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Geochemistry of Grass Biocenoses: Biogenic Cycles of Chemical Elements at Contamination of the Environment with Heavy Metals

V. S. Bezel'[a], T. V. Zhuikova[a, b], and V. A. Gordeeva[b]

[a] *Institute of Plant and Animal Ecology, Ural Division, Russian Academy of Sciences, ul. Vos'mogo Marta 202, Yekaterinburg, 620144 Russia*
e-mail: bezel@ipae.uran.ru

[b] *Faculty of Natural Sciences, Mathematics, and Informatics, Nizhnii Tagil State Social–Pedagogical Academy, ul. Krasnogvardeiskaya 57, 622031 Russia*

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Abstract—The paper addresses the involvement of grass communities in biogenic cycles of chemical elements (Zn, Cu, Pb, Cd, Mn, Co, Cr, Ni, and Fe). Both the species composition and the suprasoil phytomass of phytocenoses in the Central Urals are modified in a gradient of contamination with heavy metals. The bioproductivity and subsequent mineralization of plant remnants are discussed with reference to two soil types that differ in agrochemical parameters. The contribution of agrobotanical groups to the biological exchange of chemical elements is proved to be controlled not only by the volume of annually dying suprasoil biomass but also by the intensity of processes mineralizing plant remnants in the contamination gradient. This modifies the cycles of chemical elements in natural contaminated biocenoses. The reaction of grass communities on environmental contamination can be viewed as partial counterbalancing of the adverse effect of chemical stress via maintaining a high enough level of the biological exchange of chemical elements.

Keywords: heavy metals, chemical contamination of environment, biogeochemical cycles, grass biocenoses, grass phytocenoses, bioproductivity, mineralizing processes

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INTRODUCTION

Biological cycles of chemical elements in ecosystems of various types and changes in these cycles under the effect of human activity is one of the most pressing problems of modern geochemistry.

Extensive studies of the chemical degradation of natural environment conducted over the past years were mostly limited to studying the accumulation of certain chemical compounds in various components of natural ecosystems. However, even population effects characterize only certain aspects of biocenoses and thus fail to comprehensively and adequately reflect their state as a integral system. The fate of biocenoses as a complex of living, biologically inert, and inert components under anthropogenic effects of any type is predetermined by the levels of material, energy, and information exchanges that can be maintained by the systems within themselves and with related biocenoses (Vernadsky, 1978; Shvarts, 1980). In this formulation of the problem, chemical contamination of environment can be analyzed as anthropogenic deformation of these exchanges. Studies of this type are actively conducted by several scientists and research teams (Pokarzhevskii et al., 2000; Bezel', 2006; Bezel' and Zhuikova, 2007; Grimshaw et al., 1958; Lindquist and Block, 1997; Dmowski and Karalewski, 1979). It is reasonable to a priori expect that the intensity of the cycles at various contamination levels of soils can be controlled by the following factors:

* increasing soil concentrations of certain elements in species biologically accessible to plants;

* variations in the species composition of the phytocenoses under the effect of contamination and related specifics in the accumulation of elements and their toxicity to various species of plants;

* a decline in the overall productivity of the community under the effect of contamination, first of all, because of a decrease in the supra- and subsoil phytomasses;

* changes in the processes mineralizing plant remnants in the soils that return chemical elements in the biological exchange.

Difficulties in solving the problem stem from the highly labor-consuming character of the studies, the paucity of data on reactions to elevated concentrations of toxicants in certain links of the phytocenoses, and the direct effect of these concentrations on the mineralizing processes in plant remnants.

Our research was dedicated to exchange of phytomas and chemical elements in the soil–phytocenosis system in grass communities in the Central Urals, which are affected by various levels of contamination with aerial emissions from metallurgical plants. The study was carried out to analyze the roles of the afore-

mentioned factors in the biogenic cycles of chemical elements.

METHODS

Our research was carried out in the southern taiga subzone of the taiga geographic zone in the Central Urals. The study area (town of Nizhnii Tagil, Sverdlovsk oblast, 60° E, 58° N) hosts steel and iron metallurgical plants, and the major contaminant is dust of metals in the form of their oxides (Cr, Ni, Fe, Cu, Zn, Cd, Pb, etc.) and S.

The studies were carried out at plots outlined within the study area in its agrogenic and anthropogenic landscapes, which are widespread in the vicinities of the town of Nizhnii Tagil. Occurring at different distances from the metallurgical plants, these plots allowed us to analyze the gradient in the chemical contamination of the soils. The phytocenoses are grass communities.

Soil and plant samples to be analyzed for heavy metals were collected in compliance with the requirements in (Alekseenko, 1990; Il'in, 1991; *Methodical...*, 1992). Metals were extracted with 5% HNO_3 from soil samples and with 70% HNO_3 or mixture of HNO_3 and HCl from samples of plants. The acid extracts from soils were analyzed for Cr, Mn, Fe, Co, Ni, Cu, Zn, Cd, and Pb by flame atomic absorption spectrometry on an AAA 300 (PerkinElmer) spectrophotometer (Khavezov and Tsalev, 1983). The chemical composition of soils was analyzed in compliance with the certified analytical routine at the accredited laboratory of the Institute of Plant and Animal Ecology, Ural Division, Russian Academy of Sciences, (Accreditation Certificate ROSS RU.0001.515630). Anthropogenically disturbed soils had been identified earlier (Kaigorodova et al., 2013).

In order to evaluate the primary production of the poiums during the 2009–2012 vegetation periods, ten discount areas 25 × 25 cm were outlined using the random sampling technique during the maximum herbage time. Sampling was conducted by means of block sampling (Shalyt, 1960). Plants within any discount area were subdivided by species. Upon preparatory treatment, atmospheric dry supra- and subsoil phytomasses (g/m^2) were determined for each species.

To evaluate the actual decomposition rate of plant remnants, we have sampled the suprasoil phytomass of the grass communities. The exposed material used in our field experiments was atmospheric dry phytomass of discrete agrobotanical groups (legumes, graminoids, and forb). Then samples were placed into the uppermost 5-cm soil layer along transects spaced 30 cm apart in areas where plant phytomass was sampled. On each transect, ten samples of legumes, graminoids, and miscellaneous herbs were placed. The samples were exposed for 12 months.

Upon expiry of the exposure time, the samples were rid of soil particles and thin roots and were dried to absolutely dry mass at a temperature of 105°C (Vorobeichik, 2007; Vorobeichik and Pishchulin, 2011). The decomposition rate of the exposed material was evaluated by the decrease (in %) of the material mass (Vorobeichik, 2007).

RESULTS AND DISCUSSION

Soil characteristics and concentrations of chemical elements. Soil as the central link of biological exchange processes operates as a depot of chemical elements and maintains their translocation at trophic levels. The closedness of the production cycle in the soil–plant system is also maintained by processes mineralizing plant remnants. Thereby the intensity of biological production and mineralization of the plant litter directly depends on such major soil parameters as pH, contents of aluminosilicates and humus, and the composition and abundance of the soil biota (Sochava et al., 1962; Titlyanova, 1977, 1979; Roginskaya and Kazantseva, 1982; Kharitonov and Boikov, 1999; and others).

In the gradient of the chemical contamination of the soils, meadow-type areas were distinguished in the agrogenic and anthropogenic landscapes. The average contents of mobile species of heavy metals in the regional-background soil and anthropogenically disturbed territories differed by factors of several tens (Table 1). The major soil contaminants in the territories are Cd, Co, Zn, and Cu, whose concentrations in the soils and the background and anthropogenically disturbed territories differ by more than 75, 20, 19, and 10 times, respectively. The differences in the Pb, Mn, and Fe concentrations vary insignificantly or are small (for Ni and Cr).

The integral toxic load on the soil was evaluated as

$$S_n = \frac{1}{n}\sum_i^n \frac{C_i}{C_f}, \text{ relative units,} \quad (1)$$

where C_i and C_f are the concentrations of elements whose concentrations in the soils are higher than the background values (minimal in our situation). As indicated by this parameter, the toxic load in our gradient increases by a factor of >20.

The following two soil types were distinguished using the main agrochemical parameters in the contamination gradient (Kaigorodova et al., 2013).

Soils of type I (areas of toxic load of 1.00 and 3.33 relative units) are derelict land not cultivated for 17–20 years. The density and thickness of the grass sod and sod horizon vary. The soils are agro-podzolic, gleyic, prograded by the sodding process. The soils are weakly acidic, with exchange and hydrolytic acidity in the lower part of the profile. The exchange complex contains 57–90% exchange bases, is dominated by Ca, and bears a relatively high Mg concentration, particularly in the lower part of the profile. This is explained by the fact that the soil is formed on talc schist, which is a magnesian rock. High concentrations of plant food elements occur only in the uppermost organic horizon, the nitrogen and phosphorus concentration rapidly

Table 1. Concentrations of mobile species of heavy metals in the soil samples ($M \pm m$)

Area	S_i, rel. units	Concentrations of trace elements, ppm								
		Zn^{2+}	Cu^{2+}	Cd^{2+}	Pb^{2+}	Co^{2+}	Ni^{2+}	Mn^{2+}	Cr^{2+}	Fe^{3+}
Soils of type I										
Background	1.00	17.51 ± 1.61 (20)	12.56 ± 0.89 (20)	0.15 ± 0.96 (17)	8.10 ± 0.91 (20)	6.50 ± 0.85 (20)	12.95 ± 0.86 (20)	291.60 ± 27.17 (19)	13.06 ± 0.78 (20)	788.90 ± 50.87 (15)
Buffer 1	3.33	58.05 ± 1.10 (19)	38.62 ± 0.59 (20)	1.26 ± 0.00 (20)	13.17 ± 0.18 (20)	16.76 ± 0.21 (20)	17.97 ± 0.32 (20)	359.01 ± 9.37 (20)	20.03 ± 0.46 (20)	964.48 ± 1.65 (15)
Soils of type II										
Buffer 2	6.19	262.65 ± 39.56 (20)	101.57 ± 11.13 (20)	0.90 ± 0.08 (20)	38.81 ± 4.91 (19)	14.53 ± 3.63 (20)	7.40 ± 1.37 (20)	375.18 ± 54.02 (20)	7.78 ± 1.08 (20)	841.11 ± 13.23 (15)
Impact	22.78	390.96 ± 125.92 (5)	951.49 ± 236.10 (4)	1.54 ± 0.47 (5)	12.38 ± 3.88 (4)	124.23 ± 17.79 (5)	7.75 ± 1.30 (5)	2364.9 ± 93.52 (5)	7.14 ± 2.26 (5)	– –

S_i is the total toxic load, M is the average values, m is the arithmetic mean error, numerals in parentheses indicate the sample size, and dashes mean no data.

Table 2. Amounts of organic matter in the blocks of the biocenoses, t/ha per dry weight ($M \pm m$)

Annual suprasoil production	Toxic load, relative units				
	soil of type I		soil of type II		A.A. Titlyanova's data*
	1.00	3.33	6.19	22.78	
Phytomass	3.08 ± 0.54	2.46 ± 0.43	2.02 ± 0.36	2.30 ± 0.41	2.95–14.50
Dead grass*	4.16 ± 0.73	3.32 ± 0.58	2.73 ± 0.49	3.10 ± 0.55	1.20–3.80
Ground litter*	3.57 ± 0.63	2.85 ± 50	2.34 ± 0.42	2.67 ± 0.47	1.31–3.84
Suprasoil production*	10.81 ± 1.89	8.63 ± 1.53	7.09 ± 1.26	8.07 ± 1.44	1.51–11.4

* Calculated values obtained based on data from (Titlyanova, 1977, 1979) for forb–leguminous–graminoid meadows.

decrease down the soil profile to very low values, and the potassium concentration gradually decreases to intermediate and low values.

Soils of type II. (areas of toxic load of 6.19 and 22.78 relative units) are young and formed thanks to the sodding process on talc schist rubble and silt. This source material can be classified as transitional from anthropogenic surface materials (lithostrates and artindustrates) to young soils that are formed according to the brown soil type. The soils are weakly acidic, rich in bases (70–98%), their exchange complex is dominated by Ca, although the Mg concentrations are also high. The soils are rich in plant food elements, first of all, in P and K. The concentration of easily hydrolyzable nitrogen is high. Practically no nitrates have ever been found at any of the sites.

Phytomass of the grass communities. The annual suprasoil production of the phytocenoses includes the living green material, dead material (shoots in the process of dying and those dead this year), and ground litter (dead plant remnants of various age and variably decayed on the soil surface). Table 2 reports data on the suprasoil phytomass of the communities at the plots with different levels of chemical contamination. It is known that hay cuttings taken during the maximum herbage development in plant associations always yield a lower mass than the overall suprasoil production, because the masses of the dead material and ground litter are thereby not taken into account (Sochava et al., 1962). Several authors argue that the masses of the dead material and ground litter are controlled by combinations of such important factors as the species composition of the cenosis, temperature during the vegetation season, availability of moisture, inorganic nutrition, etc. (Kozlova, 1971; Titlyanova, 1977; Roginskaya and Kazantseva, 1982). Detailed analysis of a forb–leguminous–reedgrass meadow similar to our biocenoses has shown that the average volume of the material dying during a season and the ground litter formed by this

Table 3. Suprasoil phytomass of agrobotanical groups of the meadow communities ($M \pm m$)

S_i, rel. units	Suprasoil phytomass, g/m^2			
	legumes	graminoids	forb	total
1.0	95.76 ± 43.87	92.06 ± 10.71	120.26 ± 11.44	308.08 ± 46.64
3.33	52.98 ± 6.98	58.64 ± 10.08	137.73 ± 20.72	249.35 ± 28.79
6.19	62.90 ± 21.83	53.23 ± 11.90	87.41 ± 14.53	203.53 ± 27.06
22.78	73.65 ± 18.54	106.37 ± 26.74	49.86 ± 8.77	229.88 ± 44.75

material are 1.35 and 1.16 times, respectively, higher than the phytomass volume during the maximum herbage development (Titlyanova, 1977, 1979). Analogous data on the proportions of the maximum cutting volume and pure production for tall-grass communities were reported in (Roginskaya and Kazantseva, 1982). Table 2 presents evaluations for the volumes of the dead material and ground litter calculation from these data.

Chemical contamination affects the suprasoil phytomass of the agrobotanical groups (Table 3). In type-I soils in a contamination gradient, the contributions of forb increases and those of leguminous and graminoids decrease at decreasing total phytomass. However, the situation with the soils of type II is different. The total phytomass increases insignificantly with increasing toxic load, and the contribution of the forb thereby declines, while the phytomass of graminoids and legumes conversely increases (Fig. 1).

Mineralization of plant remnants. The intensity of the destruction processes and related return of chemical elements to biological exchange are principally important parameters of the state of natural ecosystems. Most researchers admit that an excess of certain elements (first and foremost, heavy metals and sulfur) in a soil inhibit the decomposition of the dead biomass (Vorobeichik, 2002, 2007; Parshina, 2007; Freedman and Hutchison, 1980; Baath, 1989; Berg et al., 1991; Chew et al., 2001; and others) via suppressing the activity of the microflora and possible elimination of the most susceptible destructor species (Ivshina et al., 2014). The mineralization processes of plant material in agricultural soils can even be intensified (Pomazkina et al., 1999, 2008).

With regard for the similarity in the chemical compositions of the suprasoil phytomass (dead plant material and ground litter) dying off during a season, it is reasonable to assume that the rates of the mineralizing processes of these components are similar in any

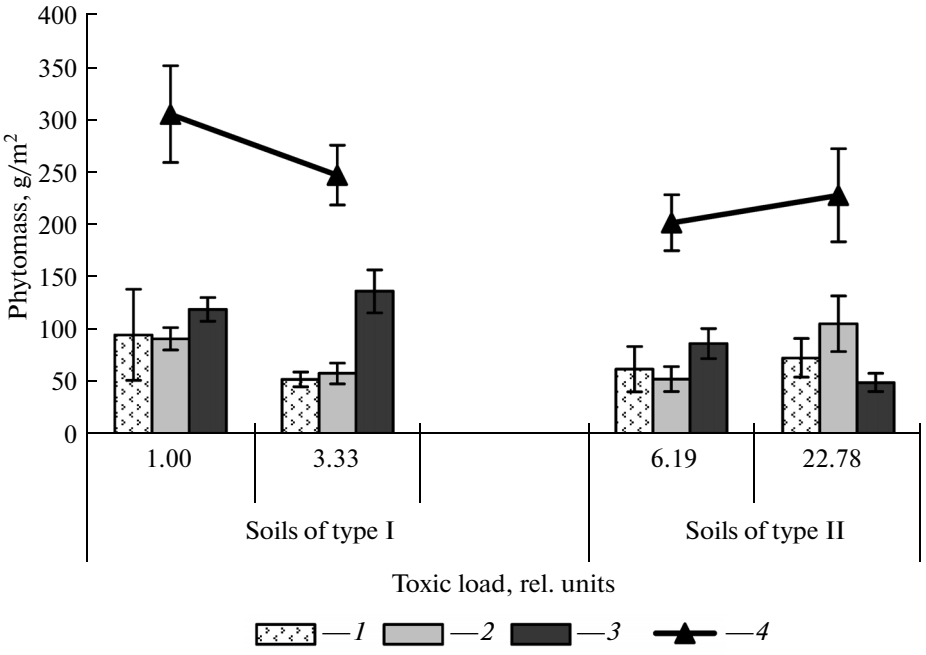

Fig. 1. Massif suprasoil phytomass of various agrobotanical groups in a toxic load gradient. Phytomass: (*1*) legumes, (*2*) graminoids, (*3*) forb, (*4*) total.

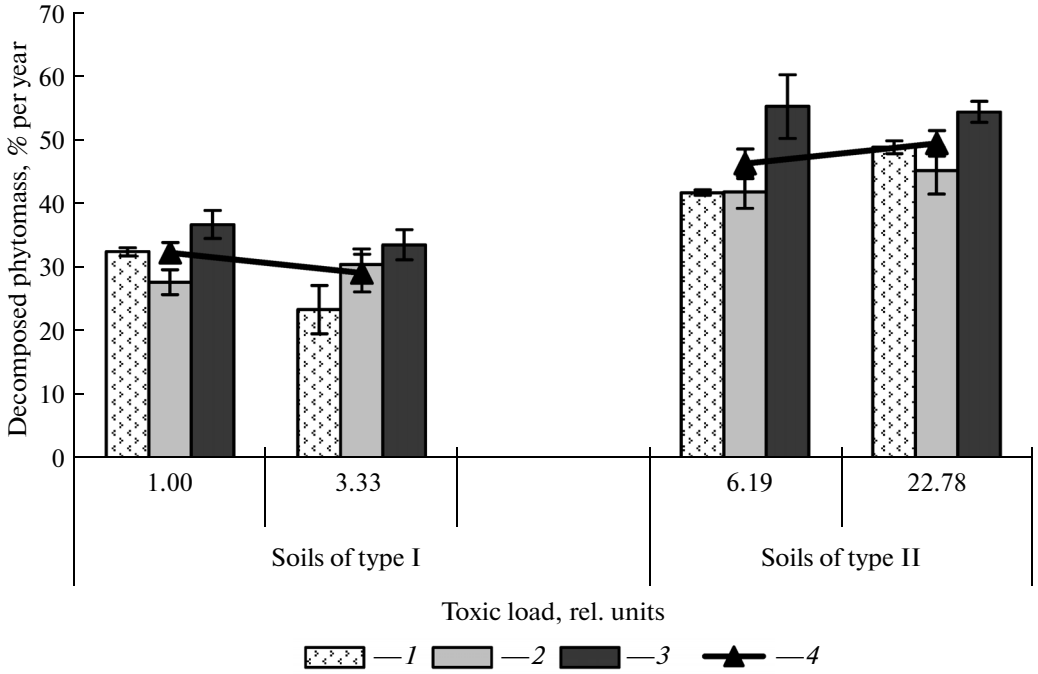

Fig. 2. Intensity of the annual mineralization of plant remnants on soils of type I and II. Decay rates: (*1*) legumes, (*2*) graminoids, (*3*) forb, (*4*) average.

agrobotanical group and can be evaluated from our experimental data.

Figure 2 shows the percentage of the total biomass and the phytomasses of certain agrobotanical groups mineralized per annum. The maximum destruction rate of organic matter was detected for forb: 32–37% at plots with type-I soils and up to 55% for type-II soils. Legumes and graminoids are mineralized less intensely in a contamination gradient. The evaluated average decomposition rates of suprasoil phytomass with regard for the various agrobotanical groups in it indicates that the mineralizing processes depend on the types of the soils and are at a maximum in type-II soils in our situation. It should be emphasized that an increase in the toxic load is generally associated with a weak tendency toward suppressing mineralizing processes in soils of type I and their intensifying in soils of type II.

Similar data were also reported by other researchers. According to Miroshnichenko (1978), the intensity of the early mineralizing processes of plant remnants is at a maximum and may reach 60–80% of the initial one, and the decomposition of graminoids thereby proceeded, similar to our situation, more slowly than for legumes and forb. Analogous estimates for the decomposition rate of plant remnants (up to 70%) at poiums is reported in (Titlyanova, 1977).

Provided that the proportions of the mineralized phytomass in the soils of types I and II remain unchanging during the ensuing years, then the suprasoil phytomass accumulated during one year at soils of type I can be completely mineralized within four years. This period of type for the type-II soils is approximately three years.

Our evaluations of the mineralizing processes make it possible to calculate the overall phytocenosis production that can be mineralized during one year (including the dead plant material and ground litter variably contaminated with heavy metals) (Fig. 3).

Concentrations of chemical elements in the grassland vegetation. Table 4 reports the average concentrations of chemical elements in the suprasoil phytomass of the fractions we distinguished in the grass vegetation. These data show a certain tendency: the greater the soil contamination, the higher the concentrations of chemical elements in the suprasoil phytomass in all variants. The concentrations of Fe, Mn, and Zn are at a maximum in the suprasoil phytomass in a contamination gradient, and the Cd concentration is at a minimum.

The effect of soil contamination on plants is most obvious if the toxic load level is evaluated by formula (1) for agrobotanical groups using concentrations of elements in them. Along with an evident increase in this parameter in the contamination gradient, it is also worth mentioning that the maximum toxic impact is suffered by the forb suprasoil phytomass (Fig. 4), and this explains why the productivity of plants of this group dramatically diminishes in a contamination gradient (Table 3).

It follows that the biological exchange of chemical elements controlled by plants in a gradient of soil contamination depends on the increasing concentrations of elements in the soils in and plants and on changes in the composition of the phytocoenosis. This is associated

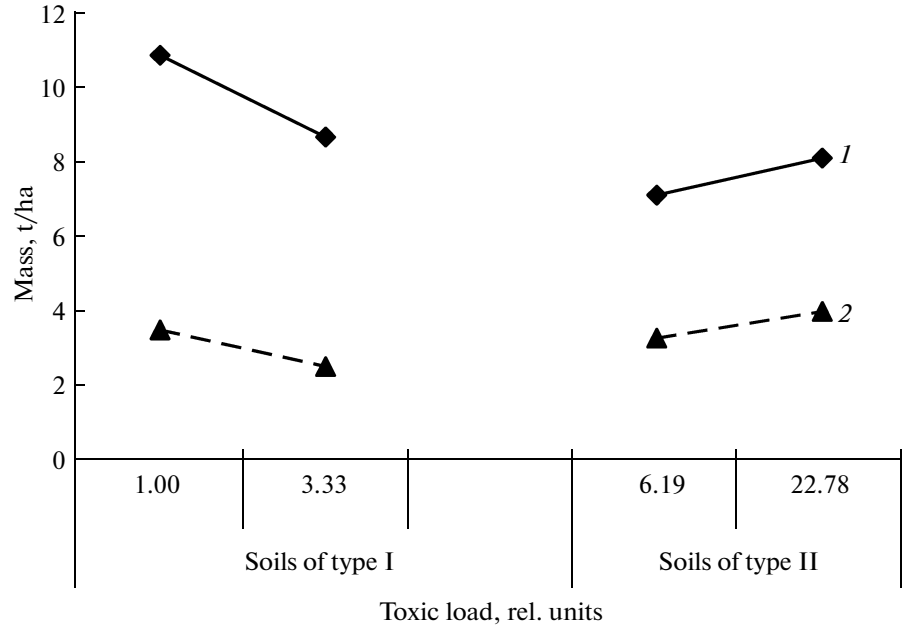

Fig. 3. (1) Total and (2) annually mineralized suprasoil phytomass.

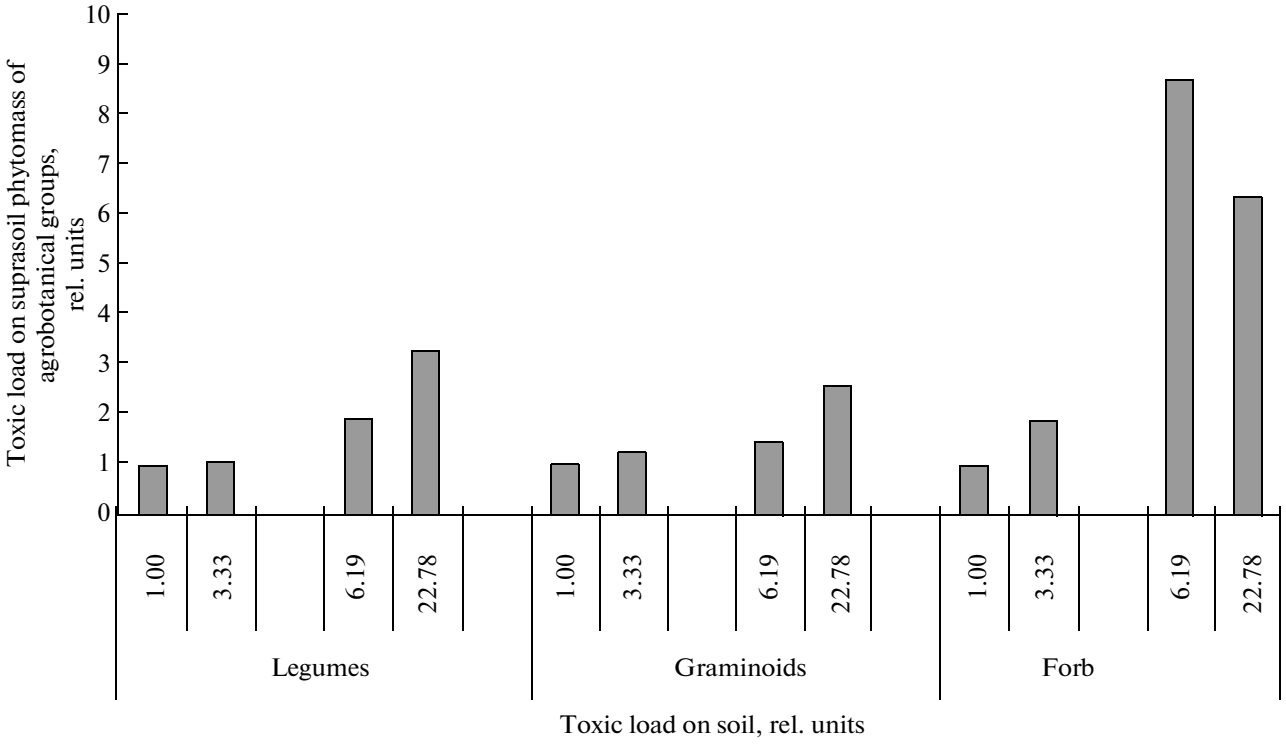

Fig. 4. Toxic load on the suprasoil phytomass of various agrobotanical groups.

with a decline in the total phytomass, an increase in the amount of accumulating species, and the disappearance of species most susceptible to excess concentrations of certain elements. It is also necessary to bear in mind that the intensity of the mineralizing processes of the dying off plant remnants can also vary. These parameters oppositely vary in a contamination gradient at any soil types.

Data in Table 4 on the concentration of elements in the suprasoil phytomass, with regard for the phytomasses of discrete agrobotanical groups (Table 3), allowed us to evaluate the overall amounts of chemical

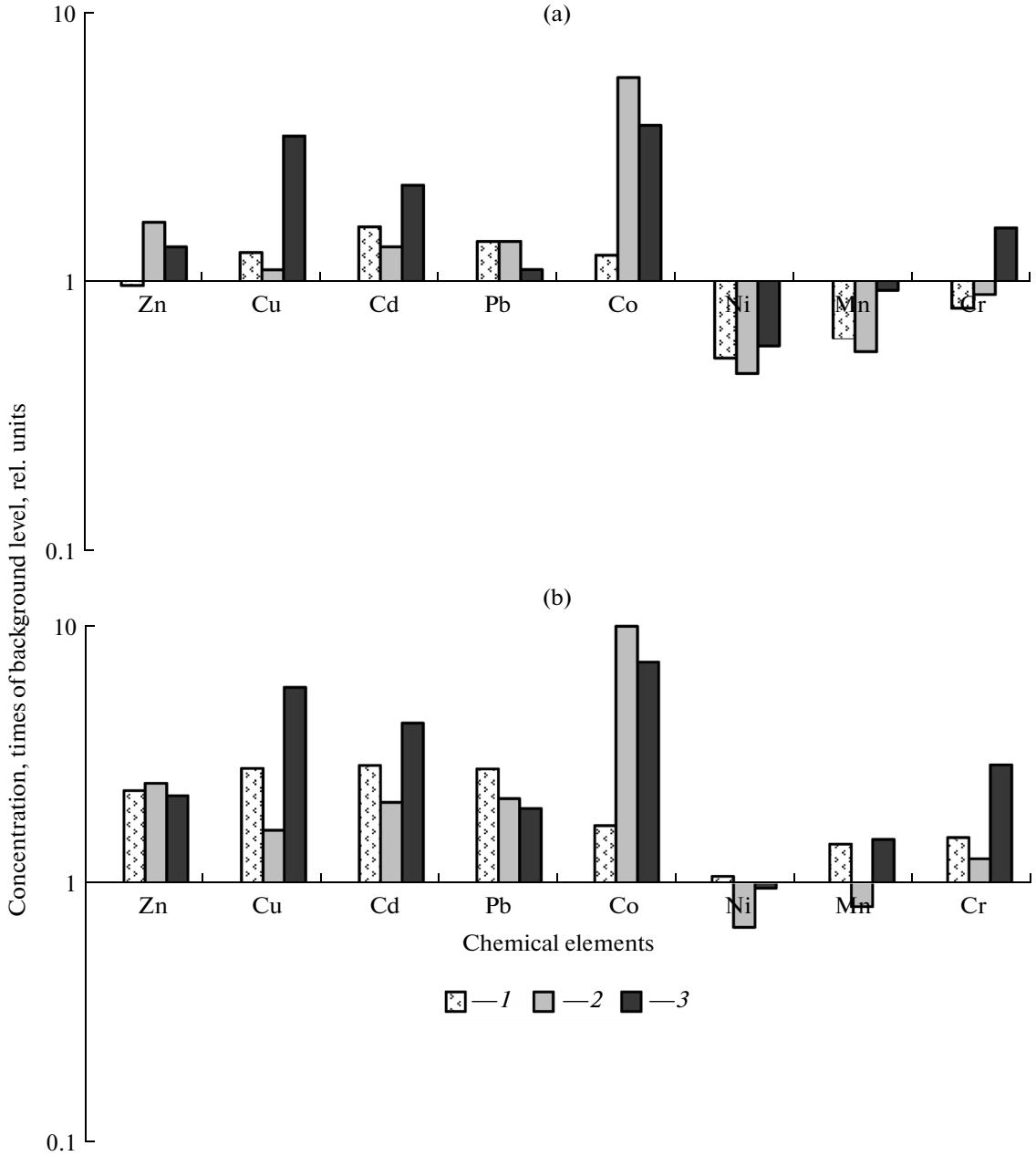

Fig. 5. (a) Rate of increase in the concentrations of metals annually involved in biological exchange in the suprasoil phytomass and (b) annual return of chemical elements in biological exchange. Data are normalized to those on the reference background territory (S_i = 1.0 relative units). Areas of certain toxic load, relative units: (*1*) 3.33, (*2*) 22.78.

elements in the suprasoil phytomass and the variations in the soil contamination gradient (Table 5).

The effect of soil contamination on the total transfer of elements into the suprasoil phytomass can be most vividly visualized by comparing these data with those on the reference background territory (S_i = 1.0 relative units). As follows from Fig. 5a, the concentrations of most of the elements (Zn, Cu, Cd, Pb, and Co) in the suprasoil phytomass in a contamination gradient increase and become involved in biological exchange during the yearly dying off of the phytomass. The production of phytocenoses most significantly incorporates such elements as Zn, Co, Cu, Cd, and Pb, whereas Mn and Ni are involved much less intensely in forb-mediated biological exchange mediated at all of the territories.

Data on the rates of the mineralizing processes in plant remnants can be utilized to quantify the level of the annual return of the elements to the biological exchange due to the annually dying off and mineralized suprasoil phytomass of grassland vegetation (Table 6, Fig. 5b).

Table 4. Concentrations of chemical elements in in the suprasoil phytomass ($M \pm m$)

S_i, rel. units	Agrobotanical groups	Concentrations of chemical elements, μg/g of dry weight								
		Zn^{2+}	Cu^{2+}	Cd^{2+}	Pb^{2+}	Co^{2+}	Ni^{2+}	Mn^{2+}	Cr^{2+}	Fe^{3+}
1.00	Legumes	34.44 ± 2.89	5.55 ± 1.06	0.15 ± 0.029	4.82 ± 0.82	10.55 ± 18.13	5.51 ± 2.09	41.38 ± 3.03	3.10 ± 0.51	220.19 ± 138.91
	Graminoids	20.86 ± 6.79	2.02 ± 0.36	0.16 ± 0.09	3.71 ± 1.61	22.59 ± 16.46	1.98 ± 1.02	71.07 ± 7.35	2.55 ± 0.71	138.51 ± 83.1
	Forb	40.52 ± 77.89	12.03 ± 6.98	0.40 ± 0.04	7.82 ± 1.46	2.66 ± 1.71	6.65 ± 1.49	58.94 ± 6.58	4.50 ± 0.75	489.72 ± 41.77
3.33	Legumes	39.67 ± 3.29	7.03 ± 1.33	0.30 ± 0.06	6.96 ± 1.18	2.15 ± 1.65	2.53 ± 0.96	39.50 ± 2.76	2.29 ± 0.37	444.95 ± 280.32
	Graminoids	21.69 ± 2.09	3.82 ± 1.72	0.37 ± 0.22	4.48 ± 1.04	20.12 ± 18.51	1.41 ± 0.65	31.41 ± 20.20	2.21 ± 0.28	–
	Forb	46.09 ± 6.19	10.54 ± 2.42	0.65 ± 0.19	13.73 ± 3.61	15.12 ± 11.13	4.08 ± 1.59	50.05 ± 6.42	4.34 ± 0.83	796.15 ± 152.05
6.19	Legumes	82.73 ± 6.87	8.63 ± 1.64	0.26 ± 0.05	7.64 ± 1.30	40.77 ± 34.47	2.29 ± 0.87	38.25 ± 2.68	2.71 ± 0.43	–
	Graminoids	49.51 ± 15.14	4.58 ± 1.52	0.19 ± 0.09	7.20 ± 1.10	19.60 ± 4.84	1.43 ± 0.11	47.18 ± 20.39	6.66 ± 7.14	302.92 ± 181.74
	Forb	106.28 ± 28.2	11.151 ± 1.53	0.92 ± 0.14	18.85 ± 5.03	133.00 ± 31.85	5.26 ± 0.95	52.98 ± 5.71	4.84 ± 0.88	630.21 ± 104.38
22.78	Legumes	68.91 ± 5.93	22.30 ± 4.24	0.55 ± 0.10	7.58 ± 1.29	77.34 ± 59.55	5.12 ± 1.94	43.34 ± 3.03	0.55 ± 0.09	–
	Graminoids	45.55 ± 14.58	17.38 ± 3.30	0.31 ± 0.18	4.42 ± 1.90	8.27 ± 5.95	–	80.99 ± 8.10	–	1149.07 ± 689.4
	Forb	78.56 ± 13.47	40.14 ± 4.62	2.16 ± 1.18	21.84 ± 18.84	70.99 ± 61.02	6.57 ± 3.31	85.13 ± 15.57	19.60 ± 3.33	675.03 ± 405.5

Dashes mean not analyzed.

Table 5. Total annual involvement of chemical elements in the suprasoil phytomass of grassland vegetation in the phytocenoses, g/m^2 ($M \pm m$)

S_i, rel. units	Chemical elements							
	Zn^{2+}	Cu^{2+}	Cd^{2+}	Pb^{2+}	Co^{2+}	Ni^{2+}	Mn^{2+}	Cr^{2+}
Soils of type I								
1.00	35.42 ± 9.37	5.46 ± 1.84	0.27 ± 0.04	6.12 ± 1.47	9.18 ± 6.06	5.30 ± 1.56	61.75 ± 9.66	3.77 ± 0.79
3.33	33.77 ± 8.95	7.13 ± 2.69	0.44 ± 0.19	8.79 ± 2.48	11.70 ± 6.75	2.71 ± 1.14	37.59 ± 3.07	2.95 ± 0.71
Soils of type II								
6.19	59.83 ± 8.96	6.15 ± 2.08	0.37 ± 0.14	8.79 ± 2.32	53.33 ± 20.08	2.38 ± 0.75	33.38 ± 10.15	3.32 ± 1.30
22.78	48.57 ± 12.38	19.28 ±7.39	0.63 ± 0.46	6.91 ± 2.58	35.5 ± 13.06	3.01 ± 1.23	56.34 ± 16.52	6.06 ± 2.18

Table 6. Annual involvement of chemical elements in biological exchange due to the suprasoil phytomass of grassland vegetation, g/m^2 ($M \pm m$)

S_i, rel. units	Chemical elements							
	Zn^{2+}	Cu^{2+}	Cd^{2+}	Pb^{2+}	Co^{2+}	Ni^{2+}	Mn^{2+}	Cr^{2+}
Soils of type I								
1.00	13.36 ± .53	2.06 ± 0.69	0.10 ± 0.02	2.29 ± 0.55	4.54 ± 3.00	2.09 ± 0.62	21.43 ± 3.35	1.39 ± 0.29
3.33	35.40 ± .38	7.08 ± 2.67	0.36 ± 0.15	6.93 ± 1.95	6.06 ± 3.50	2.65 ± 1.12	36.74 ±12.77	2.26 ± 0.55
Soils of type II								
6.19	32.38 ± 10.26	3.43 ± 1.16	0.21 ± .08	4.67 ± 1.24	30.09 ± 1.33	1.33 ± 0.42	17.45 ± 5.31	1.66 ± 0.65
22.78	30.78 ± .85	13.56 ± 5.20	0.51 ± 0.37	4.43 ± 1.66	23.23 ± 8.55	1.76 ± 0.72	36.39 ± 0.67	4.45 ± 1.60

Of course, the increase in the concentrations of chemical elements in the suprasoil phytomass shown in the figures does not exactly correspond to our estimates of the annual return of elements to the biological cycle. Operating together, a decrease in the forb phytomass, whose mineralization rate is higher, in a contamination gradient and an increase in the phytomass of graminoids, whose destruction intensity is lower, in the same gradient enhance the return of most trace elements to biological exchange more significantly than it follows from the content of these elements in the suprasoil phytomass.

For example, the concentrations of Cu and Cd in a suprasoil phytomass in a contamination gradient increase by factors of 3.5 and 2.7, respectively, whereas the amounts these elements mineralized *per annum* increase by factors of 6.6 and 5.0, respectively.

The influence of soil chemical contamination on the intensity of biological exchange of chemical elements can be most obviously illustrated by comparing the ratio of the increase in the toxic load to the examined plots with the corresponding changes in the multiplicity of their involvement in biological exchange (Fig. 6). The fact that the data points plot beneath the bisectrix indicate that an increase in the toxic load is associated with a disproportionate decrease in the involvement of all of the examined chemical elements in biological exchange owing to the annually dying off phytomass. In other words, biological exchange of chemical elements in a contamination gradient is suppressed less strongly than it follows from the integral contamination of the soil.

It is known that up to 90% of the roots of grass is hosted in the uppermost 20–30 cm soil layer (see, for example, Pomazkina et al., 1999, 2008). Bearing this in mind, one can evaluate the total amount of chemical elements available for plants from a soil volume beneath an area of 1 m^2. In this instance, the intensity of mineral exchange in the soil–suprasoil phytomass system can be characterized by the ratio of the amount of elements deposited in the suprasoil phytomass to the content of their biologically available species in the soil horizon.

Figure 7 shows that much elements (more than their concentrations in soils) can be removed into the suprasoil phytomass at low concentrations of these metals but continuous replenishment of their mobile species.

The situation is changed if the soils are intensely contaminated. For example, at Zn and Cu concentra-

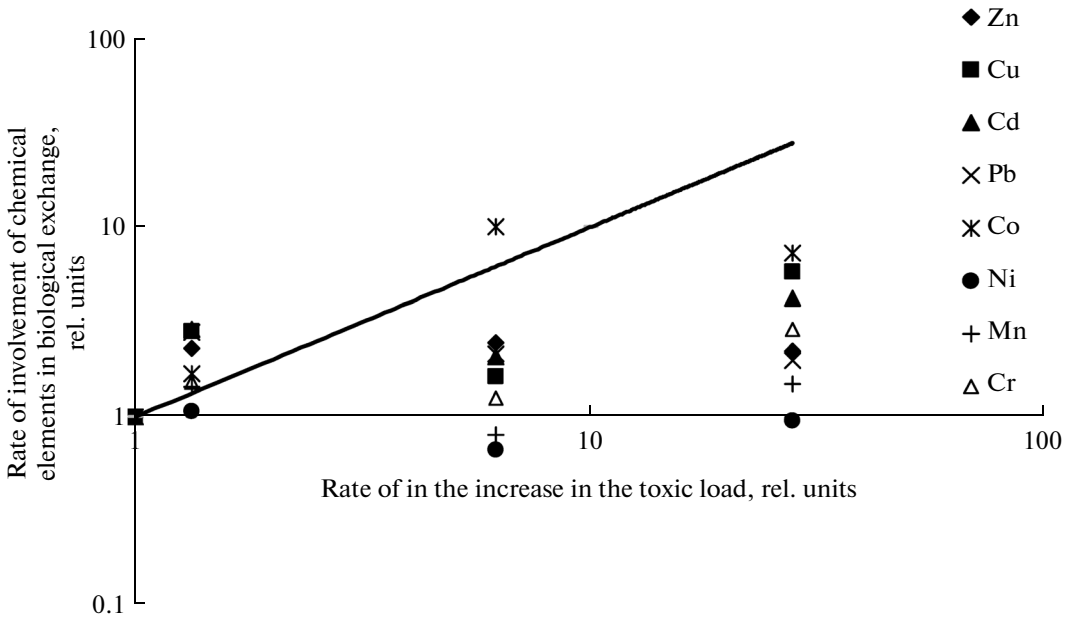

Fig. 6. Rate of biological exchange suppression for chemical elements in a contamination gradient of the soils.

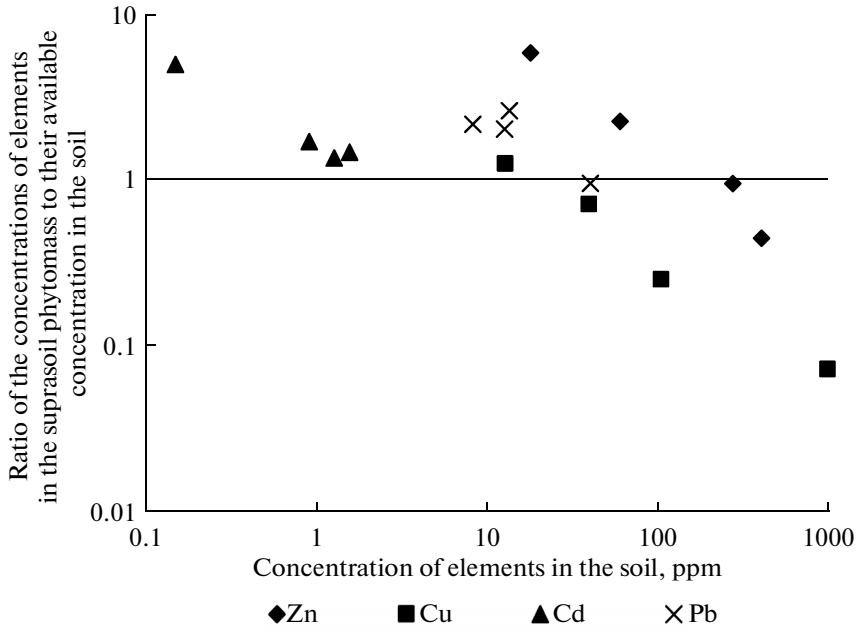

Fig. 7. Amounts of metals involved in annual biological exchange in the suprasoil phytomass relative to the available concentrations of these metals in the soil horizon.

tions in a soil greater than 300 and 40 ppm, respectively, the concentrations of these elements in the suprasoil phytomass are as low as only fractions of their available concentration in the soil. This is explained by the aforementioned partial change in the species composition and a decline in the suprasoil phytomass, which leads to that phytocenosis fails to involve all available elements contained in the soil in biological cycle.

Hence, as the anthropogenic load on natural grass phytocenoses increases, the reaction of the latter tends to partly counterbalance the negative impact by changing the species composition and modifying the mineral-

izing processes in natural phytocenoses to maintain a certain level of biological exchange of chemical elements to preclude excess involvement of toxicants in the biocenosis and thus maintaining the possibility of their operation for a long enough time under anthropogenic contamination of the environment.

CONCLUSIONS

Consequences of chemical contamination of soil are not only elevated concentrations of chemical elements in the suprasoil phytomass of the grass community but also the ensuing suppression of the biological production of the phytocenosis. The fate of the biocenosis is then controlled by the intensity of disturbance of the biological cycle of ash elements, including elements contaminating the environment.

Various agrobotanical groups of a biocenosis differently respond to elevated concentrations of contaminants. The most resistant plants are graminoids and the most susceptible ones are forbs. According to this, the contribution of each group to the intensity of biological exchange is modified in a gradient of chemical contamination.

A principally important factor predetermining the intensity of a biological cycle is processes mineralizing plant remnants. Soils with high concentrations of fertilizer elements of plants, first of all, P, K, and easily hydrolizable N (these are soils of type II in our case) can maintain a high level of mineralization and thus largely compensate the adverse impact of elevated concentrations of contaminants on these processes.

A certain level of biological exchange of chemical elements may be maintained by modifying the species composition and mineralizing processes, thus ensuring the possibility of the long-lasting activity of the biocenoses under chemical stress. The deformation of biological exchange discussed above reflects the conditions under which grass communities occur in the Central Urals in areas contaminated with heavy metals. We discuss the contamination of certain soil types with certain chemical elements, the levels of contamination these soils, specifics of the biocenoses, etc. At the same time, our results are of more general interest and can be analyzed with reference to other natural climatic conditions, qualitative composition of the vegetation, and the intensity of contamination.

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