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CRITICAL LOADS-BASED ECOLOGICAL CONTROL OF HEAVY METAL DEPOSITION IN NATURAL AND ANTHROPOGENIC ECOSYSTEMS: TRIAL STUDY

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ABSTRACT

The limits of acceptable emission of heavy metals are determined on the basis of the method of critical loads into ecosystems of one of the regions of Central Russia. The notion of integral parameter of critical loads elaborated in frames of the theoretical base of classical Russian landscape science is introduced. This parameter essentially supplements currently used methodological approaches to the assessment of critical loads and allows assessing the geochemical stability of ecosystems more fully.

Keywords: Heavy metals, ecosystems' geochemical resistance, method of critical loads, landscape science

INTRODUCTION

Critical loads (CL) define upper limits of the deposition of pollutants into ecosystems, below which no disturbance of natural geochemical balance occurs. CL-based ecological control of pollution is an advanced approach to nature management regulation as opposed to traditional hygienic control based on "anthropocentric" MPC strategy, which does not allow one to take the diversity and spatial mosaicity of ecosystem assimilation potential into consideration (Bashkin, 1997; Manual on Methodologies…, 2004).

The algorithms of calculating critical loads were mostly defined for sulfur, nitrogen and heavy metals' (HM) compounds. However, HM tend to have more intricate routes when passing through ecosystems. As a consequence, at the current level of scientific knowledge, the most models attempting to elaborate the processes of HM's soil and biogeochemical transformation result unfortunately into even more uncertain predictions. This is why the simplified mass-balance equation is commonly used to estimate the geochemical tolerance of baseline forest ecosystems (Conservancy. Urban ecosystems…, 2004):

$$M_{dep} = M_{upt} + M_{leach}, \ldots\ldots\ldots\ldots\ldots (1)$$

where M_{dep} is total deposition of metal, M_{upt} is the uptake of metal due to the woody plants' increase over a year, M_{leach} is the metal drain leaching.

The HM uptake by wood biomass (net uptake) is calculated using the following equation:

$$M_{upt} = G_{an} \times C_{back\,M}, \ldots\ldots\ldots\ldots\ldots\ldots (2)$$

where G_{an} is annual wood biomass production, C_{backM} – the maximum allowable (critical) background concentration of metal in wood. The annual wood biomass increment data calculated for major types of forest ecosystems are usually obtained from books (Bazilevich et al., 1986; Rodin, Bazilevich, 1965; Usoltsev, 2002). Assuming that the accumulation of metals by non-forest species is transient, followed by the mineralization of dead organic matter and its inclusion into biogeochemical cycle, the HM uptake by wood biomass is not taken into consideration within the forestless background territories. Within an agrocenosis G_{an} refers to the volume of agricultural product yielded during harvesting.

The acceptable level of metal drain leaching from an ecosystem is calculated using the following equation:

$$M_{leach} = Q_{runoff} \times C_{waterMPL}, \ldots\ldots\ldots\ldots (3)$$

where Q_{runoff} stands for annual runoff, $C_{waterMPL}$ – maximum permissible level of metal concentration in water.

Considering the HM critical load *(CL(M))* the upper limit of M_{dep} – total deposition of metal in the ecosystem, we obtain:

$$CL(M) = M_{upt} + M_{leach}, \ldots\ldots\ldots\ldots\ldots 4)$$

Hence, the equation (4) suggests that *CL(M)* in the "classic" sense is the sum of permissible biological uptake and permissible drain leaching. The physical interpretation of this equation implies the necessity of total 'recycling' of exogenic elements in the course of geosystem functioning, i.e., during biogenic and abiogenic migration, and any disturbance of established zonal geochemical equilibrium leads to adverse consequences. It is obvious that the excessive exogenic HM can be accumulated in soil, but the entropy of such a system will be permanently increasing, as the lasting demobilization of HM by soil assimilating complex will be unattainable.

Evaluation of maximum permissible concentrations of heavy metals in natural water and phytomass ($C_{waterMPL}$ and C_{backM})

is one of the weak points in CL-modeling. In the context of agrocenosis, one should use the relevant MPCs, but this approach makes no sense when applied to background ecosystems: estimating the HM uptake by wood in terms of health-based exposure limits set for feed and forage is obviously absurd. The "environmental demand" of critical loads methodology implies assessing the anthropogenic influence on natural components based on the information about natural limits of pollutant uptake. Such limits can be determined from samples of sufficient size that describe the geochemical features of landscape environments under conditions as much close to background as possible.

The representative data describing certain geochemical conditions have been collected during field landscape-geochemical research and were used for that purpose. The studies were conducted in the limits of the Ryazan region territory model area within the radius of 50-60 km from the central city of the region. The mentioned area represents a "geochemical focus" of the whole region since 1) all the main soil-and-geochemical environments of the East-European Plain center are represented within the boundaries of this area and the sharpest geochemical boundaries are located here ("boreal ecotone" effect) and 2) this region accommodates major industrial pollutant emitters and emission exposed areas. All these factors allow studying not only the baseline landscape-and-geochemical structure but the extent of anthropogenic transformation of this structure as well. The area of the region covered by the scope of the study made 1,900 km^2; the area of sandy and light loamy soils of Meshchera Lowland (north and northeast) accounts for 41.5% of mentioned 1,900 km^2 and 51.5% is the share of landscapes of the Oka-Don Plain and Central Russian upland south of the Oka river where medium and heavy loamy parent rocks prevail (including 2% of the territory of City of Ryazan), 7% is the area of the Oka river floodplain and its largest tributaries.

In the course of sampling carried out in 2010-2011 97 specimens of wood and 143 samples of surface water were collected (the last sampling was divided into two due to fundamental differences in water exchange and geochemical processes in the context of sand and clay-loam soils). Yield and heavy metal-composition of agrocenotic saleable product were calculated from the data of the state agrochemical monitoring system (annual sampling of reference plots).

Another aim of our study was to validate the acceptable level of annual HM deposition in soil, so 325 soil samples were analyzed. Identification of background soil conditions in old-cultivated areas is impeded by vast atmotechnogenic impact and transformation of soil processes caused by ploughing. Due to that, the procedure of background soil-geochemical sample areas mapping was carried out involving widely used method of zoning objectification – cluster analysis (Ward's method, Euclidean distances). In the course of clusterization and subsequent mapping of the results 4 major background pedogenic regimes were identified (sandy and peat soil; sandy loam soil; sandy clay loam and sandy clay soil; alluvial soil) that are fundamentally different in terms of HM behavior and speciation (*Cu, Zn, Pb, Cd*). "Technogenic" clusters are markedly separated from these areas. Their microelement composition does not always outstand with MPC and EPC of metals, but is unambiguously determined by consequences of pollution (including long-distance air migration of HM); however, related data were certainly ignored in the course of natural soil-geochemical variability limits definition. Nevertheless, composition and configuration of technogenic soil pollution areas (black in Fig.1) is of certain interest, as they represent the heterogeneity of landscape response to external influence. In particular, speaking of long-range impact of industrial emitters, peat soils, downslope and transaccumulative geochemical positions (areas of gullies, swales and lower parts of extended slopes) are particularly vulnerable to pollution. Within bottomland landscapes HM tend to accumulate more intensively in soils of meander scars, areas between low ridges, and sites of superimposed (transgressive) flood plains.

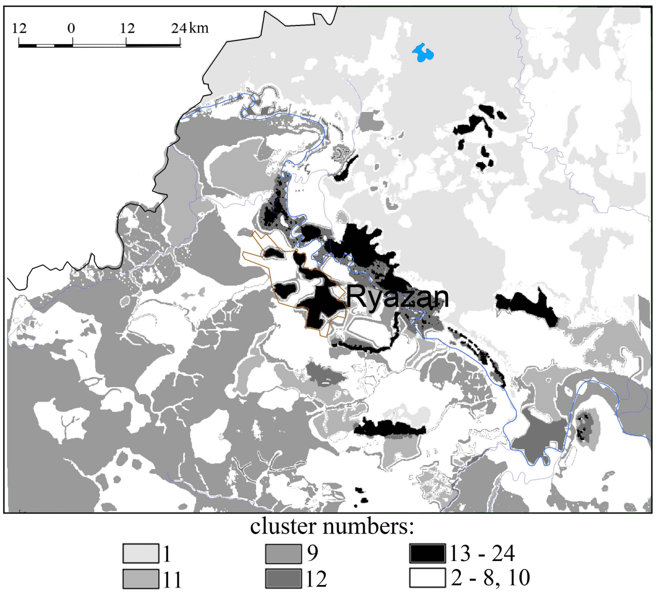


Fig.1. Mapping of background and technogenic soil-geochemical conditions in the centre of the Ryazan region based on the results of cluster analysis

Note. **Cluster 1 - sandy and organic (peat) soils background of Meshchera Lowlands. Cluster 11 - sandy loam soil background of Meshchera and northern Oksko-Donskaya plain. Cluster 9 - sandy clay loam and sandy clay soil background of southern part of the area (chernozemic and gray forest soils). Cluster 12 – flood plain accumulation background. Clusters 13-24 – areas of technogenic soil pollution with heavy metals. Clusters 2-8 and 10 – genetically heterogeneous.**

Variance analysis was applied to the results of natural components' sampling ("technogenic constituent" has been excluded); normalization was carried out if necessary (Puzachenko, 2004). Normalization allowed us to apply the classic "three sigma rule" to the distributions. According to this rule, the probability that one of the values of a normally distributed variable lies beyond three standard deviations of the mean is small enough to be neglected. If such a situation is observed when measuring some parameters of an existing natural object, this object is considered to belong to some other population. We have avoided using this rule as is, considering that the landscape environments vary in their dynamic levels, which, as we see it, should lead to different prior probability thresholds and corresponding normalized deviations. For example, the maximum threshold of an element uptake in soils was associated with the value of $X + 2.576\sigma$ given p=0.99; such dynamic landscape environments as surface waters and phytomass required a higher significance level – p=0.999 and corresponding uptake level of $X + 3,291\sigma$.

The calculated upper limit of uptake was subsequently referred to as "ecological standard" of the element concentration – ES (as an alternative to health-based standards – MPC). The exceedance of ES is the evidence of natural or – in most cases – technogenic geochemical anomaly (Table 1). Sometimes ES turned out to exceed MPC; in this case priority was given to the inferior value. Fig.2 shows the procedure of ES determination, performed using STATISTICA 6.0 software.

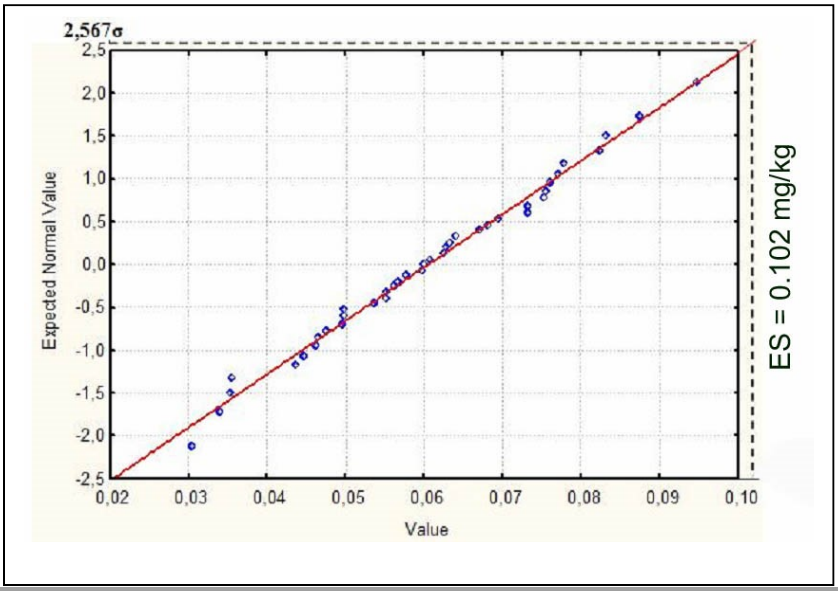


Fig.2 – An example of determining an ecological standard: concentration of cadmium mobile forms in loamy soils

Table 1. Ecological standards of HM concentrations in major landscape components of the Ryazan region and adjacent regions of Central Russia

Object, units			*Speciation*	*Cu*	*Zn*	*Pb*	*Cd*
Surface waters, μg/L (p=0.999)	"sandy" watersheds		dissolved	(1.05)[1]	4.4	8.0	(1.40)[3]
			suspended	3.90	160.0	9.5	0.250
			gross	3.65	160.0	(17.5)[2]	(1.55)[3]
	"loamy" watersheds		dissolved	(1.50)[1]	(50.0)[2]	3.6	(1.26)[3]
			susoended	8.70	49.0	5.0	0.300
			gross	9.70	51.5	4.9	(1.50)[3]
Trunk wood, mg/kg (p=0.999)			gross	3.65	42.0	0.375	0.130
Soils, mg/kg(p=0.99)	*automorphic*	sandy	mobile	0.24	6.0	3.7	0.055
			gross	2.35	165.0	11.4	0.340
		sandy loam	mobile	0.29	6.9	1.4	0.094
			gross	18.20	114.0	15.5	0.315
		sandy clay loam and sandy clay	mobile	0.51	14.0	3.0	0.102
			gross	23.50	(255.0)[4]	17.4	0.365
	peat		mobile	0.32	2.45	5.4	0.078
			gross	17.50	57.0	8.6	0.400
	alluvial loamy		mobile	0.87	(27.3)[5]	4.3	0.210
			gross	28.00	214.0	40.0	0.440

Notes. **HM soil speciation: mobile – extraction with ammonium acetate buffer, pH 4.8; gross – extraction with mixture of concentrated nitric, hydrochloric, sulfur acids and hydrogen peroxide according to Riņķis (Arinushkina, 1971).**

Values in the parentheses represent ecological standards exceeding corresponding MPCs. The following health-based standards should be used instead of them:

[1] 1.0 *μg/L* (MPC_{aq});

[2] 10.0 *μg/L* (MPC_{aq});

[3] 1.0 *μg/L* (MPC_{rw}), standards applied in aquaculture (0.5 *μg/L*) seem to be unattainable in old-cultivated regions;

[4] 220 *mg/kg*, if pH_{KCl}<5.5 – 110 *mg/kg* (EPC);

[5] 2.0 *mg/kg* (MPC).

Therefore, ES make up the most acceptable and unbiased basis to standardizing the concentrations of pollutants in the components of background ecosystems. Specifically, critical background concentrations of metals in wood (C_{backM}) can only be properly estimated using their ES (Table 1). On the contrary, corresponding parameter of agrocenosises must be linked to MPCs. We have used the most "general" standards existing for crops, fodder grain and feeds. Maximum permissible limits of HM in surface waters ($C_{waterMPL}$) were also defined basing upon their ES (that were determined on the basis of hydrochemical sampling data, carried out in July 2010). In case of the latter exceeded the corresponding MPC, priority was given to the smaller value, as shown in Table 1. Otherwise, annual average "effective" HM concentrations were calculated on the basis of ES and zonal summer-to-annual conversion factors (F_c: Table 2), that have been determined by the authors (Krivtsov, Tobratov et al., 2011), were used:

$$C_{waterMPL}^{"eff."} = ES \times Fc \text{ ...... } \textbf{(5)}$$

Note that the annual wood biomass production G_{an} was determined with the help of own biometrical measurements rather than overgeneralized data from published studies, which allowed us to fully take into account local features of existing forest ecosystems.

Table 2 – Conversion factors for transition from low-water (July) HM concentrations in surface waters to annual average "effective" concentrations

Terrestial ecosystem	*Dissolved forms*				*Solidphase forms*				*Gross concentrations*			
	Cu_d	Zn_d	Pb_d	Cd_d	Cu_s	Zn_s	Pb_s	Cd_s	Cu_{gr}	Zn_{gr}	Pb_{gr}	Cd_{gr}
Sub-boreal forest (sandy soils)	2.75	7.29	0.85	0.96	0.44	1.50	4.35	0.79	0.98	2.23	1.15	0.93
Forest steppe (loamy soils)	1.70	2.85	2.60	0.85	0.54	1.55	2.26	1.08	0.73	1.75	2.44	0.88

Note. **"Effective" concentrations, unlike the results of a single sampling, include hydrochemical and hydrodynamical features of all phases of hydroregime.**

It has been emphasized that certain level of soil deposition of metals – the excess of the summation of biological uptake and drain leaching – can be considered acceptable. According to the critical loads method approach, the acceptable deposition rate obviously must not disturb soil-geochemical equilibrium for a "long period of time", which is usually linked to typical duration of ecological succession in the temporate forest ecosystems (about 100 years). At the same time, the possible increase in excessive soil HM mobile forms deposition should be considered, whereas the deposition threshold is set equal to the corresponding ES (Table 1). Hence, we get the following model:

$$SD(M)_{an(acc)} = \frac{ES^{mob} - C_i^{mob}}{100} \times \rho \times 2000,$$

(6)

where $SD(M)_{an(acc)}$ – acceptable annual soil deposition of metal in the upper (0-20 cm) layer, g×ha^{-1}×year^{-1}; ES^{mob} – ecological standard for HM mobile forms soil concentrations, mg/kg; C_i^{mob} - actual HM mobile forms concentration in particular soil, mg/kg; ρ – natural soil density, t/m^3; 2000 – surface area conversion factor.

The idea of acceptable soil deposition of HM allows substantial updating to the critical loads concept by introducing the integral parameter of CL *IPCL(M):*

$IPCL(M) = CL(M) + SD(M)_{an(acc)}$. (7)

This idea is significantly distinct from the "classical" CL concept, as it includes not only biotic and water, but soil component of geosystems as well. The authors of basic CL method have not taken it into consideration, as the western methodology of science is based on the functional, ecosystemic approach, when the targets of research are thought to be "non-dimensional", and their genesis, peculiarities, natural boundaries, unique and typical features are not included into the model. Thus the scientific argumentation of $SD(M)_{an(acc)}$ turns out to be troublesome, so this parameter is ignored, and HM deposition tolerance is underrated. Meanwhile, it is easy enough to justify $SD(M)_{an(acc)}$, using essentially distinct landscape methodology, an important achievement of the Russian science, and integrating it with advanced quantitative zoning methods (cluster and variance analysis).

As we have already mentioned, the most acceptable way to standardize the HM deposition in phytomass is to apply the differentiated approach using health-based standards (MPCs) within agrocenosises and ecological standards within background ecosystems. Despite the significant differences between MPCs and ES (Tables 1 and 3), they are used in the same way during ecological evaluation, which allows us to analyze conditions of both natural ecosystems and agrocenosises with the help of consistent methodological approach.

Table 3 – HM concentrations in phytomass of agrocenosis in the central Ryazan region compared to MPC and average deposition in trunk wood of forest ecosystems, mg/kg.

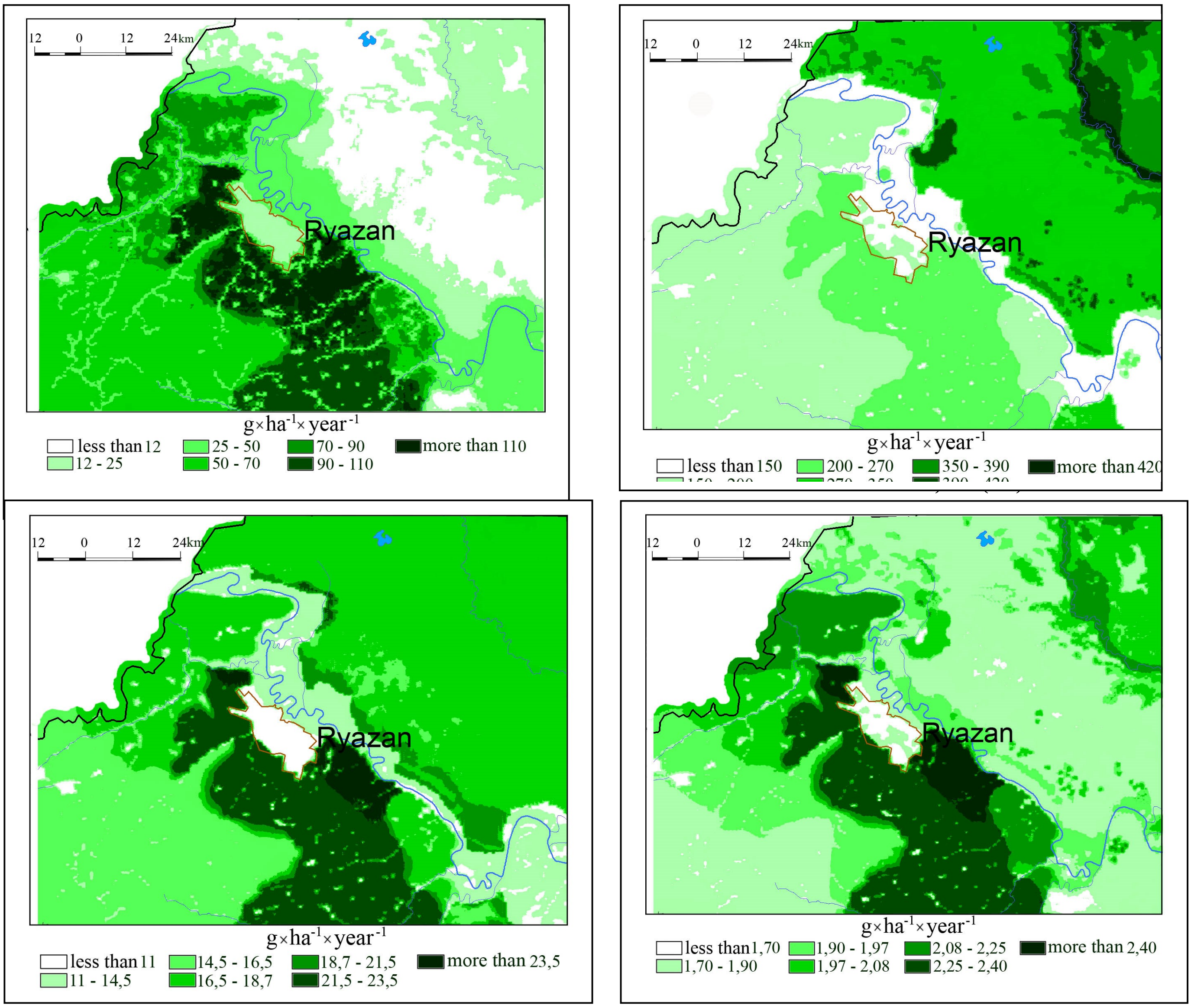
Object; health-based standard		***Cu***	***Zn***	***Pb***	***Cd***
MPC in feeds **(grain and fodder grain, roughage and succulent forages)**		**30.00**	**50.00**	**5.00**	**0.300**
Agricultural production	**wheat, grain**	**2.87**	**22.02**	**0.41**	**0.139**
	barley, grain	**4.19**	**29.84**	**0.40**	**0.088**
	clover, silage	**2.83**	**16.92**	**0.98**	**0.479**
	perennial grasses, haylage and silage	**8.05**	**22.21**	**0.90**	**0.272**
	annual grasses, haylage and silage	**21.01**	**18.19**	**1.03**	**0.460**
	species of flood meadows, motley grass-grasses hay	**5.58**	**28.11**	**3.21**	**0.178**
Trunk wood**, average in the study area**		**1.17**	**13.58**	**0.37**	**0.051**

Spatial patterns of *CL(M)* are largely determined by geochemical specialization of plants. As Table 3 shows, HM concentrations in crop plants are usually several times higher than those in trunk wood of background associations; the most significant differences are observed in concentrations of biophile copper, and cadmium – toxic, but mobile and available for root absorption. Clover and annual grasses (i.e. plants with low barrier characteristics) are biological *Cd*-concentrators; a 1.5-fold level of cadmium compared to MPC is observed within their biomass. It should be noted that *Cd* is the only metal observed concentrations of which may exceed MPC. Despite their geochemical differences, toxic *Pb* and *Cd* share one important feature of their absorption – their transfer into grasses' caryopses is effectively blocked (particularly in loamy soils), reaching the concentrations comparable to trunk wood deposition levels. This data demonstrate the ability of grasses to accumulate elements and serve biological barriers. Crop plants, just like background ones, strive for minimization of *Pb* deposition (which is promoted by its low mobility in soil solutions), therefore the probability of exceeding MPC of *Pb* is virtually the lowest one (unlike another toxic element – cadmium). Motley grasses of flood meadows may be an exception, not least due to the increase in HM fallout at the Oka flood plain, which promotes the *Pb* accumulation; however, the levels of *Pb* still do not exceed its MPC.

We have shown (Tobratov et al., 2007; Krivtsov, Tobratov et al., 2011) that *Zn* can serve as an effective indicator of humidocationic plant species, i.e. adapted to humid conditions and showing high uptake rate of those cations that are mobile in acid medium. As the dominating zonal background vegetation in the region has prominent humidocationic features, the regular zinc concentrations in wood are high. This "high-base effect" is the reason to relatively small difference in *Zn* levels between background and agrocenotic phytomass. Zinc is the geochemical equivalent of cadmium, but it is an active biophilic element, which is the reason to the signs of antagonism in deposition of given elements in cultivated plants (r=-0.88).

Analyzing the values of *CL(M)* one should keep in mind that, unlike nitrogen critical loads, biological deposition is by no means the most crucial factor, as the HM drain leaching plays just as important (or, speaking of toxic elements, even more significant) role. For that reason *CL(M)* values highly depend on the zonal surface waters' ES for metals. As Table 1 shows, the acceptable level of *Pb* accumulation in Meshchera streams is 2-fold (3-fold for *Zn)* increased compared to the conditions of "loamy" watersheds in southern part of the region; *Cu* has inverse ratio, and *Cd*-ES are identical throughout the region and are equal to MPC_{rw}. These aspects result into 2-fold *CL(Pb)* for background ecosystems of Meshchera compared to those south of Oka, *Zn* parameters are as much as 3 times different, but Meshchera landscapes are 2-3 times less tolerant to copper pollution (particularly in the "central trough" of Meshchera plain hydromorphic conditions); no significant interregional variations in *CL(Cd)* are observed.

Agrogenic transformation of *CL(M)* is generally less prominent than of sulfur and nitrogen CLs: substitution of background vegetation for more productive agricultural plants usually makes geosystems more tolerant to metal pollution; however, the maximum gain is usually no greater than 2-fold raise, and sometimes it is just several percent of background values. The highest increase typically occurs in copper tolerance, as its deposition in cultivated plants has the most contrast pattern. Negative agrogenic transformation of *Zn* critical loads may sometimes be observed, especially if birch and other humidocationic plant species have been widespread in background ecosystems. The most prominent increase in tolerance is typical for the most productive "suburban" agrocenosises compared to zonal background of forest ecosystems.



c) *CL(Pb)* d) *CL(Cd)*

Fig.3 – Critical loads of heavy metals *CL(M)* in natural and natural-anthropogenic geosystems in the central Ryazan region

So, the *CL(M)* values integrally represent the influence of hydrodynamics, trunk wood and agrocenosises' production, and upper threshold of HM deposition in phytomass and surface waters, which have been analyzed and considered to be acceptable on the basis of ES or MPCs. Fig.3 shows landscape spatial patterns of *CL(M)* resulting from combined impact of the above-noted factors.

Table 4 contains data on *CL(M)* exceedance in the study area. The table shows that lead and cadmium are the essential pollutants in the region. Average input for biophilic *Cu* and *Zn* is 2-3 times lower than CL, but it is, by contrast, more than 2-fold higher for *Pb* and *Cd.* Hence, neither biological barriers nor leaching systems of the landscapes are able to recycle the annual amounts of *Pb* and *Cd* coming from atmosphere without any aftereffects. Unfortunately, every long-cultivated area of central Russia faces these issues.

Table 4 – Estimated values of HM critical loads CL(M) in the landscapes of the central Ryazan region compared to average annual HM fallout in the area, $g \times ha^{-1} \times year^{-1}$

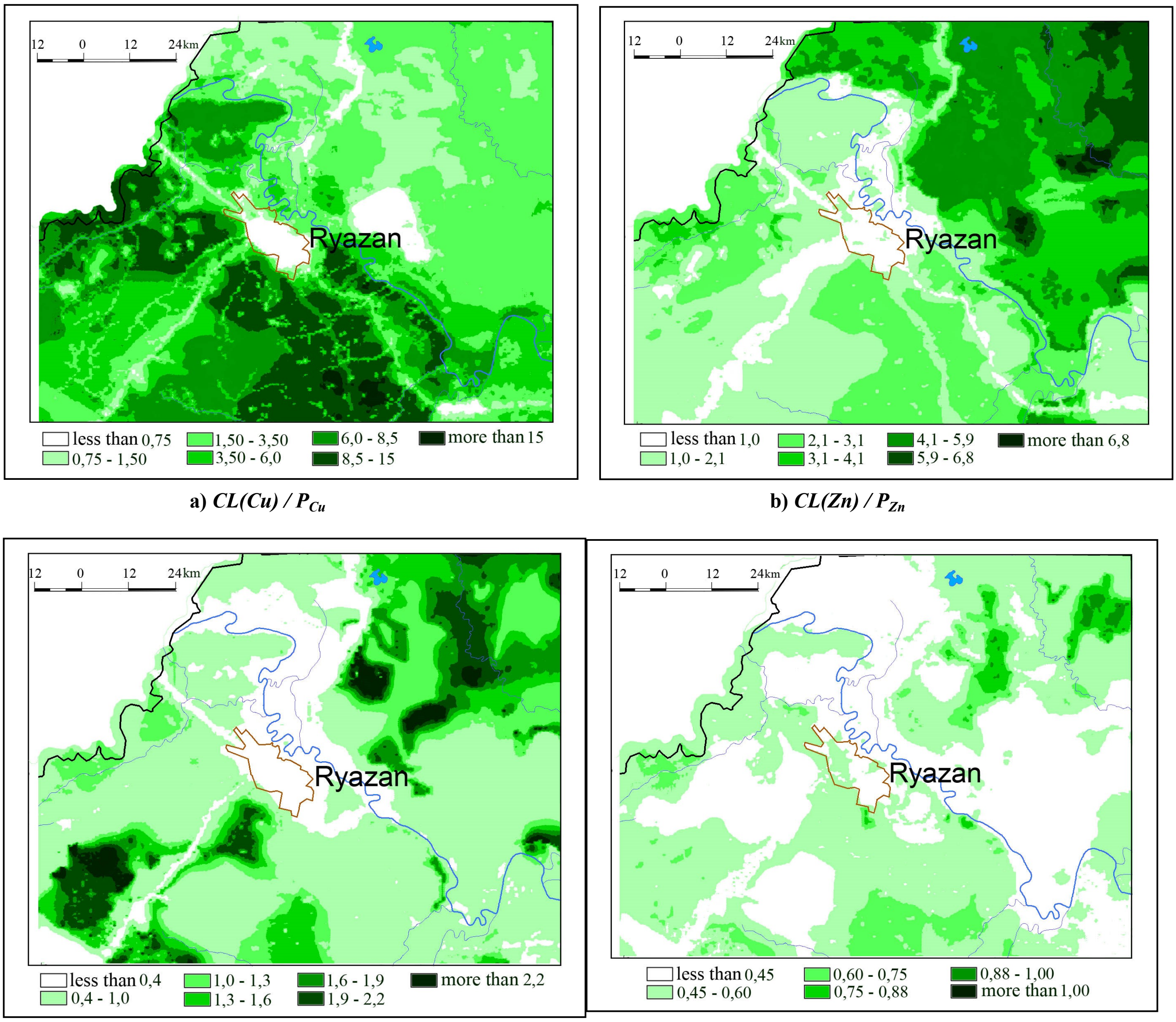
Parameter	Cu	Zn	Pb	Cd
CL(M)	**46.72**	**234.74**	**17.00**	**1.98**
Gross atmospheric input	**12.50**	**100.80**	**36.90**	**4.25**

As Fig.4 shows, HM fallout is higher than the corresponding *CL(M)* for all the elements studied, but the exceedance rates substantially vary. Generally, the observed situation results from the same technogenic factors within identical areas, thus tracing major routes of pollutant atmospheric migration. The list of such territories includes the regional center area, zones of major highways' impact, the site within the Oka flood plain adjacent to Ryazan, which is canalizing main technogenic routes, and neighboring territories, south-eastern Meshchera (light-color bordered zones in Fig.4). The total area of such zones is minimal for *Cu* and *Zn,* whereas for *Pb* these areas are dominant with almost no sites of $CL(Cd)>P_{Cd}$.

Thus, current levels of *Pb* and *Cd* – absolutely non-biophilic elements – fallout is obviously excessive both for background and for man-modified landscapes. Certain increase in *CL(M)* in agrocenosises compared to forest ecosystems cannot compensate for this excessive technogenic input. The potential risk of negative geochemical impact of *Cu* and *Zn* is much lower, and the areas of their excessive input are quite limited.

However, the *CL(M)* calculated on the basis of "classical" model (4) undervaluate the geochemical tolerance in ecosystems, as the soil stabilization capacity is not taken into consideration.

There is a good reason to calculate the integral CL, as shown in equation (7), which includes an additional summand $SD(M)_{an(acc)}$ - acceptable annual soil HM deposition.



c) $CL(Pb) / P_{Pb}$ d) $CL(Cd) / P_{Cd}$

Fig.4 – HM critical loads *(CL(M))* to matching annual fallout *(P_M)* ratio in natural ecosystems and agrocenosises of central Ryazan region (higher values represent higher geochemical tolerance).

According to the data shown on fig.5 and in table 5, $SD(M)_{an(acc)}$ inclusion actually leads to substantial increase in HM critical loads, which is comparable to the values of *CL(M)* themselves in case of *Zn* and *Cd,* is 4 times higher in case of *Pb* and 4 times lower in case of *Cu*. The reason to these differences is the deficiency of *Cu* – one of the scarcest biogenic elements in central Russia – which leads to high rates of its uptake by plants and its consolidation within the NTCs, hence only total *Cu* levels higher than 30 *mg/kg* may increase the element's mobility (Kasimov et al., 1995).

Therefore, significant concentrations of mobile copper are not typical for soils in the area, so the ES^{mob} – C_i^{mob} differences are low. Lead is much more labile and, at the same time, its mobile forms can reach quasi-steady state even in light sandy soils, whereas in aqueous media an active transition to colloid and suspended migration phase is observed as the ionic strength of solution increases (Krivtsov, Tobratov et al., 2011). The main reason for these features of *Pb* behavior is its distinct capacity for specific adsorption by humic acids and effective competition with other HMs for reactive centers (Ladonin, Margolina, 1997). Therefore, the acceptable annual soil deposition of *Pb* is particularly high (up to 80% of *IPCL)*, so the integral tolerance of geosystems to its input, considering $SD(M)_{an(acc)}$, is significantly higher – twice as its average atmospheric input levels. However, the "degree of safety" of geosystems in case of *Pb* fallout is still much lower than those of biophilic *Cu* and *Zn*, which, in turn, are very similar in this parameter (Fig.5). Both CL and IPCL for cadmium, which is the most mobile and toxic element, turn out to be lower than the average fallout magnitude, which means that *Cd* is the first-place pollutant in the landscapes of the region.

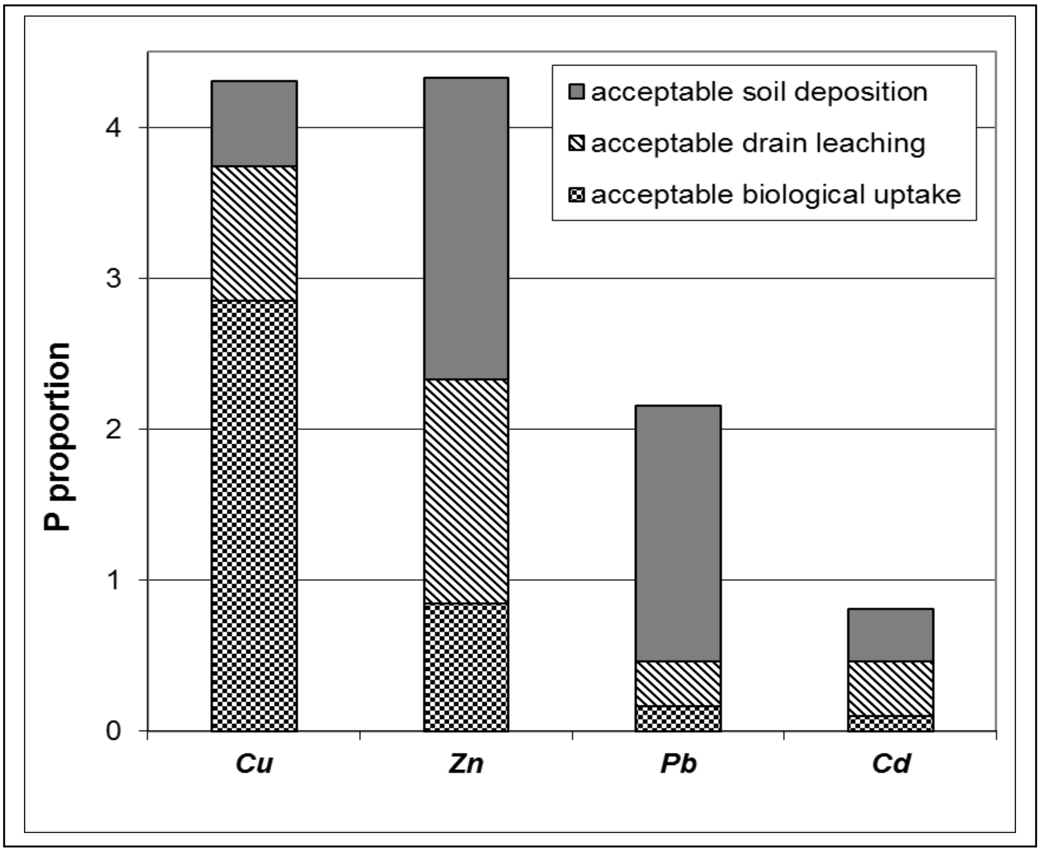


Fig.5 – Proportions of heavy metals' *IPCL* components expressed in terms of average levels of their atmospheric input in the region *(P)*.

Table 5 – Integral parameters of HM critical loads *IPCL* (average for the landscapes of central Ryazan region) compared to average fallout levels in the area, $g \times ha^{-1} \times year^{-1}$.

Parameter	*Cu*	*Zn*	*Pb*	*Cd*
IPCL	**53.77**	**435.74**	**79.56**	**3.45**
Gross atmospheric input	**12.50**	**100.80**	**36.90**	**4.25**

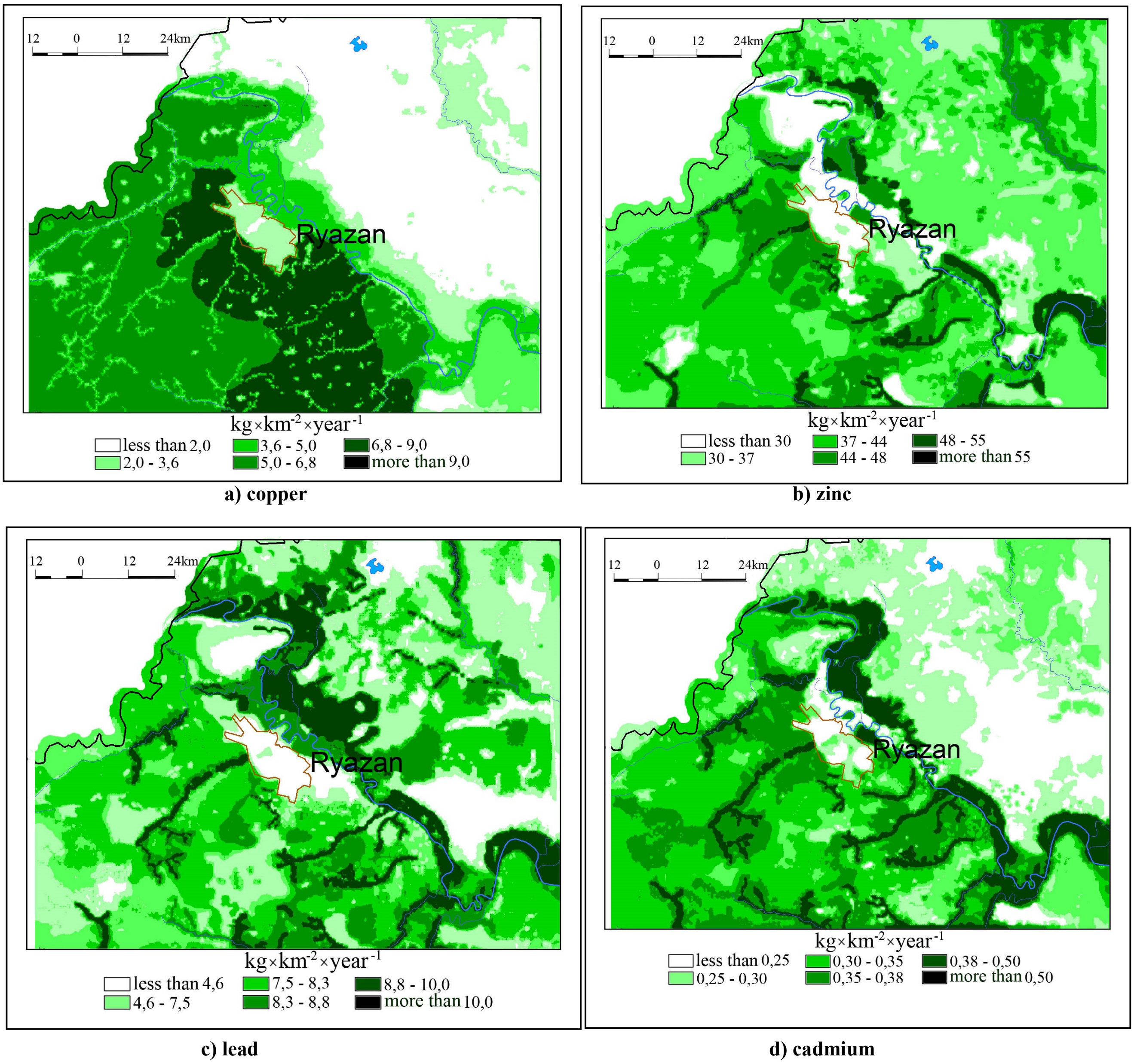


Fig. 6 – Integral parameters of critical loads of heavy metals: *IPCL(M)*

IPLC(M) expressed in terms of specific acceptable technogenic input of metals (Fig.6) give the opportunity to plan the management of natural resources in the central Ryazan region considering the spatial heterogeneity of landscape assimilatory potential. According to the cybernetic basics of landscape theory, both scale and rate of transformation are important, which, in case of transition from evolutionary to revolutionary development, will probably lead to irreversible degradation of systems and loss of valuable information (Armand, 1975). This is the reason why natural ecosystems avoid "revolutionary" transformations, which gives them an opportunity for progressive development. Therefore, sustainable technological development must not impose such "revolutionary entropy" to the system and must stay within natural limits of annual pollutant input, defined by *IPCL(M)* values. Data shown in Fig.6 represent such annual fallout levels that will allow avoiding disturbance of the soil-geochemical equilibrium, degradation of landscapes' biological components and pollution of surface and subsoil waters.

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