Working Paper

Anthropogenic and Technogenic Stress Factors to Forests in Siberia

V.I. Kharuk, S. Nilsson and E. Samarskaia

WP-96-104
December 1996

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Foreword

This is the time Siberia's forest sector has recently gained considerable international interest. IIASA, the Russian Academy of Sciences, and the Russian Federal Forest Service, in agreement with the Russian Ministry of the Environment and Natural Resources, signed agreements in 1992 and 1994 to carry out a large-scale study on the Siberian forest sector. The overall objective of the study is to focus on policy options that would encourate sustainable development of the sector. The goals are to assess Siberia's forest resources, forest industries, and infrastructure; to examine the forests'economic, social, and biospheric functions; with these functions in mind, to identify possible pathways into policy options for Russian and international agencies.

The first phase of the study concentrated on the generation of extensive and consistent databases for the total forest sector of Siberia and Russia. The study has now moved into its second phase, which encompasses assessment studies of the greenhouse gas balances, forest resources and forest utilization, biodiversity and landscapes, non-wood products and functions, environmental status, transportation infrastructure, forest industry and markets, and socio-economic problems. This report, by Dr. V.I. Kharuk from the Institute of Forestry, Russian Academy of Sciences, Krasnoyarsk, and Prof. S. Nilsson and Dr. E. Samarskaia from the core-team of the study have attempted to quantify the anthropogenic and technogenic stress factors to the forests in Siberia. This study is related to the area of "environmental status" mentioned above.

Anthropogenic and Technogenic Stress Factors to Forests in Siberia

V.I. Kharuk, S. Nilsson and E. Samarskaia

1. Introduction

Anthropogenic stress is considered to be one of the primary causes of current forest decline in the boreal forests. The overall task of this paper is to try to quantify and illustrate the impacts of technogenic stress to the Siberian forests.

In order to fulfill this objective, the following sub-tasks have to be carried out;

- identification of the location of the pollution sources in Siberia
- estimation of emissions by major pollutants (sulfur dioxide, nitrogen oxides, and heavy metals) and their deposition
- estimation of critical loads for the major pollutants
- analyses of the radio nuclide contamination
- analyses of the forest decline in Siberia.

The basic information used in this work are scientific reports, reports by different State committees and research carried out by the authors. Available data lack consistency, but the data used can be regarded as "state-of-the-art" knowledge on the studied problem.

2. The Siberian Forests

There are three major forest types in Siberia; coniferous forests (including marshes and taiga meadows), mixed coniferous-deciduous forests (including marshes and meadows), and pure deciduous forests. The main types of taiga forests are stands formed by so-called "black needle" species (spruce, fir, and Siberian pine), pine, larch, birch and aspen. "Black needle" stands are typical for the mountainous areas of East Siberia and the Altai mountains. Larch stands are found mainly in East Siberia, with its strong continental climate and carbonate soils. Pine is widely spread in West Siberia on alluvial planes, in sandy river valleys, and on sphagnum swamps. In East Siberia, pine is growing on sands, loams, and detritus slopes. Stripes of birch and aspen stands are present in West Siberia. In the taiga, birch and aspen are widely distributed and they occupy earlier burnt and logged areas. Aspen is more frequent in southern regions and on good soils. Mixed hard and soft deciduous are typical for the southern Far East.

The variety of species, types of stands, climatic and soil conditions make Siberia highly heterogeneous with respect to responses to different pollutants.

3. Natural Conditions for Accumulation of Pollutants in Siberia

It is well-known that impacts of pollutants on ecosystems depend on a whole set of factors, like: a) factors which promote accumulation of pollutants; b) factors which increase/decrease pollution impact or neutralize pollutants; c) intensity of pollution. Some pollutants could be destroyed by UV radiation, in other cases UV promotes creation of even more harmful substances (i.e., ozone); pollutants could be moved (or imported) with prevailing winds or river flows; they could be bounded and neutralized by atmospheric aerosols or soil components; high humidity and fogs may increase damage, and, on the contrary, soil fertility may increase plant resistance. In the latter case, the resistance depends on air temperature and temperature inversions, wind speed, level of precipitation, elevation, exposition and direction, duration of solar irradiation etc. There are attempts by some investigators to introduce integral indices, which could characterize the whole set of natural conditions of a given area. There are also several attempts of mapping the Russian territory for this purpose, and, in particular, for estimation of the potential ability of a given territory to accumulate pollutants (or the ability of "self cleaning", or to neutralize and to remove pollutants). All indices are produced them are more qualitative than quantitative, and may give good food for criticism.

To estimate the level of air pollution, two main indices are used in Russia: a) "Index of Atmospheric Pollution (IAP)" and "Air Pollution Potential (Ap)". IAP is the criterion on actual air quality. It is calculated (in relative units) on the basis of the following equation:

$$\mathrm{Im} = \sum_{i=1}^{m} = (q_i / CC)S_i,$$

where Im is the total IAP, q_i is the mean concentration of the i-th pollutant, CC is the critical concentration of the i-th pollutant, S_i is a coefficient which depends on the "class of danger" of the i-th pollutant [which varies within the range 0.85 (low danger) to 1.5 (high danger)].

The IAP values are calculated on the basis of the five main pollutants for given territory/city. The IAP is to be considered as moderate for values <7, as high for values 7...10, as very high for values 10...14, and as extremely high for values >14. The IAP values are published regularly by Regional Ecological Committees as an index of the air quality for cities. In 1994, among 272 cities in Russia, 41 had IAP >14 and 20 of them were located in Siberia. Since 1989, values of IAP >30 have been observed in 8 cities in, and 4 of them are in Western Siberia (Ecologicheskaya obstanovka..., 1995).

The potential ability of the atmosphere in a given territory to accumulate pollutants is indicated by the air pollution potential (Ap) in Russia. It is an integral indicator (in relative units) of the conditions which promote pollutant accumulation (based on the probabilities for number of days with no wind, temperature inversions, precipitation, direction and intensity of atmospheric circulation). Ap is calculated on the basis of a so-called physical-statistical method:

$Ap = q/q0,$

where q_i is the mean concentration of pollutants in a given region, and $q0$ is the mean concentration for a reference region. If the concentration of pollutants in the aboveground air column is described by a normal logarithmic, AP could be calculated as:

$Ap = 2.5 \times exp[0.04/(z2 - zl)\ 2 - 0.4\ zl/(z2 - zl)],$

where zl and $z2$ are the arguments for the probability of the integral. Throughout the Russian territory the Ap values vary between 2.1...4.0. For Siberia the Ap values under "low temperature emission" conditions are presented in *Figure 3.1* and in *Table 3.1*. The levels of precipitation and solar irradiation were not taken into account in these calculations. According to *Figure 3.1*, the largest part of Siberia, especially East Siberia, is characterized by high Ap values, which exceed those for the European part of Russia. The dominating part of European Russia is characterized by Ap values <2.7 (low and moderate). For West Siberia the Ap values are 2.7-2.85, and for East Siberia they are high (>2.85) and very high (>3.3).

The migration of pollutants by air and rivers are presented in *Figure 3.2*. It shows that Siberia is receiving depositions of pollutants from the European part of Russia with prevailing winds and, partly, by rivers. Within Siberia the pollutants are transported mainly from the Western to the Eastern parts. In some areas the directions of air and water transportation are not corresponding, but opposite (Glasovskaya, 1989). There are limited quantitative data for these mixed streams of pollutants. Therefore, *Figure 3.2* presents a more general overview of the problem of pollution transfer.

An attempt to classify the whole territory of Russia with respect to self-cleaning ability was presented by Glasovskaya (1989). In *Figure 3.3*, the territory of Russia is ranked according to its self-cleaning ability by air and river dissipation. This is also a more qualitative orientated approach, and the territory was ranked on the basis of the probability of days with no wind (in %) and run-off precipitation. The foundation of this regionalization is rather elementary: the higher extent of days with no wind, the less self-cleaning ability, and with higher annual precipitation, the more pollutants will be removed. According to available data based on these calculations, the dominating part of Siberia is characterized by a low probability for self-cleaning. Glasovskaya *et al.* (1989) also tried to compile a generalized map of the self-cleaning ability (*Figure 3.4*). This map summarizes the data on potential ability of the Russianterritory for self-cleaning for "hard" organic, "liquid" mineral, gaseous and aerosol pollutants. The basic difference in the results compared with *Figure 3.3* is that in *Figure 3.4*

it is assumed that UV radiation could destroy some pollutants. Five grades of self-cleaning ability have been suggested, from "very weak" (1) to "very intensive" (5) (Glasovskaya *et al.*,....1989). This approach should be considered as a rough approximation of the real self-cleaning conditions within Siberia.

In general, the natural conditions in Siberia, especially in the Eastern part, are more favorable for accumulation of pollutants in comparison with European part of Russia.

Figure 3.1 Air Pollution Potential (Ap) Mean Values.

Table 3.1. Air Pollution Potential (Ap), Minimum (Apmin), Maximum (Apmax) and Mean Values.

Ecoregion Code[1)2)]	Apmin	Apmax	Apm
11011	2.8	3.0	2.9
11012	2.4	2.9	2.6
11013	2.7	2.9	2.7
11014	2.7	2.9	2.8
11014	2.8	2.9	2.8
11041	2.4	2.9	2.6
11041	2.4	2.7	2.6
11042	2.4	2.9	2.6
11043	2.7	3.0	2.8
11044	2.4	2.9	2.6
11045	2.7	3.0	2.8
11046	2.7	3.0	2.8
11047	2.8	2.9	2.8
11048	2.8	3.0	2.9
11051	2.7	2.9	2.7
11052	2.7	2.9	2.7
11053	2.7	2.9	2.7
11081	2.7	3.0	2.8
11082	2.7	2.9	2.7
11083	2.7	2.9	2.7
11084	2.7	2.9	2.7
11085	2.7	2.9	2.8
11101	2.9	3.7	3.2
11102	2.8	3.0	2.9
11103	2.8	2.9	2.8
11251	2.9	3.0	2.9
11252	2.9	3.3	3.0
11253	2.9	3.6	3.2
11254	2.9	3.3	3.0
11255	2.9	3.3	3.0
11301	2.7	3.0	2.8
11302	2.7	2.9	2.7
11303	2.7	2.9	2.7
11304	2.7	2.9	2.7
11305	2.7	2.9	2.7
11321	2.7	2.9	2.7
11322	2.7	2.9	2.7
11323	2.7	2.9	2.8
11323	2.7	2.9	2.7
11441	2.7	3.0	2.8
11441	2.7	2.9	2.8
11442	2.9	3.0	2.9
11443	2.7	2.9	2.7

Ecoregion Code[1)]	Apmin	Apmax	Apm
11443	2.7	2.9	2.7
11501	2.4	2.7	2.5
11502	2.4	2.7	2.5
11521	2.4	2.7	2.5
11522	2.4	2.7	2.5
11641	2.7	2.9	2.7
11642	2.7	2.9	2.7
11691	2.4	2.7	2.5
11692	2.4	2.9	2.6
11711	2.4	2.7	2.5
11712	2.7	2.9	2.7
11713	2.4	2.9	2.6
11714	2.4	2.9	2.6
11715	2.4	2.7	2.5
11761	2.8	3.7	3.2
11762	2.8	3.0	2.9
11811	2.8	3.3	3.0
11812	2.8	2.9	2.8
11931	2.9	3.0	2.9
11932	2.9	3.0	2.9
11981	2.4	2.9	2.6
11981	2.7	2.9	2.7
11982	2.7	3.0	2.8
11983	2.9	3.0	2.9
11984	2.9	3.0	2.9
11985	2.8	3.7	3.2

[1)] Code for Ecoregions used in the database of IIASA's Siberian Forest Study.

[2)] In some cases there are more than one observation or source of data for individual ecoregions and all observations are presented in the table.

Figure 3.2. Regionalization of the Russian territory with respect to transportation of pollutants. Region boundaries: 1 - river flow; 2 - atmospheric transfer; 3 - subregions of river flow and atmospheric transfer.

Migration direction: 4 - with river flows; 5 - with atmospheric transfer, 6 - with sea streams.

Migration direction of different flows: 7 - directions do coincide; 8 - directions do not coincide to any large extent; 9 - directions are opposite; 10 - no river flows. Source: Glasovksaya. 1989.

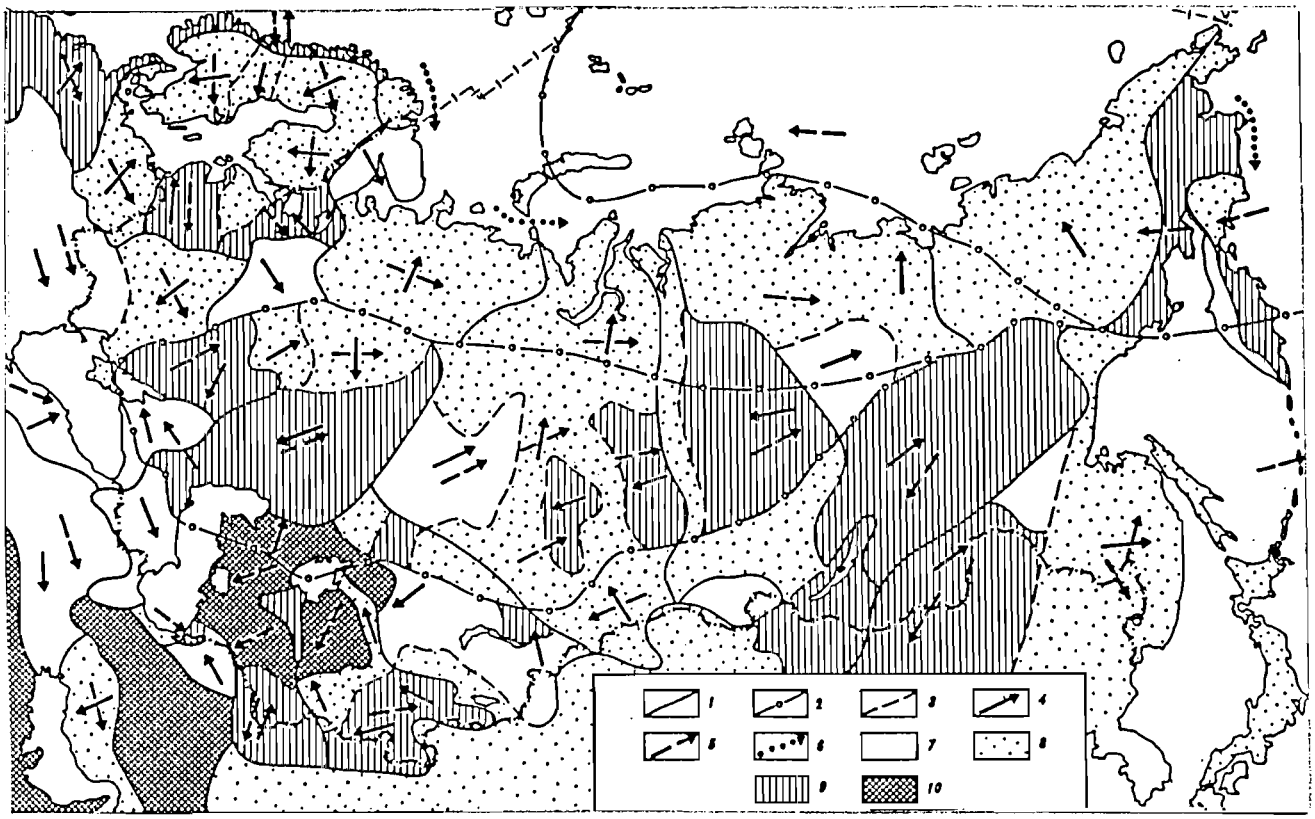


Figure 3.3. Regionalization of the Russian territory concerning the self-cleaning ability by air and river flow dissipation.

Number of days with no wind (%): III_1 >60; 50<III_2 <59; 30<III_3<49; 25<III_4<29; 20<III_5<24; 13<III_6<19; III_7<12.

Annual water flow in streams and rivers (in mm): c_1 - 10; c_2 - 10...100; c_3 - 100...200; c_4 - 200...300; c_5 - 300...400; c_6 - 400...800; c_7 - 800. Source: Glasovskaya, 1989.

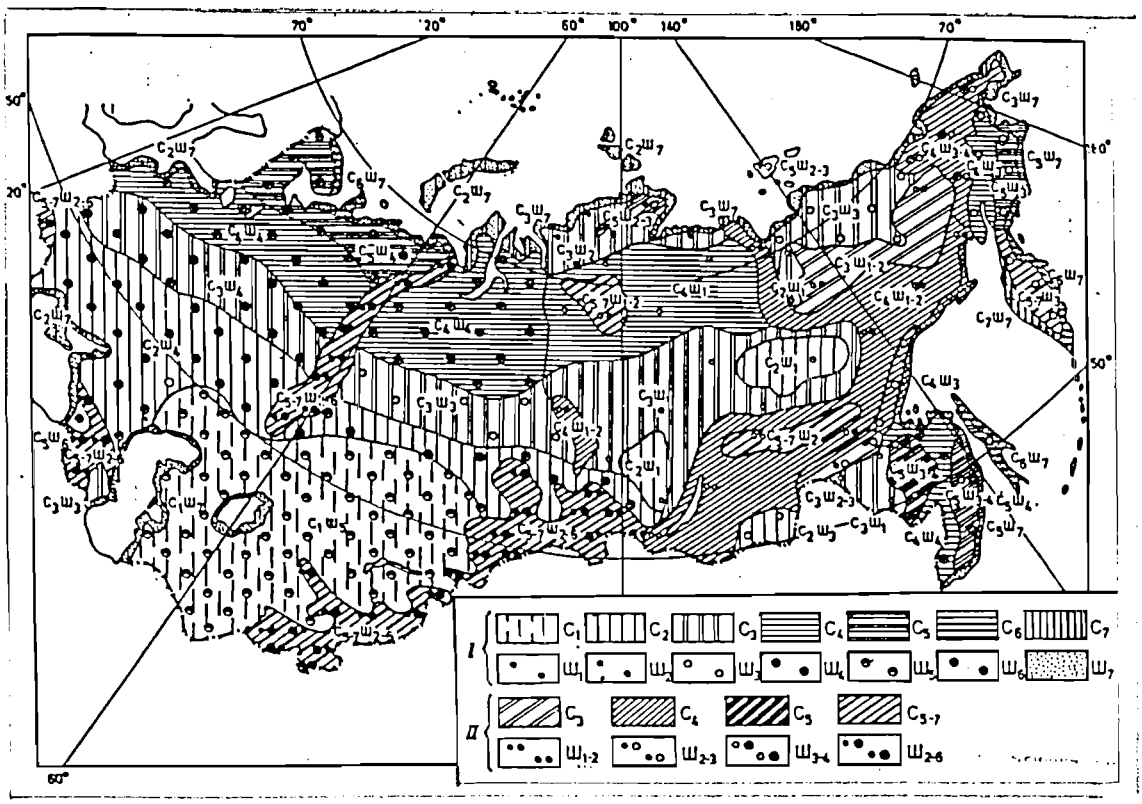
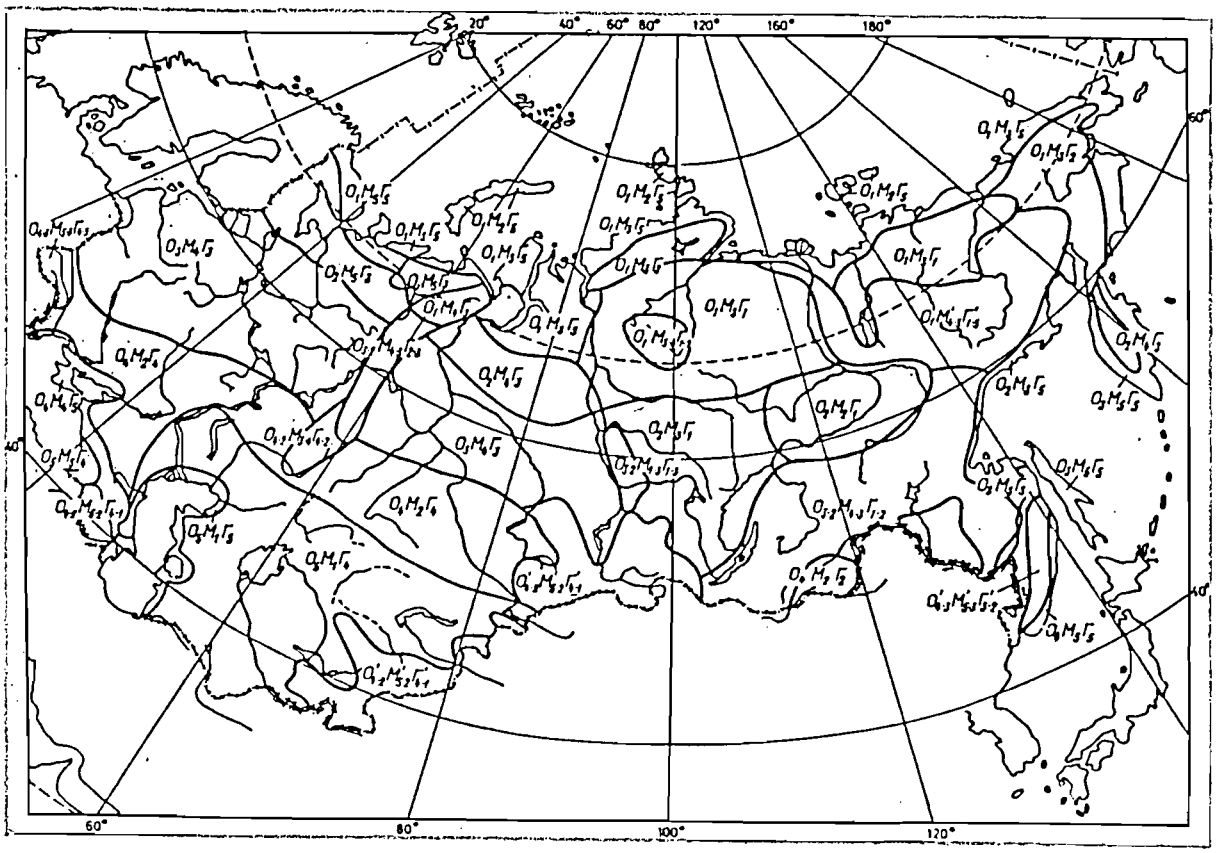


Figure 3.4. Regionalization of the Russian territory concerning the self-cleaning ability from gaseous and aerosol (Г), Liquid mineral (M), and "hard organic" (O) pollutants.
Self-cleaning ability: 1 - very weak; 2 - weak; 3 - moderate; 4 - high; 5 - very high.

Source: Glasovskaya, 1989.



4. Main Sources of Emissions and Pollution Status in Siberia

The task of this section is to give a general geographical view of pollution sources and pollution status in Siberia. The main sources of emissions are centers for steel, copper, cobalt, zinc, lead and nickel production, and a number of big coal power stations. Their locations are presented in *Figure 4.1*. The map is based on official data. All industrial centers are in southern Siberia with the exception of the Norilsk region. An objective source of information on the location of pollution sources over the vast Siberian territory is winter space images. Zones of chronicle contamination are clearly detectable on the snow. Winter images may also reveal sources of pollution, which for some reason may not be included in official reports. *Figure 4.2* is made on the basis of winter images interpretation and presents an aggregated overview of contaminated areas in Siberia (Prokacheva *et al.*, 1992). Dark areas on the map represent all sources of pollutants. These dark areas also reflect the zones of potential risk of forest decline due to pollution. All reported data on pollution impact on forest stands (See Section 9) are within the dark areas of the map in *Figure 4.2*.

Figure 4.2 shows that a) Siberia is comparatively less contaminated by pollution than the European part of Russia, b) contaminated areas in Siberia are located mainly in the south, c) the dominating part of Siberia is not suffering from any pollution impact.

Another approach for compiling a generalized overview of pollution impact was elaborated by Glasovskaya (1989). They suggested to rank the "anthropogenic pressure" of a territory according to a regional total coefficient of noospheric (geochemical impact by humans on a given territory) concentration (C_{ns}):

$$C_{ns} = D_1 C_{n1} + \ldots + D_k C_{nk},$$

where D_1, ..., D_k - technogenic pressure of different kinds of the technogenic products on a territory, C_{n1}, ..., C_{nk} are noospheric coefficients of technogenic pressure for those technogenic products. This approach was used for compiling an aggregated map (*Figure 4.3*) of technogenic pressure on the territory of Russia. The regional total coefficients of noospheric concentration were calculated on the basis of coal, oil and gas consumption per unit area studied and the technogemic emissions were assumed to be proportional to the population density (Issledovaniya..., 1990). The map generated by this approach also supports the idea that the anthropogenic pressures are concentrated to Southern Siberia. Pryde (1994) has published, based on data from the Institute of Geography of the Russian Academy of Sciences, a map of areas having critical environmental conditions in the early 1990s in the former USSR (*Figure 5.4*). Feshbach (1995) has produced a map (*Figure 5.5*) showing an integrated evaluation of anthropogenic transformations of the natural ecosystems. This map also shows the strongest transformation on the border to Mongolia and China, respectively in the southeastern part of the Far East. The evaluation of the transformations takes into account the losses of biomass and bioproductivity caused by integrated anthropogenic disturbances. Information presented in *Figures 4.1-4.5* coincide in general. Siberia is comparatively less contaminated than the European part of Russia. Contaminated areas in Siberia are located mainly in the southern and southeastern (Kemerovo and Irkutsk regions, Krasnoyarsk and Primorksy krai). The dominating part of Siberia is not suffering from pollution impacts. The emissions of sulfur and nitrogen oxides and heavy metals are presented in Sections 5 and 6.

Figure 4.1. Map of Main Pollution Sources in Siberia.

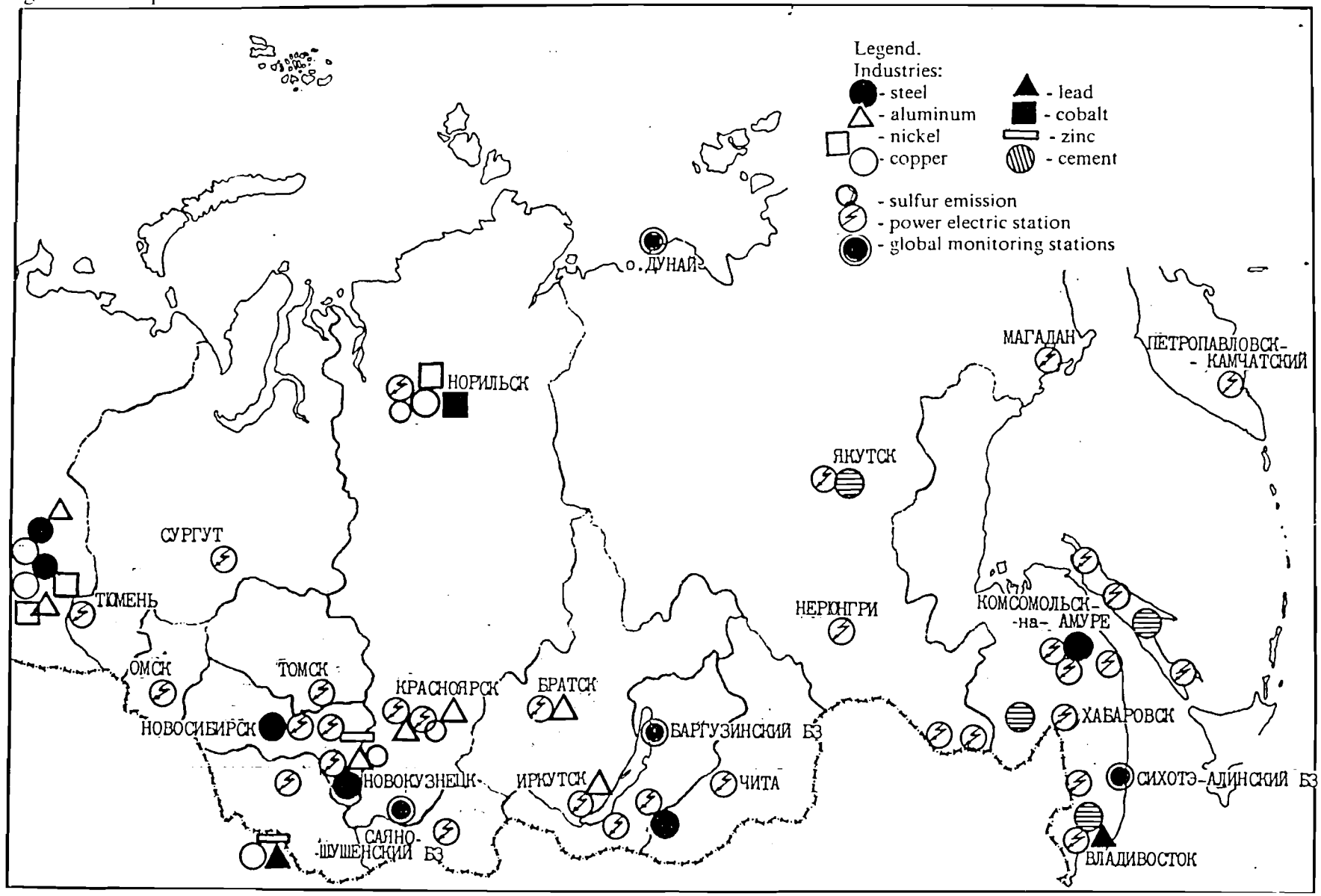


Figure 4.2. Contaminated areas in Russia. Source: Prokacheva *et al.*, 1992).

Figure 4.3. Aggregated map of technogenic geochemical pressure. Regional total coefficient of noospheric concentration module, $n10^4$: *1* - <1; *2* - <1...5; *3* - 5...10; *4* - 10...--25; *5* - 25...50; *6* - 50...100; *7* - >100. Source: Isslledovaniya, 1990.

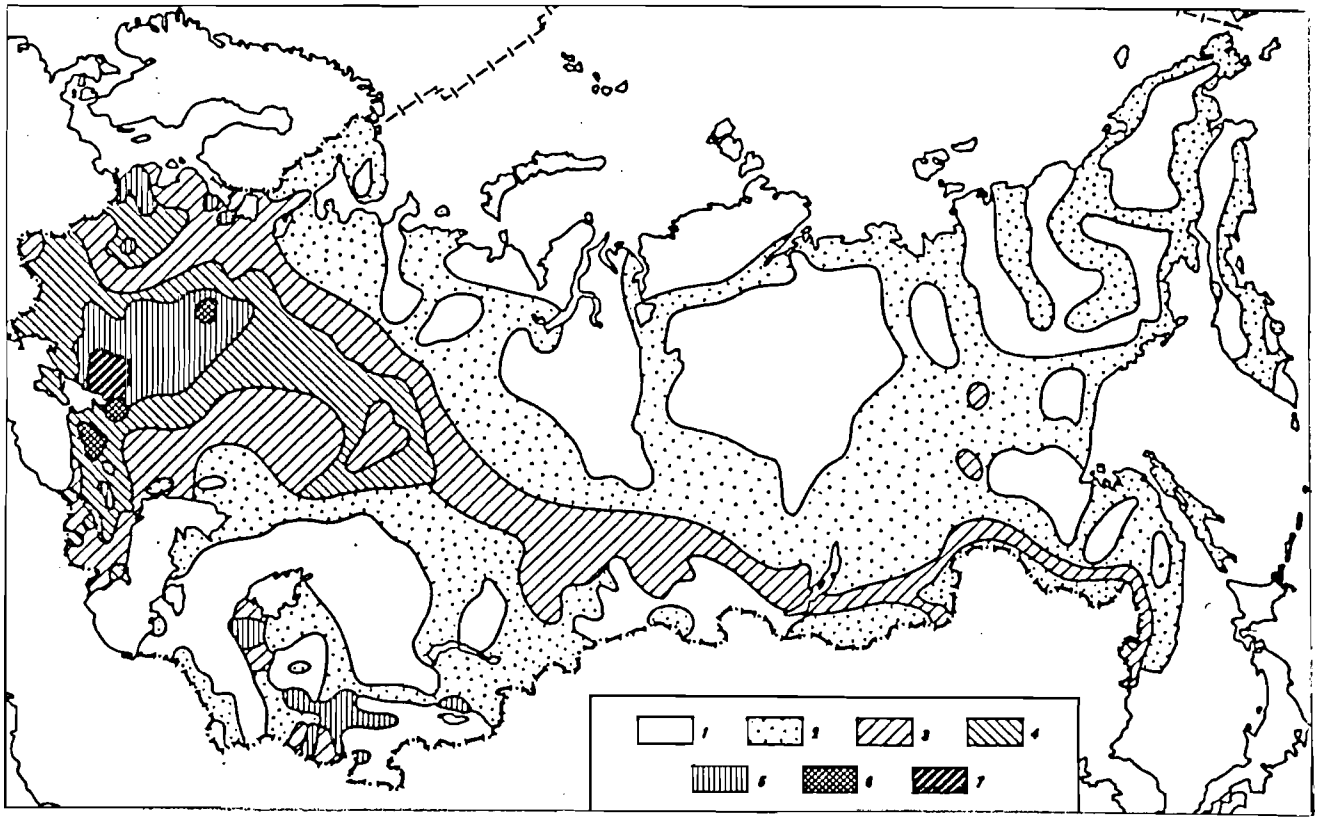


Figure 4.4 Critical environmental areas in the former USSR. Source: Pryde (1994).

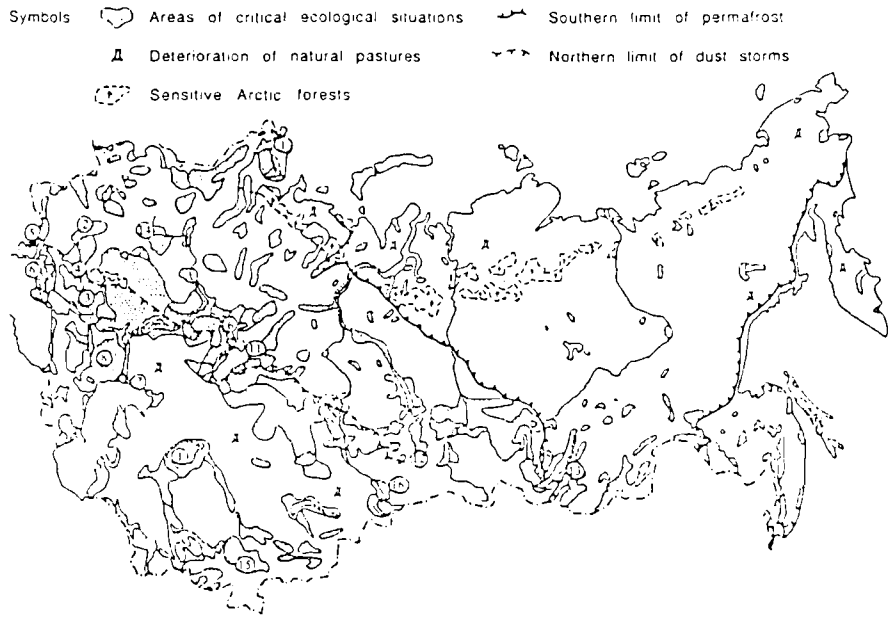
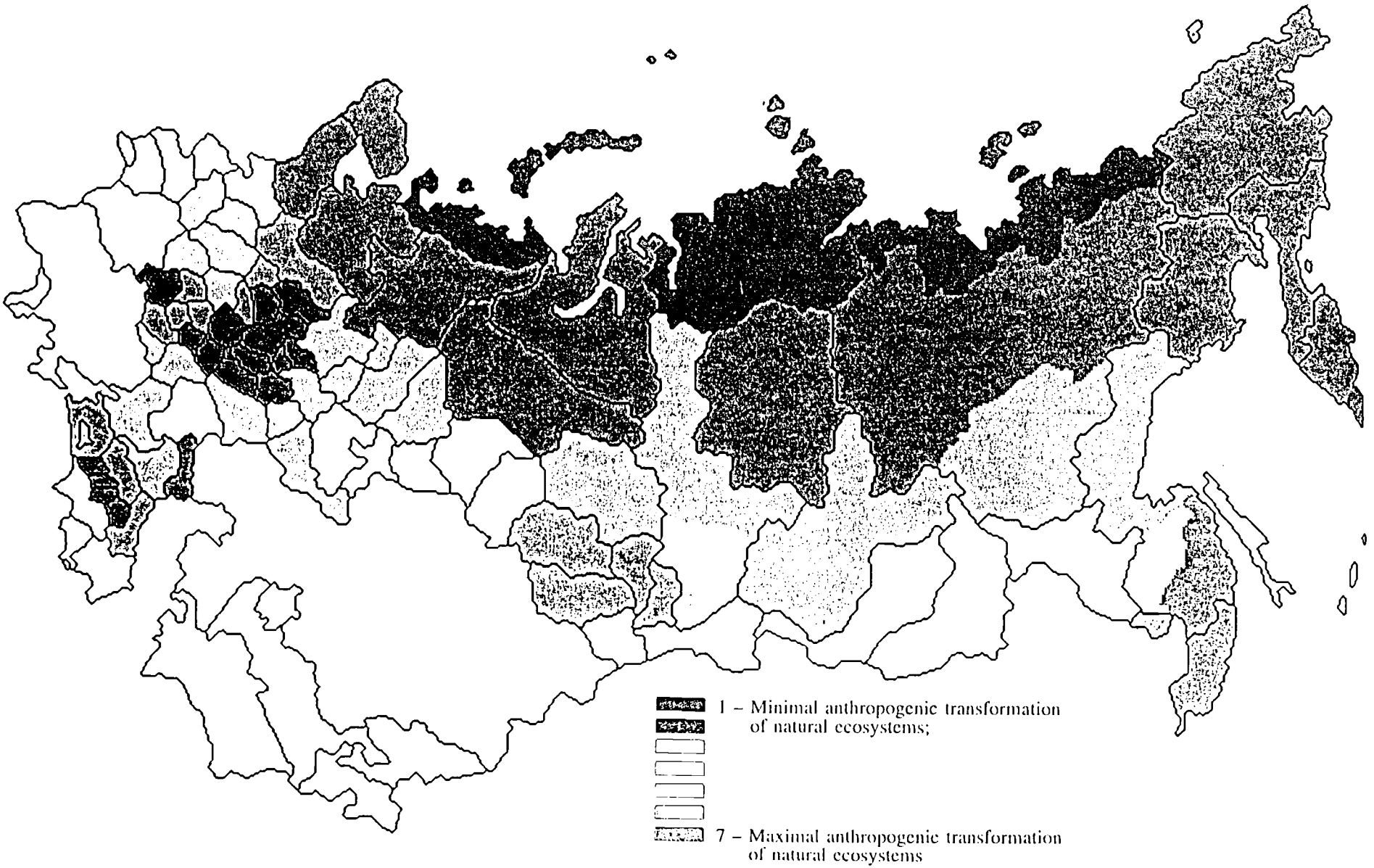


Figure 4.5 Anthropogenic transformations of the natural ecosystems. Source: Feshbach (1995).



5. Sulfur and Nitrogen Emissions

The total emissions of pollutants by the industry declined between 1988 and 1993 by 41% in West Siberia, by 20% in East Siberia, and by 37% in the Far East (Okruzhayuchaya prirodnaya..., 1995). This decline was due to decreased industrial production. The impact of these declined emissions on the ecology has not been pronounced because: 1) a decreased efficiency of air pollution scrubbing. In 1989 the Norilsk industrial complex cleaned ~20% of the SO_2 emissions, in 1994 3.5% were cleaned; 2) increased air pollution caused by traffic (for Krasnoyarsk region, the pollution by traffic has increased by 64% since 1992, and now contributes to the total pollution by more than 17%). It should also be noted that the emissions from the total industry were not included in the official statistics. Emissions caused by railway, air and water transport, as well as by military transportation, fuel waste burning by people were not included in the official statistics (Sostoyanie okruzhayuschey, 1995). Official data on the emissions of sulfur and nitrogen oxides in 1993 are presented in *Figures 5.1* and 5.2 and in *Table 5.1*. Data on the deposition of sulfates (SO_4) and total nitrogen (nitrate + ammonia) in 1993 are collected from Okruzhayuschaya sreda (1995), and are presented for ecoregions in *Figures 5.3* and 5.*4* and in *Tables 5.2* and 5.*3*. Data in *Table 5.2* (and the following tables) include maximum, minimum and mean values. Feshbach (1995) has presented aggregated maps of the sulfur sulfate burden and the total nitrogen loads for the early 1990s in Russia (*Figures 5.5* respectively *5.6*). This latter set shows somewhat lesser depositions in comparison with the first set.

The largest sulfate depositions take place in southern West Siberia, Irkutsk region and the Norilsk area, where the depositions reach 1000-3000, and in some locations >3000 kg $km^{-2}yr^{-1}$. For the dominating part of the forested areas, the deposition level is in the range of 30-500 kg $km^{-2}yr^{-1}$. The distribution of nitrogen depositions has a similar pattern with maximum loads in southern West Siberia and in the Norilsk zone (500-1000 respectively >1000 kg $km^{-2}yr^{-1}$) For the dominating part of the forested areas the depositions do not exceed 100-300 kg $km^{-2}yr^{-1}$.

Figure 5.1. SO_2 emissions, thousand tons/yr.

Figure 5.2. NO_x emissions, thousand tons/year.

Table 5.1. SO_2 and NO_x emissions, thousand tons/year.

Ecoregion Code	SO_2	NO_x
11011	1.4	1.4
11012	74.4	27.3
11013	155.5	107.2
11014	74.4	27.3
11041	2002.3	73.4
11042	28.9	276.7
11043	2002.3	73.4
11044	2002.3	73.4
11045	2002.3	73.4
11046	2002.3	73.4
11047	20.7	6.8
11048	2002.3	73.4
11051	112.3	25.8
11052	112.3	25.8
11053	112.3	25.8
11081	71.7	22.2
11082	71.7	22.2
11083	71.7	22.2
11084	71.7	22.2
11085	4.1	1.8
11101	32.0	9.8
11102	32.0	9.8
11103	32.0	9.8
11251	183.2	90.4
11252	183.2	90.4
11253	183.2	90.4
11254	183.2	90.4
11255	183.2	90.4
11301	17.5	9.4
11302	17.5	9.4
11303	17.5	9.4
11304	17.5	9.4
11305	17.5	9.4
11321	155.5	107.2
11322	64.4	38.4
11323	155.5	107.2
11441	n.a.	n.a.
11442	12.0	6.7
11443	12.0	6.7
11501	64.4	38.4
11502	64.4	38.4
11521	132.6	40.4
11522	132.6	40.4
11641	19.2	14.2

Ecoregion Code	SO_2	NO_x
11642	19.2	14.2
11642	n.a.	n.a.
11691	18.2	14.2
11692	18.2	14.2
11711	28.9	276.7
11712	28.9	276.7
11713	28.9	276.7
11714	28.9	276.7
11715	132.6	40.4
11761	61.8	17.1
11762	61.8	17.1
11811	36.4	10.2
11812	36.4	10.2
11931	4.1	2.5
11932	4.1	2.5
11981	12.7	29.1
11982	12.7	29.1
11983	12.7	29.1
11984	12.7	29.1
11985	12.7	29.1

Table 5.2. Nitrogen and SO_4 depositions, kg/km^2/yr, maximum (Nmax, Smax), minimum (Nmin, Smin) and average (Nm, Sm) values.

Ecoregion Code	Nmin	Nmax	Nm	Smin	Smax	Sm
11011	500	1000	750	150	500	325
11012	500	1000	750	500	1000	750
11013	500	1000	750	500	3000	1750
11014	500	1000	750	150	1000	575
11014	500	1000	750	150	500	325
11041	1	1000	500	10	3000	1505
11041	100	500	300	30	150	90
11042	100	500	300	30	150	90
11043	50	500	275	30	150	90
11044	30	1000	515	150	500	325
11045	100	1000	550	150	3000	1575
11046	300	500	400	500	1000	750
11047	300	500	400	30	1000	515
11048	100	300	200	150	500	325
11051	100	300	200	30	150	90
11052	100	500	300	30	1000	515
11053	100	300	200	500	1000	750
11081	50	300	175	30	150	90
11082	50	300	175	30	150	90
11083	100	300	200	30	150	90
11084	100	500	300	30	500	265
11085	300	500	400	150	1000	575
11101	50	300	175	30	500	265
11102	100	300	200	30	1000	515
11103	100	500	300	150	500	325
11251	50	300	175	30	150	90
11252	50	100	75	30	150	325
11253	100	300	200	150	3000	1575
11254	100	300	200	500	3000	1750
11254	100	500	200	500	3000	1750
11255	100	300	200	150	500	325
11255	100	300	200	500	3000	1750
11301	50	300	175	30	150	90
11302	100	300	200	30	150	90
11303	100	300	200	30	150	90
11304	100	300	200	30	150	90
11305	100	500	300	30	3000	1515
11321	500	1000	750	500	1000	750
11322	500	1000	750	500	1000	750
11323	500	1000	750	150	1000	575
11323	500	1000	750	500	3000	1750
11441	1	100	50	10	150	80
11441	1	100	50	30	150	90

Ecoregion Code	Nmin	Nmax	Nm	Smin	Smax	Sm
11442	1	100	50	30	1000	125
11443	100	300	200	30	500	265
11443	100	300	200	150	100	125
11501	500	1000	750	500	3000	1750
11502	500	1000	750	500	1000	750
11521	30	1000	515	150	3000	1575
11522	300	1000	650	500	1000	750
11641	100	300	200	30	500	265
11642	100	300	200	150	1000	575
11642	100	300	200	30	150	90
11691	500	1000	750	30	500	265
11692	500	1000	750	500	1000	750
11711	100	300	200	30	150	90
11711	50	300	175	30	150	90
11712	300	500	400	30	150	90
11713	300	1000	650	30	1000	515
11714	300	1000	650	150	100	125
11715	300	1000	650	150	500	325
11761	50	300	175	30	150	325
11762	100	300	200	150	500	325
11811	50	500	275	150	1000	575
11812	100	300	200	150	3000	1575
11931	100	500	300	150	500	325
11932	100	300	200	150	500	325
11981	1	100	50	10	30	20
11981	1	50	25	10	30	20
11982	1	100	50	10	150	80
11983	50	100	75	30	150	90
11984	50	100	75	30	1000	90
11985	50	100	75	30	500	265

Figure 5.3. SO_4 depositions, kg/km^2/yr, mean values.

Figure 5.4. Nitrogen depositions, kg/km^2/yr, mean values.

Figure 5.5 Distribution of the sulfur sulfate burden in Russia (in kilograms per square kilometers per year)

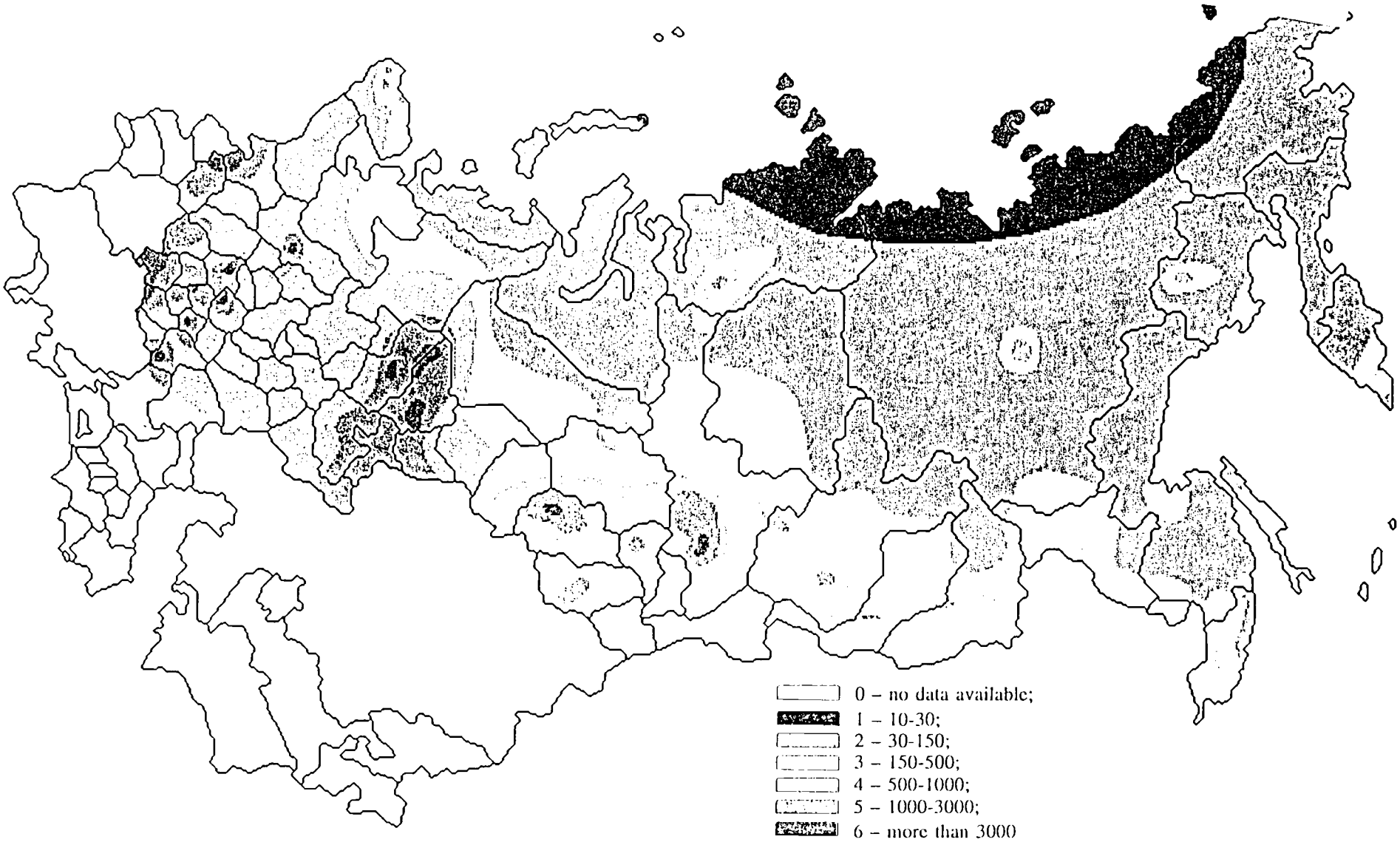
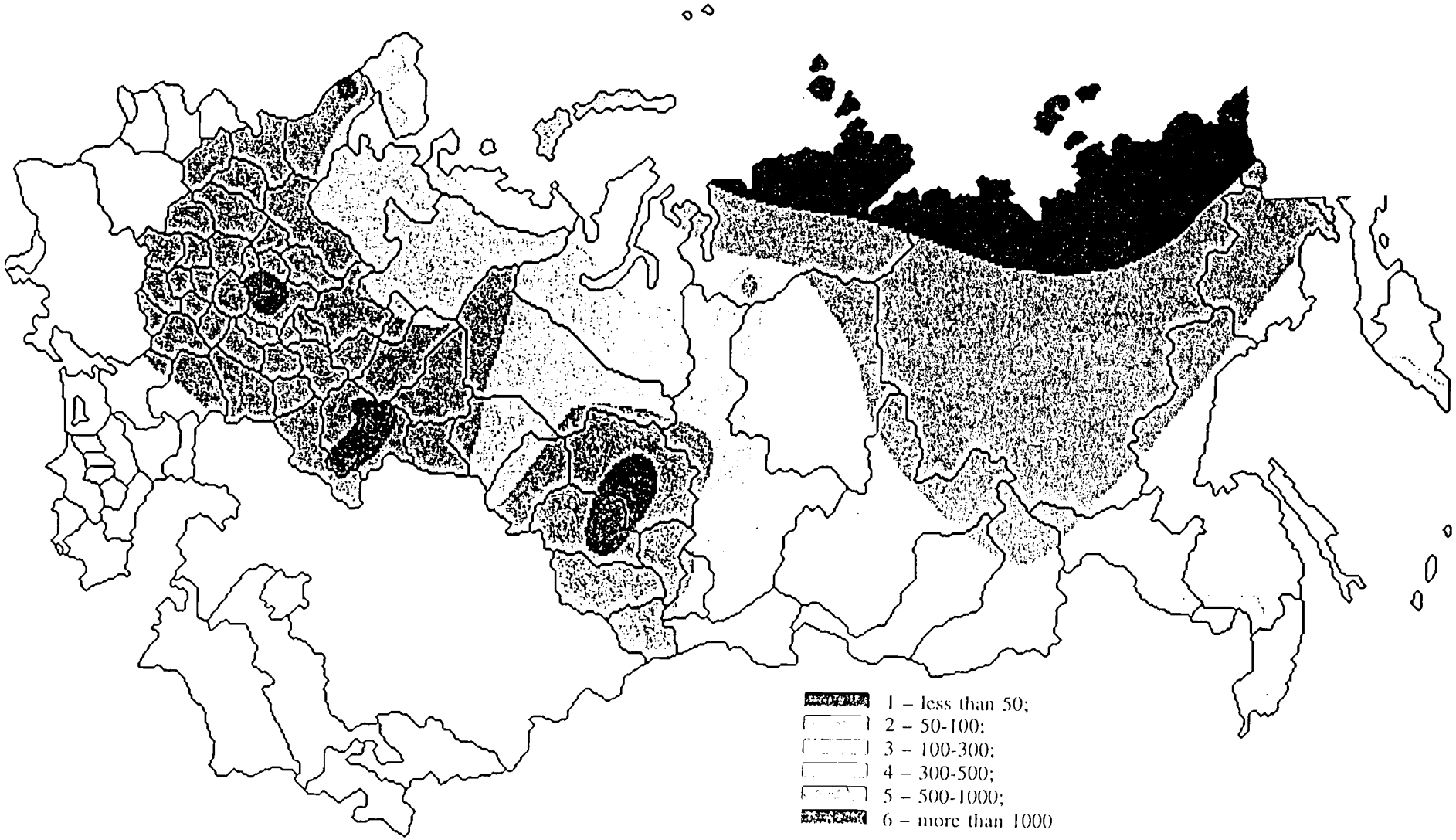


Figure 5.6 Distribution of total nitrogen load on the territory of Russia (in kilograms per square kilometer per year)



6. Emissions of Heavy Metals

The major heavy metal (HM) pollutants are considered to be Pb, Cd, Cu, Zn, Ni, V, As. Pb, V and partly Ni and they originate from many sources and are concentrated to urbanized territories. The sources of As, Cu, Zn and Cd are not abundant and have fixed geographic locations. The main sources of HM emissions in Siberia are concentrated to the latitudinal belt between 52 and 60 degrees. They are copper, nickel, cobalt, aluminum, lead, steel, and cement industries, and a large number of coal power stations (See *Figure 4.1*). The main sources of HM emissions are listed in *Table 6.1*. The world main source of Ni, Cu, V and Co emissions is the "Norilsk Nickel" industry in the north of Krasnoyarsk region. Emitters of Pb are Komsomolsk-on-Amur, Novosibirsk and Kemerovo; of Zn is Krasnoyarsk; of Cr are Barnaul, Krasnoyarsk and Komsomolsk-on-Amur, of V are Omsk and Khabarovsk. All of those HM sources are high temperature emitters of HM in the form of aerosols or "vapors". In some cases, the "vapor" fraction could be up to 50% of the HM emissions.

The process of the HM dispersion depends on the emission source parameters (stack height, volume and temperature of emissions), and meteorology (wind speed and precipitation). Around point sources of emissions local (0-40 km), medium (50-100 km) and distant (>100 km) zones of depositions are formed. In the local zone, the HM depositions could be as high as 10-30% of the total emissions. The depositions strongly depend on the precipitation. For example, with a radius of 40 km around a copper industry the extent of Ni, Pb, Zn and Cd depositions were estimated to ~3% on "dry" days, and reached 10-100% of the emissions on days with precipitation. Thus, a considerable amount of HM emissions are involved in long distance transportation, and are subject to "wet" and "dry" depositions. On the Siberian territory there are four special stations for pollution deposition analyses. They are located in remote areas far from the sources of the pollutants: in Sayano-Shushensky biosphere reserve (BR) in the south of Krasnoyarsk region, Bargusinsky BR in Irkutsk region, Sikhote-Alinsky BR, and Dunay station at the Lena river mouth. The HM depositions at these four stations in 1993 are presented in *Table 6.2*. The table shows that the wet deposition values of Pb, Cd, and Ni for Siberia are of the same order as those of the USA, Canada and Japan. The depositions of Cu seem to be lower and the depositions of Zn seem to be higher in Siberia in comparison with the mentioned countries.

The data in *Table 6.2* were used for calculation of the 1993 HM "wet" deposition. The annual HM deposition caused by precipitation was calculated according to the following equation:

$P_i = V_i * I$,

where P_i is the annual deposition of the i-th metal, mg/m^2/yr,

V_i is the i-th metal concentration in atmospheric precipitation, mg/l,

I - annual precipitation, mm.

The HM "wet" depositions of 1993 are presented in *Figure 6.1* and *Table 6.3*). "Wet" depositions are dominant in areas where the annual precipitation is >600 mm. For those areas the dry depositions are estimated to ~one third of the total depositions. This is true for the dominating part of Siberia, except for Yakutia Republic (with an annual precipitation of ~300 mm) and Tuva Republic (~250 mm). In Siberia the HM depositions are highest in Norilsk Ikrutsk region and in Siberian Far East. But even the highest depositions (30-41 kg/km^2/yr) are below the critical loads (Table 7.4) for trees. Temporal variations of the HM depositions are represented by data from the Bargusinsky reserve: during 1990-1994 the ratio between the highest and lowest annual deposition was 2 for Cu, 4 for Ni, and 2.6 for Zn. All data discussed above concern deposition outside the local zones of depositions; within the local zones the HM depositions are 1-3 orders higher (See Section 9.2). Based on current knowledge, it can be concluded that in Siberia the HM depositions are below critical loads for trees except in the vicinities of local emitters of HM.

Studies of metal accumulation in seabirds from northeastern Siberia show increased concentrations (Kim *et al.*, 1996). The authors assume that measured increased concentrations of Cd may arise from local natural sources rather than by anthropogenic sources. The high measured Hg-concentrations is explained by the sampling reason and the migration. Increased measured concentrations of Fe, Mn, Zn and Cu are explained by species-specific bioaccumulation or high natural background emissions and unlikely to pollution.

Table 6.1. Emissions of Heavy Metals, tons/yr.

Region/city	Pb	Cd	Ni	Cu	Zn	Cr	V	Mn	Fe
Omskaya	.08				.6	.49	211.3	11.50	
Novosibirskaya	1.62								
Tomskaya	0.2								
Kemerovskaya	1.30								18.0
Irkutskaya	.12								
Chitinskaya							1.1		
Barnaul	.10		543.00			1.10	2.0	.01	
Krasnoyarsk	.75		.32		17.9	5.30	3.9	2.00	2.5
Norilsk			4008.00	1813			65.0		
Khabarovsk	.18						37.0	1.14	
Amursk	.50						3.3	.40	
Komsomolsk	13.80				.1	1.10	5.2	1.80	
Vladivostok	.03	.1		.02		.06		.22	

Table 6.2. Mean Heavy Metal Concentrations in "Wet" Atmospheric Deposition, milligrams/1.

Location	Pb	Cd	Cu	Zn	Ni	As	Co	Cr
"Dunay" station*	6.000	.1200	1.400	3.0	1.100	.60		
"Sayano-Shushensky" BR**	1.600	.2500	2.500	24.0	2.200			
"Bargusinsky" BR***	2.500	.3600	3.100	53.0	2.000	.26	.1	.3
"Sikhote-Alinsky" BR****	3.000	.2000	2.300	48.0	2.000			
Japan			30.000	20.0	2.000			.5
Canada	3.000	.1200	4.000		7.000			
USA	3.000	.1800	19.000	20.0	17.000		5.0	1.7
Arctica	.013	.0004	.013		.022			

* – Lena river mouth

** – South Krasnoyarsk

*** – Baikal region

**** – Far East

BR – Biospheric Reserve

Figure 6.1. Total Heavy Metals "Wet" Deposition, kg/km^2/yr.

Table 6.3. "Wet" Deposition of Heavy Metals, kg/km^2/yr.

Ecoregion Code	Pb	Cd	Ni	Cu	Zn	Co	Cr	As	Sum
11011	.76	1.19	1.05	1.19	11.5	.05	.14	.12	15.90
11012	.76	1.19	1.05	1.09	11.5	.05	.14	.12	15.90
11013	.89	.14	1.23	1.39	13.4	.06	.17	.15	17.40
11014	.76	1.19	1.05	1.19	11.5	.05	.14	.12	15.90
11041	.74	.12	1.01	1.15	11.0	.05	.14	.12	14.33
11042	.74	.12	1.01	1.15	11.0	.05	.14	.12	14.33
11043	.74	.12	1.01	1.15	11.0	.05	.14	.12	14.33
11044	.74	.12	1.01	1.15	11.0	.05	.14	.12	14.36
11045	.74	.12	1.01	1.15	11.0	.05	.14	.12	14.40
11046	.74	.12	1.01	1.15	11.0	.05	.14	.12	14.40
11047	.74	.12	1.01	1.15	11.0	.46	.14	.12	14.80
11048	.74	.12	1.01	1.15	11.0	.46	.14	.12	14.80
11051	2.20	.15	1.47	1.69	35.2	.07	.22	.19	41.20
11052	2.20	.15	1.47	1.69	35.2	.07	.22	.19	41.20
11053	2.20	.15	1.47	1.69	35.2	.07	.22	.19	41.20
11081	1.80	.12	1.20	1.38	28.8	.06	.18	.16	33.70
11082	1.80	.12	1.20	1.38	28.8	.06	.18	.16	33.70
11083	1.80	.12	1.20	1.38	28.8	.06	.18	.16	33.70
11084	1.80	.12	1.20	1.38	28.8	.06	.18	.16	33.70
11085	1.60	.11	1.07	1.23	25.6	.05	.16	.14	30.00
11101	1.60	.11	1.70	1.23	25.6	.05	.16	.14	30.60
11102	1.60	.11	1.70	1.23	25.6	.05	.16	.14	30.60
11103	1.60	.11	1.07	1.23	25.6	.05	.16	.14	30.00
11251	1.12	.16	.89	1.38	23.6	.05	.14	.12	27.40
11252	1.12	.16	.89	1.38	23.6	.05	.14	.12	27.40
11253	.15	.16	.89	1.38	23.6	.05	.14	.12	26.50
11254	1.12	.16	.80	1.38	23.6	.05	.14	.12	27.40
11255	1.12	.16	.89	1.38	23.6	.05	.14	.12	27.50
11255	1.12	.16	.89	1.38	23.6	.05	.14	.12	27.40
11301[1]									
11302									
11303									
11304									
11305									
11321	.89	.14	1.23	1.39	13.4	.06	.17	.15	17.40
11322	.58	.09	.80	.91	8.8	.04	.11	.10	11.40
11323	.89	.14	1.23	1.39	13.4	.06	.17	.15	17.40
11323	.89	1.39	1.23	1.39	13.4	.06	.17	.15	17.40
11441									
11442									
11443									
11501	.58	.09	.80	.91	8.8	.07	.11	.10	11.40
11502	.58	.09	.80	.91	8.8	.04	.11	.10	11.40
11521	.64	.10	.88	1.00	9.6	.04	.12	.10	12.50
11522	.64	.10	.88	1.00	9.6	.04	.12	.10	12.50
11641									

Ecoregion Code	Pb	Cd	Ni	Cu	Zn	Co	Cr	As	Sum
11642									
11691	.80	.13	1.10	1.25	12.0	.05	.15	.13	15.60
11692	.80	.13	1.10	1.25	12.0	.05	.15	.13	15.60
11711	.78	.12	1.08	1.23	11.8	.05	.15	.13	15.30
11712	.78	.12	1.08	1.23	11.8	.05	.15	.13	15.30
11713	.78	.12	1.08	1.23	11.8	.05	.15	.13	15.30
11714	.78	.12	1.08	1.23	11.8	.05	.15	.13	15.30
11715	.64	.10	.88	1.00	9.6	.04	.12	.10	12.50
11761	.95	.10	.78	1.17	20.1	.04	.11	.10	23.40
11762	.95	.14	.76	1.18	20.1	.04	.11	.10	23.40
11811	.95	.10	.78	1.17	20.1	.04	.11	.10	23.40
11812	.95	.14	.76	1.18	20.1	.04	.11	.10	23.40
11931	.40	.06	.55	.63	6.0				7.64
11932	.40	.06	.55	.63	6.0				7.64
11981	1.80	.01	.30	.40	.9			.20	3.61
11982	1.80	.01	.30	.40	.9			.20	3.61
11983	1.80	.01	.30	.40	.9			.20	3.61
11984	1.80	.01	.30	.40	.9			.20	3.61
11985	1.80	.01	.30	.40	.9			.20	3.61

[1] Blanks indicate that there are no observations available.

7. Critical Loads

Russian literature is dominated by the general idea that there is no ecologically based normative for regulation of emissions and depositions of pollutants (Sadykov, 1991; Stepanov, 1991; Kompleksnaya, 1992; and Vorobeychic, *et al.*, 1994).

There are several definitions of allowable loads or critical loads (CL) in Russia. Israel (1984) considered that allowable ecological loads should prevent undesirable changes of an organism or ecosystem and preserve the quality of the environment. He suggested the following criterion for allowable load: the load could be considered as allowable if the resulting deviation from a normal state do not exceed the average level of natural fluctuations.

Sadykov (1991) argued that CL should correspond to the level of load observed on the boundary between degraded and still stable ecosystems. He stated that there should be several temporal levels of ecological normatives: current, perspective, and noospherical. Current level is set by ecological normatives, which are practically reachable now at existing level of economical development. It could correspond to the definition of target loads (Nilsson, *et al.*, 1992). Perspective levels are aimed to a gradual improvement of the environment. Noospheric levels should allow to the restoration of a high quality of the environment in industrialized regions.

In international literature CL are defined as quantitative estimates of an exposure to one or more pollutants, below which significant harmful effects on specific sensitive elements of the environment do not occur, according to our present knowledge. Target loads are less sensitive with respect to deposition loads in that they incorporate consideration for other pollution-control factors, such as economic ones (e.g., Nilsson *et al.*, 1991).

Most models for critical loads of sulfur and nitrogen developed in the west are based on Simple Mass Balance (SMB) equations (Posch *et al.*, 1995). These models have gone through a continuous development since the late 1980s (e.g., Nilsson and Greenfelt, 1988; Sverdrup *et al.*, 1990; Greenfelt and Thörnelöf, 1992; Downing *et al.*, 1993; and Hornung *et al.*, 1995).

The modern single critical load calculation (for individual pollutants like sulfur and nitrogen) include calculations on;

- critical loads of acidifying sulfur and nitrogen
    - critical loads of sulfur and nitrogen for constant sinks
    - critical loads for deposition-dependent sinks of nitrogen
- critical load of nutrient nitrogen
- calculation of alkalinity leaching and nitrogen quantities
    - critical alkalinity leaching
    - nitrogen transformation processes
- The exceedance of critical loads (Posch *et al.*, 1995).

The trend is now moving away from single critical loads to multiple critical thresholds. This has resulted in the generation of protection isolines, which illustrate combinations of sulfur and nitrogen depositions at which damage is ensured. Different isolines correspond to different protection levels. A five percentile protection isoline identifies where more than 5 percent of the studied area is at risk (Hettelingh *et al.*, 1995).

In the late 1980s, a Working Group of the Economic Commission for Europe and Nordic Council made recommendations on single critical loads for sulfur and nitrogen deposition for terrestrial ecosystems. Their actual recommendations both for sulfur and nitrogen ranged between 0.2 to 2.0 tons km^{-2} yr^{-1}, depending on ecosystem and its sensitivity (UN-ECE, 1988; The RAINS..., 1991).

Official reports of Russian Ecological Committees operate with values of critical loads taken from foreign sources (Atmospheric..., 1989; Critical..., 1986; Critical....,1988). Critical loads for sulfur identified in these reports (in form of sulfates) are 2 tons S $km^{-2}yr^{-1}$ (Q_{sc}) and 1 ton N km^{-2} yr^{-1} (Q_{nc}). In this Russian literature the analyses of the ecological consequences of sulfur and nitrogen depositions are based on a comparison of the actual depositions with critical loads values. In the case of combined S + N (SN) depositions the total effect is esteemed by the equation $Q_{sn} = Q_{si}/Q_{sc} + O_{ni}/Q_{nc}$, where Q_{si} and O_{ni} are actual values of the depositions on the *i*-th site (Atmosfernie..., 1991).

Nilsson *et al.* (1992) used the following single target loads for forests in European Russia: sulfur depositions in coniferous forests 0.5 g $m^{-2}yr^{-1}$ (high sensitivity), 1.0 (medium), 2.0 (low). For depositions in deciduous forests: 1.0 (high), 2.0 (medium) and 4.0 (low). For nitrogen depositions, the corresponding values are: 0.3; 1.0; 1.5 (conifers); 0.5; 1.2; 2.0 (deciduous). Nilsson *et al.* (1992) found that in 1985 about one-third of the coniferous forests in the European USSR suffered from exceedance by sulfur depositions and about one-fourth of the coniferous forests suffered from exceedance by nitrogen depositions.

Later on, within the Convention on Long-range Transboundary Air Pollution Program, Russia has carried out a single critical loads estimate for European Russia based on the "Guidelines for the Computation and Mapping of Critical Loads and Exceedances of Sulfur and Nitrogen in Europe" (Posh *et al.*, 1993; Downing *et al.*, 1993). However, the calculations carried out for European Russia are strong simplifications in relation to the guidelines given by the convention. However, the distribution of critical loads achieved for European Russia are presented in *Table 7.1.* From these analyses it can be concluded that in the taiga forests of European Russia the critical loads are seldom exceeded at existing levels of atmospheric depositions, but the region has a low buffering capacity. Potential dangerous effects were identified for entrophication.

Table 7.1. Distribution of values of critical deposition of sulfur and nitrogen in the European part of Russia. Figures are given as percentage of total area. Source: Downing *et al.*, 1993.

Ranges	**Critical deposition**		**Exceedances**	
(eq ha^{-1} yr^{-1})	*CD(S)*	*CD(N)*	*CD(S)exc*	*CD(N)exc*
<200	4.8	7.0	87.1	99.4
200-500	36.0	22.0	5.3	0
500-1000	26.9	15.1	5.3	0
1000-2000	19.9	22.0	2.3	0
>2000	12.4	33.9	0	0.6

In 1995, Russia presented new calculations on single critical loads for European Russia based on Posh *et al.* (1993) guidelines, but still with strong simplifications in the calculations in relation to original guidelines. However, in this report the exceedance of the critical loads for sulfur and nitrogen are not presented.

Hettelingh *et al.* (1995) show that, based on a combination effect of both sulfur and nitrogen depositions of 1990, that the dominating part of the European Russian ecosystems are protected against pollution of sulfur and nitrogen with the current depositions.

About 20 different groups in Europe are currently producing critical loads and policies for pollution by heavy metals and persistent organic pollutants (POP) (Sliggers and de Jager, 1993; de Wries and Bakker, 1995). The critical loads for heavy metals and POPs are based on the same concept as for sulfur and nitrogen, namely that an ecosystem has only a limited capacity to cope with pollutants without unavoidable damage, but the critical loads differ in some aspects from those of sulfur and nitrogen (*Table 7.2*).

Table 7.2. Differences in various aspects related to critical loads for acidity and for heavy metals/POPs. Source: de Wries and Bakker (1995).

Aspect	Acidity	Heavy metals/POPs
compounds	two	several/many[1)]
system	natural	natural and man-influenced
input loads	deposition	deposition and other
Effects:		
Soil	- root system - soil stability	- soil organisms - production loss - acceptable daily intake fauna and humans
water	fish	aquatic organisms including fish
time to steady state	intermediate to long	long/short[1)]

[1)]The first item refers to heavy metals and the second item to POPs.

The current development of critical loads for heavy metals and POPs in Europe are based on mass balance equations. Work along the above principals is being done in Russia, but nothing has been published yet (Posh *et al.*, 1995).

Bashkin *et al.* (1995) have made a first and simplified attempt to produce critical loads for northern Asia including Siberia for nutrient and acidifying nitrogen and for sulfur and acidity, based on the international approaches described above. This approach is based on strongly simplified steady state mass-balance equations stemming from geoinformation and an expert-modeling system (Bashkin *et al.*, 1993).

The critical loads for nitrogen and the exceedance of the critical load with current depositions of nitrogen are presented in *Figure 7.1.* Exceedance of the critical load for nitrogen mainly exists in the Ural mountains, in the boundaries to the steppes of Kazakhstan, in the Norilsk area and in the Far East. The corresponding information for sulfur is presented in *Figure 7.2.* The most serious exceedances are taking place in the Ural and Altai mountains, for the boundary regions with Kazakhstan, the Norilsk area, the Far East, Sakhalin and the Southern Kurilean islands.

Thus, critical loads for forests depends on a number of ecological factors, such as soil fertility, temperature, humidity, stand's density, landscape, etc. Some Russian investigators claim that all normatives should be local and ecologically based. This means that the values of critical loads should be calculated not on the basis of "in door" chamber experiments, but on direct field experiments. There are a number of industrial "experiments" with nature throughout Russia with different kinds of emissions, and based on these experiments there is a possibility to get at least some scientifically significant results.

To get dose-response effects, an Integral Saving Coefficient (ISC) for ecosystems was suggested (Kompleksnaya, 1992). This coefficient is founded on the following assumptions. In general, chemical substances stimulate living organisms at low concentrations and suppress as high. Experimental data show that toxic impacts have a significant "threshold" mode which justifies the use of a "critical load". The "threshold" originates from classical toxicology, where science deals with one agent, one object and a number of responses (1-1m). In ecology there are generally a number of agents, a number of objects, and a number of reactions (m-m-m). In most field studies we face some intermediate situation, where it is necessary to analyze consequences of one damaging agent (1-m-m), or consequences of a number of damaging agents (which is more close to reality). Therefore, "dose response" impacts on an ecosystem level should be used as background for critical load evaluations. This task is even more difficult than similar tasks in classical toxicology, and is not yet solved. Stepanov (1991) argued that basic field measured parameters of this kind of investigations should be expressed as an Integral Saving Coefficient of an ecosystem in the following way:

$ISC = 1/nAij/Aik*100\%$

where Aij is the value of the i-th parameter, Aik is the maximal value of the same parameter, j is the number of test areas. For forest ecosystems such parameters could be woody biomass (m^3/ha), leaf/needle biomass (ton/ha), and stand vigor (in relative units). ISC is intended to be considered as a generalized index for the ecosystem status. This approach is based on old ideas by Kayama (1961) and Pandeya (1961). A detailed description of this method of estimation in Russia is presented by Stepanov (1991) and Kompleksnaya (1992). A similar approach has been further developed by Vorobeichic, *et al.* (1994). An appropriate polygon for evaluation of the ISC, and consequently a critical load, is point sources of emissions. Test areas should be placed on transects along the depositions from this point source and along the direction of prevailing winds.

Russian scientists (Stepanov, 1991; Sadykov, 1991; Vorobeichic *et al.*, 1994) have tried to introduce ecologically based normatives for critical load estimates, and to elaborate on ecological critical loads (ECL). These normatives have been presented for only a limited number of heavy metals and for few regions. Concentration limit values and critical loads for trees based on this Russian approach are presented in *Tables 7.3* and 7.*4*. Thus, the Russian approach presented here is quite different from the critical load approach taken on at the international scene. In a comparison with international development of critical loads, these Russian estimates are not real critical loads but more concentration related limit values (mg/m^3) (*Table 7.3*).

Table 7.3. Concentration limit values of some substances for trees, mg/m^3. Source: Vorobeichic, *et al.* (1994).

Substance	CL
NO_x	0.04-0.02*
SO_2	0.3-0.016
NH_4	0.1-0.04
Benzol	0.1-0.05
Industrial dust, cement	0.2-0.05
Methanol	0.2-0.1
CO	3.0-1.0
H_2SO_4 vapor	0.1-0.03
H_2S	0.008-0.008
Fluorides	0.02-0.003
Formaldehydes	0.02-0.003
Cl	0.025-0.015
Cyclogeksan	0.2-0.2

*The first value is for single episodes and the second value is for mean daily concentration.

Table 7.4. Critical loads (CL) of some heavy metals for trees, $kg/km^{-2}yr^{-1}$. Source: Vorobeichic, *et al.* (1994).

	CL, current	CL, prospective
Cu	91.1-187.7	59.4-122.4
Pb	20.2-45.3	13.2-30.9
Cd	1.4-3.7	0.9-2.4
Zn	94.8-213.9	61.8-139.5

The Bashkin *et al.* (1995) data on critical loads of sulfur and nitrogen depositions discussed above have been applied to the ecoregions of the Siberian Forest Study. Bashkin *et al.* (1995) based the mapping of critical loads on "modified simple steady state mass-balance equations, the critical loads for nutrient and acidifying nitrogen as well as for sulfur and acidity have been calculated... using simplified expert modeling GIS and grid cells 150 x 150 km." The initial information consisted of geobotanic, soil, and biogeochemical hydrological data. For each elementary taxon (150 x 150 km) the major links of biogeochemical cycles of N, S have been characterized quantitatively. The authors suggested an algorithm for computer calculations of critical loads for nitrogen; parameters for mass-balance equations (coefficients of biogeochemical turnover, nitrogen mineralizing capacity, denitrification and leaching) were taken from available experimental case studies. Generally, this work was made on a basis of

the Manual on Mapping of Critical Loads (Task..., 1993) but in a simplified mode. The calculations on sulfur and nitrogen depositions were made on the basis of meteorological data and emissions for 1991. Critical loads for sulfur and nitrogen (based on this approach) and for the ecoregions of IIASA's Siberian Forest Study are presented in *Tables 7.5* and *7.6* and in *Figures 7.3* and *7.4*. The exceedances of the critical loads are presented in *Figures 7.5* and *7.6*. The values are given in equivalents/ha/yr. According to Bashkin *et al.* (1995), ecosystems of the arctic, subarctic and permafrost areas are very sensitive to excessive input of atmotechnogenic N (the critical load of nitrogen [CL(N) <100 eq/ha/yr). For southern Siberia the CL(N) is >300 eq/ha/yr and this region is characterized by rather high anthropogenic pressure (See Section 4). The exceedances of CL(N) are shown mainly in the boundary region with Kazakhstan steppes, in the Far East and in the lower part of the Yenisey river. The lowest values for the CL(S) are identified predominantly in the northern part of East Siberia and in Kamchatka peninsula. In the area between Yenisey and Ob rivers CL(S) values increase up to 50-100 eq/ha/yr and the highest values are observed for ecosystems with neutral and alkaline soils. The corresponding exceedances are shown for ecosystems in the northern part of Asia with the highest exceedance for Altai mountains, for the boundary regions with Kazakhstan, the lower parts of Yenisey river, the Far East, Sakhalin and the South-Kurilean islands.

Thus, Bashkin *et al.* (1995) have made the first quantitative attempt for mapping critical loads and their exceedances for vast areas of Siberia. It is evident that the number of experimental case studies backing up data for the basic calculations in these kind of investigations have to be increased, and further progress in this respect depends on additional site measurements.

Results of investigations in the Norilsk region did not reveal any signs of nitrogen eutrophying in spite of substantial nitrogen oxide emissions since 1944 (Monitoring..., 1992). Also in spite of extremely heavy S pressure in that region (~2 million/tons/year), there is no evidence of any considerable soil acidification. This could be a result of an alkali reaction of the soils and of the bedrocks in the Norilsk region. According to Menzhikov et al. (1990), the snow pH along a gradient up to a distance of 140 km from the smelters was nearly neutral or slightly alkalic. It could be explained by 1) the CaO emissions by the Norilsk cement industry, and 2) absorption by aerosols: in Russia emissions are not purified by aerosols as much as in Western countries. As a consequence, aerosols actively absorb acidity and create hard solution compounds. However, depositions of those compounds in soils are normally dangerous for plants. Available data show that there is no considerable acidification of the precipitation as well as of the snow cover (Okruzhayuchaya prirodnaya.., 1995).

It is important to note that the primary cause of forest decline is not sulfur depositions, but SO_2 or its derivatives in gaseous or aerosol forms. Stand vigor correlates better with needle sulfate-ion concentration than with S depositions. This has been proven for the Norilsk area (Monitoring..., 1992). Menzhikov *et al.* (1990) also reported a poor correlation between soil sulfur content and stand vigor.

Thus, the problem of acid rain in Russia is not as acute as in western countries. Generally, sulfur and nitrogen depositions in Siberia are considerably lower than in the European part of Russia. On the other side, the critical loads for many ecosystems of Siberia are significantly lower than those in Europe or in Southern Asia. Based on current knowledge it can be concluded that there are no dramatic "acid rain" problems in Siberia. This conclusion is based on results from analyses with traditional Russian approaches and by analyses employing simplified international methodologies.

Figure 7.1. Critical loads of nitrogen (A) and their exceedances (B) in northern Asia (free space cell means zero exceedance). Source: Bashkin, *et al.* (1995).

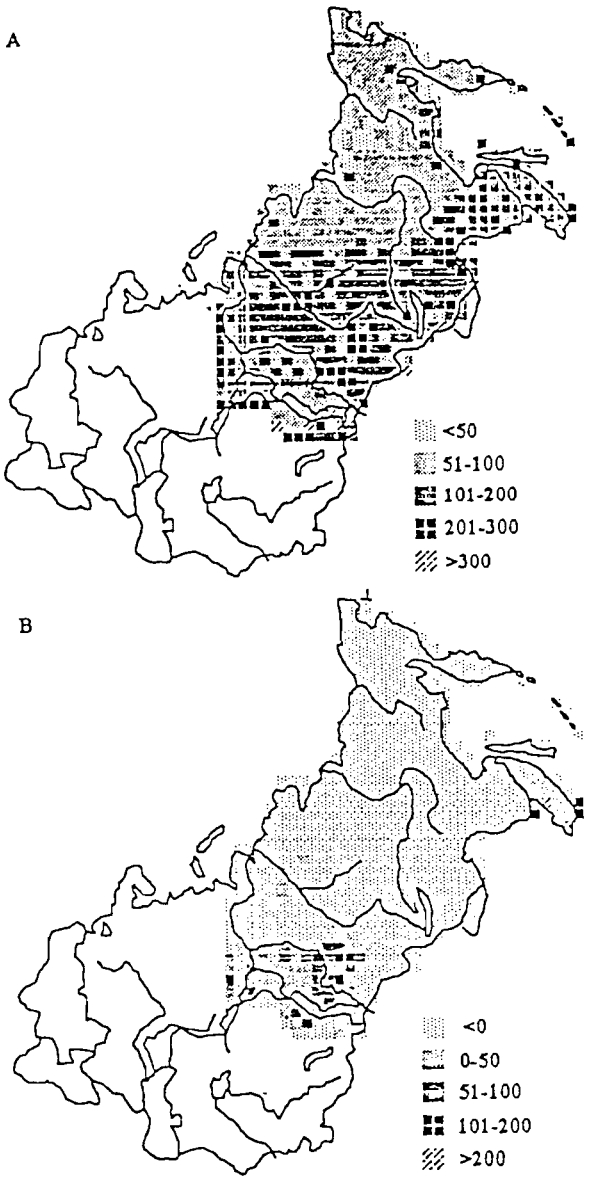


Figure 7.2. Critical loads of sulphur (A) and their exceedances (B) in the northern Asia (free space cell means zero exceedance). Source: Bashkin, *et al.* (1995).

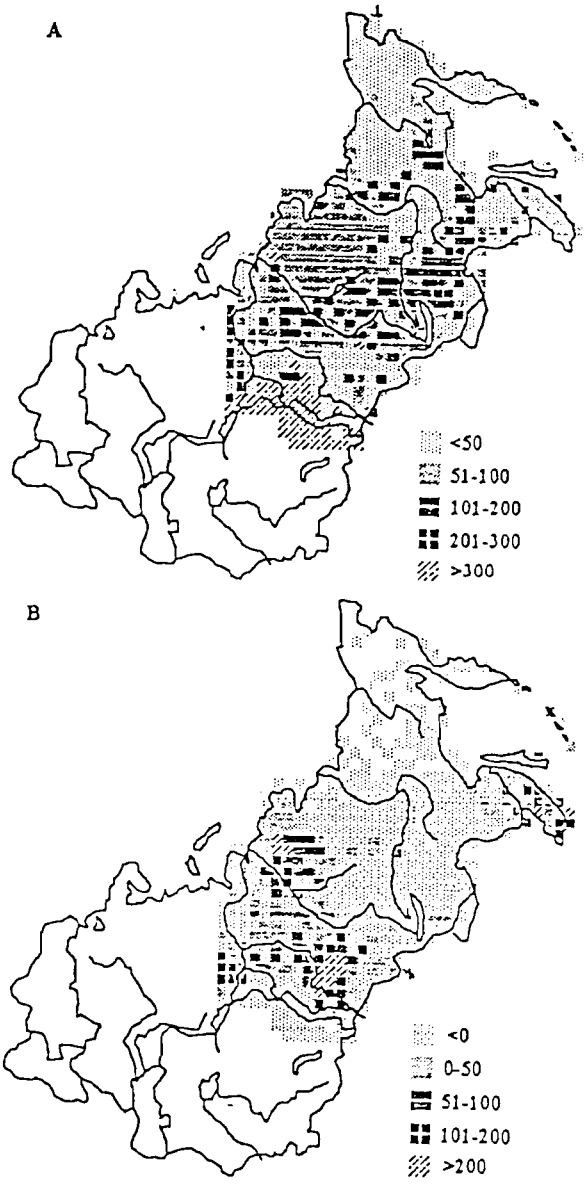


Figure 7.3. Critical loads of sulfur depositions, eq/ha/yr, mean values.

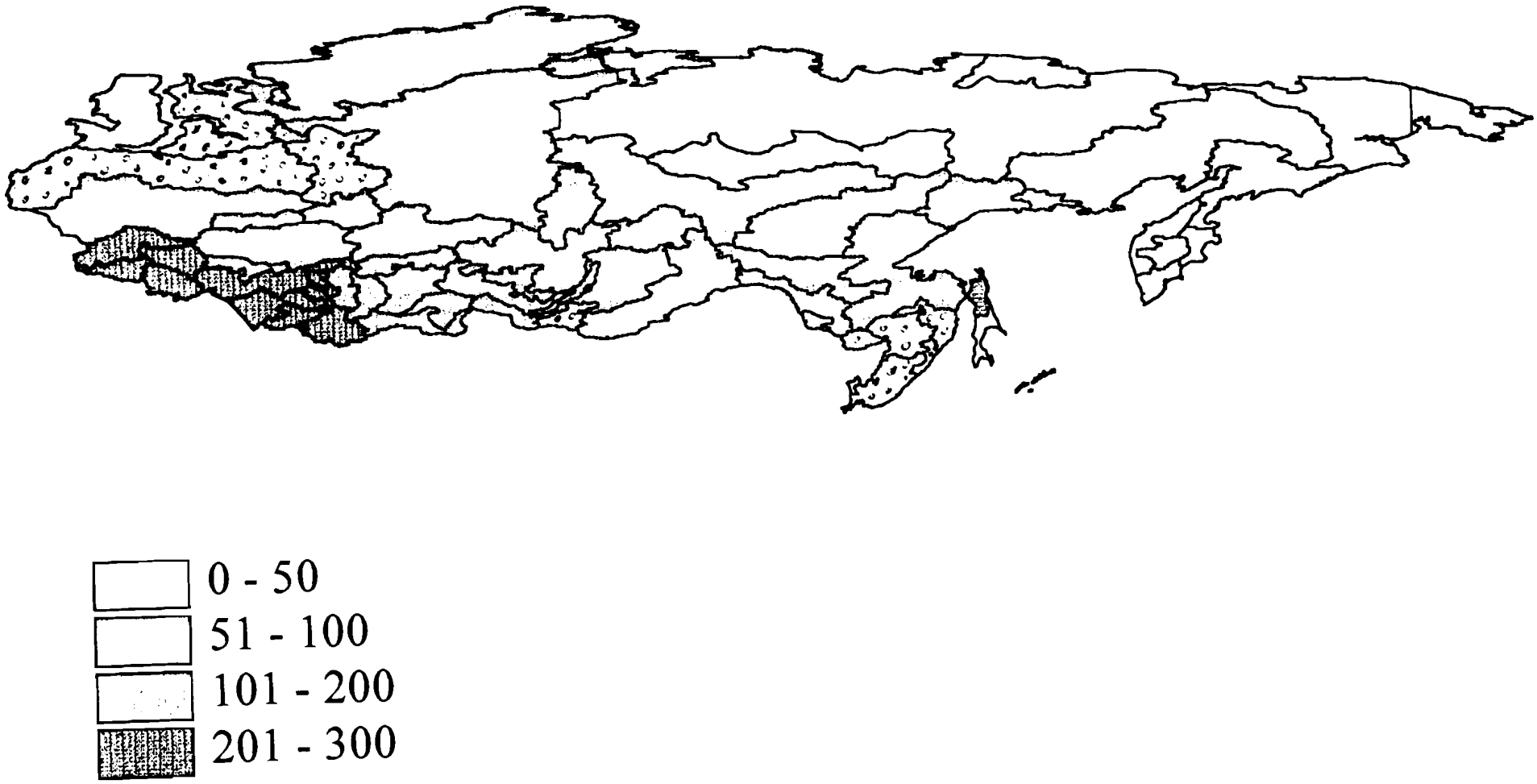


Table 7.5. Critical loads of sulfur deposition, eq/ha/year, maximum (SmaxCL), minimum (SminCL), and average (SmCL) values, and their exceedances (SmaxEX, SminEX, SmEX).

Ecoregion Code	Smin CL	Smax CL	Sm CL	Smin EX	Smax EX	Sm EX
11011	200	300	300	0	200	0
11012	200	300	300	0	50	0
11013	200	300	300	100	200	150
11014	200	300	300	51	200	0
11014	200	300	300	0	200	100
11041	101	300	200	0	200	100
11041	51	100	75	0	200	100
11042	51	100	75	0	200	100
11043	51	300	175	0	100	50
11044	101	200	150	0	100	50
11045	101	300	200	0	50	25
11046	100	300	200	0	200	100
11047	51	200	125	0	200	100
11048	100	300	200	0	200	100
11051	50	100	75	51	100	75
11052	50	50	50	51	200	125
11053	50	50	50	51	200	125
11081	50	200	125	0	50	25
11082	50	50	50	0	50	25
11083	51	100	75	0	200	100
11084	50	100	75	51	200	125
11085	50	50	50	0	100	50
11101	100	300	200	0	50	0
11102	50	200	125	0	50	0
11103	50	50	50	0	50	0
11251	101	200	150	0	50	0
11252	50	200	125	0	50	0
11253	51	200	125	0	50	25
11254	51	200	125	0	50	25
11254	51	200	125	0	5	25
11255	51	200	125	0	50	25
11301	50	50	50	0	50	0
11302	50	50	50	0	50	0
11303	50	50	50	0	50	0
11304	50	50	50	0	50	0
11305	50	50	50	0	50	0
11321	200	300	300	0	200	100
11322	200	300	300	51	200	125
11323	50	300	175	51	200	125
11323	200	300	300	100	200	150
11441	50	50	50	0	0	0
11442	50	50	50	0	0	0
11443	50	50	50	0	50	25

Ecoregion Code	Smin CL	Smax CL	Sm CL	Smin EX	Smax EX	Sm EX
11443	50	50	50	0	0	0
11501	100	300	200	0	50	25
11502	200	300	300	51	200	125
11521	200	300	300	0	50	25
11522	200	300	300	0	50	25
11641	200	300	300	0	50	0
11642	50	50	50	0	50	0
11642	50	50	50	0	50	25
11642	50	50	50	0	50	25
11691	51	200	125	0	50	25
11692	50	300	175	0	100	50
11711	50	50	50	0	0	0
11711	50	100	75	0	0	0
11712	51	100	75	0	50	25
11713	50	200	125	0	200	100
11714	101	300	300	50	200	125
11715	200	300	300	0	50	25
11761	51	200	125	0	50	25
11762	50	50	50	0	50	25
11811	50	300	175	0	50	25
11812	50	100	75	0	50	25
11931	50	200	125	0	50	25
11932	50	200	125	0	50	25
11981	51	100	75	0	0	0
11981	50	50	50	0	0	0
11982	50	200	50	0	100	0
11983	50	200	50	0	0	0
11984	50	200	125	0	100	50
11985	50	300	150	0	50	0

Figure 7.4. Critical loads of nitrogen deposition, eq/ha/yr, mean values.

Table 7.6. Critical loads of nitrogen deposition, eq/ha/yr, maximum (NmaxCL), minimum (NminCL), and average (NmCL) values, and their exceedances (NmaxEX, NminEx, NmEX).

Ecoregion Code	Nmin CL	Nmax CL	Nm CL	Nmin EX	Nmax EX	Nm EX
11011	51	200	125	0	50	25
11012	50	100	75	0	100	50
11013	101	300	200	51	100	75
11014	51	300	75	0	50	25
11014	101	200	150	0	0	0
11041	51	300	175	0	50	25
11041	101	200	150	0	0	0
11042	101	200	150	0	0	0
11043	101	300	175	0	0	0
11044	101	200	150	0	0	0
11045	51	300	175	0	0	0
11046	300	300	300	0	200	100
11047	51	200	150	0	100	50
11048	101	300	200	0	50	25
11051	101	300	200	0	50	25
11052	101	300	200	51	200	125
11053	101	300	200	0	50	25
11081	50	200	100	0	50	25
11082	51	300	175	0	50	25
11083	101	200	150	0	50	25
11084	101	200	150	0	50	25
11085	101	200	150	51	200	125
11101	100	300	250	0	0	0
11102	101	300	200	0	0	0
11103	101	200	150	0	0	0
11251	101	200	150	0	0	0
11252	51	200	125	0	0	0
11253	101	200	150	0	0	0
11254	101	200	150	0	0	0
11254	51	300	150	0	0	0
11255	50	200	125	0	0	0
11255	101	200	150	0	0	0
11301	50	100	75	0	50	25
11302	51	100	75	0	50	25
11303	50	100	75	0	50	25
11304	51	300	175	0	50	25
11305	51	100	75	0	50	25
11321	300	300	300	51	200	125
11322	300	300	300	51	200	125
11323	300	300	300	51	200	125
11323	101	300	200	51	100	75
11441	51	100	75	0	50	25
11442	51	100	75	0	50	25

Ecoregion Code	Nmin CL	Nmax CL	Nm CL	Nmin EX	Nmax EX	Nm EX
11443	50	200	125	0	50	25
11443	50	100	75	0	50	25
11501	51	200	125	0	100	50
11502	101	200	150	0	100	50
11521	51	200	125	0	50	25
11522	101	200	150	050	25	
11641	50	100	75	0	0	0
11642	101	200	150	0	0	0
11691	51	200	125	0	100	50
11692	51	200	125	0	200	100
11711	50	50	50	0	0	0
11712	51	200	125	0	0	0
11713	51	300	175	0	50	25
11714	51	300	175	0	50	25
11715	51	100	75	0	100	50
11761	101	300	250	0	0	0
11762	101	300	200	0	0	0
11811	101	300	200	0	0	0
11812	50	200	125	0	0	0
11931	51	200	125	0	50	25
11932	51	100	75	0	50	25
11981	50	50	50	0	0	0
11981	51	100	50	0	0	0
11982	50	50	50	0	0	0
11983	51	100	75	0	0	0
11984	51	200	125	0	0	0
11985	50	100	75	0	0	0

Figure 7.5. Exceedances of critical loads of sulfur deposition, eq/ha/yr, mean values.

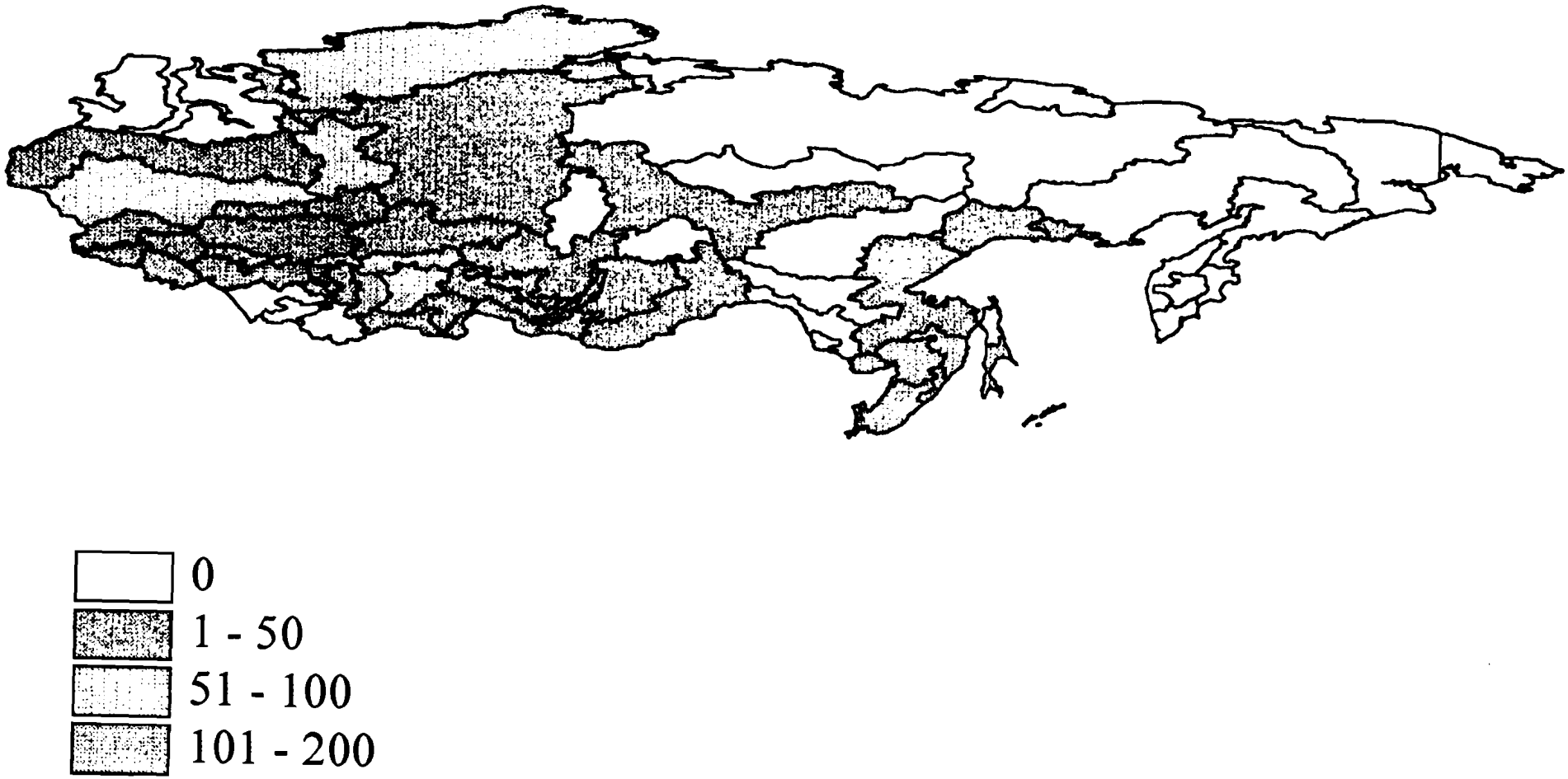


Figure 7.6. Exceedances of critical loads of nitrogen deposition, eq/ha/yr, mean values.

8. Radio Nuclide Contamination

Official data on radio nuclide contamination in Siberia are of different sources and with different reliability. This problem became open in discussions and by independent investigations in the late 1980s. Data from official reports and non-official measurements do not always coincide. This is especially true for estimates on the potential risks of the nuclear industry and waste storage. Generally it is considered, that the radioactive conditions in Siberia are determined by: - global radioactive background; - natural radioactivity; - contamination by radio nuclides due to radioactive outbreaks and underground nuclear explosions; - and by the nuclear industry and nuclear waste storage.

Table 8.1 presents the main sources of nuclear contamination in Siberia (Bulatov *et al.*, 1994).

Table 8.1 Main Sources of Nuclear Contamination in Russia and Siberia. Source: Bulatov *et al.*, 1994. Number of sources.

	Russia	**Siberia**
1. Nuclear materials mining	12	8
2. Nuclear fuel and nuclear weapon industry	15	8
3. Nuclear power stations (NPS)	12	4
4. Power units and reactors, which are equivalent to NPS (no longer in operation)	38(5)	10(2)
5. Research nuclear reactors	39	4
6. Underground nuclear explosions	93	35
7. Radio nuclide storage ("cemeteries")	21	7

8.1. Data Reliability

Published data on radio nuclide contamination have been obtained by using different instruments, different techniques and differently trained personnel. Data obtained by specialized departments at nuclear plants seem to be the most reliable, since the personnel is highly qualified, and the instruments used have acceptable precision (Lebedev, 1995; Malyshkin *et al.*, 1995; Zidkov, 1995). But there is also room for some doubt concerning the completeness of reported data. The reported data by the State Ecological Committees have been obtained annually on test sites in different regions of Siberia (Gosudarstvenny doklad..., 1995; Ecologicheskoe sostoyanie..., 1995; Ecologicheskaya obstanovka..., 1995). The accuracy of the data presented is not always clear. The standard deviation (in cases where it is reported) is of ±10-14% (Gosudartsvenny doklad..., 1995). The accuracy of aircraft gamma-spectrometry (Nazarov *et al.*, 1983) is estimated to be ±10-20%.

8.2. Atmospheric Deposition as a Source of Nuclear Contamination

In West Siberia (Tumen, Omsk, Novosibirsk regions) the atmospheric depositions of ^{137}Cs in 1994 were 5-6 Bq/m^2/day, which is ~2 times less than in 1991. The ^{90}Sr concentration of the atmospheric deposition was below detectors sensitivity (Sostoyanie okruzhayuschei..., 1994; Sostoyanie okruzhayuschey..., 1995; Ecologicheskoe sostoyanie..., 1995). In East Siberia (Irkutsk region, Yakutiya republic) the rate of depositions were similar. In Irkutsk region the highest values of beta activity was 14.6 Bq/m^2/day (Ecologicheska ya obstanovka..., 1995). In Yakutia republic the mean daily depositions were 1.0-9.0 with maximum values of 40.0 Bq/m^2/day. Depositions of ^{137}Cs was practically zero. Depositions of ^{90}Sr were at a stable level during the last 5 years (0.43 Bq/m^2/month). The total air beta-activity concentration in 1994 was at the average level for the past 5 years (30.8 x 10-5 Bq/m^3), which corresponds to the background level. The ^{137}Cs mean air concentration was 0.04 x 10-5 Bq/m^3 and the ^{90}Sr concentration was below instrumental sensitivity (Gosudarstvenny..., 1994). These values seem to be representative for the dominating part of the Siberian territory (Ecologicheskoe..., 1995; Ecologicheskaya...1995; Obzor..., 1994; Gosudarstvenny..., 1995).

The influence of the Chernobyl (1986) and the Kishtim (Chelyabinsk region, 1957) catastrophes on the air contamination in Siberia are considered to be negligible. According to the Russian State Committee of Sanitary and Epidimiologee, the mean levels of air contamination by ^{137}Cs and ^{90}Sr in 1994 were at the same level as before these two catastrophes.

8.3. Soil and Bedrock Natural Radioactivity

Nazarov, *et al.* (1983) have published specific data on soil radioactivity for the entire territory of the USSR. The geographic distribution of radioactive elements (uranium, thorium, potassium) is complex. This is due to the plentiful causes which control the concentration of radioactive elements in soils and rocks. The main factors are types and origin of the soils and bedrock, soil genesis and its mechanical composition, relief and water content. Concentration of the different elements varies between (0.1...4.5) x 4-10% for uranium, (0.1...16) x 4-10% for thorium, and 0.1-3% for potassium. In West Siberia the mean concentrations of radioactive elements gradually increase from the north to the south. This general trend interferes with irregularities caused by local alluvial and bedrock depositions. In East Siberia a large part of the bedrock is open (~40% of area). Thus uranium, thorium and potassium concentrations reflect the initial radioactivity of the bedrock. The concentrations vary within the following limits: (0.5...3.5) x 4-10% for uranium, (1...14) x 4-10% for thorium, 0.3-3.0% for potassium. The concentration of uranium in soils of the Chita region (east of Baikal lake) is the highest in Russia. The average concentrations of uranium, thorium and potassium in mountain soils are of the same levels as in European and North American mountains (Nazarov *et al.*, 1983).

Recent data give additional information on natural Siberian soil radioactivity. In West Siberia the average values for the isotopes, K-40, Th-232 and Ra-226 were 464, 38 and 32 Bq/kg respectively. The background levels for those isotopes are equal to 370, 25 and 38 Bq/kg respectively (Sostoyanie okruzhayuschey..., 1995; Ecologicheskoe sostoyanie..., 1995). In

East Siberia (Yakutia republic) the background level of natural gamma ray power dose was measured to be in the range of 8-19 μR/hour. Anomalous values (>65 μR/hour) were observed only within very local areas (<10 m^2) (Gosudrstvenny doklad..., 1994; 1995).

Throughout Siberia there are a number of sites with increased soil/bedrock natural radioactivity caused by depositions, but this kind of information is poorly described in available literature.

8.4. Soil Radio Nuclide Contamination

The mean ^{137}Cs deposit in Siberia is 2.3 x 109 Bq/km^2 (except in high mountainous areas) with the variations of 5.6 x 108 - 7.4 x 109 Bq/km^2 (Nazarov *et al.*, 1983). The ^{137}Cs geographic distribution is characterized by latitudinal zones with a large amount of "spots" of increased radioactivity due to local specificity of different regions. The maximum levels of contamination have mainly been observed in the latitudinal belt between 50-60 degrees and with deposits of (3.7...6.5) x 109 Bq/km^2. South and north of this belt the rate of contamination decreases. The miminum rates of contamination are detected at the latitudes >70 degrees and <45 degrees (9.2 x 108-1.85 x 109 Bq/km^2). In mountainous regions the contamination is higher. East of Yenisey river the deviations from the latitudinal zones correlate with the precipitation levels, which is considered to be an effect of the mountainous landscapes. The power gamma ray dose of ^{137}Cs for the ecoregions of IIASA's Siberian Forest Study (at the height of 1 m) is presented in *Figure 8.1* and *Table 8.2*. The power dose varies in the range of 0.1-1.25 μR/hour with a mean value of ~0.5 μR/hour, which is considerably less than the natural background of soil radioactivity (see above). The power gamma ray dose geographic distribution is similar to the ^{137}Cs deposition (Nazarov *et al.*, 1983).

The data presented above are higher than the data presented later by Regional Ecological Committees. In West Siberia (Tumen region) the regional data show a ^{137}Cs and ^{90}Sr soil contaminations (in the layer of 0-20 cm) which do not exceed 1.3 x 109 Bq/km^2 and 0.9 x 109 Bq/km^2 correspondingly (e.g., within global background levels). The levels of soil contamination within some river valleys are higher. For the Iset river, for example, the contamination is 7.4 x 109 Bq/km^2 for ^{90}Sr and (1.5-5.5) x 109 Bq/km^2 for ^{137}Cs. Since 1962 the contamination has decreased by 5-25 times (Ecologicheskoe..., 1995).

In Irkutsk region the ^{137}Cs soil contamination exceeds the background levels by 280 Bq/kg. More than 95% of the radiation activity is concentrated in the 6 cm top soil layer. The maximum level of ^{137}Cs contamination exceeded the background level by 5-6 times (Ecologicheskaya obstanovka..., 1995). This indicates that the contaminations within regions cab be caused by man-made depositions. In Siberian Far East the soil contamination was within 0.022-0.041 Ci/km^2. The soil radio nuclide content was 4.1-6.3 Bq/kg for ^{90}Sr, and <6.7 Bq/kg for ^{137}Cs, which is within the background levels.

8.5. Nuclear Industry

Two of the three nuclear military industries in Russia are located in Siberia. They are Krasnoyarsk Mount Chemical Plant (KMCP) (so called Krasnoyarsk-26, at Yenisey river some 60 km south of the city of Krasnoyarsk), and the Siberian Chemical Plant (SCP) (so called Tomsk-7, 15 km north of the city of Tomsk).

KMCP in Krasnoyarsk-26 has been in operation since 1950. The final products of KMCP are PuO2 and uranilnitrate. Currently, two of the three existing nuclear reactors are stopped. The total are of the KMCP plant (together with sanitary defense zone) is ~13,000 ha. Within this area there are spots of nuclide contamination (Lebedev, 1995). Currently, the atmospheric contamination by the plant is considerably below the critical limits (7.6% of allowed beta-activity, for example). Since 1992, the emission of inert gases to the atmosphere have decreased by 27 times. After closing the so called direct current reactors, the atmospheric radioactivity in the sanitary defense zone has decreased by 8 times. It is stated that the KMCP impact on the atmospheric contamination is considerably below the global background radioactivity (Zidkov, 1995). Radioactivity of berries in the site, where smoke touches the soil, is below critical values. The direct current reactors have caused radioactivity of the river bottom and sand banks of islands; mean values of radioactivity are 7.4 x 109 Bq/km^2, and with a maximum of 5.2 x 1010 Bq/km^2. A radioactive "patch" is detected 1500 km down the river. The radioactivity 500 km down the river is 0-15 μR/hour with spots of ~200 μR/hour. Currently the radioactive emission into water is <6% of allowed limits (Okruzayuschaya prirodnaya..., 1995; Zidkov, 1995). The general radioactive condition in the Yenisey valley down the river has been classified as not dangerous for the people's health (Zidkov, 1995). Data published in a report by the Krasnoyarsk Regional Ecological Committee (Sostoyanie okruzhayuschey..., 1995) differ from the data cited above. In the bottom sediments of the river the activity of 51 Cr reached 1.3 x 1012 Bq/km^2, for 60 Co 1.2 x 1011, for 65 Zn 9.2 x 1010, for ^{137}Cs 8.1 x 1010 Bq/km^2. The Pu activity was measured to 20-27 Bq/kg. The territory of contamination includes ~300 km^2 of islands and river banks and the maximum values of ^{137}Cs islands contamination is (2.3-6.3) x 1011 Bq/km^2.

The nuclear plant in Tomsk-7 is one of the five largest plants in the world. The total nuclear radioactivity of the plant's nuclear wastes is ~526 million Ci. In Tomsk-7 the radioactivity around the plant has been considered to be "stable" for many years and with a background level of radioactivity of 7-15 μR/hour. The radioactive emission to the atmosphere is considered to be 0.1-6.0% of the allowed limits. Within the so called "industrial zone" of the plant the levels of ^{137}Cs and ^{90}Sr soil contamination are within the background level ((1.1-3.7) x 109 Bq/km^2). North of the plant (7-8 km along the wind rose) ^{137}Cs activity is 2-3 times higher in comparison with the global background activity. This is caused by pollution which has occurred over decades (Malyhkin *et al.*, 1995). Currently the emission of nuclides to the Tom river is <80% of the allowable limits. ^{90}Sr and ^{137}Cs contaminations are currently not detected. During 1990-1992 two of the five nuclear reactors were closed in Tomsk-7.

8.6. Radioactive Outbreaks

There are official reported data on several accidents or incidents in the nuclear plants in Siberia, although it was also stated that the influences of the accidents were within acceptable limits. In Tomsk-7, there were 24 incidents of different intensity reported (Bulatov and Chirkov, 1994). Only one incident has been reported as potentially dangerous. A chemical explosion occurred in a technical unit on April 6, 1993 in the Siberian Chemical Plant (SCP). Before its destruction, this unit contained 25 m^3 of solutions with 8773 kg of uranium and 310 kg of plutonium, as well as short-lived isotopes like 103 Ru, 106 Ru, 95 Ni, and 95 Zr. Its total radioactivity was 559.3 Ci. Different sources give more or less a similar picture of the consequences of this incident. The leakage of activity was ~5% of the total beta and gamma activity (Sostoyanie okruzhayuschei..., 1994). The length of the radioactive footprint with a gamma ray power dose of 60 μR/hour was 15 km and with a maximum width of 3 km. The contaminated area was ~3500 ha and the extremely contaminated area (>1000 μR/hour) was ~600 ha. The total length of the radioactive footprint with an activity of >15 μR/hour was 123 km^2. The total contaminated (by short lived isotopes mainly) area was estimated to ~150 km^2 (Okruzayuschaya prirodnaya..., 1995). The total amount of radioactive products was estimated to 40-50 Ci. The maximum density of plutonium contamination was 0.2 x 109 Bq/km^2. According to Bulatov and Chirkov (1994) the total impact of the accident was estimated to 115 Ci. Malyshkin *et al.* (1995) reported that the territory of radioactivity with values >20 μR/hour was 43 km^2 caused mainly by 103 Ru, 106 Ru, 95 Ni and 95 Zr. The total radioactivity outside the "industrial zone" (33 km^2) was 115.3 Ci. The plutonium contamination was in line with the global background level. Observations in 1994 showed that there is no contamination outside the "industrial zone" with the exception of several spots along an emission path with activity of 22-24 μR/hour, which were caused by Ru-106. The activity of this isotope has decreased (since the accident by 1.2-2 times, and is currently (0.52-4.9)x1010 Bq/km^2. It is stated that the incident in 1993 did not worsen the radioactive conditions around the SCP. According to data of independent ecologists (Checketkin, 1995), the surroundings of SCP (within a radius of 10-20 km) were contaminated already before the accident. The contamination were up to 3100 Bq/km^2 for Pu, up to 13,700 for ^{90}Sr, and up to 15,300 Bq/km^2 for ^{137}Cs. After the accident, the level of vegetation radioactivity reached 6.5 Bq/kg.

It is stated that in the KMCP plant (Krasnoyarsk-26) there has not been any leakage of nuclear materials during the last 17 years.

8.7. Nuclear Minerals Mining

In zones of radioactive minerals mining [Zabaikal'ye (East of Baikal lake), Tuva republic, Chukotka peninsula] there are spots of nuclear contamination. In Chita region (East of Baikal lake, in Zabaikalsky and Priargunsky mining industries) the radioactivity is measured to 50-150 μR/hour due to mining and open storage of radioactive and uranium tailings (Bulatov and Chirkov, 1994; Okruzhayuchaya..., 1995). On sites in the Yakutiya republic the level of the power dose has been measured as highly variable: 13-650 μR/hour, and with a uranium concentration of 0.0001-0.1%. The maximum values of depositions were measured to >12,000 Bq/kg (Gosudarstvenny..., 1994).

8.8. Nuclear Underground Explosions

There has been a set of nuclear underground explosions in Siberia during the period 1960s-1980s. They were so called "geophysical" explosions whose main purpose was for the reconnaissance of mineral resources, and, in some cases, for preparing underground storages (of oil, for example). The explosions were located in West Siberia, in Evenkiya (Krasnoyarsk region) and in Yakutiya republic. No radioactive leakage was reported from nine underground nuclear explosion sites in Krasnoyarsk region. In the Tumen region, 8 nuclear explosions were carried out and currently only one of them (Sredne-Balykskoe oil deposition, so called "Benzol") is in operation. The purpose of that explosion (dated to 1985) was to increase the oil output. There is no nucleotide leakage reported from any of the explosion sites in Tumen (Ecologicheskoe..., 1995).

In Yakutiya, the consequences of 12 underground nuclear explosions (which were carried out during 1974-1987) have been investigated. The nuclear explosions on the sites called "Kristall" and "Kraton-3" caused radioactive leakage. On the "Kristall" site, the contaminated zone with a $^{239, 240}$Pu activity of >10 Bq/kg (the background level is ~1 Bq/kg) have the dimensions of 0.25 x 0.05-1.0 km. On the "Kraton-3" site, the contaminated territory is 3.0-3.5 km in length and is 1.0-1.2 km wide (northeast of the site). On the dominating part of this territory the contamination was >5.6 x 1011 Bq/km^2. The maximum ^{90}Sr activity in soil near the center of the explosion was 3.0 x 1013 Bq/km^2, and the maximum ^{137}Cs activity was 101,442 Bq/kg. The maximum activities (^{90}Sr + ^{137}Cs) in soils reached 500,000 Bq/kg. The concentration of $^{239, 240}$Pu in one test site (a creek bottom; the creek flows through the explosion location) exceeds the global background level by 1590 times. This value exceeds the mean values of the Pu concentration in soil within the 30 km Chernobyl accident zone (630 Bq/kg) by 2.5 times. The ^{137}Cs contamination of soils in Yakutiya was of the same order or lower than the global level (53 Bq/kg). In 46 tests it was <10 Bq/kg, in 10 tests <50, and in 3 tests 62.4, 70.8, 89.8 Bq/kg, respectively. In vegetation the concentrations were higher: in 34 tests <100, in 23 tests <250, and in 3 tests 272.9, 348.0, 459.3 Bq/kg respectively (Gosudarstvenny doklad..., 1994; 1995).

8.9. Nuclear Waste Storage

In Siberia there are several sites for nuclear waste storage. Recently, reports by State Ecological Committees include information on the safety of these storage sites. Officially, the storages have been considered as reliable with no leakage (e.g., the nuclear waste storage in Irkutsk region (Ecologicheskaya obstanovka..., 1995)). Liquid nuclear wastes of low and medium levels of radioactivity from the Krasnoyarsk Mount Chemical Plant (KMCP) are stored at the polygon "Severny" (12 km from the KMCP) at the depth of 200-500 m. "Hard" wastes were stored in special storages. During the time of the KMCP operations also 6500 m^3 of radioactive pulp were stored in special tanks with a total activity of ~110 million Ci. In open "water pools" there were 50,000 m^3 of radioactive pulp with a total activity of 20,000 Ci. There are no data on nuclide leakage from this storage. The state of the polygon is considered satisfactory with no negative influence on the ecological conditions (Lebedev, 1995).

However, it has been reported that along a transportation tube (water transport) there are spots of radioactivity and the maximum values of ^{137}Cs were 3,000,000 Bq/m^2, of $^{239, 240}Pu$ 180,000 Bq/m^2. It has been stated that those spots originated from the initial exploitation of the storage (>25 years back) (Sostoyanie okruzhayuschey, 1995). The vegetation contamination on spots was 800-6800 Bq/kg for ^{137}Cs and 13-61 Bq/kg for Pu (Chechetkin, 1995). Outside the sanitary defense zone there are no limitations on agriculture production due to the contamination (Nosukhin and Revenko, 1995).

Some experts link an additional risk of contamination to the so called RT-2, a specialized plant for temporary storage and recycling of nuclear waste. The nuclear waste originates from nuclear power stations. The RT-2 was built at the KMCP in 1977 (Lebedev, 1995).

8.10. Other Sources of Contamination

The nuclear power stations in Siberia (Beloyarskaya nuclear power station in Tumen region, Bilibinskaya nuclear power station in Magadan region) as well as the affiliated nuclear waste storage are considered to be absolutely safe (Ecologicheskoe..., 1995).

Radio nuclide containing instruments, which are used for industrial purposes, such as tube seam control, are also a source of radioactive contamination. It is stated that some of them could be out of control, and several cases have been reported where this was the cause of nuclear contamination of urban areas.

Oil industry could also be a source of contamination. In some cases, oil filtrates have levels of radioactivity of ~400 μR/hour (the maximum value measured is 5600 μR/hour). In one reported case, the oil exploitation caused radioactivity of the soils of 280 μR/hour. The cause is natural radioactivity, which leaks in the process of oil extraction.

Building materials with a high natural level of radioactivity have caused radioactivity, in some cases (i.e., road construction), 20-55 μR/hour (Ecologicheskoe..., 1995). In some cases it also caused an increased level of radioactivity in urban areas. In Siberian cities, the mean power exposition gamma radioactivity is 5-10 μR/hour, and the concentrations of radioactive elements: Uranium (0.5-1.5) x 10-4, Thorium (3-5) x 10-4, Potassium-40 0.6-1.5 (%). The average ^{137}Cs contamination is (0.15-0.55) x 1010 Bq/km^2, which corresponds to the average global contamination (the mean ^{137}Cs contamination density in middle latitudes of the

Northern hemisphere is (0.4-0.48) x 1010 Bq/km^2). The power gamma ray dose of ^{137}Cs (at the height of 1m) is presented in *Figure 8.1* and *Table 8.2*.

Figure 8.1. ^{137}Cs Gamma Ray Power Dose (μR/hour) on 1m above ground.

Table 8.2 ^{137}Cs Gamma Ray Power Dose (μR/hour) at 1 m above ground level.

Ecoregion Code	μR/hour
11011	n.a.
11012	.50
11013	.50
11014	.50
11041	.35
11041	.75
11042	.50
11043	.75
11044	.75
11045	.75
11046	.50
11047	1.00
11048	.50
11051	.50
11052	.50
11053	.50
11081	n.a.
11082	.75
11083	.50
11084	.50
11085	1.00
11101	1.00
11102	1.00
11103	.75
11251	.60
11252	n.a.
11253	.75
11254	.75
11255	.75
11301	.25
11302	.25
11303	.25
11304	n.a.
11305	.25
11321	.50
11322	.50
11323	.50
11441	.50
11441	.25
11442	.50
11443	.25
11501	.75
11502	.50
11521	.75
11522	.50
11641	.75

Ecoregion Code	μR/hour
11642	1.00
11642	.50
11691	.50
11692	.75
11711	.50
11712	.75
11713	.50
11714	.75
11715	.50
11761	1.00
11762	n.a.
11811	1.00
11812	n.a.
11931	n.a.
11932	n.a.
11981	.25
11982	.50
11983	.35
11984	.50
11985	1.00

8.11 Accumulation of Radionuclides in Forest Ecosystems

In the area of "radioactive contamination of the environment, its assessment and mitigation" the problem of accumulation of radionuclides in forest ecosystems is an important one. The forest represents a specific natural object, in which the distribution, accumulation and migration of radionuclides have their own specifics. It is connected with the special role of a forest ecosystem as a strong barrier against radionuclides dispersion. For example, aboveground pine and spruce stands absorb about 30% of the annual precipitation, deciduous stands-about 10-15%. Forests play a major role in stabilizing the radiation situation as a significant part of radionuclides is accumulated in tree canopies and in understory. Depending on the type and age of forest stands, meteorological conditions, chemical composition of precipitation the intensity of radionuclide contamination in the forest may be from 6 to 12 times higher than in open fields.

Forests are capable to prevent a widespread radiocontamination of the environment by accumulating it. This capability might lead to a dramatic forests' contamination. Wash-out of radionuclides from the aboveground forest biomass is from 10 to 25 times less than from crops. In comparison with crops contamination, radionuclides are accumulated over years in aged stands of forests. A primary reservoir for radionuclides is the forest soil. The uptake of radionuclides through the root system permanently increases radionuclides concentration in needles, branches and wood.

Radionuclides Migration in Forest Ecosystems

Usually in the year immediately following an emission, surface deposition is the primary mechanism of the radioactive contamination of forests, while in later years direct uptake from the soil via the root system predominates. Normally, as a result of radioactive emissions a wide range of radionuclides is deposited. Many of these have a relatively short half-life time or are present in such small amounts, that they are only of short-term radiological significance. From the radiological point-of-view the most important radionuclides are ^{90}Sr, Pu isotopes and ^{137}Cs as they are long-lived isotopes. They are of greatest concern with respect to contamination of land, forests, soil-to-plant and are the most important contributors to the radioactive dose. Thus radionuclides can contaminate the plant via three main pathways:

1. uptake through the soil solution into the root system;

2. uptake through the alv;

3. direct external contamination (direct deposition, rain, resuspension).

Each of these routes should be adressed, but the main emphasis is on the soil-root pathway.

Here we present some results of ecological impacts studies and conclusions, that have been made on the basis of data collected on the radioactive contaminations of forest vegetation after the Chernobyl nuclear accident (Parfenov and Yakushev, 1995) and the Kyshtym accident in Chelyabinsk region of Russia (Sokolov and Krivoluzkii, 1993), which created the so-called East-Ural Radioactive Track (EURT).

8.11.1 Initial Period of Vertical Radionuclides Migration

Analysis of experimental data shows (Tikhomirov, 1993) that the foliage of deciduous forests (during the vegetation period) and needles of coniferous forests (during the whole year) absorb from 50 to 100% of deposed radionuclides, depending on the density of stands and meteorological conditions at the period of deposition. Deciduous forests without leaves absorb about 25% of disposed radionuclides (*Table 8.3*).

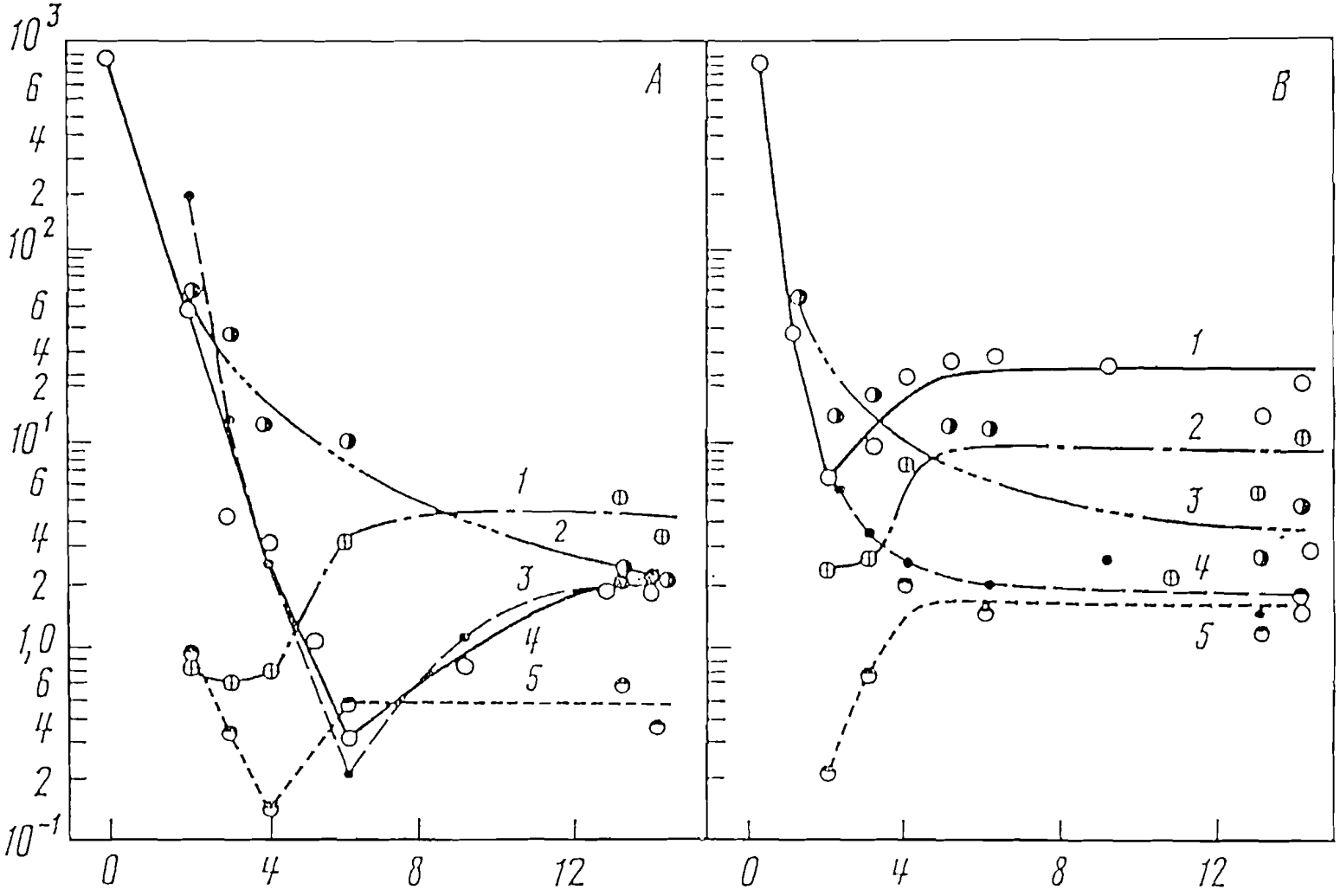
Table 8.3. Radionuclides Absorption by Pine and Birch Stands, % of total deposition.

Type of Forest Stands	Type of Isotopes deposition	Coefficient of Absorption, %
6 to 10 years old pine stands	Spraying of ^{90}Sr solution	90-100
60 years old pine stand, density - 0,9	Particle deposition (size up to 50 mcm (EURT)	80-100
25 years old pine stand, density - 0,8	Particle deposition (size up to 100 mcm (EURT)	70-90
30 years old pine stand, density - 0,8	Secondary Particle deposition (moved from soil by wind)	40-60
40 years old pine stand, before budding, density - 0,8	Secondary Particle deposition (moved from soil by wind)	20-25
50 to 60 years coniferous and deciduous stands, density - 0,7-0,9	Radioactive deposition with particle's size up to 20 mcm (Chernobyl accident)	60-90

Immediately after radionuclides deposition the process of their migration from tree canopies starts. It is determined mainly by the biogenic fall. Migration with precipitation is not significant in comparison with the biogenic fall - only a few percentages of the total migration. Therefore the intensity of radionuclides migration in forests depends on the physiological state of trees, determined by seasonal conditions. In physiologically active state decontamination of tree canopies is fast even without precipitation. In autumn the process of decontamination of standing trees is slower than in summer. During winter periods this process is 4-5 times slower.

Figure 8.2 shows the dynamics of ^{90}Sr in pine (A) and birch (B) biomass after ^{89}Sr aerosol deposition. The period of radionuclides half-life time in aboveground biomass for pine is from 8 to 10 months, for birch - about 2-3 weeks (due to leave dropping).

Figure 8.2 Dynamics of ^{90}Sr in the biomass of pine (A) and birch (B) (dose- $1 Mbq/m^2$).
A: 1 - floema, 2 - inner bark, 3 - twigs, 4 - needles, 5 - wood;
B: 1 - leaves, 2 - floema, 3 - bark, 4 - twigs, 5 - wood;



The period of vertical radionuclides migration (95% in deciduous forests and 99% in coniferous forests of the total deposition) to the litter layer is 2 years for birch stands and about 5 years for pine stands. During this period the main source for contamination of the aboveground stands are radionuclides, taken up through the above ground part of the trees. In a few weeks after a radionuclide deposition not only leaves/needles and branches are contaminated, but also living bark with cambia wood layers, which are protected from direct contamination by the bark layer. Few years after the deposition of radionuclides, nuclides, absorbed by the stem and branches, are the main sources of ^{90}Sr and ^{137}Cs migration to young sprouts and wood.

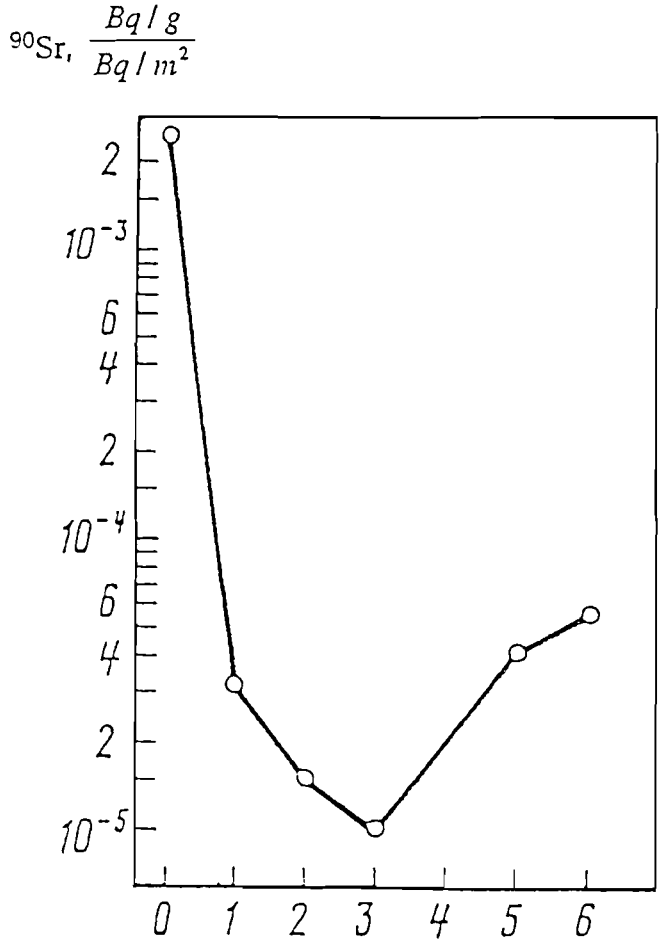
The level of radionuclide contamination increases in leaves/needles, bark and wood after the initial period of migration and stays later constant for a few years: 4-6 years for deciduous stands, 10-12 years for coniferous stands (after the deposition). Periods differ due to different processes of natural decontamination for conifers and deciduous trees.

8.11.2. Radionuclide Migration in Different Components of a Tree

Radioactivity of birch leaves

Figure 8.3 shows that within a period of a few years, after the deposition, specific radioactivity of the leaves have decreased. During this period, the contamination was mainly via radionuclides redistribution in tree canopies and contamination by aerosols from other trees. The curve also reflects the vertical migration of radionuclides from tree canopies to the forest floor and their accumulation in the litter layer and in the soil. After 3 years, the ^{90}Sr concentration starts to increase as a result of uptake from the soil. In the following 4 years the equilibrium state is reached with deviations of no more than 30%. In the case of strong radiation damage the radioactivity of leaves reaches the minimum one year later and in this latter case it takes 6-8 years to reach the equilibrium state.

Figure 8.3 Dynamics of ^{90}Sr in birch leaves (years after deposition).



Radioactivity of pine needles

During the first years after the deposition the distribution of radionuclides in pine canopies is not uniform. The highest specific radioactivity is in needles in the lower layers of the canopy. The vertical gradient of the radionuclides distribution is determined by vertical migration of the radionuclides. For needles, developed after the deposition, the distribution of radionuclides is uniform. The processes of decontamination in alive needles are much faster than in dead needles (*Figure 8.4*). During the first four years after the deposition the processes of vertical migration of radionuclides dominate. This migration is explained by the fact that old (more contaminated) needles drop. During this period, alive needles are contaminated by radionuclides only through the process of redistribution between different parts of the tree and therefore radionuclides' concentration is decreasing. After 4 years it has decreased by more than 10^3 times.

Radioactivity of branches and bark

During the first years after the deposition, the radioactivity of branches and bark is dependent on the uptake through the above ground part. Therefore, the distribution of radionuclides in branches and bark and its dynamics (vertical migration) are caused by washing-out by rain, blowing-out by wind, and bark scaling.

The absolute values of specific radioactivity are strongly different for pine and birch. This fact might be explained by two reasons:

- the different life time of leaves and needles and therefore they have a different role as a source for the second wave of contamination;
- difference in roughness of branches. Due to the roughness, the specific radioactivity of pine branches is 2-3 times higher than of birch branches. In 6 years the radioactivity decreased by 1000 times in branches. On the seventh year radioactivity of branches started to increase due to the ^{90}Sr uptake through the soil. After 13 years the equilibrium state was reached.

The process of decontamination of bark is approximately the same for pine and birch. In 13-14 years the ^{90}Sr radioactivity became close to the equilibrium state. During the process of vertical migration some parts of the washed-out radionuclides were absorbed by bark and lower parts of the stem.

Radioactivity of bast and wood

During the initial period of contamination, bast and wood are contaminated via biological transport of radionuclides from needles, leaves and branches. It's known that the mobility of ^{90}Sr, ^{144}Ce, ^{106}Ru is restricted and their transport to other parts of a tree is not very significant. Therefore, the radioactivity in bast and wood during the first years is much less than in needles, leaves and bark. A 4 years' accumulation in birch via uptake through the roots, increased the ^{90}Sr concentration by 4 times and became close to equilibrium. For pine the equilibrium state was reached in 6 years after the radioactive deposition, because the root uptake of radionuclides became significant 2-3 years later than for birch.

As a result of radionuclides vertical migration from tree canopies to the understory the main part of radionuclides is concentrated in the litter layer and later on reaches the root layer of the soil.

8.11.3. Migration of Radionuclides within Forest Soils

Following a nuclear plant accident, the primary reservoir for radionuclides deposited in a forest ecosystem is the soil. The forest soil is a very strong absorbent of radionuclides. It is composed of organic matter, inorganic matter, water, gas and soil fauna. Particles enter the forest ground through the litter layer which consists primarily of organic matter. There are two possible states of particles: soluble and non-soluble. Particles that reach the forest floor with needles or leaves are non-soluble and therefore they are immobile and remain in this state until either their decomposition or leaching by precipitation. After the release of particles from the non-soluble state they enter a soluble form and are mobile with water. Particles that are transferred to the litter layer by leaching of the canopy are already in soluble form, and enter directly into the soil solution. Particles in the forest soil may also exist in its inorganic layers. Three physical states can be considered: soil solution, absorbed by the soil, and fixed in soil. Particles primarily migrate in the forest soil by the soil solution. Radionuclides that migrate past the root zone of the forests are assumed to be lost in terms of cycling within the ecosystem; i.e., the radionuclides will either remain in the inorganic soil, leave the forest through the underground aquifer, or decay to an inactive form. Depending upon the particular radionuclide, particles in the soil solution may sorb to the inorganic matter. This process includes adsorption, chemisorption, and ion exchange (Berg and Shuman, 1994).

An investigation of the radioactivity of the soil, layer by layer, has shown that the boundary of radionuclides penetration into the ground is not planar or a straight line but has a complicated, fingered surface. This is basically related to the way in which atmospheric precipitation soaks through woodland soils.

8.11.4. Radioactivity of Soil

The most important factors influencing the uptake of radionuclides from the soil include:

- the level of contamination in the soil;
- the soil and forest type.

Chernobyl Nuclear Accident

It is clear that both now and in the future, it is ^{137}Cs and ^{90}Sr which are and will continue to be of greatest concern with respect to contamination of land and soil-to-plant transfer. A study of territorial distribution of long-lived radionuclides of ^{90}Sr, ^{137}Cs and Pu isotops in natural plant complexes of Belarus has shown that Cs isotops were scattered everywhere. Large quantities of ^{90}Sr isotopes were found within a 10 kilometer zone. The farther from the damaged reactor, the number of ^{90}Sr sharply decreased: at the distance of 250 km the ratio was 1:10 for Pu - 1:1000, in relation to the ^{137}Cs concentration (Parfenov and Yakushev (1995).

Both field and laboratory studies have demonstrated the very low mobility of radioactive particles in soils. The rates of leaching of strontium and caesium from particles are significant but their subsequent behavior is dependent on the soil type. Vertical profiles measured in disturbed and undisturbed land around Chernobyl showed that even several years after the accident the particles had not penetrated the 5-6 cm layer.

Predictions for forest soils forecast that even after 50 years, no more than 10% of the ^{90}Sr and ^{137}Cs will have reached the 5-6 cm layer (Petrayev *et al.*, 1991) Similarly, Silantev *et al.*, (1989), predicted that on grassland and in the forest, where vegetation and litter contribute to the hold-up, only about 1% of the two radionuclides would penetrate to 5 cm. On wet meadows a few percent would reach the 5 cm layer.

Konoplev *et al.*, (1993), examined soil cores around Chernobyl and showed that the vertical distributions of Cs, Ru and Ce (but not Sr) were the same despite the fact that they all differ with respect to fixation in soil. Differences between migration of ^{137}Cs and ^{90}Sr are significant. Migration of ^{90}Sr is largely determined by its exchangeable form and its movement with the infiltration flow. From measurements of vertical distribution, the diffusion-like migration coefficients for different soil type were obtained. The parameters deduced by Konopolev *et al.* (1993) show that the loss from the upper layers of the soil profile is slow for both strontium and caesium.

Plants absorb active forms of radionuclides that are distributed in the root area. The amount of radioactive elements, biological characteristics of plants and active parts of elements define the level of radioactive accumulation in plants. Annenkov and Yudintseva (1991) point out that the level of ^{90}Sr adsorption by soil is much higher than for ^{137}Cs (5-10 times).

Following the deposition, the level of radionuclides uptake by the soil strongly depends on the soil type. If caesium associates with clay mineral fractions of soils, it becomes progressively unavailable for uptake by plants. (Schultz et al. 1960; Tamura, 1964). The rate and degree of fixation will depend on the presence of suitable exchange sites and will vary with the type of

soil (Cremeres *et al.*, 1988). Organic matter has been shown to be less effective in the fixation of caesium in a manner that makes it unavailable for plant uptake (Barber, 1964). Peat soils, characterized by a high organic matter content, low pH, an absence of clay minerals and a deficiency in potassium, have been shown to be particularly vulnerable following contamination with radiocaesium (Livens and Loveland, 1988).

The Kyshtym Accident

In the area of the Kyshtym accident, the radionuclides chemical activity was in the following order $^{137}Ce < ^{90}Sr < ^{144}Ce < ^{106}Ru$. The natural migration activity of ^{90}Sr and ^{137}Ce in EURT also depends on the soil type and is presented in the following order: sod-podzolic > grey forest > chernozem. Vertical migration is determined by processes of diffusion and mass transportation. Linear velocity vertical migration in 15 cm soil layer is 0,2-0,4 cm/year for ^{90}Sr and 0,15-0,3 cm/year for ^{137}Ce. During a period of 30 years a redistribution of radionuclides in the soil profile took place according to *Table 8.4.*

Table 8.4 ^{90}Sr and ^{137}Cs content in different soil layers in 1988, in percentage.

Soil Type	^{90}Sr			^{137}Ce		
	0-2cm	0-10cm	10-50cm	0-2cm	0-10cm	10-50cm
Chernozem leached	31	94	6	41	93	7
Grey forest	11	84	16	51	89	11
Sod-podzolic	8	54	46	44	84	16

8.11.5. Radionuclides Accumulation by Trees

Accumulation and Isotopic Structure

The question of accumulation and distribution of different radioisotopes in plants is very complicated. Results of observations show that in the second year after the Chernobyl accident the main part of radionuclides accumulated by the litter layer moved to the upper layer of the soil and became the main source for the radionuclide contamination of the aboveground biomass. While some amount of radioactive elements still had been accumulated in different parts of the trees through other pathways the process of foliage decontamination was very noticable. During the subsequent years this natural process continued. After three years the specific radioactivity of different parts of the trees decreased by 10-15 times, but still some parts of the trees were contaminated, especially the bark. There is a direct connection between soil radioactive contamination and radioactive accumulation in the forest biomass: with increased radioactive elements in the soil, the accumulation in trees increases. But according to Gulyakin and Yudinceva (1962), there is a limit to the radioactive

concentration in trees. This is connected with the fact of decreasing tree vitality with strongly increased radiation.

The most significant contamination of trees is caused by ^{90}Sr and ^{137}Cs. Under certain conditions they are accumulated by trees through the root system and therefore may influence the process of the forest vitality and determine the possibility of future utilization for commercial purposes. The major part of the other radioactive isotopes (^{103}Ru, ^{106}Ru, ^{144}Ce) accumulated through the root system constitutes small amounts and therefore they are not significant. Results of observations showed that ^{137}Cs and ^{134}Cs have the highest level of uptake through the root system among the radioactive isotopes. Knowledge and understanding of the process driving the uptake of $^{134,137}Cs$ by vascular plants and fungi is strongly limited. For example, the root uptake of radiocaesium by vascular plants, and its subsequent role in plants is not well understood; i.e. there exists no known biological requirement for radiocaesium in the growth process.

For the comparison of radionuclides transport in the system soil-to-plant., coefficients of accumulation (CA) are used (the ratio between concentrations of the element in the plant and in the soil). This varies for different tree species (*Table 8.5*). The highest values of CA of ^{137}Cs are for birch (2.8-3.8). For oak and aspen approximately the same (1.39-1.56 and 1.42-1.44 respectively). The highest level of ^{90}Sr accumulation is in oak (0.79).

Table 8.5 Coefficients of accumulation (Perfenov and Yakushev, 1995).

Tree species	^{90}Sr	Pu	^{144}Ce	^{144}Pr	^{106}Ru	^{134}Cs	^{137}Cs
Birch	0.50	0.30	1.44	2.79	1.52	2.85	3.82
Aspen	0.60	0.09	1.66	-	-	1.42	1.44
Oak	0.79	0.18	1.37	2.72	0.83	1.39	1.56
Alder	0.60	0.22	1.12	0.29	0.53	0.53	0.71
Pine	0.45	0.19	0.73	0.73	0.88	0.88	0.74

Available data show that under similar conditions, deciduous forests accumulate more radiocaesium and radiostrontium than coniferous forests. However the processes of absorption and accumulation of radioactive elements through the soil for coniferous and deciduous forests have a similar character:

1. direct dependence between the radioactive accumulation and the concentration of soil contamination;
2. radioactive concentration varies in different parts of trees and the highest level is in the photosynthetic part of the trees.

8.11.6. Biological Specifics of Tree Species by Radionuclides Accumulation.

It is known that radionuclides uptake through the soil and the level of accumulation in trees strongly depend on the chemical conditons, physio-chemical characteristics of the environment, climatic and topographic conditions, structure of the biogeocenosis, but also on the biological characteristics of the phytocenosis's types (activity of the physiological processes, type of root systems, intensity of growth development, vegetation period, productivity, etc.). Differences in radionuclide accumulation through the root system vary by 10-30 times for different tree species.

Table 8.6 shows ^{90}Sr accumulation through the soil in different parts of pine and birch stands 13 years after the Kyshtym accident.

Table 8.6. ^{90}Sr accumulation through soil (Sokolov and Krivoluzbii, 1993).

	Tree Part	^{90}Sr, Bq/g	Percentage
B	Leaves	21,6	37.1
I	Bark	13	22.3
R	Wood	1,75	3
C	Small Branches	15,5	26.2
H	Large Branches	6,4	11
P	Needles	1,21	22.1
I	Bark	1,80	33
N	Wood	0,20	3.7
E	Small Branches	1,4	25.6
	Large Branches	0,85	15.6

Radionuclides are mainly accumulated by foliage and needles. High concentration in the bark and sprouts is explained by aerial contamination. The lowest radionuclide concentrations are measured in the wood. However, results of measurements show, that the concentration of ^{90}Sr in the wood of birch and pine increased by 10 times within a period of 10 years after the radioactive deposition in the under story (Annenkov and Yudinzeva, 1991).

The comparison of ^{90}Sr accumulation dynamics in the aboveground forest biomass for different densities of soil radioactive contamination shows that the relative ^{90}Sr root uptake (Bq/g:MBq/m^2) in tree phytomass is decreasing for higher levels of radioactive contamination at which tree canopies were damaged by radiation. *Figure 8.5* shows the dynamics of ^{90}Sr accumulation in birch leaves as a function of the ^{90}Sr density in soil. One can see that the equilibrium of relative ^{90}Sr accumulation by leaves of damaged trees (100 MBq/m^2) is one order lower than equilibrium for healthy trees (0,1-1 MBq/m^2). The same results as for the dynamics of ^{90}Sr accumulation in birch leaves were obtained for pine needles and pine and birch wood. These results show that in the case of higher levels of radioactive depositions it will cause tree canopy radiation decline (> 10MBq/ m^2) and that the main part of roots in the upper (5 cm) layer are damaged as well. Damage of the root system in this layer might lead to a decrease in mineral elements uptake, including ^{90}Sr. During the following period, characterized by the redistribution of radioactive elements and decay processes, the root system recovers and after 12-13 years the level of radioactive contamination by damaged and

healthy trees is the same. To predict maximum concentrations of ^{90}Sr in aboveground biomass the following formula is used (Aleksakhin and Narishkin, 1977)

$$\frac{Crad}{Csoil-rad} = \frac{Cst}{Csoil-st},$$

where $Crad$, Cst and $Csoil-rad$, $Csoil-st$ are concentrations of ^{90}Sr and stable strontium in tree species and soil respectively.
There are data available for one case in Siberia where the possible cause of forest damage was radioactivity. In Yakutiya Republic, on the site "Kraton-3" after the underground nuclear explosion (See Section 8.8) a forest area of 0.5 x 5 km died. The direct cause of this is not clear, and it is doubtful that the dieback was only a consequence of direct radioactive impact. Larch bark, bushes and mosses were strongly contaminated. Within the zone of dead forests the ^{90}Sr activity in mosses varies between 10,300 (marginal areas) and 44,000 (central site) Bq/kg (Gosudarstvenny doklad..., 1994; 1995).

Sviderskaya (1996) have investigated *Pinus silvestris* growth development on nuclear test areas in Semipalatinsk poligone (Kasakhstan Republic). The test sites are located at ~60 km from the nuclear test area. The radial growth increment and the xilem cell structure were studied. Special attention was paid to disturbances after 1949 (the beginning of nuclear tests in the atmosphere). No deviations from the reference areas, which could be attributed to radioactivity were discovered. The radioactivity of the samples were also within the background level. Measurements of the increment of Pinus silvestris in the vicinity of "water pools" (nuclear storage in Krasnoyarsk-26, See Section 8.9) have been made. No deviations from the reference areas concerning the increment were discovered and no deviations were observed at the cell level.

Main similarities of radionuclides accumulation in different parts of tree species:

- The dominating part of the radionuclides (up to 80-90% of the deposition) is usually accumulated in aboveground parts, mainly in the foliage (to 40-45%) and in the branches (up to 20%).
- Small roots accumulate radionuclides by 2-4 times more than larger roots. Roots up to 5 mm in diameter taken for analysis from the top 5-cm layer of soil had a radioactive cesium content of 920-2200 Bq/kg; larger-diameter roots contained less - 540-1380 Bq/kg.
- The wood itself has the lowest radionuclide content. The contamination level in bark, lime, and cambium is 10-15 times higher than (ecpecially, of oak) that of wood.
- The level of radioactivity in living bark and inner bark is significantly higher than in the wood of coniferous and deciduous trees. The range of variation of radioactivity in the bark is within limits of 1150-1760 and 240-370 Bq/kg for coniferous and deciduous stands respectively.
- The radioactivity of the stem wood decreases from outer layers to the center.
- The wood contamination increases from the base of steam to the top.

Tree species specifics in accumulation of radionuclides from the soil:

- Birch accumulates ^{137}Cs and ^{90}Sr 2-9 times more than pine. The accumulation of ^{137}Cs by birch foliage is 7 times, and for ^{90}Sr - 19 times higher than in pine needles.
- The specific radioactivity for ^{137}Cs of birch branches is 2 times less than for pine branches.
- The maximum concentration of ^{90}Sr for pine is in the roots, for birch -in the foliage. The ^{90}Sr accumulation by branches of pine is 2.8 times higher than for birch.
- The roots of conifers and alders contain more radionuclides than those of birch trees.
- Birch wood accumulates less radioactive cesium than pine and alder in the vicinity of contaminated land or on land contaminated to the same degree: pine wood 130-200 Bq/kg; alder - 80-130 Bq/kg, birch - 30-40 Bq/kg.

Figure 8.4 Dependence of ^{90}Sr in pine needles on the density of contamination (1-1959, 2-1960, 3-1961).

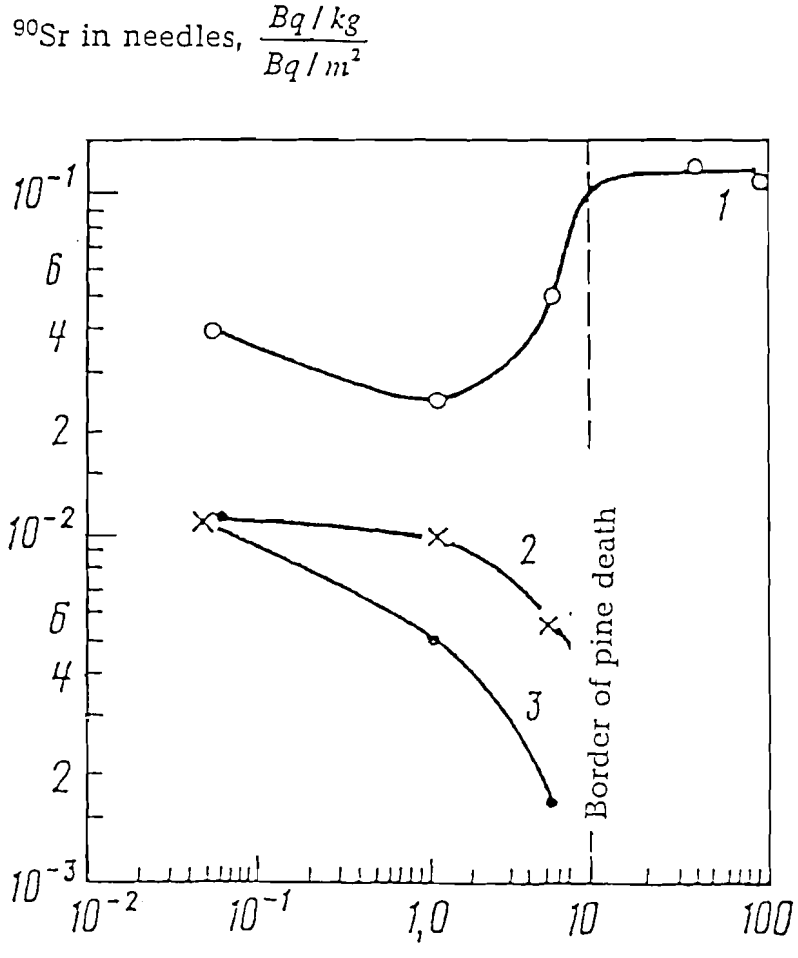
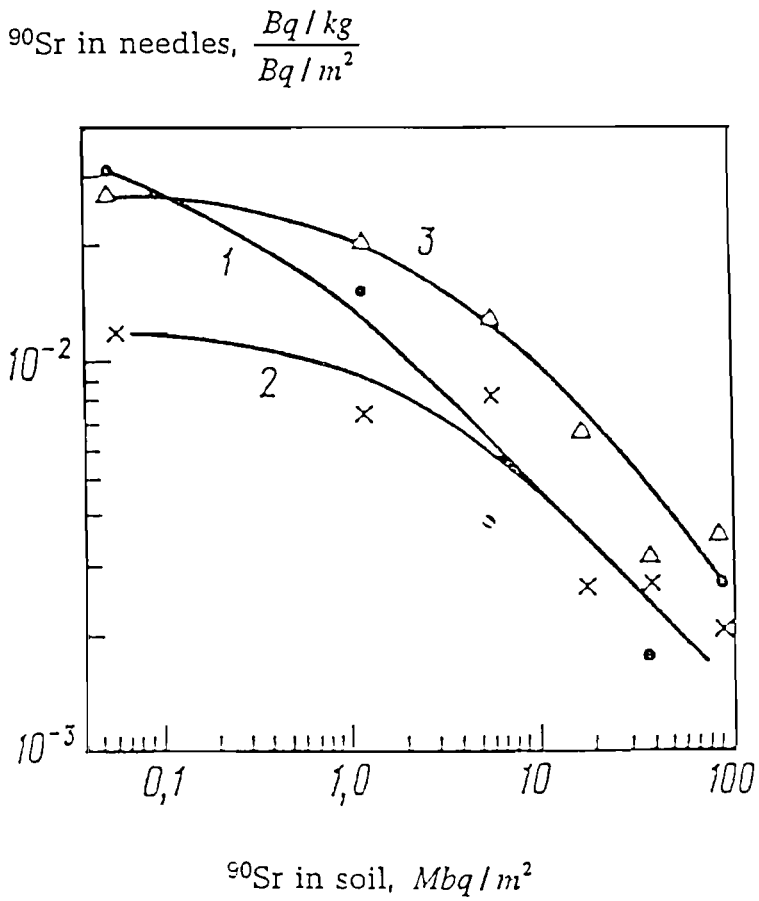


Figure 8.5 Dependence of ^{90}Sr in pine needles on the density of contamination (1-1959, 2-1960, 3-1961)



8.12. Conclusions

1. Atmospheric deposition in Siberia is not regarded as being a major source of nuclear contamination due to the fact that the contamination is composed mainly by natural isotopes.

2. The soil's natural radioactivity in Siberia is considered to be within the background level except for areas with naturally increased levels of radioactivity and sites of nuclear minerals mining. The mean concentrations of uranium, thorium and potassium in mountain soils are at the same level as in European and North American mountains.

3. Soil contamination by ^{137}Cs and ^{90}Sr through natural radioactivity is highest in the latitudinal belt of 50-60 degrees north with deposits of (3.7...6.5) x 109 Bq/km^2. In addition, there are spots of radioactive contamination within urban areas caused by nuclear waste, and by building materials with naturally increased levels of radioactivity. Spots of contamination are identified in the vicinity of nuclear waste storage and along nuclear waste transportation tubes. Underground explosions have caused local soil contamination in at least two cases.

4. Currently the nuclear industry is not considered as a significant source of nuclear contamination in Siberia. But until 1992, the so called direct current reactors caused problems with nuclear contamination in the Yenisey river valley. The accident in the Siberian Chemical Plant (1993) is not considered to have caused any dramatic impact on the ecology of the affected area. However, questionmarks remain concerning the severity of the problems for the Yenisiy river basin and Irkutsk region. But the nuclear waste storage is a high potential risk for future radioactive accumulation in the Siberian forest ecosystems. This high risk condition requires further investigations.

5. There are no data which support the idea that forest ecosystems in Siberia currently are under substantial risk of radioactive damage, and that currently there seems to be a negligible amount of radioactive accumulation in the woody parts of the Siberian forests.

9. Forest Decline in Siberia

In Russia the reported total forest decline in the form of dead forests up to 1988 was about 50,000-150,000 ha/yr (*Figure 9.1*). Since 1989, a strong increase has been reported in the areas of decline due to the fact that data on fire damage were included in the reports. The impact of the main causes for forest decline during the last several years has been rather stable. The variations of the decline reported and caused by pollution during this period (*Figure 9.2*) are mainly due to changed data collection (Obzor sanitarnogo..., 1994). Mean annual forest mortality caused by anthropogenic (MantMa) factors and pollution (MpolMa), as well as forest mortality intensity (in ha/yr/1000 ha forests) caused by all stress factors are presented in *Figures 9.3-9.5* and *Table 9.1*. In Russia the officially reported total accummulated area of dead or severely damaged forests by pollution in 1993 is estimated to be 832,461 ha (Obzor sanitarnogo..., 1994). In 1993 and in total Russia anthropogenic factors caused a mortality of 2718 ha from which 1142 ha were caused by pollution. Vast spruce and fir stands decline in the Far East is considered to be of biotic origin (Manko and Gladkova, 1995). In Siberia the intensity of pollution-induced mortality (Mpol) varies between 0.01 >Mpol>5 ha/1000 ha forests with the highest values in Irkutsk region and the Norilsk area. Thus, special emphasis should be given to those territories.

9.1. Irkutsk Region

The polluted area in Irkutsk region is ~31,300 km^2. The polluted area around Irkutsk city is about 3400 km^2 (in comparison: around Kemerovo ~55,600, around Novosibirsk ~1750 km^2). The pollution caused dieback of pine stands in the territory is ~110,000 ha, and the pine stand decline is additionally >600,000 ha (Reznikov, 1995). The most dramatic situation is around Bratsk city, where pine stands have died on an area of >100,000 ha due to emissions of fluorides by the aluminum plant. In the first years of pollution the decline rate was very rapid due to a "shock" poisoning effect of the trees. Currently, a process of more steady decline caused by pollution is observed. According to existing forecast, an additional >150,000 ha of pine forests are estimated to die up to year 2000. There is also evidence of pine decline around Shelekhov city caused by fluorides.

Around Baikalsk city the SO_2 emissions were reduced by 38% during the last 10 years. This reduction decreased the sulfur content in fir needles by 33%. The forecast for the forest vitality around Lake Baikal is the following:

- in the case of a stabilization of the pollution rate at today's level, there will be progressive fungi damage to fir stands;
- in the case of a considerable reduction of SO_2 emissions, the fir stands will restore their vigor.

Irkutsk scientists (Siberian Institute of Plant Physiology and Biochemistry) have analyzed the vigor of the pine stands in the territory (~500,000 ha), and discovered weakened stands all over the territory. On the dominating part of the surveyed territory, the stands are in a medium stage of decline. Increment reduction has been observed since the 1960s. Reforestation is poor in this area due to a decreased production and poor quality of seeds.

Pine stands in the upper part of the Angara river valley are considered to be in the IV level of decline ("severely damaged ecosystems" according to a 5-grade scale). According to the most likely scenario, within 20 years there will be a massive forest die-back and stands have moved into the phase V level of decline (destroyed ecosystems). Pine stands will also be substituted by birch and aspen stands (Ekologicheskaya obstanovka..., 1994).

The main cause of forest decline in the Irkutsk region is fluoride emissions, and to a lesser extent, SO_2 emissions. The precipitation acidity is within the range 5.0...7.3.

9.2. Norilsk Zone

The Norilsk mining industry has been affecting the so called "pretundra forests" since the 1940s. Pretundra forests are formed by *Larix sibirica, Larix dahurica, Picea obovata, Betula verrucosa*, and by a variety of willow species. These stands form the northern tree line. The main Norilsk pollutants are SO_2, NO_x, and heavy metals. Currently, the annual emission of the main pollutant, SO_2, exceeds 2 millions tons. In comparison, a similar plant on the Kola peninsula, near the Finnish border, produced only 0.25 million tons/year (the total SO_2 emission in Russia is about 20 million tons/yr). The history of forest decline in the Norilsk area goes back to the time of the construction of the Norilsk industry. From the late 1930s till the beginning of the 1950s, the pretundra forests in this area were an important source of regional wood supply. The earliest available data on forest decline caused by pollution in the Norilsk area go back to the late 1960s, and the area of dead forest was estimated to ~5,000 ha. Interviews with local aboriginal people established the beginning of the degradation to the middle of the 1950s.

The forests in the region grow in the Norilsk basin (the plane is about 150-170 km in length, 20-60 km in width with many bogs and small lakes). They are mainly oldgrowth (200-300 years), of low density (0.4 in average, 0.8 in the river valleys). The soils are often very poorly drained brown, cryaquept, and tundra histosoils. The average depth of soil thawing during the summer is 0.4-0.6 m. The forest floor is composed by a thick layer of mosses and lichens. The precipitation is 400-500 mm and the average air temperature during the growing season is above +10 C. The growing season is approximately 60 days. The smoke from the Norilsk industry is transferred mainly in the Southeastern direction (following the wind rose) along the Norilsk basin. The main smelter smoke stacks are of the following heights: 138m, 150m (constructed in the 1940s), 180 (1950s), and 250m (constructed in 1980).

The emission dynamics of the main pollutant, SO_2, is presented in *Figure 9.6*. The annual SO_2 emission in the beginning of the 1990s was ~2 million tons. The emission of NO_x does not exceed 20,000 tons/year. The typical signs of forest decline are discoloration (loss of chlorophyll in the case of chronic SO_2 influence), "needle burn" (direct impact of pollution smoke), defoliation, changes in the shape of the tree crown, and decline of increment. The sensitivity to pollutants is estimated to decrease in the following order: larch-spruce-birch-willow. This is in contradiction to earlier reported data of higher resistance of larch in comparison to evergreen spruce (Trenshow, 1988). In the areas of direct pollution impact, even the most resistant species (willows) are damaged. Outside the direct impact zone the level of damage depends on prevailing winds, landscape, and soil fertility. Outside the wind rose direction the stands are not eliminated even in the vicinity of the smelters (5-10 km). The stands on the wind-protected exposures are considerably less affected, as well as stands in the

river valley. The healthiest stands grow on more fertile soils with an organic matter content of 1.5-4.0 times higher, and with a nitrogen content of 2.8-8.0 times higher than in soils with more damaged stands.

The concentrations of main pollutants along a pollution gradient is represented in *Figure 9.7*. The soil sulfur content varies between 30-540 kg/ha for the upper 0-2 cm, and between 100-250 kg/ha for the 0-20 cm layer. In spite of an intensive pollution rate, the changes in the soil pH have not been dramatic. The pH values are never below 4.4 even in the direct impact zone; and normally they are in the range of 5.3...7.0. The process of acidification is controlled by a high soil buffer capacity and the emissions of CaO by the cement industry in Norilsk. The concentration of heavy metals (Cu, Ni, Co) in the upper soil horizon (0-10 cm) exceed the background level by 10-1000 times within a distance of 30 km from the smelters (*Figure 9.7*). The maximum values on the graph correspond to the areas where the smoke "touches" the ground. The ratios (Ca+Mg)/Cu and (Ca+Mg)/Ni are used as indexes of soil toxicity. The critical values are 10 and 5, correspondingly. According to these indexes, soils are toxic within a radius of 20-25 km from the smelters. The concentration of some biogenic elements (Mg, K, Ca, Fe) in the larch needles is 1.3-5.0 times higher in the affected areas.

Satellite image analyses have been used for evaluating the forest decline. The images dated 1979, 1982 and 1984 were used for generating simple maps of the affected area (*Figure 9.8, A,B,C*). The process of forest decline spread in the south-eastern direction, indicating that the main pollutant transfer took place along the Norilsk valley (*Figure 9.8A*). The images obtained in 1982 and 1984 showed considerable increases in the forest decline rate (*Figure 9.8 B,C*). The total emission of SO_2 reached its maximum at this time (2.3 million tons/year; *Figure 9.6*).

Figure 9.8 show forest mortality at a distance up to 80-100 km, forest decline at a distance up to 200 km. Data showed a decrease of the increment and an increase of the forest mortality rate by 5-15 times during the period 1965-1985 (Ivshin, 1993). The expansion of the pollution impact to the southern direction is limited by mountains (the average altitude is 800-900 m). Lichens beyond the ridge showed signs of damage at a distance of 200-250 km (Monitoring..., 1992).

Due to the fact that forest damage has spread to a distance of ~200 km, the SO_2 emissions are probably the main factor of the forest decline. A good correlation was found between needle sulfur content and the level of the stand damage (Monitoring..., 1992). In comparison with sulfur dioxide emissions, the NO_x emissions (20,000 tons/year) is negligible. The heavy metal impacts are limited in the vicinity of the smelters (*Figure 9.7*). The current area affected by decline in the Norilsk region is ~2 million ha.

An important question is how resistant the pretundra species are to sulfur depositions. There are no data on the SO_2 concentration of the air along the pollution gradient, except for the Norilsk city air (*Figure 9.6*). The pollution sources are at a distance of 2-10 km from downtown. Reported sulfur dioxide concentrations in the city air are about 0.1-0.2 mg/m^3, and maximum ("shock") concentrations could reach 20 mg/m^3. The average concentration exceeds the human allowable concentrations (0.05 mg/m^3) by some 40 times.

There are no data for the Siberian larch resistance to SO_2. Analyses of similar data for other larch species (Trenshow, 1988) allow us to estimate the critical values for "shock" concentrations to 0.15 mg/m^3, and to 0.03 mg/m^3 for the daily average concentrations, and 0.01 mg/m^3 for the growing season.

Ivshin (1993) made investigations on increment reduction due to pollution. Graphs in *Figures 9.9* and *9.10* show the difference in increments for different test areas. The coordinates of test areas are shown on *Figure 9.8A*. The data show a considerable reduction of the increment during 1965-1985. In that period the Norilsk industry started to use a high sulfur content in the production. The pollution effect was increased due to unfavorable climatic conditions in that period (synergism). In spite of the fact that the climatic conditions improved in the mid 1970s, the increment decrease continued (*Figures 9.9, 9.10*). The decline can be attributed to the huge increase of SO_2 emissions in this period caused by the "Nadezda" industry; this conclusion is supported by independent data (*Figures 9A,B,C*). On test area #3 the growth reduction was ~60%. Increment decrease was also discovered in areas with no visual symptoms of decline. On the basis of the data above, it was concluded that in the near future the dominating part of the pretundra forests in Norilsk basin will die out.

In addition to pollution, there are also natural factors causing forest decline within the studied area. These factors are overmaturing, poor regeneration, and fungi diseases. Natural regeneration is limited by thick lichens and "pillows" of mosses. Wild fires are the natural promoters of the regeneration process, since they mineralize the soil. On post-fires territories the amount of natural regeneration is increased by 10-20 times, reaching 10-20,000 seedlings/per ha. There also exist dendrochronology data indicating that climate is a driving force of forest decline (Vaganov *et al.*, 1994). In general, the total impact of non-pollution factors on the forest decline has been estimated to be 20% of the total decline.

Forest decline in the Norilsk area is considered to be the greatest pollution-induced ecological catastrophe in the boreal region. Effects of similar industries in North America (Copper Hills, USA and Sudbury, Canada) have had considerably less impact on the environment (Hutchinson and Whitby, 1977). The "Norilsk phenomena" is a unique "man-made" experiment in the boreal biome.

Thus, the primary cause of forest decline in the Norilsk area is SO_2 emissions. Negative impacts of heavy metals is restricted to a radius of ~20-30 km, and even within that radius the dominant damage is caused by SO_2 or its derivatives.

However, based on the above overview on forest decline caused by air pollutants, we may have an extent of 3-3.5 million ha in Siberia, which is 3-4 times more than official reports for total Russia. Thus, there is a strong need for an improved monitoring system in Siberia and Russia on the vitality of the forest resources.

Figure 9.1. Forest Mortality (in ha) Caused by Fires and Other Agents.

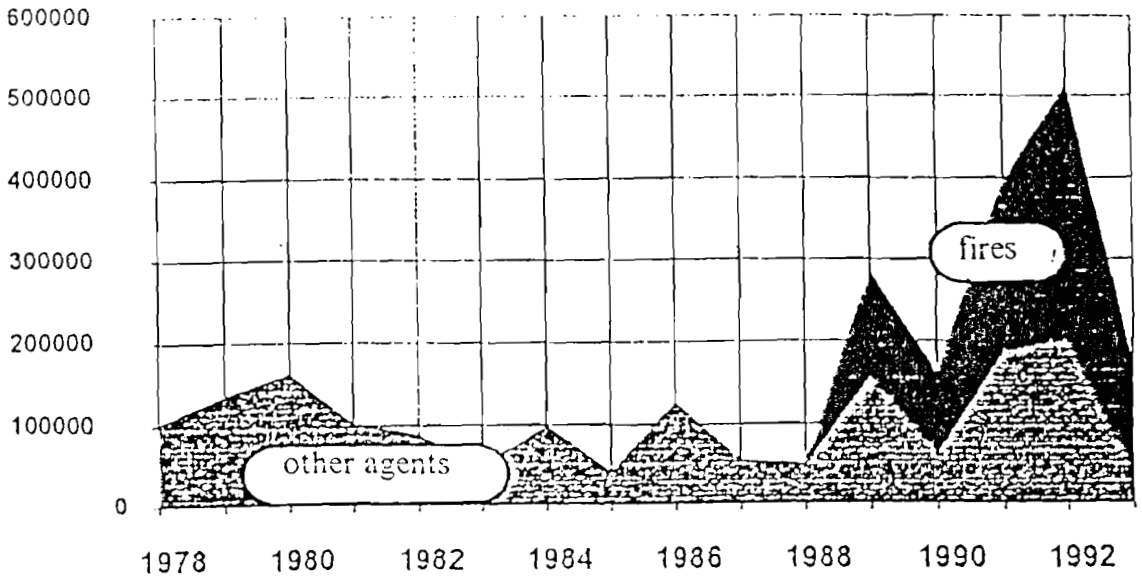


Figure 9.2. Relative Importance (in %) of Different Agents (except fires) for Forest Mortality.

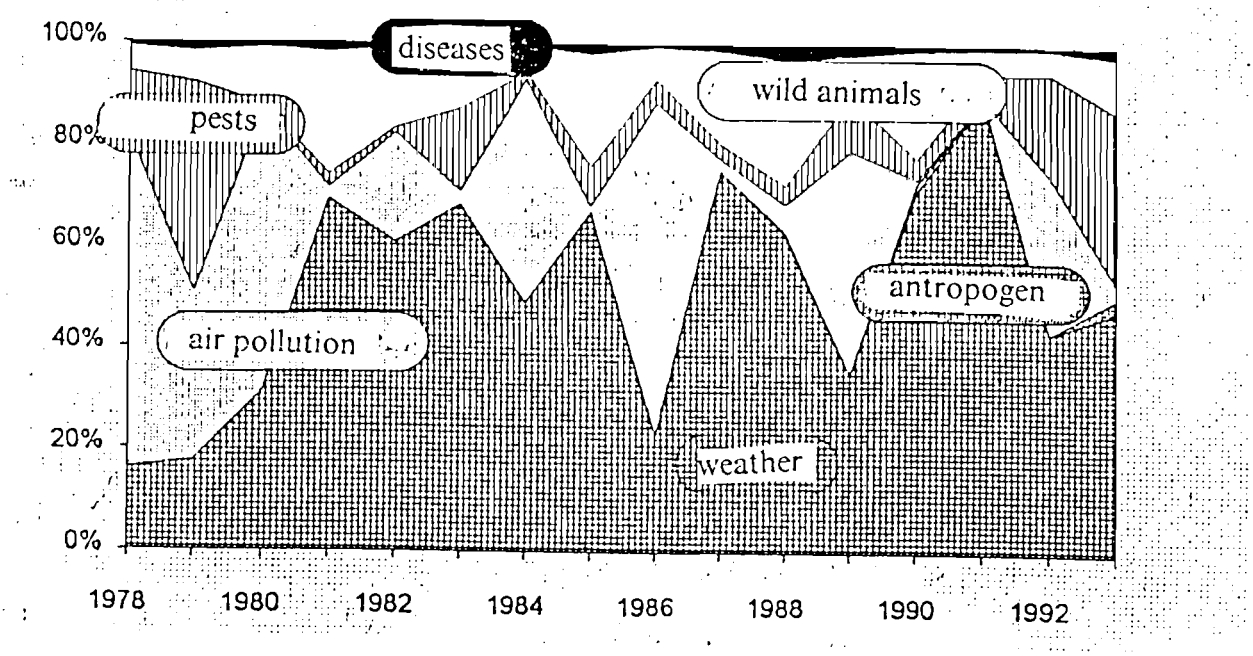


Figure 9.3. Anthropogenic Induced Mean Annual Forest Mortality (Mant MA), in ha.

Figure 9.4. Pollution Induced Mean Annual Forest Mortality (Mpol MA), in ha.

Figure 9.5. Intensity of Forest Mortality, in ha/yr/1000 ha forests.

Table 9.1. Forest mortality for 1992, 1993, average annual (MA);
Mant - anthropogenic induced mortality, ha/yr;
Mpol - pollution induced mortality, ha/yr
(Mant includes Mpol);
Fm - intensity of forest mortality ha/yr/1000 ha caused by all stress factors.

Ecoregion Code	Mant '92	Mant '93	Mant MA	Mpol '92	Mpol '93	Mpol MA	Fm '92	Fm '93	Fm MA
11011	3	0	2	0	0	2	.21	.25	.23
11012	0	0	1	0	0	1	1.36	0.00	.73
11013	0	0	20	0	0	1	.15	.01	.29
11014	0	0	1	0	0	1	1.36	0.00	.73
11014	3	0	2	0	0	2	.21	.25	.23
11041	61303	0	27981	61303	0	29781	1.12	.22	.37
11042	61303	0	27981	61303	0	27981	1.12	.22	.37
11043	61303	0	27981	61303	0	27981	1.12	.22	.37
11044	61303	0	27981	61303	0	27981	1.12	.22	.37
11045	61303	0	27981	61303	0	27981	1.22	.22	.37
11046	61303	0	27981	61303	0	27981	.12	.22	.37
11047	0	0	0	0	0	0	0.00	0.00	0.00
11048	61303	0	27981	61303	0	27981	.12	.22	.37
11051	0	0	70	0	0	70	.19	.12	.64
11052	0	0	70	0	0	70	.19	.12	.64
11053	0	0	70	0	0	70	.19	.12	.64
11081	0	0	0	0	0	0	.41	.10	.37
11082	0	0	0	0	0	0	.41	.31	.37
11083	0	0	0	0	0	0	.41	.31	.37
11084	0	0	0	0	0	0	.41	.31	.37
11085	0	0	0	0	0	0	0.00	0.00	0.00
11101	0	0	0	0	0	0	.37	.22	.05
11102	0	0	0	0	0	0	.37	.22	.05
11103	0	0	0	0	0	0	.37	.22	.05
11251	0	674	1736	0	674	1736	2.21	.20	.34
11252	0	674	1736	0	674	1736	2.21	1.20	.34
11253	0	674	1737	0	674	1737	2.21	1.20	.34
11254	0	674	1736	0	674	1736	2.21	1.20	.34
11255	0	674	1736	0	674	1736	2.21	1.20	.34
11255	0	674	1736	0	674	1736	2.22	1.20	.34
11301	0	0	0	0	0	0	.01	0.00	.02
11302	0	0	0	0	0	0	.01	0.00	.02
11303	0	0	0	0	0	0	.01	0.00	.02
11304	0	0	0	0	0	0	.01	0.00	.02
11305	0	0	0	0	0	0	.01	0.00	.02
11321	0	0	20	0	0	20	.15	.01	.29
11322	0	0	1	0	0	1	.19	.21	1.59
11323	0	0	20	0	0	20	.15	.01	.29
11323	0	0	20	0	0	1	.15	.01	.29
11441	0	0	0	0	0	0	0.00	0.00	0.00

Ecoregion Code	Mant '92	Mant '93	Mant MA	Mpol '92	Mpol '93	Mpol MA	Fm '92	Fm '93	Fm MA
11442	0	0	0	0	0	0	.48	.14	.27
11443	0	0	0	0	0	0	.48	.14	.27
11443	0	0	0	0	0	0	.48	.14	.47
11501	0	0	1	0	0	1	.19	.21	1.59
11502	0	0	1	0	0	1	.19	.21	1.59
11521	0	0	6	0	0	6	.35	.10	.36
11522	0	0	6	0	0	6	.35	.10	.36
11641	0	0	0	0	0	0	0.00	.29	.22
11642	0	0	0	0	0	0	0.00	.29	.02
11642	0	0	0	0	0	0	0.00	0.00	0.00
11691	0	0	3	0	0	3	.11	.53	.10
11692	0	0	3	0	0	3	.11	.30	.13
11711	0	0	16	0	0	16	.92	.03	.14
11712	0	0	16	0	0	16	.92	.03	.14
11713	0	0	16	0	0	16	.92	.03	.14
11714	0	0	16	0	0	16	.92	.03	.14
11715	0	0	6	0	0	6	.35	.10	.36
11761	0	0	0	0	0	0	.12	.19	.05
11762	0	0	0	0	0	0	.12	.19	.05
11811	0	0	15	0	0	1	.22	.20	.04
11812	0	0	15	0	0	0	.22	.20	.04
11931	0	0	0	0	0	0	.05	0.00	.17
11932	0	0	0	0	0	0	.05	0.00	.17
11981	0	0	0	0	0	0	.94	0.00	.06
11982	0	0	0	0	0	0	.94	0.00	.06
11983	0	0	0	0	0	0	.91	0.00	.06
11984	0	0	0	0	0	0	.91	0.00	.06
11985	0	0	0	0	0	0	.91	0.00	.06

Figure 9.6. The pollution dynamic: [SO_2] is the sulfur dioxide concentration in the Norilsk city air, milligram per cubic meter; SO_2 is sulfur dioxide annual output, million tons; Dust is the annual industrial dust output, thousand tons.

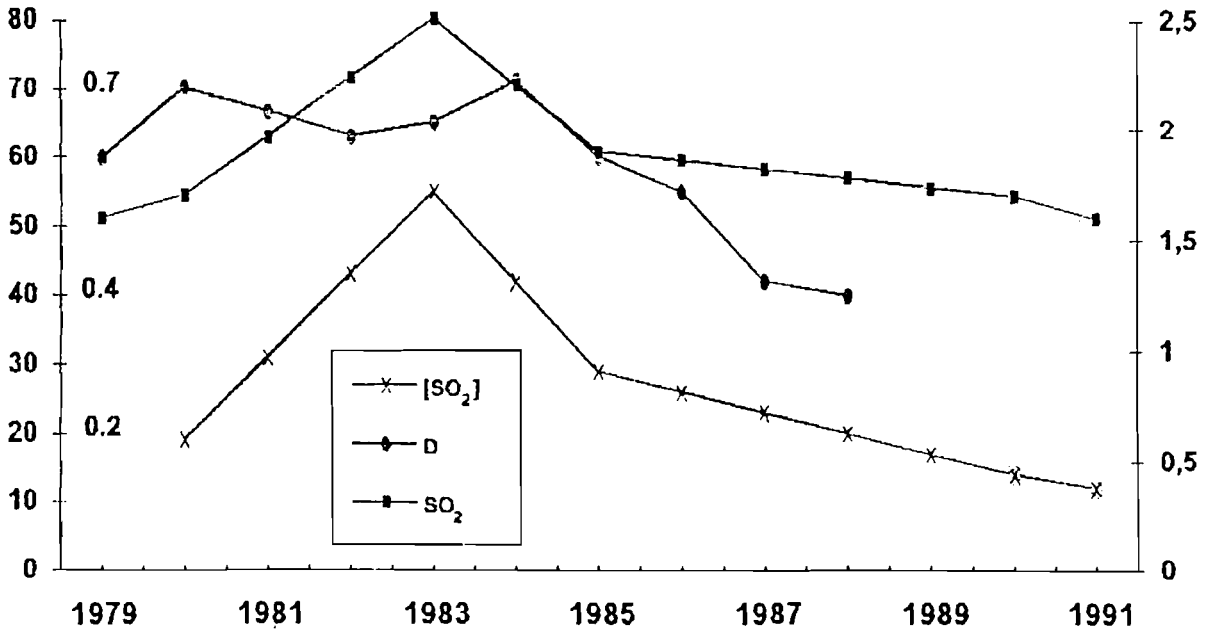


Figure 9.7. Content of pollutants in the soil along Norilsk valley, expressed in mg of pollutant per kilogram of soil.

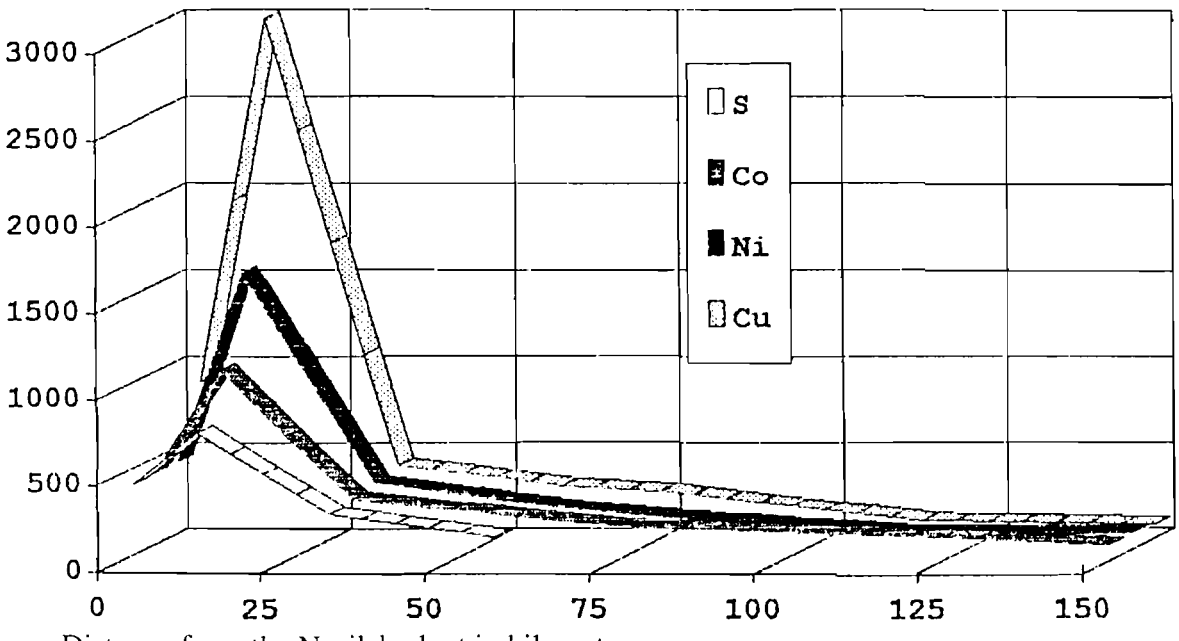


Figure 9.8 A. Temporal series of maps over damaged area based on satellite images.
A.-1979.

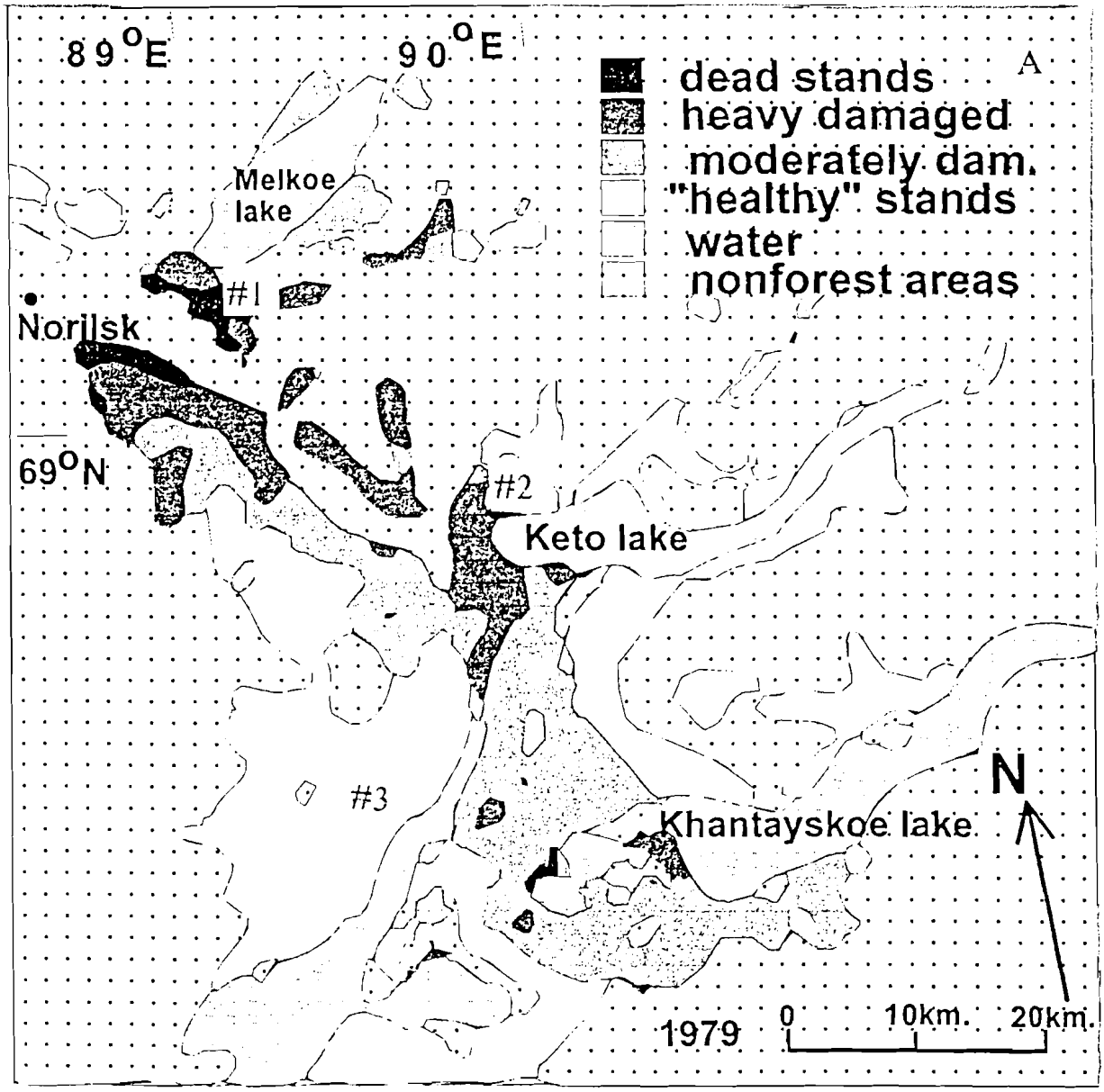


Figure 9.8 B. Temporal series of maps over damaged area based on satellite images.
B - 1982.

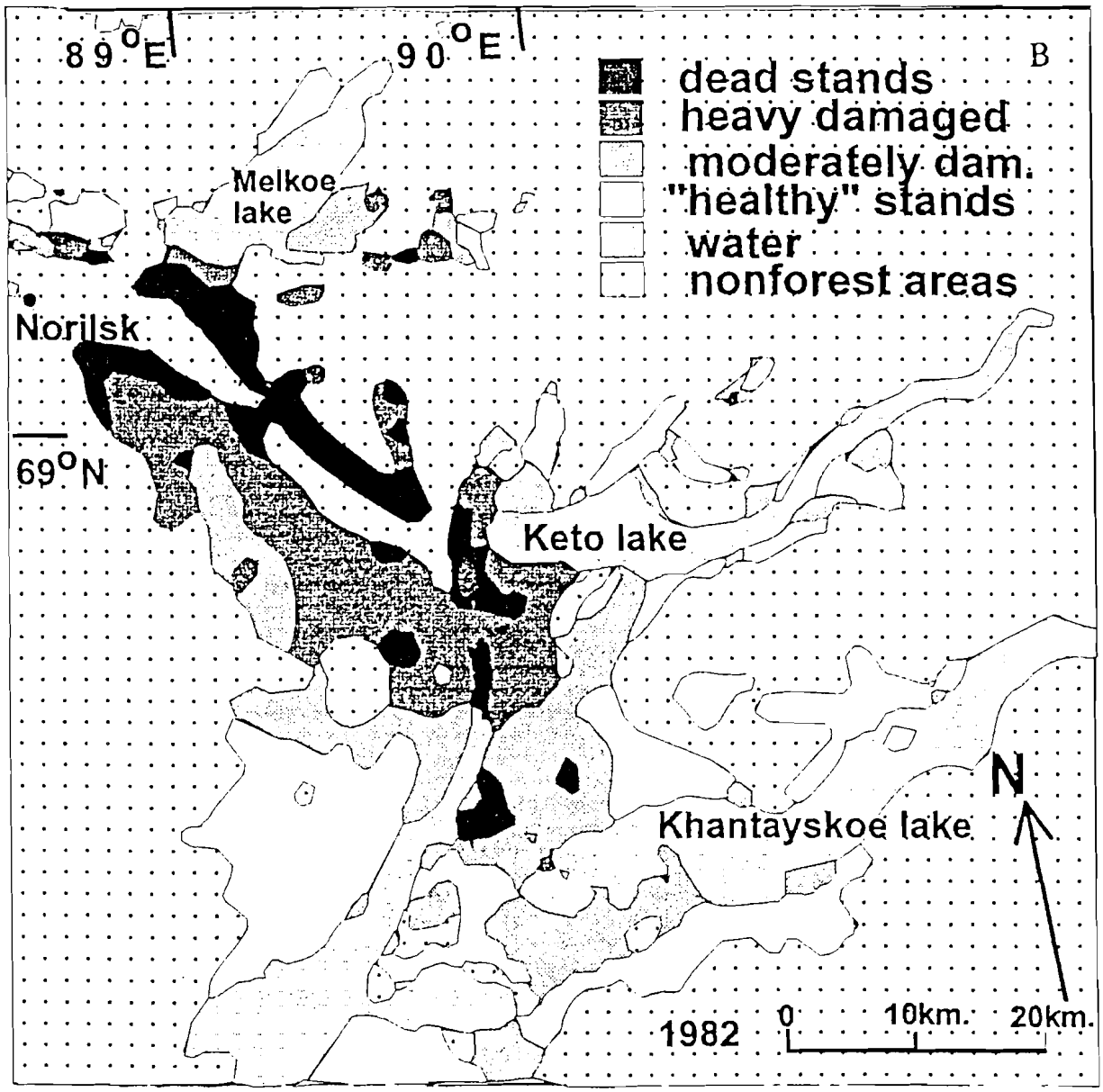


Figure 9.8 C. Temporal series of maps over damaged area based on satellite images.
C - 1984.

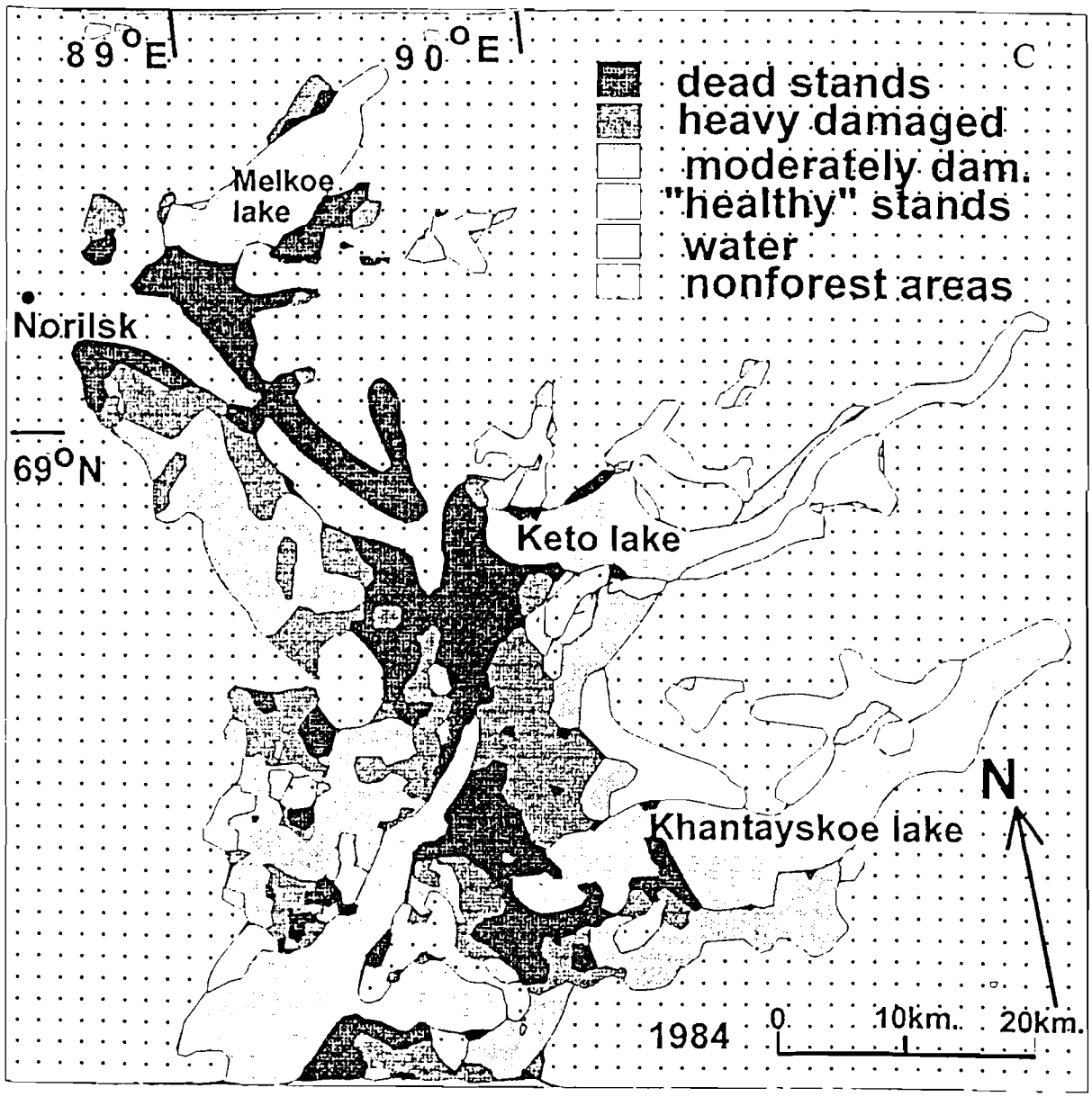


Figure 9.9. Increment indexes: - - - increment indexes of trees of I-II levels of vigor; ... of III-IV levels; —— I-IV levels; —— forecasted increment index. Study sites #1 and #2 (Figure 9.8A).

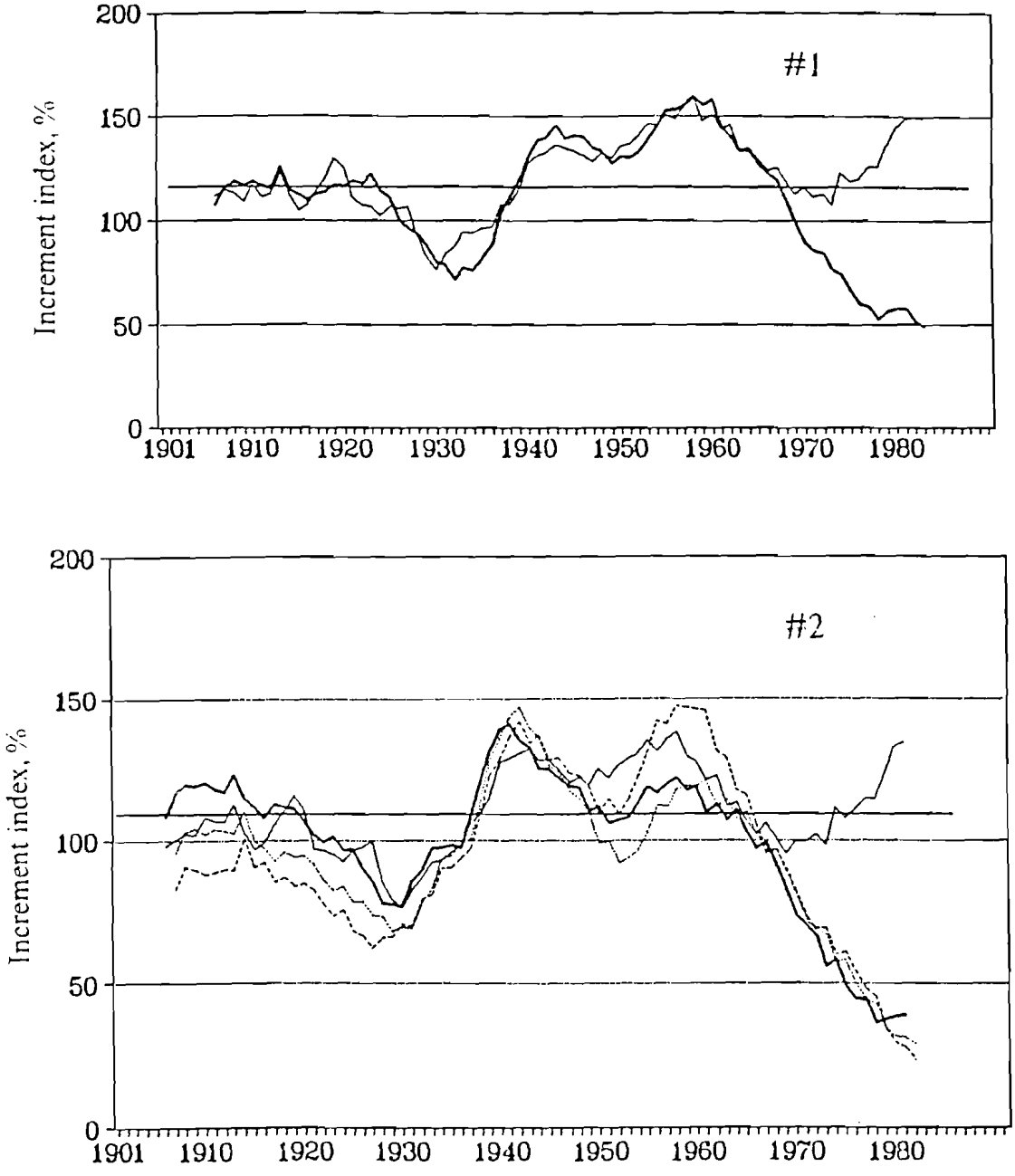
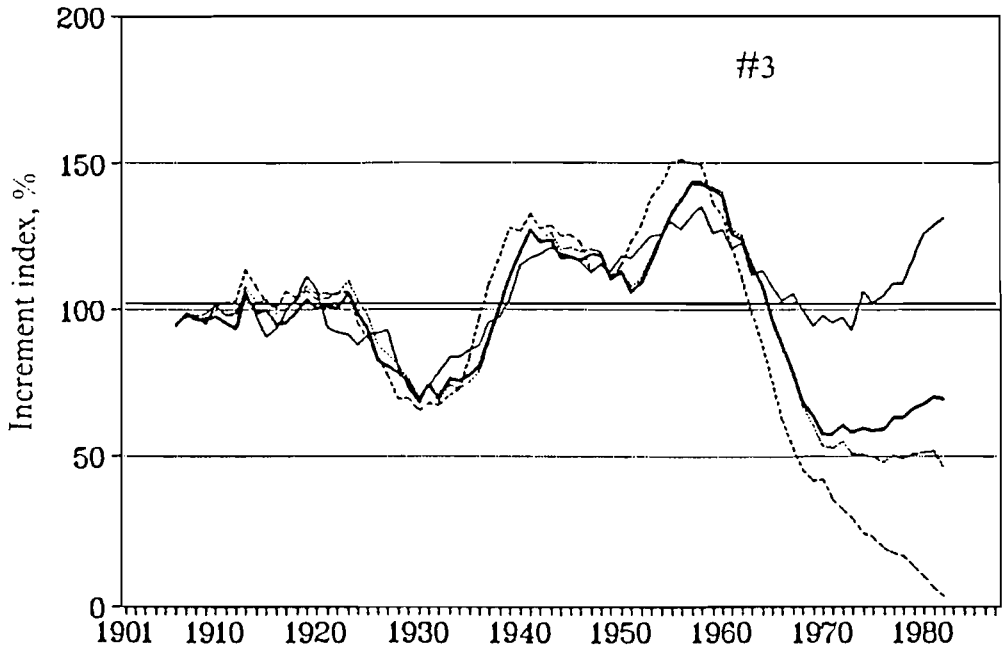


Figure 9.10. Increment indexes: - - - increment indexes of trees of I-II levels of vigor; ... of III-IV levels; ——— I-IV levels; —— forecasted increment index. Study site #3 (Figure 9.8A).



10. Conclusions

1. The Siberian territory, especially the Eastern part, is considered to possess a rather high ability of accumulation of pollutants (higher than in the European part of Russia). This is due to a higher buffering potential, but also due to a low river drainage in some territories, and low level of UV radiation.

2. During the last five years (1988-1993), the emissions of the major pollutants were considerably lowered due to a general industrial decline. Sulfur dioxide and nitrogen oxide emissions declined during this period by 42% in West Siberia, in East Siberia by 20%, in Siberian Far East by 37%.

3. In Siberia the average SO_4 depositions are of ~30-500 kg/km^2/yr and with maximum depositions of ~1000-3000 kg/km^2/yr in the south of West Siberia, Irkutsk and Norilsk regions. The average nitrogen depositions are of ~100-300 kg/km^2/yr and with maximum depositions of 500-1000 kg/km^2/yr. Generally, sulfur and nitrogen depositions in Siberia are less than in the European part of Russia.

4. The level of "wet" heavy metal (HM) depositions is below the critical levels except for the vicinities of big industrial centers (30-40 km), where the HM concentrations are ~2-3 orders higher than the critical levels. The "wet" deposition levels in undisturbed areas (biosphere reserves) are the same or less as in Japan, Canada and the USA.

5. The existing normatives or critical loads for pollutants in Siberia are empirical and require improvements based on field experiments with respect to ecological impacts.

6. There are no data in Siberia supporting any soil damage by precipitation acidity and reported pH-values of rain and snow, are within allowable ranges.

7. The major cause of forest decline in Siberia is not HM, S or N depositions, but gaseous and aerosol fractions of SO_2, fluorides and their derivatives.

8. The intensity of pollution-caused forest mortality *(Fpol)* in Siberia based on official reports varies between 0.01-5 ha/thousand ha forests and with maximum values in the zones of Norilsk and Irkutsk. In Norilsk the affected area is ~2 million ha and at a distance of 150-180 km along the prevailing wind direction the forests are dead. Currently, in Irkutsk region, pine stands are dead on an area of >100,000 ha, and on an area of >500,000 ha trees are heavily damaged. The forest decline in the Baikal region is synergetic by origin (pollution + biotic agents). The official reports indicate a forest decline due to air pollutants in total Russia of some 850,000 ha. But inofficial reports indicate that this may be an underestimate by 3-4 times only with respect to Siberia.

9. The potential sources of radioactive contamination in Siberia are the following: a) deposits and mining of radioactive materials, b) nuclear industry, c) nuclear waste storage, d) accidents in nuclear industries, f) underground nuclear explosions made during the 1970s-1980s, g) nuclear waste, e) natural material with high radioactivity used for industrial purposes, k) atmospheric depositions.

10. Currently, the nuclear industry can not be considered as a significant source of nuclear contamination in Siberia. But the nuclear waste storage is a high potential risk for future radioactive accumulation in the Siberian forest ecosystems. This high-risk condition requires further investigation.

11. There are no data, which support the idea that forest ecosystems in Siberia currently are under substantial risk of radioactive damage, and that there seems to be a minor amount of radioactive accumulation in the woody parts of the Siberian forests.

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