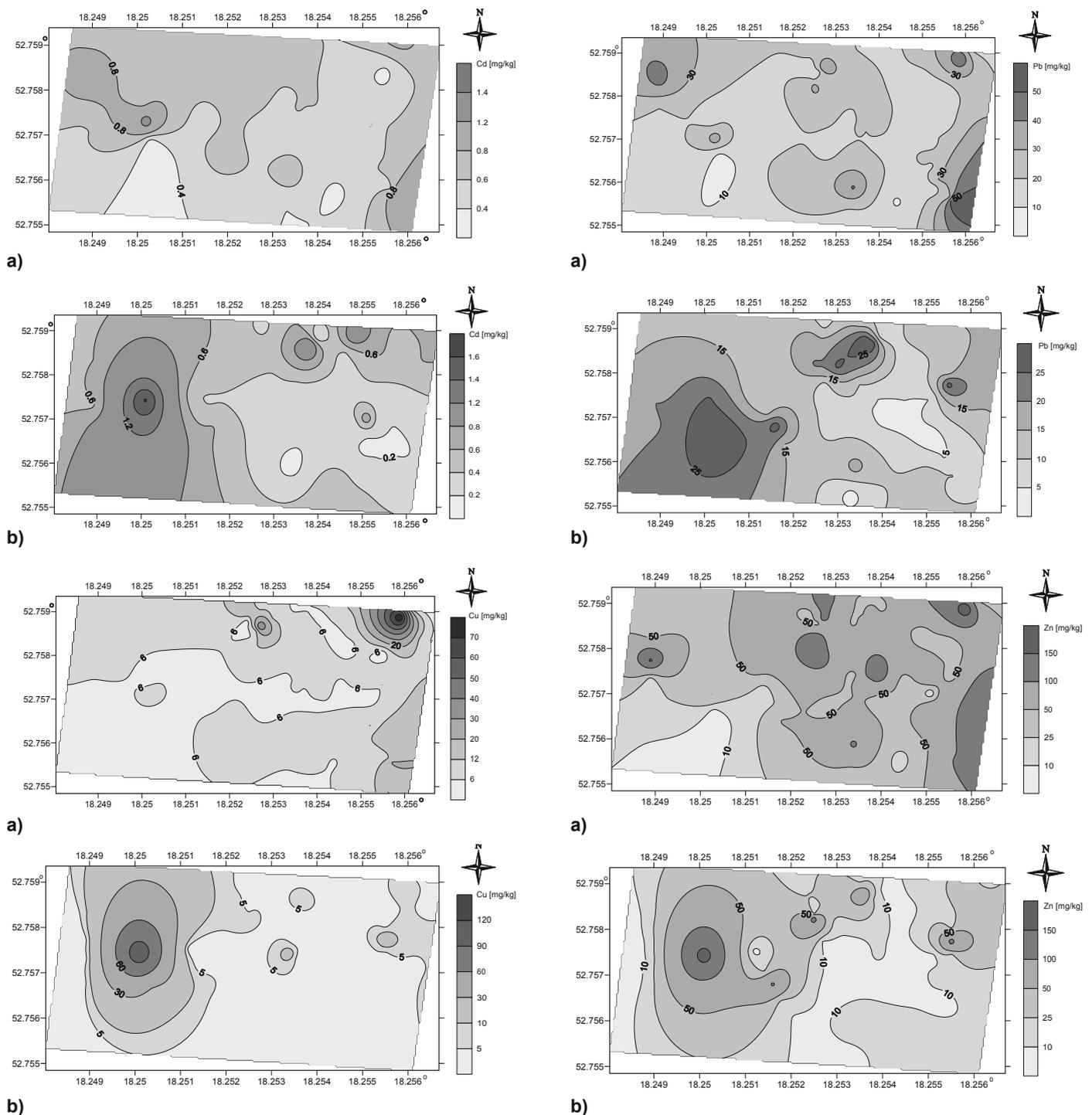| Metal	Depth	0 – class	1 – class	2 – class	3 – class	4 – class	Mean value
		Percentage of samples in given class					
Cd	0–20 cm	3.1	64.1	31.3	1.6	0	1.31
Cu		68.8	26.5	1.6	1.6	1.5	0.41
Pb		79.7	18.8	1.6	0	0	0.22
Zn		51.6	37.5	10.9	0	0	0.59
Cd	80–100 cm	40.0	43.3	15.0	1.7	0	0.78
Cu		40.0	43.3	15.0	1.7	0	0.17
Pb		91.7	8.3	0	0	0	0.08
Zn		90.0	8.3	1.7	0	0	0.12

Fig. 6. Maps of the spatial variability of heavy metals in a). 0–20 cm layer and b). 80–100 cm layer

The correlation analysis (R^2 at $p < 0.05$) revealed that heavy metals content, except Pb, in deeper layer (80–100 cm) was positively and significantly correlated with soil salinity: Cd (0.36), Zn (0.38) and Cu (0.59). Nonetheless, when both surface and subsurface samples were included in the correlation analysis, a statistically significant relationship was found only for Cd (0.19) and Cu (0.42). This confirms a previous assumption that waste ponds influence Cu and Cd content by percolation of precipitation, which transports soluble contaminates to saline groundwater. Our results are in agreement with earlier reports of Du Laing et al. (2008)

and Acosta et al. (2011), who described different behaviours of heavy metals when an increase of salinity takes place. The complexation process of salt derived anions with Cu and Cd increase their mobility while, Pb^{2+} undergo precipitation. This could explain the lack of correlation between soil salinity level and Pb content.

Enzymatic activity

The dehydrogenases activity in the area studied varied between 0.05–2.93 TFFg^{-1} d.m. h^{-1} (Tab. 2) and were in a similar range to data presented by Telesiński (2012) for Mollic Gleysols of
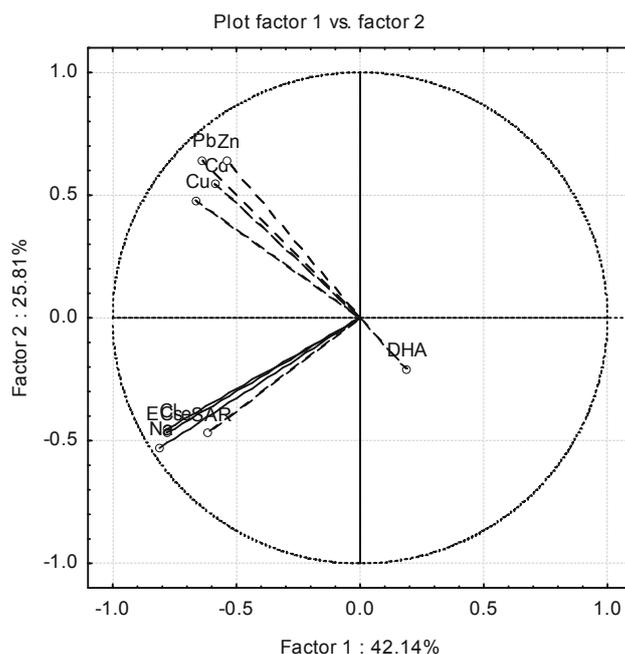
Fig. 7. Graph of a principal component analysis

the Gumienicka Plain. In contradiction to data presented by Telesiński (2012), Guangmining et al. (2017) and Zheng et al. (2017) the correlation and the principal component analyses (PCA) of this research showed that activity of this enzyme was not influenced by soil salinity (Fig. 7). The parameters used for soil salinity evaluation were not correlated with DHA activity (at $p < 0.05$), R^2 values for Cl^-, ECse SAR were -0.10, 0.00 and -0.06, respectively. This could be caused, on the one hand, by low values of ECse noted during the period studied, which were in most cases typical for non saline and slightly saline soils. On the other hand, the sensitivity of soil enzymes varies and dehydrogenases are more resistant to soil salinity (Gracia and Hernandez 1996). Additionally, the microbial communities have the ability to adapt both physiologically and evolutionarily to the presence of contaminants in soil. After the long period of pressure, there is an increase in enzymes activity as a result of acclimated microbial communities, which were capable to adapt and proliferate in the presence of stress factors (Markiewicz et al. 2016). Also Singh (2015) points out that salt-tolerant microbes produce enzymes, which are more resistant to higher salt concentration than corresponding enzymes produced by non-tolerant microbes. Nonetheless, it was observed that the activity of dehydrogenases was inhibited by heavy metals. The strongest inhibition effect was noted by Zn content and the least one by Cu.

Conclusions

The study conducted in the vicinity of the soda plant in Mątwy revealed that the salinity of this area is caused by shallow saline groundwater. As a consequence, moderate to strong salinity is observed in the subsurface layer, which decreases towards the surface. A high concentration of Cl ions created favourable conditions for heavy metal mobility, especially for Cu, Zn and Cd. This led to an emergence of spots with high

salinity and heavy metals content. The spatial distribution of heavy metals in the surface layer suggests that other sources (probably atmospheric deposition) of these elements other than the waste ponds of the soda plant were responsible for increasing their content in soil. According to the results, soil salinity in this region did not inhibit dehydrogenases activity. However, completely different results may be obtained during the summer period.

On the basis of these results, it was established that during the winter period soil quality in the surface layer was not affected by the activity of the soda plant. However, in the case of deeper layers the impact of saline groundwater led to the deterioration of soil quality.

The soil salinisation in the vicinity of the soda plant is reversible and human-induced. Infiltrating water from the waste ponds raises the level of groundwater and increases the quantity of soluble contaminants. Lowering the groundwater table by land amelioration would eliminate the salinity problem of this area.

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