ISSN 1067-4136, Russian Journal of Ecology, 2014, Vol. 45, No. 6, pp. 498–507. © Pleiades Publishing, Ltd., 2014.
Original Russian Text © E.L. Vorobeichik, M.R. Trubina, E.V. Khantemirova, I.E. Bergman, 2014, published in Ekologiya, 2014, No. 6, pp. 448–458.

Long-term Dynamic of Forest Vegetation after Reduction of Copper Smelter Emissions

E. L. Vorobeichik, M. R. Trubina, E. V. Khantemirova, and I. E. Bergman

Institute of Plant and Animal Ecology, Ural Branch, Russian Academy of Sciences, ul. Vos'mogo Marta 202, Yekaterinburg, 620144 Russia

e-mail: ev@ipae.uran.ru

Received May 7, 2014

Abstract—The state of tree and ground vegetation layers in spruce–fir forests around the Middle Ural Copper Smelter (Revda, Sverdlovsk oblast) has been repeatedly evaluated in 25 permanent sampling plots at 5- to 10-year intervals (1989–2013). The results have been used to characterize the dynamics of plant communities in the period of reduction of emissions from the smelter. Although the annual amount of emissions has decreased from $150–225 \times 10^3$ t in the 1980s to less than 5×10^3 t after 2010, the vegetation in the impact zone (1 and 2 km from the smelter) remains severely suppressed: the trees continue to die off, and the diversity of ground vegetation layer is very low. In zones with low and moderate levels of industrial pollution (30 and 4–7 km from the smelter), natural factors associated with windfall disturbance after the 1995 windstorm with snow have played a more important role in the dynamics of forest communities than the reduction of emissions itself.

Keywords: industrial pollution, air pollution, heavy metals, copper smelter, reduction of emissions, forest ecosystems, plant communities, tree stand, ground vegetation, biodiversity, biomass, dynamics of recovery, southern taiga, the Middle Urals

DOI: 10.1134/S1067413614060150

In the past few decades, the amounts of industrial emissions in many economically developed and some developing countries have decreased due to the improvement of technologies and reduction in the volume of production or relocation of production facilities. The consequent decrease in the toxic load on ecosystems provides the possibility to analyze processes of their natural recovery, which have not been sufficiently studied to date. Special publications on demutational successions are relatively few. Some of them deal mainly with the tree stand (Vavrova et al., 2009; Zverev, 2009; Jonard et al., 2012), including tree radial increment (Juknys et al., 2003; Danek, 2007; Chernen'kova and Bochkarev, 2013); others, with the ground vegetation layer (Trubina and Makhnev, 1997; Chernen'kova et al., 2001, 2011; Vidic et al., 2006; Vavrova et al., 2009; Lyanguzova and Maznaya, 2012).

Such studies are of theoretical significance, since they provide material for analyzing mechanisms of resilience as the ability of a system to return to the initial state. They are also important from the applied aspect, since knowledge of trends and rates of ecosystem recovery dynamics is required for choosing the optimal strategy of nature management and making valid decisions concerning the necessity and type of recultivation measures. These factors account for increasing interest in the problem at issue (Gunn et al., 1995; Vavrova et al., 2009; Kalabin and Moiseenko, 2011; Chernen'kova and Bochkarev, 2013).

However, there are some principal methodological limitations to studies on the recovery dynamics. The main limitation is the absence or deficiency of data on the state of the biota prior to reduction of technogenic impact. The correct comparison of such data for different periods of time is possible only if they have been recorded strictly at the same points; otherwise, the spatial variation of test parameters in space may be erroneously interpreted as their change with time. This aspect is especially important in impact territories, since spatial variation both in the amount of toxic load and in the species composition and structure of plant communities sharply increases under heavy pollution (Trubina and Vorobeichik, 2012).

Another limitation is that it is difficult to reveal the causes of observed changes. A rigorous proof that the dynamics of vegetation have been conditioned by reduction of emissions rather than by other factors (internal or external) is an extremely complex task. To accomplish this task, it is necessary to analyze changes in toxic load, taking into account not only the input of pollutants from the atmosphere but also their deposition and translocation in the soil. Second, analysis of vegetation demutation dynamics should be combined with characterization of climatic changes and weather anomalies, which can markedly modify the course of succession and even play a more important role as its drivers that the reduction of emissions itself. In other words, it is necessary to perform a coupled analysis of several time series: of the status of vegetation, toxic load, and weather conditions. In an ideal case, such series should consist of large numbers of points to allow the researcher not only to describe trends in the observed changes but also to analyze relationships between them.

With rare exceptions (Vavrova et al., 2009; Zverev, 2009; Lyanguzova and Maznaya, 2012), sampling plots for repeated surveys do not coincide with the initial plots, and information on the state of the biota before the reduction of emissions is sometimes absent. Analysis of vegetation recovery dynamics in available publications is usually based on comparisons of only two time sections, before and after the reduction of emissions.

There is as yet no consensus on the rate of demutational successions triggered by the reduction of emissions. According to some authors, decrease of technogenic impact is followed by rapid recovery of ecosystems, but others consider that the biota "by inertia" remains suppressed for a long time, even in the absence of emissions. The inertial hypothesis was formulated on the basis of experiments with simulation models (Tarko et al., 1995). Empirical data confirming or disproving it are scarce and controversial. Some studies provide evidence for the absence of any positive trend after the reduction of emissions, e.g., in demographic parameters of birch (Zverev, 2009) and bilberry (Lyanguzova and Maznaya, 2012); in other studies, conversely, the authors have noted relatively rapid recovery of parameters such as tree radial increment (Juknys et al., 2003; Chernen'kova ad Bochkarev, 2013) and the abundance of ground vegetation (Chernen'kova et al., 2001, 2011).

The purpose of this study was to analyze the dynamics of forest plant communities after the reduction of emissions from a large point polluter. In its course, we checked two hypotheses: (1) in zones with a low or moderate pollution level, natural factors may produce a stronger effect on the dynamics of vegetation than the reduction of emissions itself, and (2) the vegetation in the zone of heavy pollution continues to deteriorate or remains suppressed for a long time, even after almost complete cessation of emissions.

Our studies on forest ecosystems in the region of the Middle Ural Copper Smelter, one of the most powerful sources of pollutant emissions to the atmosphere (which have been strongly reduced to date), were started in the late 1980s. One of their purposes was to assess the state of forest vegetation (Vorobeichik et al., 1994; Vorobeichik and Khantemirova, 1994) and soils (Kaigorodova and Vorobeichik, 1996). The state of plant communities was repeatedly evaluated in permanent sampling plots at 5- to 10-year intervals over an almost 25-year period. The dynamics of toxic load were analyzed taking into account not only the

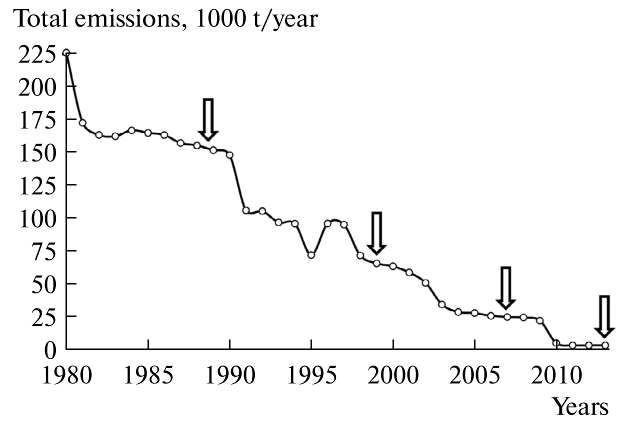

Fig. 1. Dynamics of total emissions from the MUCS over the period of 1980 to 2013. Arrows indicate years when the state of ground vegetation was evaluated.

amount of emissions to the atmosphere but also changes in the concentrations of heavy metals in the soil (Trubina et al., 2014). Data on weather conditions over the study period were also included in analysis.

STUDY REGION

Studies were performed in the region exposed to emissions from the Middle Ural Copper Smelter (MUCS) located in the suburbs of Revda, 50 km from Yekaterinburg. The main toxic components of emissions are gaseous compounds of sulfur, fluorine, and nitrogen and dust particles with adsorbed heavy metals (Cu, Pb, Zn, Cd, Fe, Hg, etc.) and metalloids (As). The MUCS has been in operation since 1940. The total amount of emissions in the 1980s reached 150–225 × 10^3 t/year, but it was reduced to 65 × 10^3 t/year at the turn of the century and to 25 × 10^3 t/year by the mid-2000s (decade). A crucial overhaul of the smelter in 2010 resulted in its further reduction to less than 5 × 10^3 t/year (Fig. 1).

The territory west of MUCS (opposite to the prevailing wind direction) was surveyed. By the criteria of heavy metal contents in accumulating substrates and the state of higher vegetation, three zones with different toxic loads were delimited: the impact, butter, and background zones (up to 2 km, 7 km, and father than 7 km west of the polluter) (Vorobeichik et al., 1994).

According to physiographic zoning, the study region is in the southern taiga subzone of the low-mountain province of the Middle Urals with elevations of 100 to 450 m a.s.l. (Prokaev, 1976). Climatic conditions (1960–2010) are as follows: annual average temperature +1.7°C, the lowest monthly (January) temperature –15.0°C, the highest monthly (June) temperature +17.7°C, frost-free period less than 90 days; annual average precipitation 540 mm, maximum snow depth 40–50 cm. The amount of forests in the region exceeds 60%, with prevalence of dark conif-

Table 1. Dynamics of tree stand parameters in zones with different toxic loads, $M \pm SE$ (sample plot is an accounting unit, $n = 5$)

Parameter	Year	Zone (distance from smelter, km)				
		background (30)	buffer (7)	buffer (4)	impact (2)	impact (1)
Stand density, ind./ha	1989	2048 ± 251	2086 ± 108	1318 ± 448	1450 ± 125	822 ± 126
	1998	858 ± 86	1226 ± 49	701 ± 86	1094 ± 206	365 ± 60
	2008	1104± 117	1155 ± 69	1184 ± 262	1997 ± 133	464 ± 257
Standing volume, m^3/ha	1989	409.2 ± 53.9	418.1 ± 44.0	313.1 ± 26.4	149.3 ± 22.1	63.0 ± 9.2
	1998	253.1 ± 29.4	321.2 ± 19.6	228.5 ± 25.1	139.6 ± 24.5	38.2 ± 6.8
	2008	438.9 ± 61.5	526.0 ± 32.3	394.5 ± 23.5	301.2 ± 14.9	74.2 ± 57.2
Percentage of snags number in tree stand, %	1989	7.8 ± 4.2	7.0 ± 1.2	16.5 ± 3.2	25.2 ± 2.3	26.8 ± 4.7
	1998	7.4 ± 2.9	12.4 ± 0.7	18.7 ± 4.9	7.8 ± 2.7	34.4 ± 4.2
	2008	22.3 ± 1.2	18.7 ± 5.7	4.1 ± 1.9	–	80.4 ± 8.2

A dash indicates the absence of data.

erous and secondary birch and aspen forests (Prokaev, 1976). Brown, gray, and soddy podzolic soils prevail in the soil cover.

Surveys were made in uneven-aged spruce–fir forests with elements of the nemoral floristic complex on flat hill slopes with soddy podzolic heavy loam soils, moderately deep and acidic (in the background zone, pH_{water} 4.5–5.0 in humus horizon).

The tree stand dominated by Siberian spruce (*Picea obovata*) and Siberian fir (*Abies sibirica*) included single birch (*Betula pendula*), pine (*Pinus sylvestris*), and aspen trees (*Populus tremula*). In most cases, tree age in the overstory averaged 60–80 years (in 1989). Tree height and diameter were maximal in the background zone (18–20 m and 18–23 cm) and minimal in the impact zone (9–12 m and 9–14 cm, respectively). The undergrowth consisted mainly of rowan (*Sorbus aucuparia*), Siberian elderberry (*Sambucus sibirica*), bird cherry (*Padus avium*), and linden (*Tilia cordata*). The shrub layer was represented by scattered raspberry (*Rubus idaeus*), fly honeysuckle (*Lonicera xylosteum*), and prickly wild rose plants (*Rosa acicularis*).

In 1989, dominant and codominant species in the ground vegetation layer were as follows: *Oxalis acetosella, Aegopodium podagraria*, *Gymnocarpium dryopteris*, *Dryopteris carthusiana*, *Asarum europaeum*, *Maianthemum bifolium*, *Cerastium pauciflorum*, and *Stellaria holostea* in the background zone; *Oxalis acetosella*, *Cerastium pauciflorum*, *Maianthemum bifolium*, *Carex montana*, *Calamagrostis obtusata*, *Rubus saxatilis*, and *Linnaea borealis* in the buffer zone; and *Equisetum sylvaticum*, *Deschampsia cespitosa*, *Tussilago farfara*, *Agrostis capillaris*, *Maianthemum bifolium*, *Calamagrostis arundinacea*, and *C. langsdorffii* in the impact zone.

MATERIAL AND METHODS

In 1989, 25 permanent sample plots (25 × 25 m) were established at different distances to the west of the MUCS: 1 and 2 km (impact zone), 4 and 7 km (buffer zone) and 30 km (background zone), five plots per distance. The state of tree stand was repeatedly assessed in 1998 and 2008; of ground vegetation layer (GVL), in 1999, 2007, and 2013. Thus, our observations covered a period of almost 25 years, during which the amount of emissions from the MUCS decreased gradually (Fig. 1). This allowed us to obtain information on the status of vegetation at times of heavy emissions (1989), their significant reduction (1999), and almost complete cessation (2007 and 2013).

All test parameters were recorded strictly on the same plots, except for two plots damaged by fire in the impact zone (2 km), where two new plots were laid out in 1998. Therefore, data for 2-km distance were excluded from ANOVA comparison between 1989 and 1998 (see Tables 2, 4).

Total tree surveys with measurements of tree height and diameter were performed in 1989, 1998, and 2008 to determine stand density, standing volume, and the proportion of dead wood in it. For correct comparison of the respective data, which were obtained using slightly different techniques, mensuration descriptions made in different years were recalculated by the standard method (Usol'tsev and Zalesov, 2005).

Geobotanical descriptions (relevés) were made to evaluate the floristic composition and species richness of GVL. The latter parameter was evaluated on three spatial scales: microscale (the average number of species per 50 × 50-cm square, or 0.25 m^2); mesoscale (the average number of species per 25 × 25-m sample plot, or 625 m^2); and macroscale (the total number of species in all five plots located at the same distance from the MUCS, or in 3125 m^2). Squares for microscale analysis were randomly distributed over sample

Table 2. Results of ANOVA with repeated measures for differences in tree stand parameters between toxic load zones and between years (F-test and, in parentheses, significance level)

Parameter	Years	Source of variation		
		zone	time	zone × time
Stand density	1989–1998	30.5 (<0.001)	85.0 (0.001)	1.0 (0.419)
	1998–2008	9.0 (0.001)	5.5 (0.031)	2.2 (0.115)
Standing volume	1989–1998	132.8 (0.001)	14.7 (0.001)	0.4 (0.774)
	1998–2008	28.6 (0.001)	15.4 (0.001)	1.8 (0.165)
Percentage of snags number in tree stand	1989–1998	11.0 (0.001)	1.4 (0.252)	0.7 (0.579)
	1998–2008	6.2 (0.022)	2.3 (0.177)	8.0 (0.011)

Table 3. Dynamics of parameters of ground vegetation layer in forest communities during the period of 1989 to 2013, $M \pm SE$ (sample plot is an accounting unit, $n = 5$)

Parameter	Year	Zone (distance from smelter, km)				
		background (30)	buffer (7)	buffer (4)	impact (2)	impact (1)
S_1	1989	7.6 ± 1.2	3.9 ± 0.5	3.8 ± 0.4	0.6 ± 0.1	0.8 ± 0.1
	1999	11.4 ± 0.7	5.4 ± 0.4	4.9 ± 0.4	0.9 ± 0.3	1.7 ± 0.3
	2007	10.0 ± 0.3	4.9 ± 0.3	3.9 ± 0.2	0.9 ± 0.3	1.3 ± 0.2
	2013	11.0 ± 0.6	4.7 ± 0.4	3.2 ± 0.3	0.7 ± 0.1	1.3 ± 0.2
S_2	1989	36.8 ± 2.4	32.2 ± 1.2	24.8 ± 1.8	9.2 ± 0.6	7.0 ± 1.4
	1999	53.2 ± 1.9	39.0 ± 1.8	31.4 ± 3.6	11.6 ± 1.6	6.4 ± 1.2
	2007	58.2 ± 0.9	44.0 ± 1.8	31.8 ± 3.2	14.0 ± 2.0	7.0 ± 0.6
	2013	60.8 ± 1.2	41.4 ± 1.3	28.2 ± 2.2	12.6 ± 2.2	6.8 ± 0.3
Total biomass, g/m^2	1989	16.5 ± 3.0	7.3 ± 1.7	18.6 ± 4.8	6.7 ± 2.0	16.5 ± 7.8
	1999	30.2 ± 2.8	14.6 ± 0.3	14.3 ± 1.8	4.8 ± 2.2	15.7 ± 2.2
	2007	52.2 ± 6.9	17.9 ± 3.7	16.5 ± 1.9	2.9 ± 1.7	44.5 ± 21.1
	2013	71.6 ± 7.1	22.5 ± 6.4	9.4 ± 1.3	2.2 ± 1.2	25.8 ± 9.6

Here in Table 4, S_1 is the number of species per 0.25 m^2, and S_2 is the number of species per 625 m^2.

plots (15 squares per plot in 1989 and 1999, and 35 squares in 2007 and 2013). The aboveground biomass of GVL was assessed at the peak of development (mid-July) by hay-harvest method in 50 × 50-cm squares randomly distributed over sample plots (15 squares per plot in 1989 and 1999, and 10 squares in 2007 and 2013). The samples were sorted out by species, air-dried in an oven at 80°C for 24 h, and weighed with an accuracy of 0.01 g.

Data on weather conditions over the period of 1960 to 2010 were obtained from the nearest weather station in the city of Revda. The dynamics of emissions from the MUCS (Fig. 1) were reconstructed using information from different sources, including annual state reports on the state of natural environment in Sverdlovsk oblast (1994–2012), publications on the MUCS official site (www.sumz.umn.ru/ru/about/ecology), and the book by Yusupov et al. (1999). The amounts of emissions over the period of 1980 to 1985 were calculated from total values for the city of Revda (given in the paper by Kozlov et al., 2009), taking into account the ratio between MUCP and city's emissions in 1986.

The data were analyzed by periods: from 1989 to 1999, from 1999 to 2007, and from 2007 to 2013. Statistical significance of differences in parameters of vegetation between toxic load zones and years during each period was tested using two-way ANOVA with repeated measures. Before analysis, parameter values were transformed logarithmically ($y = \ln(x + 1)$) or by square root.

RESULTS

Dynamics of tree stand. Parameters of tree stand changed considerably between 1989 and 1999 (Table 1): its density and standing volume decreased by 20–40% synchronously in all plots (the effect of factors "zone" × "time" was nonsignificant, Table 2), while the proportion of deadwood remained unchanged. In 1999 to 2008, stand density and standing volume increased in all zones, eventually reaching the values recorded in

Table 4. Results of ANOVA with repeated measures for differences in parameters of ground vegetation layer between toxic load zones and between years (*F*-test and, in parentheses, significance level)

Parameter	Years	Source of variation		
		zone	time	zone × time
S_1	1989–1999	138.4 (<0.001)	15.8 (0.001)	0.7 (0.595)
	1999–2007	123.1 (<0.001)	3.0 (0.101)	1.9 (0.157)
	2007–2013	188.9 (<0.001)	1.7 (0.214)	3.4 (0.033)
S_2	1989–1999	187.5 (<0.001)	13.0 (0.002)	3.0 (0.062)
	1999–2007	109.6 (<0.001)	6.9 (0.018)	0.4 (0.823)
	2007–2013	181.5 (<0.001)	0.6 (0.438)	1.1 (0.412)
Biomass:				
total	1989–1999	3.1 (0.059)	5.5 (0.033)	1.5 (0.249)
	1999–2007	20. 9 (<0.001)	2.8 (0.115)	1.0 (0.427)
	2007–2013	19.3 (<0.001)	1.4 (0.257)	0.9 (0.479)
grasses and sedges	1989–1999	2.9 (0.069)	32.1 (<0.001)	3.8 (0.030)
	1999–2007	4.7 (0.010)	11.1 (0.004)	1.9 (0.152)
	2007–2013	5.4 (0.005)	7.6 (0.013)	1.1 (0.406)
herbs	1989–1999	81.2 (<0.001)	8.1 (0.012)	3.5 (0.041)
	1999–2007	171.5 (<0.001)	3.7 (0.072)	10.0 (<0.001)
	2007–2013	99.5 (<0.001)	1.7 (0.204)	2.2 (0.115)
ferns	1989–1999	12.7 (<0.001)	2.9 (0.107)	0.9 (0.447)
	1999–2007	5.6 (0.005)	8.4 (0.010)	1.3 (0.302)
	2007–2013	15.8 (<0.001)	4.0 (0.063)	1.3 (0.316)
horsetails	1989–1999	23.8 (<0.001)	2.7 (0.123)	0.3 (0.829)
	1999–2007	14.1 (<0.001)	2.1 (0.167)	4.9 (0.008)
	2007–2013	6.4 (0.002)	7.2 (0.015)	2.0 (0.140)

1989 (Table 1). The proportion of deadwood increased in all zones, especially in the impact zone (up to 80%), but this increase was asynchronous (Table2).

Dynamics of ground vegetation layer. ***During the first period*** (1989–1999), the species richness of GVL changed significantly (Table 3), with changes in the background and buffer zones differing from those in the impact zone (Table 4). A common trend in the first two zones was that the number of species proved to increase on all spatial scales (Fig. 2), whereas such an increase in the impact zone was observed on the microscale but not on meso- and macroscales. In other words, this increase occurred not in species richness itself but rather in species density within small areas (sized in tens of centimeters across), which could be due to increase of abundance and/or more uniform spatial distribution of resident plant species.

Changes in the species composition of GVL in plant communities of background and buffer zones, as a rule, were related to the establishment of species positively responding to thinning of the forest canopy and mechanical disturbance of soil cover: *Agrostis capillaris*, *Stachys officinalis*, *St. sylvatica*, *Deschampsia cespitosa*, *Calamagrostis langsdorffii*, *Chrysosplenium alternifolium*, *Dryopteris filix-max*, *Lathyrus pratensis*, *Moehringia trinervia*, *Phegopteris connectilis*, *Bistorta officinalis*, *Prunella vulgaris*, etc. The GVL species composition in the impact zone remained unchanged.

The total biomass of GVL increased significantly only in the background and buffer (7 km) zones, due mainly to the growth of grasses (*Calamagrostis obtusata*) and herbs; the biomass of other functional groups increased to a lesser extent (Fig. 3). Changes in the impact zone occurred only in the structure of biomass, with an increase in the proportion of grasses and a decrease in that of horsetails.

During the second period (1999–2007), the diversity of GVL on the microscale stabilized in all zones, while that on the meso- and macroscales increased in the background and buffer (7 km) zones, but to a lesser extent than during the first period. Changes in species composition were observed only in the background and buffer (7 km) zones. As during the first period, they resulted from the establishment of species typical of disturbed habitats (*Galeopsis bifida*, *Urtica dioica*, *Tussilago farfara*, *Chamaenerion angustifolium*) or dif-

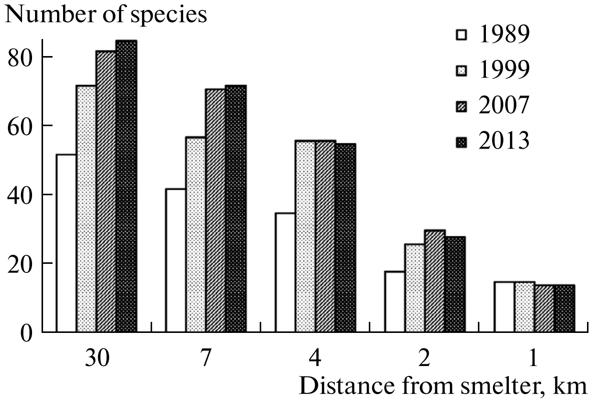

Fig. 2. Dynamics of plant species richness in different segment of pollution gradient.

ferent types of communities (*Veratrum lobelianum*, *Vicia sepium*, *Coccyganthe flos-cuculi*, *Epilobium palustre*, *Glechoma hederacea*, *Succisa pratensis*). The species composition of GVL in the impact zone remained practically unchanged.

The total biomass of GVL increased significantly in the background and impact (1 km) zones but remained approximately the same in other plots. Its increase in the background zone was accounted for mainly by ferns and, to a lesser extent, grasses and herbs; in the impact zone, only by grasses (*Agrostis capillaris*). The biomass of horsetails continued to decrease in all plots, especially in the impact zone.

During the third period (2007–2013), the macroscale diversity of communities in all zones remained at almost the same level, while meso- and microscale diversity decreased in the impact (2 km) and buffer zones, remained unchanged in the immediate vicinity of MUCS (1 km), and slightly increased in the background zone. The total biomass of GVL in the background zone increased significantly due solely to the growth of ferns, while that in other areas either remained unchanged or markedly decreased at the expense of grasses in the impact (1 km) zone and of grasses and horsetails in the buffer (4 km) zone.

Dynamics of toxic load. Despite drastic reduction of emissions from the MUCS (Fig. 1), the concentrations of heavy metals (Cu, Pb, Cd, Zn) in the upper part of humus horizon not only did not decrease by 2012, compared to 1989, but even increased (Trubina et al., 2014). In particular, lead concentrations increased from 19 μg/g in 1989 to 66 μg/g in 2012 in the background zone, from 45 to 135 μg/g in the buffer (4 km) zone, and from 280 to 380 μg/g in the impact (1 km) zone. The pH of humus horizon did not change between 1989 and 1999 but decreased by 0.5–0.7 units in all plots between 1999 and 2012, approaching the values characteristic of soddy podzolic soils of the Middle Urals (pH 4.6–5.0) (Trubina et al., 2014).

Dynamics of weather conditions. Analysis of meteorological data showed that, notwithstanding the overall trend of climate warming and humidification during the past 50 years (Fig. 4), weather conditions in the period of 1989 to 2010 could be considered constant: the correlation coefficient of linear approximation was close to zero both for the annual sum of effective temperatures ($r = 0.07$, $p = 0.774$) and for the annual amount of precipitation ($r = -0.06$, $p = 0.814$).

To our knowledge, only one extreme weather phenomenon with serious consequences for the dynamics

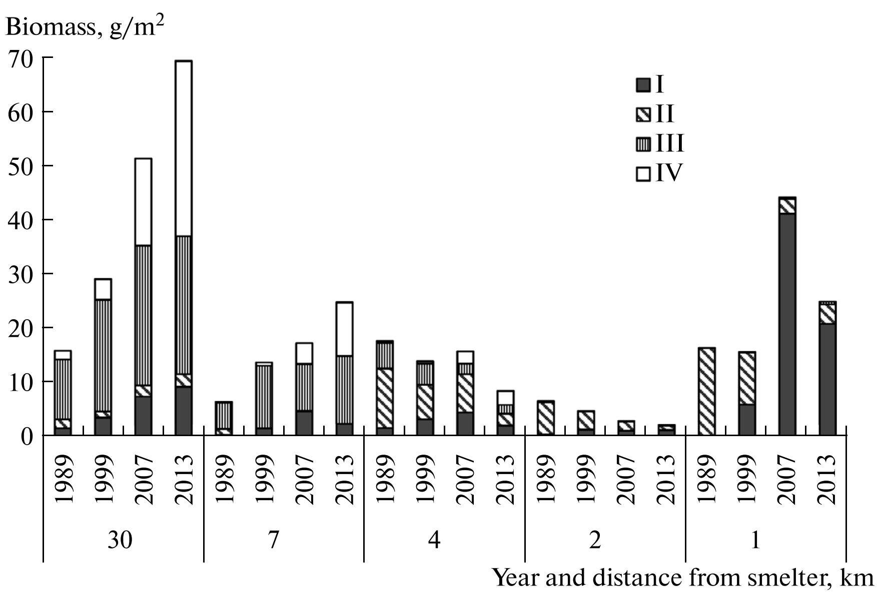

Fig. 3. Dynamics of the structure of ground layer vegetation at different distances from the MUCS: (I) grasses and sedges, (II) horsetails, (III) herbs, (IV) ferns.

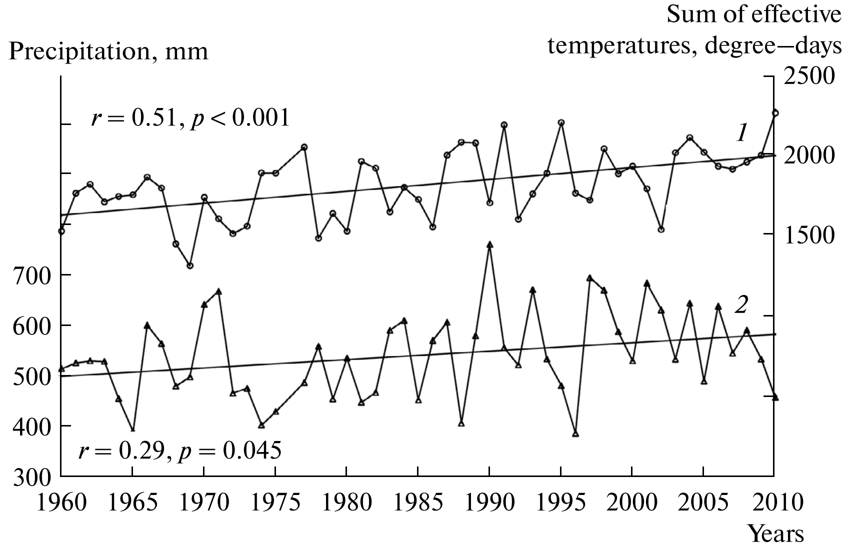

Fig. 4. Long-term dynamics of (*1*) annual sum of effective temperatures (above 10°C) and (*2*) annual precipitation. Lines show linear trends (r is linear correlation coefficient, p is significance level).

of vegetation was recorded in Sverdlovsk oblast during the period considered in this study. This was the snowstorm on June 6, 1995, which brought wind gusts as high as 40 m/s and wet snow, which amounted to 26 cm and packed on tree crowns (Uspin, 2000). This storm caused large-scale tree windfall in an area of over 360×10^3 ha in five regions of Sverdlovsk oblast (Sibgatullin and Shlykova, 2000). It should be noted that this windfall is not unique: such phenomena in the Middle Urals occur with a period of 40–50 years (Turkov, 1979). The region of our studies was at the periphery of the storm, and the extent of the damage was limited to increased uprooting of single trees and groups of trees, with snow load contributing to tree breakage and die-off.

DISCUSSION

The dynamics of plant communities in areas with a low or medium pollution level radically differed from those in heavily polluted areas, with the differences concerning both the pattern and probable mechanisms of the dynamics.

Dynamics of communities in the background and buffer zones. During the 25-year observation period (1989–2013), the vegetation of these zones has undergone significant and in some aspects positive changes, which suggest a tempting idea to regard them as a direct consequence of the reduction of atmospheric emissions. However, some facts do not allow such a conclusion.

First, the synchronous decrease in tree stand density and standing volume in all zones during the first period is an indirect indication that the dynamics of tree stand were affected by some external factor common to the entire study region and not related to pollution (based on the fact of emission reduction, an opposite reaction could have been expected, namely, an increase in standing volume). During the second period, tree stand density and standing volume increased synchronously in all zones, but this increase cannot be unequivocally explained by the reduction of emissions: it can as well reflect the process of recovery after the elimination of the external factor responsible for changes observed during the first period.

Moreover, an increase in the proportion of dead wood during the second period is evidence for further degradation of tree stand, rather than for its recovery. This may be due to high mortality in tree generations approaching the limit of their life span (Zverev, 2009) or a decline in tree viability caused by certain factors (Juknys et al., 2003; Chernen'kova et al., 2011; Jonard et al., 2012). In any case, the observed process is inverse to that expected upon the reduction of emissions.

The absence of distinct trend in climatic factors during the study period (Fig. 4) allows us to exclude their role as basic determinants of vegetation dynamics (at least in first approximation). On the other hand, a devastating weather anomaly such as the 1995 windstorm gives a key to understanding the mechanisms of the observed dynamics, since it is most likely to be the external factor responsible for changes in tree stands throughout the study region (see above). Moreover, this anomaly is also likely to account for the observed changes in the structure of plant communities in the ground layer, because its dynamics appear to directly respond to thinning of the tree stand.

As noted above, the windfall caused by the 1995 windstorm in the study region was not extensive and did not result in the destruction of forest. Moreover, its consequences for GLV were rather favorable: gaps formed in the forest canopy by fallen trees provided new microhabitats for ground-layer plants, with the consequent “explosive” growth of their diversity. Changes observed in the background and buffer zones—a sharp increase in the diversity and biomass of GLV in the first period, stabilization of these parameters during the second and third periods, and the establishment and increase in abundance of typical species of open and disturbed habitats—agree perfectly with the results of observations on forest vegetation after windfall (Skvortsova et al., 1983; Ulanova, 2000; Von Oheimb et al., 2007) and mechanical disturbance simulating the fall of single trees (Trubina, 2009). In particular, an increase in the abundance of ferns and grasses (mainly *Calamagrostis obtusata*) is a characteristic feature of post-windfall demutational successions in dark coniferous forests of the Middle Urals (Skvortsova et al., 1983; Belyaeva, 2000), and it is the abundance of these plant groups that was increased synchronously in all zones (Fig. 3). Another relevant fact is that the growth of diversity in the communities of background and buffer zones was accounted for solely by species positively responding to thinning or mechanical disturbances of soil–plant cover.

Dynamics of plant communities in the impact zone. During the 25-year observation period, the diversity of communities in plots near the MUCS (1 km) remained almost unchanged, and the increase in diversity at a distance of 2 km was markedly lower, compared to that in the background and buffer zones (Fig. 2). A very low rate of recovery after the reduction of technogenic load, especially when its initial level was high, has been observed in a number of studies not only on higher plants (Gunn et al., 1995; Vavrova et al., 2009; Chernen’kova et al., 2011) but also on other groups of biota, including hydrobionts (Havas et al., 1995; Keller et al., 1998) and lichens (Bates et al., 2001).

The dynamics of plant communities in the impact zone were actually manifested only in changes in the structure of dominance: wood horsetail (*Equisetum sylvaticum*) decreased in abundance to be superseded by common bent grass (*Agrostis capillaris*). Gradual elimination of the former species from communities may be explained by an increase in illumination level resulting from tree die-off (this species is known to prefer shady moist forests) as well as by intensification of competition from light-loving bent grass, whose live and dead shoots and roots form a thick grass sod under these conditions.

The stability of the suppressed state of vegetation in the impact zone is evidence that the capacity of local ecosystems for recovery has been impaired significantly. This is primarily explained by the fact that soil toxicity remains high, because the rate of soil self-purification from heavy metals is extremely low (Tyler, 1978). A detailed consideration of this phenomenon is beyond the scope of this study, because it should involve in-depth analysis of trends in heavy metal migration over the soil profile. However, it is important to note that, since measurements of heavy metals are made in the root soil layer, an increase in their concentrations indicates that the toxic load on plants has not only remained the same but also worsened. The observed increase in soil pH by 0.5–0.7 units appears to be unfavorable for the recovery of vegetation, because such an increase further reduces the mobility of heavy metals but does not reach the level (pH > 6.0) sufficient for reducing their bioavailability and, hence, toxicity for the biota (McBride et al., 1997). Therefore, the reduction of atmospheric emissions from the MUCS has resulted in alleviation of toxic load on plants from only one group of components, gaseous pollutants (SO_2, HF, NO_x), without affecting the impact of other components (heavy metals and metalloids accumulated in the soil). This circumstance is especially important, because the strong toxic effect of copper smelters on the vegetation is explained not only by emission of acid gases but primarily by pollution of the environment with heavy metals, especially under conditions of high soil acidity (natural or technogenic) (Kozlov et al., 2009).

The stable status of plant communities in the impact zone may also be explained by deficit in the input of diaspores (since the majority of species have disappeared) and depletion of the soil seed bank (Ginocchio, 2000; Meerts and Grommesch, 2001; Trubina, 2009). The establishment of plants in vacant sites may be impeded by a deep layer of almost undecomposed forest litter, which is characteristic of ecosystems in the impact and buffer zones near the MUCS (Vorobeichik, 1995): its thickness averages about 5 cm and may reach 10–15 cm, compared to 1–2 cm in the background zone. The negative effect of the litter on the establishment and survival of plants and on the composition and species richness of plant communities is well known (Sydes and Grime, 1981; Xiong and Nilsson, 1999; Weltzin et al., 2005).

CONCLUSIONS

Repeated estimation of the state of forest vegetation in permanent sample plots at 5- to 10-year intervals over an almost 25-year period allowed us to check the validity of several hypotheses about its dynamics during the period of strong reduction of atmospheric emissions from a large copper smelter.

A rigorous proof of the first hypothesis—about the leading role of natural factors—is hardly possible at all, but the full set of indirect data provide evidence in its favor: it is natural factors, namely, windfall disturbance after the windstorm, that have played a more important role in the dynamics of forest communities than the reduction of emissions itself. Hence follows an important methodological consequence: it is risky to use the results of short-term observations as proof that rapid recovery of ecosystems has resulted from reduction of industrial emissions, since this reduction may coincide in time with exposure to some natural factors. Therefore, long-term studies are necessary for a correct conclusion about the causes of the observed dynamics.

Despite almost complete cessation of emissions from the smelter, forest vegetation in the impact zone remains severely inhibited: the trees continue to die off, and ground vegetation has shown no signs of recovery during the study period. This may be regarded as evidence for the validity of the second (inertial) hypothesis. Apparently, the stability of the suppressed status of ecosystems is accounted for not only by the extremely low rate of soil purification from heavy metals but also by other factors: the presence of thick litter layer, deficit in the input of diaspores, biotic interactions, etc. Special studies, primarily experiments in nature are needed to evaluate individual contributions of these factors.

Regardless of the mechanism responsible for "conservation" of vegetation in the impact zone, it must be admitted that plant communities in most heavily polluted areas have very low regenerative capacity. It appears that, even in the absence of emissions, the process of natural recovery of their composition and species diversity will take hundreds of years. From a practical standpoint, this means that the transition of ecosystems into the "impact" state is irreversible on the actual time scale, with their return to the initial state being impossible without intensive recultivation measures.

ACKNOWLEDGMENTS

We are grateful to N.V. Zolotareva, R.M. Khantemirov, S.Yu. Kaigorodova, and E.A. Bel'skii for their valuable comments and discussion of the manuscript.

This study was supported by the Government of Sverdlovsk oblast; the Russian Foundation for Basic Research (project no. 13-04-96073); the Basic Research Program of the Ural Branch, Russian Academy of Sciences (project no. 12-P-4-1026); and the Scientific School Support Program (project no. NSh-2840.2014.4).

REFERENCES

Bates, J.W., Bell, J.N.B., and Massara, A.C., Loss of *Lecanora conizaeoides* and other fluctuations of epiphytes on oak in S.E. England over 21 years with declining SO_2 concentrations, *Atmos. Environ.*, 2001, vol. 35, no. 14, pp. 2557–2568.

Belyaeva, N.V., Disastrous windfall and transformations of herb–dwarf shrub and moss layers in forests of the Visim Reserve, in *Posledstviya katastroficheskogo vetrovala dlya lesnykh ekosistem* (Consequences of Disastrous Windfall for Forest Ecosystems), Yekaterinburg, 2000, pp. 46–62.

Chernen'kova, T.V. and Bochkarev, Yu.N., Dynamics of spruce stands in the Kola Peninsula under the effect of natural and anthropogenic environmental factors, *Zh. Obshch. Biol.*, 2013, vol. 74, no. 4, pp. 283–303.

Chernen'kova, T.V., Kabirov, R.R., Mekhanikova, E.V., et al., Demutation of vegetation after copper smelter shut-down, *Lesovedenie*, 2001, no. 6, pp. 31–37.

Chernen'kova, T.V., Kabirov, R.R., and Basova, E.V., Progressive successions in northern taiga spruce forests upon reduction of industrial air pollution, *Lesovedenie*, 2011, no. 6, pp. 49–66.

Danek, M., The influence of industry on Scots pine stands in the south-eastern part of the Silesia–Krakow Upland (Poland) on the basis of dendrochronological analysis, *Water Air Soil Pollut.*, 2007, vol. 185, nos. 1–4, pp. 265–277.

Ginocchio, R., Effects of a copper smelter on a grassland community in the Puchuncavi Valley, Chile, *Chemosphere*, 2000, vol. 41, nos. 1–2, pp. 15–23.

Gunn, J., Keller, W., Negusanti, J., et al., Ecosystem recovery after emission reductions: Sudbury, Canada, *Water Air Soil Pollut.*, 1995, vol. 85, no. 3, pp. 1783–1788.

Havas, M., Woodfine, D.G., Lutz, P., et al., Biological recovery of two previously acidified, metal contaminated lakes near Sudbury, Ontario, Canada, *Water Air Soil Pollut.*, 1995, vol. 85, no. 2, pp. 791–796.

Jonard, M., Legout, A., Nicolas, M., et al., Deterioration of Norway spruce vitality despite a sharp decline in acid deposition: A long-term integrated perspective, *Global Change Biol.*, 2012, vol. 18, no. 2, pp. 711–725.

Juknys, R., Vencloviene, J., Stravinskiene, V., et al., Scots pine (*Pinus sylvestris* L.) growth and condition in a polluted environment: From decline to recovery, *Environ. Pollut.*, 2003, vol. 125, no. 2, pp. 205–212.

Kaigorodova, S.Y. and Vorobeichik, E.L., Changes in certain properties of grey forest soil polluted with emissions from a copper-smelting plant, *Russ. J. Ecol.*, 1996, vol. 27, no. 3, pp. 177–183.

Kalabin, G.V. and Moiseenko, T.I., Ecodynamics of technogenic provinces around mining industries: From degradation to recovery, *Dokl. Ross. Akad. Nauk*, 2011, vol. 437, no. 3, pp. 398–403.

Keller, W., Heneberry, J.H., and Gunn, J.M., Effects of emission reductions from the Sudbury smelters on the recovery of acid- and metal-damaged lakes, *J. Aquat. Ecosyst. Stress Recovery*, 1998, vol. 6, no. 3, pp. 189–198.

Kozlov, M.V., Zvereva, E.L., and Zverev, V.E., *Impacts of Point Polluters on Terrestrial Biota: Comparative Analysis of 18 Contaminated Areas*, Dordrecht: Springer, 2009.

Lyanguzova, I.V. and Maznaya, E.A., Dynamic trends in *Vaccinium myrtillus* L. cenopopulations in the zone affected by a copper–nickel smelter complex: Results of 20-year monitoring, *Russ. J. Ecol.*, 2012, vol. 43, no. 4, pp. 281–288.

McBride, M., Sauvé, S., and Hendershot, W., Solubility control of Cu, Zn, Cd and Pb in contaminated soils, *Eur. J. Soil Sci.*, 1997, vol. 48, no. 2, pp. 337–346.

Meerts, P. and Grommesch, C., Soil seed banks in a heavy-metal polluted grassland at Prayon (Belgium), *Plant Ecol.*, 2001, vol. 155, no. 1, pp. 35–45.

Prokaev, V.I., *Fiziko-geograficheskoe raionirovanie Sverdlovskoi oblasti* (Physiographic Zoning of Sverdlovsk Oblast), Sverdlovsk: Sverdlovsk. Ped. Inst., 1976.

Sibgatullin, R.Z. and Shlykova, N.A., Impact of disastrous windfall in 1995 on primeval forests of the Visim Reserve, in *Posledstviya katastroficheskogo vetrovala dlya lesnykh ekosistem* (Consequences of Disastrous Windfall for Forest Ecosystems), Yekaterinburg, 2000, pp. 24–31.

Skvortsova, E.B., Ulanova, N.G., and Basevich, V.F., *Ekologicheskaya rol' vetrovalov* (Ecological Role of Windfalls), Moscow: Lesnaya Promyshlennost', 1983.

Sydes, C. and Grime, J.P., Effects of tree leaf litter on herbaceous vegetation in deciduous woodland: 1. Field investigations, *J. Ecol.*, 1981, vol. 69, no. 1, pp. 237–248.

Tarko, A.M., Bakadyrov, A.V., and Kryuchkov, V.V., Modeling the impact of atmospheric pollutants on forest ecosystem in the region, *Dokl. Ross. Akad. Nauk*, 1995, vol. 341, no. 4, pp. 571–573.

Trubina, M.R., Species richness and resilience of forest communities: Combined effects of short-term disturbance and long-term pollution, *Plant Ecol.*, 2009, vol. 201, no. 1, pp. 339–350.

Trubina, M.R. and Makhnev, A.K., Dynamics of ground vegetation in forest phytocenoses under conditions of chronic pollution by fluorine, *Russ. J. Ecol.*, 1997, vol. 28, no. 2, pp. 73–77.

Trubina, M.R. and Vorobeichik, E.L., Severe industrial pollution increases the β-diversity of plant communities, *Dokl. Biol. Sci.*, 2012, vol. 442, pp. 17–19.

Trubina, M.R., Vorobeichik, E.L., Khantemirova, E.V., et al., Dynamics of forest vegetation after reduction of industrial emissions: Rapid recovery or continuing degradation?, *Dokl. Biol. Sci.*, 2014 (in press).

Turkov, V.G., On windfall of trees in primeval forest as a biogeocenotic phenomenon: The example of mountain fir–spruce forests in the Middle Urals, in *Temnokhvoinye lesa Srednego Urala* (Dark Conifer Forests in the Middle Urals), Sverdlovsk, 1979, pp. 121–140.

Tyler, G., Leaching rates of heavy metal ions in forest soil, *Water Air Soil Pollut.*, 1978, vol. 9, no. 2, pp. 137–148.

Ulanova, N.G., The effects of windthrow on forests at different spatial scales: A review, *Forest Ecol. Manag.*, 2000, vol. 135, nos. 1–3, pp. 155–167.

Usol'tsev, V.A. and Zalesov, S.V., *Metody opredeleniya biologicheskoi produktivnosti nasazhdenii* (Methods for Determining Biological Productivity of Plantations), Yekaterinburg: UGLTU, 2005.

Uspin, A.A., Meteorological characterization of disastrous windfall in the Middle Urals (June 1995), in *Posledstviya katastroficheskogo vetrovala dlya lesnykh ekosistem* (Consequences of Disastrous Windfall for Forest Ecosystems), Yekaterinburg, 2000, pp. 18–24.

Vavrova, E., Cudlin, O., Vavricek, D., and Cudlin, P., Ground vegetation dynamics in mountain spruce (*Picea abies* (L.) Karsten) forests recovering after air pollution stress impact, *Plant Ecol.*, 2009, vol. 205, no. 2, pp. 305–321.

Vidic, T., Jogan, N., Drobne, D., and Vithar, B., Natural revegetation in the vicinity of the former lead smelter in Žerjav, Slovenia, *Environ. Sci. Technol.*, 2006, vol. 40, no. 13, pp. 4119–4125.

Von Oheimb, G., Friedel, A., Bertsch, A., and Härdtle, W., The effects of windthrow on plant species richness in a Central European beech forest, *Plant Ecol.*, 2007, vol. 191, no 1, pp. 47–65.

Vorobeichik, E.L., Changes in thickness of forest litter under chemical pollution, *Russ. J. Ecol.* 1995, vol. 26, no. 4, pp. 252–258.

Vorobeichik. E.L. and Khantemirova, E.V., Reaction of forest phytocenoses to technogenic pollution: Dose–effect dependences, *Russ. J. Ecol.*, 1994, vol. 25, no. 3, pp. 171–180.

Vorobeichik, E.L., Sadykov, O.F., and Farafontov, M.G., *Ekologicheskoe normirovanie tekhnogennykh zagryaznenii nazemnykh ekosistem (lokal'nyi uroven')* (Ecological Rating of Technogenic Pollutants in Terrestrial Ecosystems: Local Level), Yekaterinburg: Nauka, 1994.

Weltzin, J.F., Keller, J.K., Bridgham, S.D., et al., Litter controls plant community composition in a northern fen, *Oikos*, 2005, vol. 110, no. 3, pp. 537–546.

Xiong, S. and Nilsson, C., The effects of plant litter on vegetation: A meta-analysis, *J. Ecol.*, 1999, vol. 87, no. 6, pp. 984–994.

Yusupov, I.A., Luganskii, N.A., and Zalesov, S.V., *Sostoyanie iskusstvennykh sosnovykh molodnyakov v usloviyakh aerompromvybrosov* (The State of Young Forest Plantations under Industrial Air Pollution), Yekaterinburg: UGLTA, 1999.

Zverev, V.E., Mortality and recruitment of mountain birch (*Betula pubescens* ssp. *czerepanovii*) in the impact zone of a copper–nickel smelter in the period of significant reduction of emissions: The results of 15-year monitoring, *Russ. J. Ecol.*, 2009, vol. 40, no. 4, pp. 254–260.

Translated by N. Gorgolyuk