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Statistical modeling of metal migration in soils of oil and gas condensate fields in the north of Western Siberia

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Abstract

The article examines the mechanisms of metal migration and immobilization in the soils of the Western Siberian Subarctic resulting from contamination associated with hydrocarbon development. The research is based on the analysis of 106 soil and soil solution samples for metal content (Na, Ca, K, Fe, Mn, Cd, Cr, Zn, Sr, Cu, Ni, Ba, Pb, Co, V) and petroleum hydrocarbons in the soil, collected at oil fields and background sites in the Nadym-Pur-Taz interfluve. Statistical modeling based on geochemical data was conducted using multivariate statistical methods. The first stage involved data factorization and factor interpretation, identifying drilling cuttings, formation waters, and motor vehicles as primary sources of pollution. Discriminant analysis enabled the grouping of samples based on the nature of the impact. Multivariate analysis of variance made it possible to identify the key factors governing metal mobility potential in soils. These factors were subsequently used as predictors in multiple regression analysis (MRA). MRA revealed the primary mechanisms of metal mobilization into solution: competitive sorption (Mn, Sr, Co, Pb, Cr, Ca, Ba, Zn), solution ionic strength (Na, Mn, Zn), soil alkalinization (Cr, V), and hydrophobization by petroleum hydrocarbons (Sr, Zn). It has been established that immobilization is most effective during ion-exchange sorption processes involving amorphous oxyhydroxides of Fe (Na, Ca, Mn, Sr, Zn) and Mn (Ba, Co, Cr, Pb), as well as organic matter of the humic series (Sr, Ba). A detailed characterization of metal behavior under contamination conditions is provided. It has been shown that the lateral migration of metals under the studied conditions ranges from 150 to 200 m. The reliability of multiple regression models was assessed, and internal validation was performed using experimental data. Based on an analysis of metal mobilization and immobilization mechanisms, recommendations are provided for reducing the migration potential of pollutants.

Keywords

drilling cuttings; formation waters; competitive metal sorption; soil hydrophobization; metal mobilization; metal immobilization

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Introduction

The problem of pollution in northern territories resulting from oil and gas field development is among the most critical issues, given the high vulnerability of Arctic natural-territorial complexes. The northern part of Western Siberia is no exception. Hydrocarbon extraction has been underway in the Yamalo-Nenets Autonomous Okrug (YaNAO) for over half a century. It accounts for 98 % of the Okrug's total industrial production. The state balance of mineral reserves for the region lists 60 hydrocarbon fields currently under development and 36 under exploration*. Extraction operations on this scale have a significant impact on the environment. This is particularly true regarding well construction, which generates large volumes of drilling waste and leads to landscape contamination when requirements for waste management and the remediation of sludge pits are violated. Instances where drilling waste is dumped directly onto the soil – resulting in contamination by petroleum products

* Report "On the State and Protection of the Environment in the Nenets Autonomous Okrug in 2024". Naryan-Mar, 2025. 132 p.

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and metals – also occur [1]. The extent of contamination depends on the migration potential of pollutants, while the effectiveness of environmental protection measures depends on an understanding of the mechanisms governing their mobilization and immobilization – mechanisms that are studied using mathematical modeling [2-4].

To date, extensive research has been conducted on modeling the migration of petroleum hydrocarbons and their individual fractions in soils [5, 6]. In contrast models of heavy metal mobilization during oil and gas field development remain scarce [7, 8], however, metal contamination is a significant issue associated with the disposal of drilling cuttings and the discharge of formation waters onto the land surface, particularly in subarctic conditions. Metals readily undergo lateral migration as constituents of soil solutions and groundwater within the seasonally thawed layer (STL) [1].

Drill cuttings constitute a complex mixture of drilling fluids and drilled rock, containing solid substances as well as inorganic and organic compounds dissolved or dispersed in the aqueous phase [9]. Depending on the geological structure and the composition of the solutions, they may contain elevated concentrations of various metals. Key indicators of contamination include Na, K, Ba, Sr, Ca, V, Ni, Cr, Zn, Mn, and Pb [9-11]. These elements exhibit a wide range of properties that determine their high mobility in Arctic landscapes.

The studies were conducted within licensed areas where exploratory or production drilling had taken place, resulting in the discharge of formation waters and spills of drilling cuttings. The fieldwork was carried out between late July and the first half of August. These areas had undergone partial remediation through the application of sand-based fill. The aim of the research is to analyze the migration activity of metals using statistical modeling. To achieve this, the objectives were formulated: to group soil samples from the overall dataset based on exposure to different sources of impact; to identify metals that serve as pollution indicators and assess their potential for lateral migration within landscapes; and to develop proposals for reducing metal lability by examining the mechanisms of metal mobilization and immobilization.

Materials and methods

Studies were conducted in the YaNAO at three sampling sites (Yamburgsky, Tazovsky, and Purovsky) located within the licensed areas of oil, gas, and condensate fields (Fig.1). An additional site (Novourengoysky) was selected to assess traffic-related impact; it is situated along the Surgut – Salekhard highway and includes part of the Novy Urengoy urban area.

The study sites are located within the Nadym-Pur-Taz interfluve, which is characterized by a continuous cover of Quaternary deposits, tens of meters thick, resting on the eroded surface of Paleogene continental rocks. The upper part of the stratigraphic section comprises Upper Neopleistocene and Holocene deposits of marine, nearshore-marine, glaciomarine, lacustrine-alluvial, and alluvial origin. Soil-forming parent materials are of particular interest in the context of the research; they are represented by several facies [12]. In the northern part of the study area, the surface deposits consist of Upper Neopleistocene alluvial-marine clays of the fourth marine terrace (Tazovsky site) and light and heavy loams of the second and third marine terraces (Yamburgsky site). Further south, the rock types give way to lacustrine-alluvial light and medium loams (Purovsky site), as well as glaciolacustrine deposits (western part of the Novourengoysky study site) and alluvial deposits of the second supra-floodplain terrace of Upper Neopleistocene age, composed of sands, light loams, and peat (eastern part of the study area). The terrain of the study areas consists of gently undulating plains. Permafrost-related processes give rise to various mesorelief forms, with a widespread occurrence of thermokarst features. The dominant soil types in the study area are oligotrophic peat soils (histosols), peaty-gley soils (gleysols), podzols, and iron-illuvial podburs (cryosols).

Within the study sites, paired samples of soil and soil solution were collected in accordance with the Russian national standard GOST 17.4.3.01-2017. Samples were collected from two genetic soil horizons – the organogenic (0-25 cm) and the illuvial (25-50 cm) – at background sites and near oil and gas extraction facilities. Sampling was conducted along transects at 50 m intervals up to a total length of 300 m, while background samples were collected as point samples at a distance of more than 3 km from local pollution sources. A detailed rationale for the selection of sampling points is presented in previous studies [13, 14]. The total number of analyzed samples is 106.

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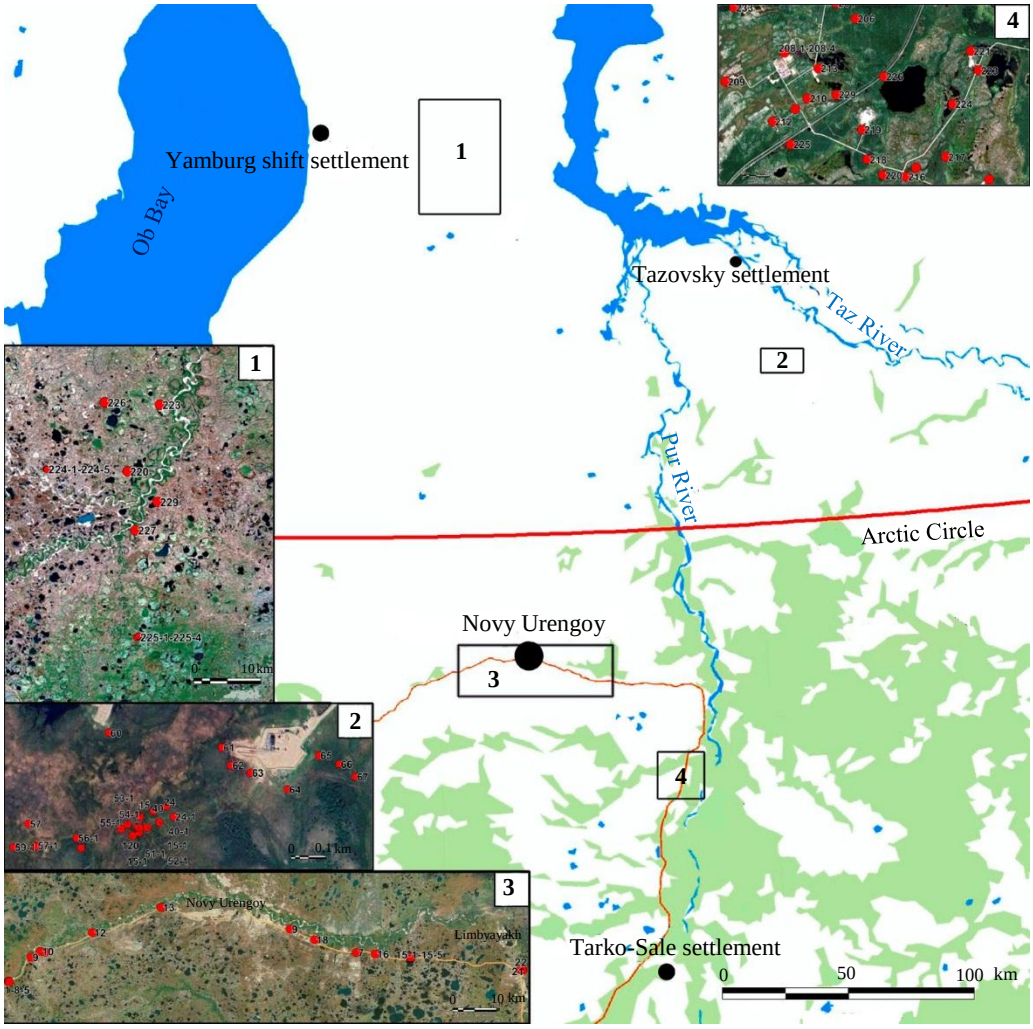


Fig.1. Location map of the sampling sites

1 – Yamburgsky; 2 – Tazovsky; 3 – Novourengoysky; 4 – Purovsky

Soil solutions from organic horizons were extracted using a PVR-1 mechanical press and then centrifuged in an SM-6M centrifuge at 3500 rpm for 30 min directly in the field laboratory. This procedure removes particles larger than 0.45-0.60 μm from the solution [15]. Soil solutions were collected from illuvial (mineral) horizons and subsequently centrifuged. After separation, the solutions were filtered through quantitative Blue Ribbon filter paper and acidified with HNO_3 (1 ml per liter of solution).

The metal concentrations (Na, Ca, K, Fe, Mn, Cd, Cr, Zn, Sr, Cu, Ni, Ba, Pb, Co, V) in solutions and soils were determined at the accredited Central Laboratory of the Karpinsky Institute using the ICP-MS method on an ELAN-6100 DRC instrument (USA). Soil samples were subjected to complete acid digestion in accordance with the Russian environmental regulatory document PND F 6.1:2.3:3.11-98.

The concentrations of petroleum hydrocarbons (PHC) in soils were analyzed at the accredited laboratory of NPO "Typhoon" using two methods – infrared spectrometry (PHC_{IR}) and fluorimetry (PHC_{FL}). Sample preparation for the analysis of total non-polar hydrocarbon concentrations in soils was carried out according to PND F 16.1:2.22-98 by subjecting lyophilized soil samples to triple ultrasonic extraction with CCl_4. The concentration of non-polar hydrocarbons was determined in accordance with PND F 16.1:2.21-98; lyophilized soil samples were extracted three times with 10 ml of hexane using ultrasonic treatment. A Fluorat-02 liquid analyzer was used for quantitative analysis. The methods of analysis and data processing are described in greater detail in a previous study [13]. The IR method detects a wide range of hydrocarbons, including those of natural origin. Fluorimetric analysis yields the total concentration of aromatic hydrocarbons, the presence of which in environmental media is primarily associated with anthropogenic activity. Solution electrical conductivity was determined using a conductometric method (COM-100), and pH was measured with a LAQUAtwin pH-22 meter (Horiba).

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Statistical modeling was based on multivariate statistical methods. The model was developed using data on the chemical composition of soils and solutions. Initially, a factor analysis (FA) of metal concentrations in soil solutions was performed using the maximum likelihood method to identify geochemical associations in solutions from both background areas and sites subject to contamination.

The next stage of modeling employed discriminant analysis to develop a predictive model for classifying objects into mutually exclusive groups: background areas and sites subject to contamination from various anthropogenic sources. This is particularly important for samples with low factor values, which otherwise cannot be reliably linked to specific sources of impact.

Next, multivariate analysis of variance (MANOVA) was applied, with contaminated soils serving as the study objects. The objective was to determine the significance of natural and anthropogenic factors in shaping soil water composition, thereby enabling the informed selection of predictors for the subsequent modeling stage. A key measure of effect size in MANOVA is partial eta-squared (η^2), which represents the proportion of the dependent variable's variance explained by a specific factor when the influence of all other factors is excluded. The final stage involved multiple regression analysis to formulate a model of metal migration activity and to identify the mechanisms governing their mobilization and immobilization. The metal mobility potential served as the quantitative characteristic for the variables; this was calculated as the percentage of dissolved metal forms relative to the total metal concentration in the soil, using the formula

$$P = \frac{100 C_W}{C_S},$$

where C_W – metal concentration in the solution, mg/l; C_S – metal concentration in soil, mg/kg.

Statistical processing of geochemical data and modeling were performed using the IBM SPSS Statistics 26 software package. Metal concentrations in soil solutions do not follow a normal distribution. All variables exhibited a log-normal distribution, as determined by the Kolmogorov – Smirnov test applied to the logarithms of their concentrations (significance level $p > 0.05$). Given this circumstance, the variables underwent logarithmic transformation for the multivariate statistical analysis. The most robust metrics were used to interpret the results. Descriptive statistics employed non-parametric methods or calculated the median concentration.

The resulting model is linear with respect to the parameters but describes nonlinear (power-law) relationships between the initial variables. The validity of the multiple regression models was assessed using the Fisher F-test. A model was considered statistically significant at the $p = 0.05$ level if the calculated value F_{calc} exceeded the tabulated (critical) value F_{crit} (k, $n - k - 1$). The Fisher test statistic was calculated using the formula

$$F = \frac{R^2}{1 - R^2} \frac{n - k - 1}{k},$$

where R^2 – coefficient of determination; n – sample size; k – number of predictors in the model.

Additionally, the significance of each predictor was assessed using Student's t-test, the value of which was characterized by the adjusted coefficient of determination,

$$t = \frac{B_i}{SE_{B_i}},$$

where B_i – unstandardized coefficient; SE_{B_i} – its standard error. The obtained values of the criteria were compared with the tabulated ones, at $t_{calc} > t_{crit}$ ($n - k - 1$) the predictor made a significant contribution to the model.

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Discussion of results

The results of the chemical analysis, including the median concentrations of metals and petroleum hydrocarbons in soils and soil solutions across the studied sites, are summarized in Table 1.

Table 1

Medians of chemical substance concentrations in paired samples of soils (mg/kg) and soil solutions (μg/l) at the study sites

Substances	Study sites							
	Tazovsky (*n* = 35)		Novourengoysky (*n* = 26)		Purovsky (*n* = 16)		Yamburgsky (*n* = 29)	
	Soil	Solution	Soil	Solution	Soil	Solution	Soil	Solution
Na	1650	1850	1500	1270	3950	18,750	7400	8680
K	2650	800	3100	2155	6600	4815	14,200	2410
Ca	5700	5900	2600	1585	4050	1520	7500	6840
V	21.5	3.41	22.6	2.92	40.2	4.64	30	3.08
Cr	18.5	4.46	20.5	2.90	28.1	4.52	24.6	4.51
Mn	155	132	72.4	72.5	132	92.2	178	173
Fe	7280	1980	9650	2820	15,300	1810	15,100	3610
Co	5.61	3.34	4.17	1.71	4.48	2.46	6.72	7.19
Ni	12.9	6.64	9.3	2.90	12.4	8.31	10.4	11.7
Cu	11.3	7.66	7.58	3.04	9.2	12.7	6.71	8.2
Zn	23.6	47	20.4	161	32.7	40.4	26.6	50.9
Sr	46.7	34.1	34	13.4	73.6	131	120	265
Cd	0.15	0.34	0.16	0.35	0.14	0.36	0.091	0.58
Ba	122	33	194	33	337	102	535	182
Pb	7.15	1.98	8.64	1.56	11.6	2.27	9.24	2.78
PHC_{IR}	428	–	220	–	1550	–	1700	–
PHC_{FL}	27	–	24	–	410	–	650	–

Note. A dash indicates a lack of data.

Factor analysis performed on the total sample of soil solutions allowed us to clarify the nature and extent of soil contamination (Table 2). The first component, accounting for 23.6 % of the sample variance, reflects the influence of parent rocks, represented by alluvial-marine deposits of the third and fourth marine terraces, on the composition of the solutions. The paragenesis combines metals of marine facies (Ca, Co, Ni, Mn, Sr), as shown in [14]. The second factor, accounting for 21.0 % of the variance, reflects soil and solution contamination by drilling cuttings. The third component (explaining 16.3 % of the variance) characterizes soil contamination by formation waters, indicated by the Cr-Pb-V-Cu metal association. The fourth factor (accounting for 7.8 %) reflects the composition of peatland soil waters, which are distinguished by high zinc and potassium contents. Additionally, high factor loadings were characteristic of samples collected within the zone of vehicular impact along the Surgut – Salekhard highway.

Factor analysis results for the soil solutions from contaminated sites indicated that the identified types of impact were the primary drivers of metal dispersion. Factor I reflects soil contamination by drilling cuttings and muds deposited on adjacent areas following the completion of drilling operations; this involves contamination by metals present in the drilling and flushing fluids as well as in the drilled rock cuttings. The composition of the sludge indicates the presence of Na^+ and Cl^- (derived from the technical-grade salt used in drilling fluids when drilling through permafrost) as well as trace elements such as Ba, Sr, Ca, Pb, and Fe (components of weighting agents and chemical additives used to adjust mud density and viscosity). Compounds of Zn, Cr, Cu, and Mn are also used to impart specific properties to the drilling muds, while K, Na, and Ca are included in the fluid formulations. High chloride levels drive an increase in the mineralization of the waste's liquid phase. This factor reaches peak values in areas where waste was discharged directly onto the land surface, a phenomenon documented during field surveys.

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Table 2

Parageneses of chemical elements in soil solutions based on generalized factor loadings (rotation method – varimax with Kaiser normalization)

Factor	Factor name	Parageneses	Proportion of sample variance, %
Soil solutions from background and contaminated sites – pooled sample (n = 106)			
I	Rock-derived	$Co_{84}Ni_{74}Mn_{74}Ca_{73}Sr_{63}Ba_{55}Cd_{52}Fe_{48}$	23.6
II	Technogenic (drill cuttings)	$Na_{90}TDS_{82}Ba_{70}Sr_{66}K_{58}$	21.0
III	Technogenic (formation water discharge)	$Cr_{87}Pb_{79}V_{75}Cu_{64}$	16.3
IV	Organogenic (acidification)	$\frac{Zn_{91}K_{49}}{pH_{30}}$	7.8
Solutions of contaminated soils (n = 57)			
I	Drill cuttings	$TDS_{90}Sr_{89}Na_{88}Co_{87}Ba_{83}Mn_{76}Ni_{72}Ca_{69}$	34.6
II	Formation water discharge	$Pb_{88}Cr_{82}V_{80}Cu_{68}$	19.2
III	Transport	$\frac{Fe_{65}Zn_{48}}{pH_{45}}$	8.4

Note. Factor loadings in the formulas are multiplied by 100.

Factor II (Cr-Pb-V-Cu) reflects the influence of formation waters. According to current understanding [16, 17], the concentrations of these metals in formation waters from hydrocarbon fields (sodium-calcium chloride brines) reach $0,n – n \cdot 10^2$ mg/l. The enrichment of formation waters in V and Cu is attributed to biogeochemical processes, whereas enrichment in Cr and Pb results from the leaching of metals from the host rocks [18]. Maximum values for this component were recorded in areas where formation waters were discharged onto the soil – a practice associated with well testing and reservoir flooding operations, as observed during fieldwork.

Factor III, which reflects the impact of transport infrastructure on soils and soil solutions, is expressed by the Fe-Zn association. Soil pH has a negative loading on factor III. This is reflected in the low pH values of solutions in contaminated areas, which are partly caused by the acidifying effect of vehicle emissions. The issue of soil pollution from traffic in the Arctic is discussed in previous work [14]. The highest values of this factor were observed in the immediate vicinity of the Surgut – Salekhard highway, in sections adjacent to Novy Urengoy. It is significant that all three types of pollution can occur simultaneously at the study sites; this was taken into account when interpreting the results.

Discriminant analysis (using stepwise selection based on Mahalanobis distance) was performed to classify soil solution samples into four distinct groups, defined by the categorical grouping variable: contamination by drill cuttings, formation waters, and transport-related sources, as well as conditionally background samples. Metals were considered as independent variables. The discriminating variables identified in the analysis were Na, V, Zn, Ca, Sr, and Pb, all of which exhibited high loadings within the associations characterizing the groups. Among the predictors, Na, V, Zn, and Pb made the greatest contribution to intergroup differences. The analysis results show that 92.8 % of the 106 original observations were correctly classified, reflecting a high degree of geochemical distinctness among the groups.

The first unstandardized canonical discriminant function (Table 3), which explains 69 % of the sample variance, reflects the gradient of soil contamination by metals; its positive values are associated with the impact of drilling waste, while negative values correspond to background conditions. Na and Ca exhibit the highest loadings on this function, whereas Sr loads most heavily on the background component. The second function (accounting for 22.9 % of the variance) groups characteristics of solutions contaminated by formation waters; Pb, Sr, and V contribute the primary loadings to this function. The third function (accounting for 8.2 % of the variance) reflects the impact of vehicular traffic, with Zn showing the highest loading.

Based on the structure matrix – which indicates the strength of the relationships between the discriminant variables and the discriminant function values – indicator metals for specific types of

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pollution were identified, while elements with weak associations (Cd, Ni, Cu, Fe, K) were excluded from further analysis. Only solution samples with chemical compositions altered by hydrocarbon extraction were included in the subsequent modeling process (Table 4).

Table 3

Main indicators of standardized canonical discriminant functions

Indicators	Functions		
	1	2	3
Functions in group centroids			
Drill cuttings	3.156	–0.592	–0.38
Formation water	0.781	2.038	0.929
Transport	–1.305	–1.815	1.02
Background	–1.742	0.275	–0.571
Standardized canonical discriminant function coefficients			
Na	1.351	–0.026	0.309
Ca	1.09	–0.414	–0.719
V	–0.565	0.443	0.326
Zn	–0.494	–0.841	0.809
Sr	–1.076	0.503	0.258
Pb	0.441	0.635	–0.183

Table 4

Statistical indicators of metal content in soil solutions within the zone of impact of the oil and gas extraction industry

Indicators	Sources of pollution					
	Drill cuttings (n = 19)		Formation water (n = 16)		Transport (n = 22)	
	Median	Min-max	Median	Min-max	Median	Min-max
Na, mg/l	83.2	10.3-973.5	33.8	1.57-525.2	3.29	0.724-119.4
Ca, mg/l	39.1	2.73-102.8	2.52	0.478-25.5	2.75	0.395-49.1
V, µg/l	1.38	0.25-10.9	9.9	3.1-98	2.8	0.22-19.5
Cr, µg/l	2.35	1.03-25.5	9.8	3.2-74	3.8	0.72-20.1
Mn, µg/l	1250	32.2-9220	139	19.6-717	165	12.4-1890
Co, µg/l	17	1.85-439	5.8	1.5-14,4	2.3	0.93-68
Zn, µg/l	53.5	17.8-881	50.8	11-114	235	15.8-1050
Sr, µg/l	376	16.4-4750	88.4	11.3-4600	22.4	8.0-481
Ba, µg/l	609	89.3-8390	116	28.5-5070	63	14.1-1590
Pb, µg/l	2.04	0.28-17.6	6.5	2.6-32.1	1.8	0.32-5.5
pH	4.94	4.27-6.80	4.90	3.90-8.30	4.6	3.80-6.90
Total dissolved solids, mg/l	476	28-6290	156	26-2020	40	19-734

A MANOVA was performed to model the migration capacity of metals. The objective was to determine the magnitude of the effect of leading natural and technogenic factors on metal dispersion in solutions. Natural factors included the facies factor (the influence of parent rock) and soil type, while anthropogenic factors comprised the identified types of impact and the ionic strength (total dissolved solids, TDS) of the solution. An exploratory analysis of the influence of these factors on metal dispersion in solutions yielded the following results (Fig.2). Effect size (Pillai's trace): facies type – 0.55; soil type – 0.27 (statistically non-significant; p = 0.118); type of contamination – 0.66; solution ionic strength – 0.47. Thus, under the conditions evaluated, the impact of the oil and gas industry becomes the dominant factor driving metal dispersion in soil solutions (whereas under

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background conditions, the primary influence is the composition of soil-forming rocks – i.e., facies type). The η^2 values for the metals fall within the range of 0.39-0.83, indicating the strong explanatory power of the model. Low effect sizes were observed for Pb (0.39) and Zn (0.50), while all other values exceeded 0.6. A statistically significant η^2 value for the pollution factor is reflected in the variance of all metals, with the exception of Co, Zn, and Ba. The effect of solution ionic strength is also significant for metals other than Ca, Cr, and Pb. The facies factor influences the variability of Ca, V, Mn, Co, Sr, and Ba concentrations in the solutions. The effect of soil types on metal dispersion is insignificant. Further model development will consider both technogenic factors and natural factors modified by industrial development.

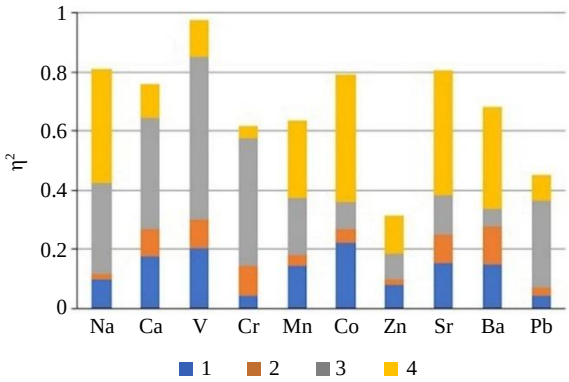


Fig.2. Intergroup effects of factors influencing metal distribution in soil solutions of disturbed sites

1 – facies; 2 – soil type;
3 – pollution source; 4 – TDS

The final stage of the modeling process involved the use of multiple regression analysis (MRA) to assess metal mobility. The objective was to identify the primary mechanisms governing metal mobilization and immobilization, as well as the potential for lateral migration. Mobility potential values (Table 5) were used to characterize the metals. The mobility series is as follows: Na > Zn > Mn > Sr > Ca > Co > Ba > Pb > Cr > V. Indicator metals of formation waters are characterized by minimal migration capacity, reflecting their immobile behavior under acidic environmental conditions. Overall, the mobility of the elements depends on the source of contamination. ANOVA revealed statistically significant differences in metal mobility associated with various pollution sources. Increased mobility was observed for metals serving as indicators of drill cuttings (Na, Ca, Mn, Co, Sr, Ba), formation waters (V, Pb), and transport-related pollution (Zn) compared to levels associated with other sources.

Table 5

Statistical indicators of metal mobility potential in soils of contaminated sites

Sources of pollution	Statistics	Na	Ca	V	Cr	Mn	Co	Zn	Sr	Ba	Pb
Drill cuttings (n = 19)	Median	1.531	0.44	0.003	0.009	0.445	0.157	0.189	0.287	0.097	0.016
	Minimum	0.117	0.053	0.000	0.002	0.007	0.015	0.045	0.041	0.016	0.001
	Maximum	10.25	1.31	0.04	0.12	7.47	9.34	2.06	3.47	1.43	0.08
Formation water (n = 16)	Median	0.328	0.043	0.039	0.043	0.049	0.087	0.171	0.108	0.043	0.051
	Minimum	0.011	0.005	0.004	0.009	0.008	0.02	0.041	0.006	0.003	0.012
	Maximum	8.35	0.58	0.32	0.31	0.36	0.72	0.37	1.22	0.55	0.26
Transport (n = 22)	Median	0.138	0.087	0.015	0.023	0.165	0.067	1.324	0.062	0.035	0.019
	Minimum	0.022	0.022	0.000	0.002	0.011	0.01	0.056	0.015	0.006	0.000
	Maximum	1.68	0.22	0.08	0.39	1.27	1.06	4.63	0.48	0.26	0.07

In the MRA, the magnitude of the mobility potential of pollution-indicator metals was treated as the dependent variable. The concentrations of Fe, Mn, and Ca in solutions (Fe_{sol}, Mn_{sol}, Ca_{sol}) and soils (Fe_s, Mn_s, Ca_s), as well as the content of organic matter (OM) in soils – representing the primary metal-bearing phases – were selected as predictors. Based on the MANOVA results, the predictors included substances and parameters whose activity is primarily driven by pollution: solution pH and TDS, as well as the concentration of chlorides (Cl^-), PHC_{IR}, and PHC_{FL}. To avoid multicollinearity when constructing models of calcium and manganese migration, the Ca and Mn concentrations in the solution were excluded from the set of predictors, respectively. The model is characterized by the parameters: the coefficient of determination R^2, representing the proportion of variance in metal

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mobility explained by the model; standardized coefficients β, reflecting the relative influence of the predictors included in the model on migration potential; and the regression equation (Table 6). The conditions for metal mobilization and immobilization were interpreted based on indicator metals.

Table 6

Model of metal migration in soils contaminated with drill cuttings and formation waters based on MRA results

Dependent variable	Model (n = 43)	Adjusted R^2	Standardized coefficient β	Regression equation
	Predictors: pH, TDS, Cl^-, OM, PHC_{IR}, PHC_{FL}, Fe_{sol}, Mn_{sol}, Ca_{sol}, Fe_s, Mn_s, Ca_s			
Na	TDS Fe_s	0.622	0.807 –0.327	$Na = 1.107 + 1.101\ TDS - 0.951\ Fe_s$
Ca	Mn_{sol} Fe_{sol}	0.518	0.791 –0.268	$Ca = -1.884 + 0.656\ Mn_{sol} - 0.179\ Fe_{sol}$
V	Fe_{sol} Ca_{sol} pH PHC_{IR}*	0.441	0.359 –0.458 0.435 0.185	$V = -4.567 + 0.164\ PHC_{IR} + 4.053\ pH - 0.491\ Ca_{sol} + 0.291\ Fe_{sol}$
Cr	Mn_s Fe_{sol} pH PHC_{IR}*	0.555	–0.522 0.283 0.271 0.247	$Cr = -1.782 - 1.100\ Mn_s + 0.177\ Fe_{sol} + 1.998\ pH + 0.172\ PHC_{IR}$
Mn	Cl^- Ca_{sol} Fe_{sol} Fe_s*	0.619	0.386 0.498 0.345 –0.211	$Mn = -2.006 + 0.298\ Cl^- + 0.51\ Ca_{sol} + 0.263\ Fe_{sol} - 0.548\ Fe_s$
Co	Mn_{sol} Mn_s Fe_{sol} OM*	0.650	0.679 –0.430 0.250 –0.202	$Co = -0.277 + 0.567\ Mn_{sol} - 0.971\ Mn_s + 0.168\ Fe_{sol} - 0.154\ OM$
Zn	PHC_{FL} Cl^- Fe_s OM	0.533	0.357 0.534 –0.268 0.281	$Zn = 0.132 + 0.152\ PHC_{FL} + 0.300\ Cl^- - 0.505\ Fe_s + 0.177\ OM$
Sr	Ca_{sol} Ca_s Fe_{sol} OM Fe_s PHC_{FL}	0.801	0.928 –0.337 0.202 –0.408 –0.283 0.229	$Sr = 1.813 + 0.842\ Ca_{sol} - 0.837\ Ca_s + 0.137\ Fe_{sol} - 0.313\ OM - 0.650\ Fe_s + 0.119\ PHC_{FL}$
Ba	Mn_{sol} Mn_s OM	0.546	0.740 –0.512 –0.300	$Ba = 0.433 + 0.605\ Mn_{sol} - 1.113\ Mn_s - 0.224\ OM$
Pb	PHC_{IR}* Mn_s pH Fe_{sol}	0.439	0.264 –0.401 0.323 0.278	$Pb = -2.434 + 0.176\ PHC_{IR} - 0.813\ Mn_s + 2.286\ pH + 0.167\ Fe_{sol}$

* The effect of the predictor on the variable is statistically insignificant.

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Assessment of model significance (F-test) confirmed their statistical validity (R^2, $p < 0.01$). Student's t-test was used to identify predictors that do not significantly influence the dependent variable (Table 6). Additionally, model adequacy was confirmed by a paired t-test comparing observed and predicted values, which demonstrated the absence of systematic forecast bias.

Among the metals under consideration, sodium is the most mobile element; its mobility potential is 2.8 % in the presence of drill cuttings contamination and 1.9 % in the presence of formation water contamination. The maximum potential value exceeds 10 %. The use of industrial-grade salt drives the active migration of the metal in the form of chlorides [10, 19]. Based on the results of the MRA, the following predictors were included in the model: TDS and soil Fe content. It should be noted that TDS is primarily driven by high chloride concentrations in the solution (up to 9500 mg/l), as evidenced by a strong positive correlation between them ($p < 0.001$). Thus, sodium mobility is driven by Na^+ migration. The negative correlation between metal mobility and soil Fe content in the regression equation suggests the occurrence of ion-exchange sorption on iron oxyhydroxides, involving the displacement of Al^{3+} and H^+ ions under acidic conditions. This is likely one of the most effective mechanisms for removing sodium from the soil solution and sequestering it within the soil adsorption complex. The R^2 value indicates that these two mechanisms account for more than 62 % of the variance in the metal mobility potential.

According to modeling results, calcium mobility is primarily regulated by the Mn^{2+} and Fe^{2+} concentration in the solution. However, these processes operate in opposite directions: an increase in Mn^{2+} concentration promotes calcium solubility, whereas Fe^{2+} reduces it. These mechanisms can be explained by the behavior of metals in the acidic environment of peat soils; Mn^{2+} can displace Ca^{2+} from exchange sites [20, 21], releasing it into the solution through the competitive sorption characteristic of cations [22, 23]. Given that calcium has a higher binding energy, this is possible given a significant increase in the Mn^{2+} concentration in the solution – a phenomenon observed in the study area. For instance, the Ca^{2+}/Mn^{2+} cation ratio is 75.0 in background soils but drops to 25.0 in contaminated soils, reflecting a relative increase in Mn^{2+} content.

Under similar conditions, at Ca-Fe interfaces in an oxidizing environment or at the boundary of reducing zones, ferrous iron from drilling waste or peat oxidizes to Fe^{3+} and hydrolyzes into amorphous oxyhydroxides, which possess variable surface charge and act as sorbents for cations, including Ca^{2+} [24]. When soils are alkalized by drill cuttings, calcium can also be incorporated into mixed Ca-Fe phases. Furthermore, Fe^{3+} complexes and colloids act as sorbents for Ca^{2+}, which can bind to the negatively charged sites of iron-organic colloids or become part of forming mixed phases-either directly in solution or during coagulation. Under actual conditions in an acidic peatland contaminated with drilling waste, these processes will occur simultaneously, and their balance will determine the net mobility of the metal. The mechanisms examined account for 52 % of the variance in calcium mobility potential.

The manganese mobility model demonstrated a dependence on the calcium concentration in the solution, the iron content in both the solution and the soil, and the chloride concentration. Soil iron content exhibited a negative standardized coefficient, highlighting the role of iron mineral phases in manganese immobilization, particularly during the anthropogenic alkalinization of solutions. However, this predictor does not have a significant effect, likely due to the localized nature of such conditions. Ca^{2+} cations exhibit the highest β value in the regression equation and exert a corresponding influence on manganese mobility; this inverse relationship is discussed above.

The direct correlation between mobility potentials likely reflects the active interaction of metals in the soil solution [25] via a mechanism of competitive displacement within a multicomponent sorption system [22]. When the manganese concentration in the solution is elevated (resulting in an increased Mn/Ca ratio due to contamination), Mn^{2+} ions actively occupy sorption sites on the surfaces of the soil adsorption complex and Mn/Fe oxides, displacing previously adsorbed Ca^{2+} into the solution [21]. An increase in Ca^{2+} concentration in the solution creates a counteracting competitive effect: Ca^{2+} blocks sorption sites, thereby preventing the readsorption of Mn^{2+}. A state of dynamic equilibrium is established in which both elements exhibit enhanced mobility, and their concentrations in the

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solution show a positive correlation. This phenomenon, known as the mutual stimulation of mobility in multicomponent systems [23], highlights the need to account for competitive interactions when predicting element migration under conditions of technogenic contamination. Another mechanism of manganese mobilization is associated with the formation of metal chlorides.

Strontium and, especially, barium form sparingly soluble sulfates; they exhibit weak complexation with organic matter and can participate in ion-exchange sorption, albeit with low binding energy. Modeling of metal mobility has shown that barium mobility is predicted by manganese concentrations in the solution and soil, as well as by organic matter content; a positive correlation with Mn^{2+} reflects competition for adsorption sites, where Mn displaces barium into the solution, whereas the inverse relationship between barium mobility and soil manganese is determined by the high adsorption potential of manganese oxyhydroxides (MnO_2, MnOOH, etc.). The negative correlation between organic matter and barium mobility indicates that an abundance of OM increases the number of sites available for metal binding [26].

The solubility of strontium compounds increases with rising calcium levels in the solution. Strontium is prone to ion-exchange sorption via a mechanism similar to that of calcium, though with lower binding energy; this involves competitive sorption, resulting in the displacement of strontium into the solution. Based on the standardized coefficient, this factor is a primary determinant of Sr^{2+} migration activity. One mechanism of metal immobilization is the isomorphous substitution of calcium in its minerals ($CaCO_3$, $CaSO_4$) – particularly as pH rises – as evidenced by the negative coefficient associated with soil calcium content. The interaction between strontium and iron is interpreted similarly and is represented in the model by two predictors – iron concentration in the solution and iron content in the soil. In the solution, a positive feedback loop within the sorption system (competitive coupling) may be inferred, whereas in the solid phase, the barrier function of newly formed iron oxyhydroxides plays a role. Like that of barium, strontium's migration potential decreases with high soil organic matter content, a result of ion-exchange interactions with the OM. The energy of the interaction between strontium and OM increases under conditions of rising pH (alkalinization).

Complexation with calcium – which is capable of forming bridging linkages with OM – may be of some significance [27]. At the same time, PHC_{IR} compete for organic matter sorption sites; they occupy these sites and displace strontium into the solution, as evidenced by the positive value of the unstandardized coefficient for PHC in the model under consideration. Studies [28, 29] have shown that the presence of petroleum hydrocarbons leads to soil hydrophobization and reduces the sorption potential of the soil adsorption complex. The relationship between strontium and iron is determined by the forms in which the latter occurs. Soil iron in the form of secondary oxyhydroxides (goethite, ferrihydrite) and Fe-organic complexes acts as a sorption barrier. Iron in solution – predominantly Fe^{2+} or possibly colloidal Fe^{3+} – competes for exchange sites within the soil adsorption complex and on mineral surfaces. A high concentration of Fe^{2+} in solution promotes the active displacement of Sr^{2+} from the solid phase.

Thus, under contaminated conditions, iron and manganese can play a key role in regulating the mobility of alkaline-earth metals (Sr, Ba). Sr mobility is controlled by iron – its content in the soil solid phase acts as an immobilization factor (sorption onto oxyhydroxides), whereas the Fe^{2+} concentration in the solution serves as a mobilization factor (competitive displacement). For Ba, a similar control mechanism operates via manganese: in soils, oxyhydroxides reduce Ba mobility, whereas dissolved manganese species increase it. Soil OM acts as a universal factor reducing the mobility of both metals under conditions of high industrial load, highlighting its buffering role even for elements that do not form stable chelates. The observed patterns reflect sorption/desorption mechanisms and competitive ion exchange, which become significant under contamination conditions. The Sr mobility potential model shows the highest explained variance (0.80), while the coefficient of determination for Ba is lower (0.55).

In acidic peat soils, cobalt tends to undergo complexation and sorption onto mineral and organic surfaces. Cobalt mobility (represented by the β-value) is primarily regulated by manganese concentrations in the soil solution and the soil itself, and to a lesser extent by the iron concentration in the

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solution and OM content; the influence of the latter on the predictor is statistically insignificant. Concentrations of Mn^{2+} and Fe^{2+} in the solution increase cobalt mobility. Here, similar to the Mn-Ca and Sr-Fe pairings, a positive feedback mechanism is evident within the soil sorption system. The negative correlation between cobalt mobility and soil Mn content is attributed to the oxidation of Co^{2+} to Co^{3+} on the surface of manganese oxides, which constitutes one of the primary fixation pathways. Potential interactions between cobalt and soil organic matter manifest as the formation of sparingly soluble complexes with humic acids and the sorption of the metal by organic matter – including inner-sphere sorption [30], a process enhanced by soil alkalinization resulting from drilling cutting contamination. The model accounts for 65 % of the cobalt variance.

In the model, zinc serves as an indicator of the impact of motor vehicle traffic. Its mobility in the presence of drilling waste contamination is primarily due to its occurrence in the form of chlorides. Immobilization occurs through adsorption onto the surface of iron oxyhydroxides. Release into solution is observed during contamination with petroleum hydrocarbons (PHC_{FL}) due to competition for organic sorption sites. This is attributable to the relatively weak bond between zinc and humus, which is largely ion-exchange in nature. The positive effect of OM content on mobility stems from the fact that zinc is the only one of the studied elements whose concentration increases in a statistically significant manner in strongly acidic solutions. Under these conditions, and given a high organic matter content, zinc is readily displaced from sorption sites by H^+ and Al^{3+} ions. The proportion of variance explained by the model is 0.53.

Vanadium, chromium, and lead serve as indicators of contamination by formation waters and exhibit the lowest mobility values; while mobility is positively correlated with PHC_{IR} concentration, the influence of this predictor proved statistically insignificant for all three variables upon testing. Therefore, this predictor is not considered in the characterization of the models. Chromium is a redox-sensitive metal; its mobility is positively correlated with Fe^{2+} content and increasing pH. The results reflect chromium's ability to form stable complexes with the carboxyl groups of humic acids. However, it can be displaced from organic complexes by Fe^{3+} (and Al^{3+}), particularly at low pH. An increase in pH, especially under oxidizing conditions, promotes the oxidation of Cr^{3+} to highly mobile Cr^{6+}, thereby increasing its concentration in the soil solution (redox mechanism) [31]. A high concentration of Mn in the soil reduces the metal's mobility, a phenomenon attributed to its adsorption by amorphous Mn oxyhydroxides. Based on the β-value, the immobilization mechanism contributes more significantly than the mobilization mechanism, indicating low metal mobility.

In the model, vanadium mobility is positively correlated with Fe_{sol} concentration and increasing pH. In contaminated peatlands, high concentrations of dissolved Fe^{2+} serve as a marker of anaerobic conditions, under which iron hydroxides dissolve and vanadium – previously adsorbed onto these hydroxides – is released into the solution. It can also migrate by becoming incorporated into dissolved Fe-organic complexes [32]. Increased mobility with rising pH is characteristic of its anionogenic properties; this process involves the oxidation of V^{4+} to the mobile V^{5+} anion (vanadate). The inverse relationship between the Ca concentration in the solution and vanadium mobility is attributed to the formation and co-precipitation of sparingly soluble calcium vanadates. The rise in soil solution pH caused by the discharge of formation waters creates conditions favoring the conversion of vanadium into an anionic form (vanadate ion). Under these conditions, high calcium concentrations trigger the precipitation of sparingly soluble phases, such as $Ca_2V_2O_7$, resulting in the effective immobilization of vanadium [33].

Lead is classified as a cationogenic element; however, its behavior in neutral and slightly alkaline media resembles that of chromium and vanadium. Lead's mobility increases with high iron concentration in the solution and rising alkalinity, whereas it decreases in the presence of high levels of manganese compounds in the soil. The influence of Fe_{sol} on lead mobility is linked to the iron's strong complexation with humic and fulvic acids via carboxyl and phenolic groups [34]. In this case, it may compete with iron – a strong complexing agent – leading to the partial displacement of lead into the solution. An increase in alkalinity does not directly cause increased mobility; rather, this effect is indirect. One possible explanation is based on the low mobility of lead, which is securely bound in the solid phase by phosphates [34]. When soils are contaminated with drilling waste or formation waters, phosphate

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levels reach high values compared to background concentrations. In turn, the behavior of phosphates in solution is governed by iron. As the pH rises, iron hydroxides adsorb phosphates from the solution, thereby increasing lead mobility. A second mechanism leading to mobilization is the formation of soluble organo-mineral Pb-fulvate complexes under alkaline conditions [35]. Amorphous manganese oxides are very strong sorbents for Pb, often involving irreversible chemisorption; this is reflected in the negative correlation between metal mobility and Mn content in the soil solid phase.

Despite the statistically insignificant mobilizing effect of PHC_{IR}, the mechanism itself is of interest. In this instance, it is specific to metals serving as indicators of formation waters. The polar fractions isolated from PHC_{IR} (resins, asphaltenes) exhibit a high affinity for the functional groups of humic acids and can compete with chromium, vanadium, and lead ions for binding sites within organo-mineral complexes. Their displacement by hydrocarbon ligands leads to the release of metal ions from the surface of the OM and their subsequent transfer into the soil solution. In addition, hydrocarbon contamination and the subsequent biochemical degradation of petroleum products cause the alkalinization of solutions, which promotes the oxidation of Cr^{3+} to highly mobile Cr^{6+} and of V^{4+} to the mobile V^{5+} anion [31] as well as the formation of soluble Pb^{2+} complexes with fulvic acids [35].

An analysis of the primary mechanisms of metal mobility under contamination conditions revealed that the pathways by which metals transition into the dissolved phase are more diverse and effective (in terms of the magnitude of unstandardized coefficients) than the processes leading to immobilization. The principal mobilization mechanisms involve competitive interactions with Fe^{2+} and Mn^{2+}, and to a slightly lesser extent with Ca^{2+} (Table 7). The ionic strength of the solution, governed primarily by Cl^- concentration, also plays a significant role. Soil hydrophobization resulting from PHC_{FL} contamination is a factor of some importance. A different mobilization mechanism, described above, is driven by the involvement of PHC_{IR}.

Table 7

Mechanisms of metal mobilization and immobilization under soil contamination conditions

Mechanisms	Metals
Mobilization	
Ionic strength of the solution (due to chlorides)	**Na**, Mn, **Zn**
Competitive contingency:	
Fe^{2+}	Mn, Sr, Co, Pb, Cr, V
Mn^{2+}	**Ca**, **Ba**, **Co**
Ca^{2+}	**Mn**, **Sr**
PHC_{IR}	Cr, Pb, V
Al^{3+} and H^-	Zn
Soil hydrophobization (PHC_{FL}) accompanied by metals displacement	Sr, Zn
Soil alkalinization	Cr, V
Reductive dissolution of Fe oxyhydroxides, V release, and incorporation into dissolved Fe-organic complexes	V
pH-dependent reduction in phosphate-mediated immobilization	Pb
Immobilization	
Ion-exchange sorption on amorphous Fe oxyhydroxides	Na, Ca, Mn, Sr, Zn
Ion-exchange interaction with humic organic matter	Sr, Ba
Isomorphous substitution of Ca^{2+} in minerals	Sr
Outer-sphere and inner-sphere adsorption by Mn oxyhydroxides	Ba, Co, **Cr**, **Pb**
Sorption and complexation with humic organic matter	Co
Formation and co-precipitation of sparingly soluble calcium vanadates	**V**

Note. Variables with dominant mobilization/immobilization mechanism (based on the magnitude of the β-coefficient) are highlighted in bold.

Mechanisms of metal immobilization are less diverse. Among them, sorption onto freshly formed Fe and Mn oxyhydroxides stands out for its high activity. Metal phase transitions are driven by ion-exchange interactions, sorption, and complexation with humic-type OM. For Sr, isomorphous substitution of Ca^{2+} in minerals is typical.

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When discussing the role of iron oxyhydroxides in metal immobilization, some clarification is required. In acidic and neutral media, the surface of iron oxyhydroxides generally carries a positive charge, which hinders cation adsorption. This phenomenon can be explained by mechanisms confirmed by modern spectroscopic studies. Organo-mineral complexes with humic acids form on the surface of Fe oxides – particularly in peat soils – imparting a negative charge to the surface and acting as cation-binding sites via carboxyl groups [36]. Such a mechanism has also been described for other mineral surfaces [26]. Furthermore, for transition metals, direct inner-sphere sorption on Fe oxides is possible; this process occurs via coordination interactions and is not limited by the sign of the charge [37]. It is possible that, under conditions of contamination, the surface of Fe oxides may possess a high density of reactive sites, including those bearing a negative charge.

Analysis of the migration model reveals another factor – the mobility mechanisms of metals associated with formation waters differ from those of metals found in drilling cuttings. First, there is a difference regarding the involvement of petroleum hydrocarbons, which – as previously noted – comprise both polar and non-polar fractions. Polar fractions (within the PHC_{IR} component) play a significant role in the mobilization of formation-water metals (Pb, Cr, V), whereas non-polar fractions are significant for metals in drilling waste. Second, pH levels influence mobility only in the first group. Third, mobilization mechanisms are the most significant factor for metals in drilling waste, whereas immobilization mechanisms are paramount for metals in formation waters.

Internal model validation, conducted using field data from two profiles – a sludge spill (transect 224) and a formation water discharge (transect 208) – demonstrated satisfactory predictions of metal mobility in the immediate vicinity of the source. Furthermore, at greater distances – as background conditions were approached – the predicted mobility potential values *P* closely matched the actual values (Fig.3, 4). The average potential observed at background sites was adopted as the background migration potential. Notably, background mobility potential values were reached for all metals at a distance of 200-300 m. When visually analyzing discrepancies between actual and calculated data – which were more frequently observed in solutions along the profile affected by formation water – it is essential to account for the mosaic nature and complexity of the soil cover in Arctic landscapes.

Differences in metal migration patterns were identified depending on the source of contamination. In the case of contamination by drilling waste, the lateral migration distance is 100-150 m for Na, Mn, Co, Zn, and Ba, and 200-300 m for Sr and Ca; for metals serving as indicators of formation waters (Cr, Pb, V), it does not exceed 50 m. With contamination by formation waters, the lateral migration distance of metals increases to 150-200 m (Mn, V, Cr, Pb, Co, Zn), exceeding 200 m for the most mobile elements (Na, Sr, Ba). Calcium exhibits low mobility under conditions of alkalinization.

Elucidating the mechanisms of metal mobilization in soil resulting from oil and gas industry contamination has enabled the formulation of recommendations for neutralizing the pollutant flux using nature-inspired technologies [38]. Under conditions of competitive interaction, soil hydrophobization by petroleum hydrocarbons, and alkalinization, many of the studied metals remain in solution and continue to migrate. Soil alkalinization-which can enhance the role of Fe and Mn hydroxides in immobilizing pollutants-yields only a temporary effect, followed by a return to an acidic environment, the dissolution of these hydroxides, and the release of metals into the solution.

Covering contaminated areas with peat appears to be the most rational approach; this increases the number of sorption sites, alleviates competitive interference, and stabilizes physicochemical conditions, thereby promoting the sequestration and immobilization of metals within the solid phase under conditions that closely mimic natural ones. Peat is known to be an effective natural sorbent for a wide range of metals. Its ability to remove up to 91 % of Cd^{2+}, 98 % of Pb^{2+}, and 90 % of Cu^{2+} ions from solutions within a short timeframe (the first few minutes of contact) has been experimentally confirmed, demonstrating high sorption capacity [39]. It is also effective against Fe^{3+} and Cr^{6+} [40]. Studies of sorption kinetics confirm the role of peat as a source of additional sorption sites. It has been established that the uptake of metal ions (e.g., Cu^{2+} and Fe^{3+}) proceeds under diffusion-controlled conditions, with their chemical interaction with the functional groups of the peat playing a

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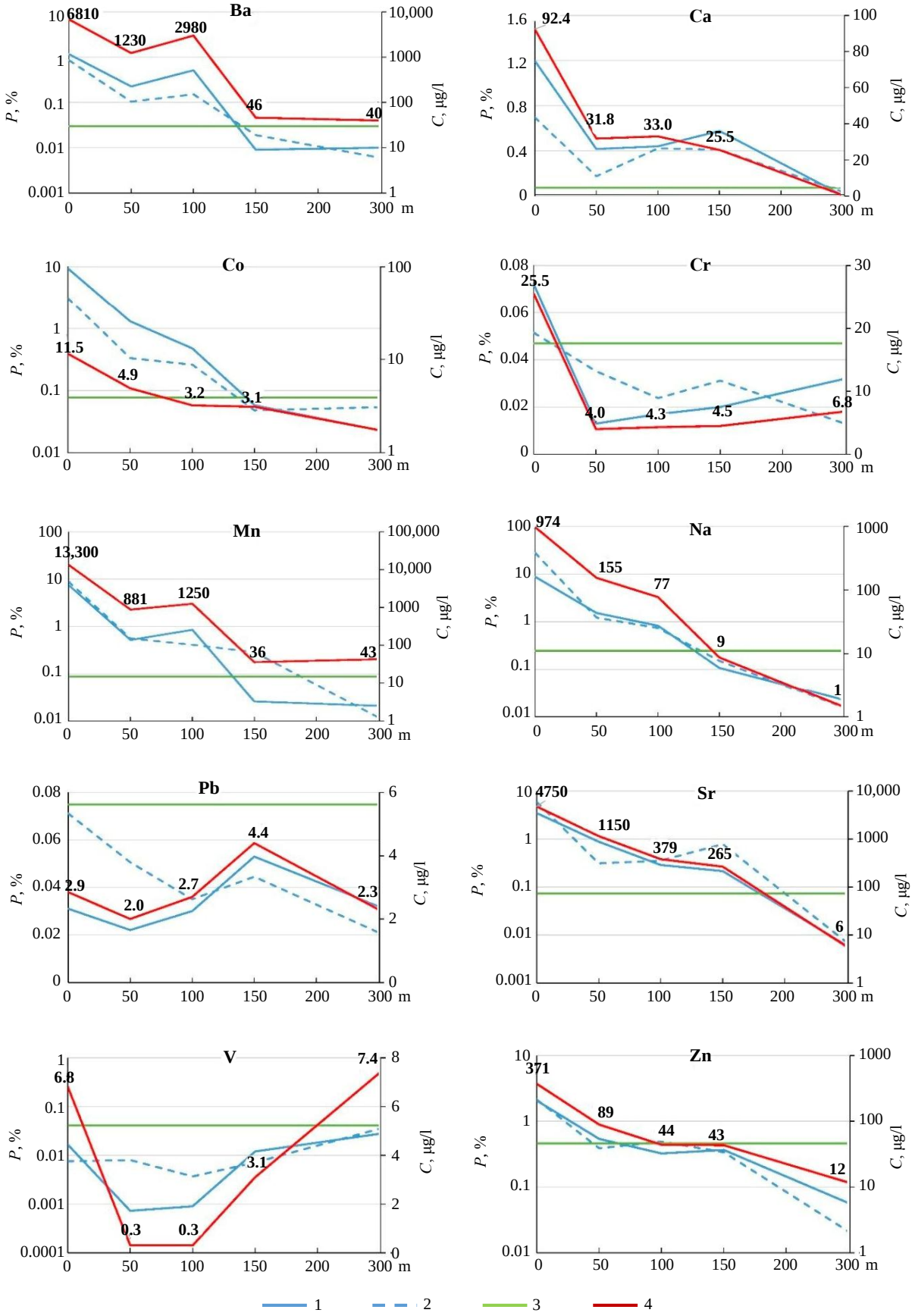


Fig.3. Measured and modeled values of metal mobility potential in the organic soil horizon along a profile extending from the drilling cuttings disposal site (Yamburgsky site, transect 224)

1 – measured *P*; 2 – modeled *P*; 3 – background *P* value; 4 – metal concentration in soil solutions *C*

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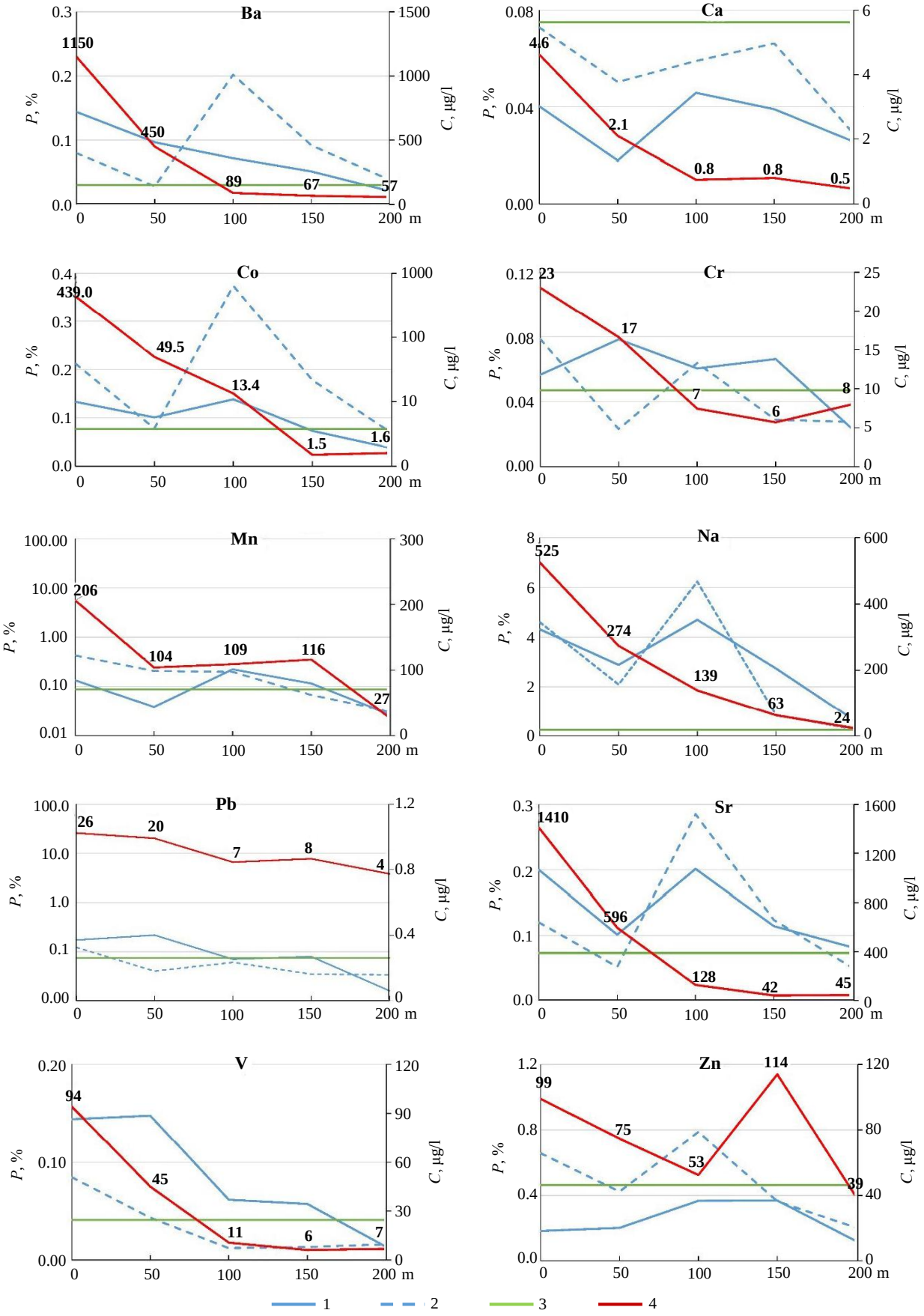


Fig.4. Measured and modeled values of metal mobility potential in the organogenic soil horizon along the profile extending from the formation water discharge site (Purovsky site, transect 208)

1 – measured *P*; 2 – modeled *P*; 3 – background *P* value; 4 – metal concentration in soil solutions *C*

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significant role [41] the peat does not merely retain the ions within its pores but chemically binds them. An effective method involves the co-application of biochar and iron to the soil, resulting in the formation of an iron-modified biochar that is stable under acidic conditions and actively precipitates both cations and anions [42].

Conclusion

Statistical modeling of metal migration in soils subjected to contamination from drilling waste, formation waters, and transport was based on the results of field studies and analyses of elemental concentration in the soil solid phase and soil solutions. The systematic application of multivariate statistical methods made it possible to identify the primary source of contamination amidst combined impacts, group samples by contamination type (drilling cuttings, formation waters, and vehicular traffic) or classify them as representing background conditions, and determine the magnitude of the effects of natural and industrial factors governing metal distribution in the soil. Given that natural factors governing metal differentiation lose their decisive role under contaminated conditions, the modeling utilized predictors that reflect the nature of anthropogenic processes or are subject to transformation.

In the final stage of modeling, multiple regression analysis was performed, revealing the key mechanisms governing metal mobilization and immobilization. Metal mobilization is significantly influenced by the competition of metal ions with Fe^{2+}, Mn^{2+}, Ca^{2+}, and petroleum hydrocarbons for sorption sites. This competitive interplay, which enhances pollutant mobility, is driven by soil contamination with metals and petroleum hydrocarbons, as well as by changes in physicochemical conditions (specifically, increases in TDS and pH). These factors led to an increase in solution ionic strength, soil alkalinization, and soil hydrophobization – processes that play a crucial role in metal mobilization. The values of the standardized coefficients indicate geochemical and physicochemical conditions favorable for the presence of metals in the solution. Immobilization is most effective during ion-exchange sorption on amorphous oxyhydroxides of Fe (Na, Ca, Mn, Sr, Zn) and Mn (Ba, Co, Cr, Pb), and on humic-series OM (Sr, Ba). Under the studied conditions, the lateral migration of metals amounts to 150-200 m. An analysis of metal mobilization and immobilization mechanisms leads to the conclusion that covering contaminated areas with peat is the most suitable method for reducing the migration potential of pollutants, while iron-modified biochar is the most effective.

The models passed the F-test, demonstrating their statistical significance. The results of t-test identified predictors that do not have a significant impact on the dependent variable. Model adequacy was confirmed by the absence of systematic forecast bias.

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