

Environmental radioactivity, population exposure and related health risks in the east Baltic region

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The paper considers radioactive contamination of the east Baltic region, population exposures, and the risk of damage to human health. Principal sources include global fallout, the Chernobyl accident, and marine transport of radionuclides. A mean annual exposure of 2–3 mSv comes from environmental radioactivity. Main contributors are primarily radon and its decay products. The Chernobyl accident brought an additional dose of about 0.5 mSv in southern Finland and 1.4 mSv in the most contaminated districts of the Leningrad region, Russia. Both external and internal exposure via contaminated food contributed. Currently, significant long-term radiological consequences of the Chernobyl accident include persistent radioactive contamination of natural terrestrial (forest) and freshwater (oligotrophic lakes) ecosystems and food products. Radiation health risks are lung cancer among the general population from indoor exposure to radon, acute radiation syndrome from occupational exposure, thyroid cancer among children in heavily contaminated non-Baltic areas, and mutations among offspring of exposed parents.

Key terms Chernobyl accident, environmental dosimetry, lung cancer, radiology, radon, thyroid cancer.

Geographically, the east Baltic region includes southern Finland, the Leningrad and Kaliningrad regions of Russia, Estonia, Latvia, and Lithuania. These areas are located on the shore of the eastern part of the Baltic sea, and its gulfs of Riga and Finland. However, radionuclides released in the environment by sources located in this region can be transported by sea water to all Baltic countries and — by even greater distances — with the air. The region and its immediate vicinity is the place of location for multiple power and research nuclear reactors, the production of radioactive substances, the storage of radioactive wastes, construction sites for ships with nuclear power units, and other nuclear facilities. Figure 1 shows the location of main nuclear objects in the Baltic region, which are described in table 1 (1).

The present paper considers principal sources of radioactive contamination of the east Baltic region and its population exposure; the levels of content of radiologically important natural and man-made radionuclides in the environment; the ways, regularities, and levels of human exposure from the environment; and the corresponding risk of damage to human health. By their origin, the sources of human exposure are divided into

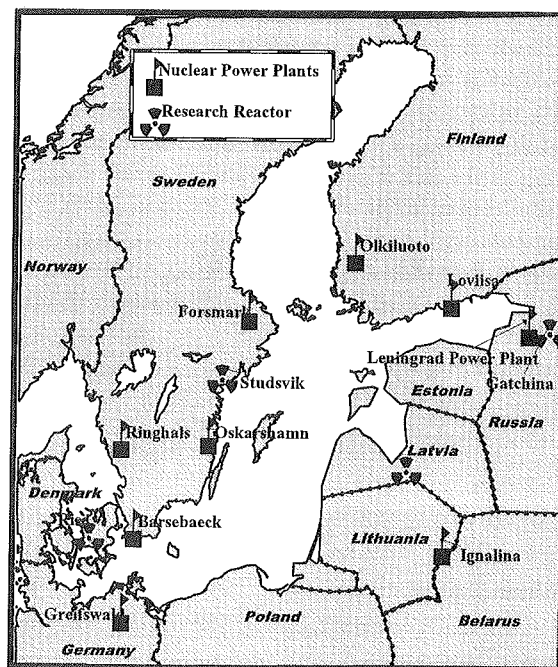


Figure 1. Location of nuclear power plants and research reactors in the Baltic region (1).

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Table 1. List of nuclear installations in the Baltic region (1). (NPP = nuclear power plant, PWR = pressurized water reactor, BWR = boiling water reactor, BPCR = big power channel reactor)

Name	Country	Object type	Number of units
Greifswald	Germany	NPP	2 PWR
Risö	Denmark	Research reactor	
Ringhals	Sweden	NPP	3 PWR & 1 BWR
Barsebäck	Sweden	NPP	2 BWR
Oskarshamn	Sweden	NPP	3 BWR
Forsmark	Sweden	NPP	3 BWR
Studsvik	Sweden	Research reactor	
Olkiluoto	Finland	NPP	2 BWR
Loviisa	Finland	NPP	2 PWR
Sosnovy Bor	Russia	NPP	4 BPCR
Gatchina	Russia	Research reactor	
Salaspils	Latvia	Research reactor	
Ignalina	Lithuania	NPP	2 BPCR
Paldiski	Estonia	Training center	

natural and artificial, as well as into global, distant and local, sources according to the distance to the considered region. Both external gamma exposure to the human body and internal exposure from radionuclides incorporated into the body with inhaled air, food, and drinking water have to be taken into account. In connection with a wide variety of exposure types and locations of its sources in the body after internal exposure, we have calculated the "effective dose" according to the International Commission on Radiological Protection (ICRP) (2) to get a dose quantity having a value that is supposed to be proportional to the risk of damage to human health under the action of low levels of radiation. Estimates of population exposure are based on the results of a prolonged study of the radiation situation in the region by the Finnish Centre for Radiation and Nuclear Safety (STUK) Helsinki, the Khlopin Radium Institute, the Institute of Radiation Hygiene in St Petersburg, Russia, and other institutions of Baltic countries.

Sources of population exposure

Natural sources

Natural sources of external human exposure include multicomponent space radiation and gamma radiation of potassium-40 (K-40) and radionuclides of uranium (Ra-226, etc) and the thorium series. Internal exposure is attributed to the intake of these radionuclides with food and water in the body. The inhalation of radon and the products of its decay in the air play a special role in population exposure.

Radon in buildings. Residential radon is the principal source of population exposure to ionizing radiation in

most countries. Two radon isotopes occur in significant amounts indoors due to the decay of radium isotopes in the environment — Rn-222 (radon) and Rn-220 (thoron). Radon is an inert gas; therefore, in contrast to other members of the radioactive families, it can migrate long distances from the place of its formation. Rn-222 — a member of a uranium radioactive family, with the decay half-life of 3.8 days — is of the highest importance. Rn-220 — a member of the thorium radioactive family — has a considerably shorter decay half-life, and, therefore, can migrate considerably shorter distances.

The ground is the major radon source. Radon accumulates in rock pores and is released to the earth surface with the air. Radon that enters buildings from rocks is mixed in a relatively small volume. This mixing causes higher radon volume activity in the air of buildings than in atmospheric air. The inflow of radon in the air of buildings depends on local inhomogeneities of the radon flux from the earth and on specific features of building constructions.

Less significant indoor radon sources include building materials and water when underground water sources with high radon content are used. However, some building materials made of soil or rock may have high concentrations of Ra-226. Water from drilled wells in regions with granite rock can have concentrations of radon of several thousand becquerels per liter (eg, in Sweden and Finland). The use of such water contributes both to indoor radiation and to exposure via ingestion (3).

Others. The population of the Baltic region spends the largest part of its time indoors (about 80%). Therefore, external gamma exposure is produced in the population mainly by radionuclides present in building materials used for construction. Alum shale for filling light concrete was widely used in Sweden for the construction of dwellings during the period from 1930 to 1975. The specific activity of Ra-226 in this concrete was up to 1300 Bq/kg. The gamma dose rate in buildings constructed with such concrete multiply exceeded the routine level and caused a considerable increase in exposure to the population of Sweden.

Intake of K-40 and radionuclides of the uranium and thorium series with food is considered background exposure and is not regulated. Intake of the radionuclides with drinking water varies widely depending on the type of water supply source. When underground sources of water supply are used, the intake of radionuclides with water can be considerable.

Man-made sources

Global sources. Globally distributed artificial sources of human exposure include radioactive products of predominantly thermonuclear weapon tests with TNT (trinitrobenzene) equivalent to shot energy over 100 kT.

Radioactive products of fission and activation lifted with the shot cloud to the stratosphere, flowed over the hemisphere mixed in high-altitude wind, and gradually fell out on its surface. The intensity of fallout from the stratosphere exponentially decreased with a half-period of about 1 year (4, 5). During the retention of the mixture in the stratosphere, the majority of radionuclides with a relatively short half-life decayed, and mostly long-life radionuclides reached the Earth's surface. Among them, Cs-137 (Cs = cesium) and Sr-90 (Sr = strontium) are the most radiologically significant for the present generation of people. The total release of these nuclides during the period of active nuclear tests performed in the atmosphere in 1952—1962 in the Union of the Soviet Socialist Republic, the United States, and Great Britain was 910 and 600 PBq, respectively (3). According to the assessment by HELCOM experts (1) based on materials of UNSCEAR (5), the total deposition of Cs-137 and Sr-90 on the surface of the Baltic Sea in 1980 was 1.1 PBq and 0.7 PBq, respectively. Separate shots performed by France and China in the 1970s did not contribute significantly. Depending on the power and conditions of the shots, a part of the radionuclides was deposited locally, near the test sites. This factor is not important for the Baltic region, where no nuclear tests were performed.

Distant sources. In the Baltic region, distant sources include West European plants for the regeneration of

nuclear fuel, whose liquid releases in the sea are transported gradually with flows to the Baltic Sea, and the Chernobyl accident.

A part of radioactive wastes from the plants in Sellafield (Great Britain) to the Irish Sea and in La-Hague (France) to the English Channel gradually penetrates to the Baltic Sea through the Denmark straits. The time of transport from the Irish Sea is 4—5 years, and from the English Channel it is about 2 years. Ingress of Cs-137 with flows to the Baltic Sea reached its maximum in 1980, 5 years after the maximum of the releases from the plant in Sellafield. Since then, the flow has considerably decreased in connection with decontamination of the plant wastes.

During the Chernobyl accident in the Ukraine, about 1900 PBq of a radionuclide mixture was released in the atmosphere to a height of over 1 kilometer during 10 days in April-May 1986. According to modern assessments (6), the total release of the most radiologically significant volatile radionuclides — I-131, Cs-137, and Cs-134 — to the atmosphere was 1500 PBq, 85 PBq, and 46 PBq, respectively. In addition, about 8 PBq of Sr-90 and 0.1 PBq of plutonium isotopes were released, which were deposited mainly in the vicinity of Chernobyl.

The map in figure 2 (7) shows that the products of the Chernobyl accident, especially Cs-137, were nonuniformly deposited in the east Baltic region. On 27—28 April, the cloud of the first release of the accident with

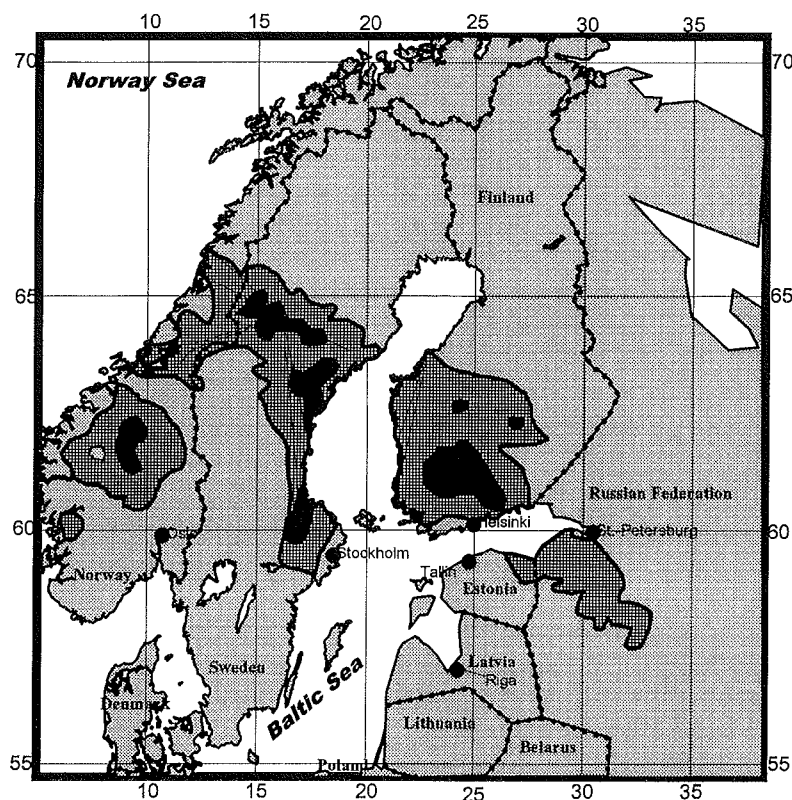


Figure 2. Cesium-137 soil contamination normalized to 10 May 1986 (7). (medium raster = 10—40 kBq/m², dark raster = 40—185 kBq/m²)

I-131 touched eastern Sweden, contaminated southern Finland, and dissipated above the Leningrad region to the southwest of St Petersburg. The deposition of Cs-137 on the surface of the Baltic sea proper, predominantly of the Gulf of Bothnia, was 4 to 5 PBq (1). This level is considerably higher than the level of global fallout in the 1960s and 1970s. It is interesting to note that the Baltic Sea, located at a distance over 800 kilometers from Chernobyl, was more strongly contaminated with the accident products than the Black Sea was, which is closer to Chernobyl and receives waters of the Dnieper with Pripyat, the river which runs close to the Chernobyl reactor.

Regional sources. The main objects that contain considerable amounts of radionuclides in the Baltic region are the nuclear power plants in Sweden, Finland, Russia, Lithuania, and Northern Germany, see figure 1 and table 1; also, research reactors in Denmark, Russia and Latvia; industrial, research and medical laboratories and the like. Radionuclides enter the environment with liquid wastes, as well as with gases and aerosols, released from these objects. Weakly toxic tritium dominated in the total activity of the releases, and the prolonged release of Cs-137 is less than the activity from the global fallouts, especially from Chernobyl. The releases increased by approximately 20% in the last 5—7 years.

A large storage site for radioactive wastes with average and low activity is located on the seashore, on the territory of the Swedish nuclear power plant Forsmark. Another storage site is located in the town of Sosnovy Bor on the shore of the Gulf of Finland about 130 kilometers from St Petersburg. Shipyards of the nuclear Russian Navy are located in St Petersburg and in its suburbs, the result being several areas of historic radioactive contamination since the 1950s. Releases of radionuclides from these objects to the environment are weakly documented but seem to be insignificant.

On the shore of the Narva Bay of the Gulf of Finland, there is a storage of wastes from the metallurgical plant in Sillamäe (Estonia), which has processed uranium- and thorium-containing ores since 1948. Up to 200 000 cubic meters of liquid wastes and bottom sediments are in a pond separated from the sea by a dam 25 meters high. The amount in the storage site is estimated at 1600 tons of uranium (U) and 850 tons of thorium. The annual release of radionuclides with soil water is estimated at 6 GBq each for U-238 and U-234, and 47 MBq each for Ra-226 and lead-210 (8).

In March 1992, an industrial accident at the third unit of the Leningrad nuclear power plant resulted in an insignificant release of radionuclides in the atmosphere; it was detected in southern Finland, however. Among the detected radionuclides were not only volatile isotopes of iodine (I) and cesium, but also refractory elements (cerium, neptunium, etc).

Current levels of radionuclides in the environment

Air concentration

Radon in buildings. Systematic investigations of the radon problem were begun only in recent years. Earlier, human exposure from radon and other natural sources was included in the notion of the natural background and not studied seriously. The radon content in the air of buildings varies considerably with time, depending on weather conditions, temperature difference between the room and atmospheric air, ventilation, and the like. The average annual radon content in the air of buildings is of importance for exposure. Systematic investigations of the radon problem became possible only after the development of integral measurements with the use of track detectors.

Investigations of the radon content in dwellings have been performed in countries of the European Community. The highest radon content in dwellings was found in Finland. The average value of the radon volume activity was 145 Bq/m³ in houses and 82 Bq/m³ in flats (9), which is 2—3 times higher than the world average value. A still higher radon content was found in the most populated southern regions of Finland adjacent to the Gulf of Finland.

No systematic investigation of the radon problem has been performed in the countries of the former Soviet Union. However, preliminary assessments have indicated that radon-dangerous areas include the northeast of Estonia, the south shore of the Gulf of Finland, and the Karelian isthmus of the Leningrad region, as well as the southwest part of St Petersburg. Recently, a systematic study of the radon concentration in Estonian dwellings was started in collaboration with Swedish experts (10). Winter measurements with track detectors were performed in 500 randomly selected buildings. In 3 districts of Tallin the radon air concentration was found to be 23 to 47 Bq/m³, which is close to the world average. In a 4th district (Pirita-Kosa), where 1-family houses dominate, the mean concentration was higher (157 Bq/m³). In 5 other Estonian towns the radon level in dwellings was relatively high in winter, the mean concentrations ranging from 86 to 151 Bq/m³ with the maximum above 2000 Bq/m³ (10).

Man-made radionuclides. Man-made radionuclides appeared in the ambient air with the beginning of operation of the first nuclear reactors, but their concentration increased drastically during the period of nuclear tests in the atmosphere. The content of radionuclides in the air of a region distant from test sites consists of tropospheric and stratospheric components. The first component is attributed to radionuclides retained in the troposphere after shots, radiation accidents, or routine releases, and they are deposited on the surface of the Earth with

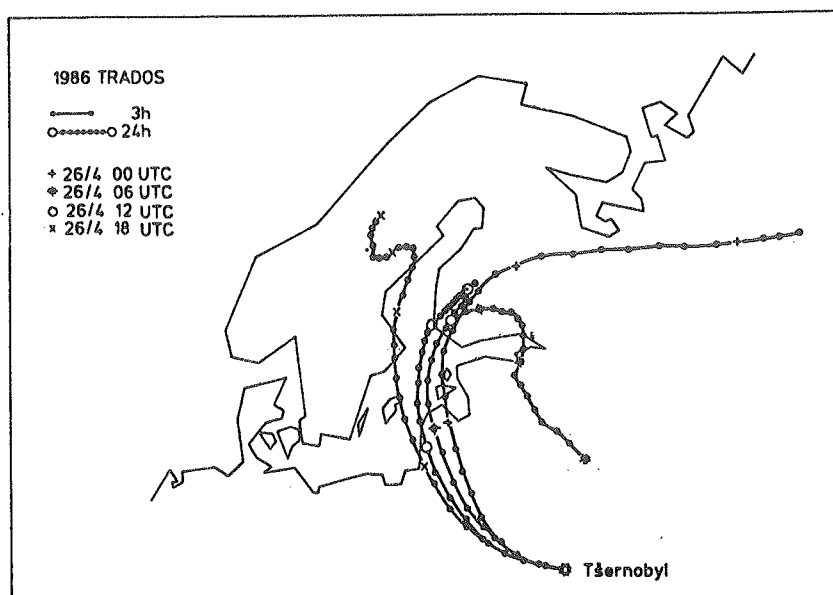


Figure 3. TRADOS trajectories from Chernobyl 26 April 1986 (13).

precipitation or by dry deposition with the typical half-period of 10 days to 1 month. On the other hand, the near-surface concentration of radionuclides thrown to the stratosphere after high-power thermonuclear shots exponentially decreased with a half-period of about 1 year with annual spring maximums. According to data, since 1959, from the Radium Institute in St Petersburg, the monthly average Cs-137 concentration in the near-surface air of this city reached its maximum of 1–2 mBq/m³ in 1962–1963, and it decreased in the next 20 years by 3 orders of magnitude and was about 1 µBq/m³ in 1983–1985 (11).

The first Chernobyl cloud appeared above southwest Finland at a height of 0.7–1.5 kilometers on 27 April 1986 (12). However, contamination of the east Baltic region started on the next day with the arrival of radionuclides in the near-ground air layer (figure 3) (13). The same cloud came to the territory of the Leningrad region on 29 April. In St Petersburg, the average Cs-137 concentration increased up to 10 mBq/m³ in May 1986 and then rapidly decreased with time. However, even in 1995 it exceeded the preaccidental level several-fold. This increased level is attributed mainly to wind resuspension of local and, to a less extent, distant Chernobyl depositions. According to data from STUK (13, 14), the monthly average Cs-137 concentration in the air before the Chernobyl accident was below 1 µBq/m³, and it reached 100 mBq/m³ in May 1986. Its further decrease is similar to that in St Petersburg; see figure 4 (14). I-131, Cs-134, ruthenium-103, ruthenium-106, and nonvolatile radioisotopes of cerium, zirconium and other products of nuclear fission were also detected in the air after the Chernobyl accident.

Information about the detection of “hot particles” in the air and on the soil surface aroused considerable

concern in populations of contaminated areas of Europe. Two main types of hot particles were found in the atmosphere after the Chernobyl accident, particles with the total composition of exposed fuel and particles with a predominance of ruthenium radioisotopes (Ru-103 and Ru-106). The particles of the second type became concentrated after the evaporation of ruthenium compounds in specific high-temperature conditions. International investigations of radiological consequences of the Chernobyl “hot particles” deposition did not indicate a particular hazard for humans (15).

Fresh products of nuclear fission, including radioisotopes of the inert gases krypton and xenon, and of iodine and other elements, were regularly detected by monitoring stations in Scandinavia and the Soviet Union after nuclear shots in the atmosphere, after releases from shafts of underground nuclear tests, in particular, in the northern test site of the Soviet Union on the Novaya Zemlya island up to 1990, and after accidents of ships of the nuclear Navy and radiation accidents at nuclear power

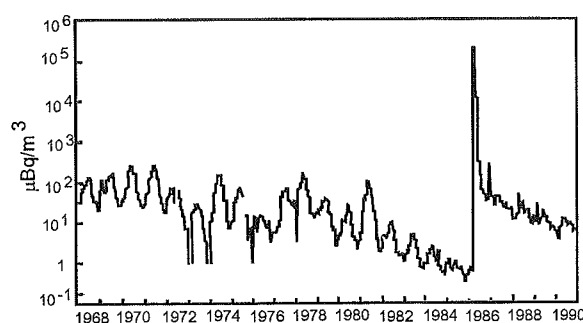


Figure 4. Monthly mean concentrations of cesium-137 (mBq/m³) in ground-level air in Nurmijärvi, Finland, 1968–1990 (14).

plants. However, dosimetric calculations show that exposure of the European population, in particular, of the east Baltic region, from tropospheric fallout of products of nuclear shots has been negligibly small. With their fallout from the stratosphere and during nuclear reactor accidents, I-131 and long-lived radionuclides play the main role after their deposition on the surface, but not due to their inhalation.

Terrestrial environment

Cs-137 and Sr-90 in soil. At the end of April and beginning of May 1986, the east Baltic region was nonuniformly contaminated by rains from the Chernobyl radioactive clouds. Both at STUK and at laboratories in St Petersburg, the entire spectrum of fission products and irradiated reactor fuel were detected in soil samples. The ratio of radiologically important radionuclides I-131, Cs-137, Cs-134, and Sr-90 in fallouts of southern Finland and of the Leningrad region was 10–20, 1, 0.5, and 0.01–0.02, respectively, at the end of April. Figure 2 shows the mapping of the radioactive contamination carried out in subsequent years with the long-lived Cs-137 (7). The typical range of soil contamination in southern Finland is 10–40 kBq/m², but in some areas it reaches 40–80 kBq/m² (12). In the Leningrad region, the most contaminated were the Volosovo and Kingisepp districts. The level of their contamination is close to that of southern Finland. According to data by the Russian Committee for Hydrometeorology, in 1986, the area of about 70 settlements in these districts was subjected to contamination above 37 kBq/m² (1 Ci/km²), including 5 villages with contamination above 80 kBq/m². These levels are 3–30 times higher than the highest contamination of soil from global fallout.

Along with Cs-137, another radioisotope of the same element, Cs-134, with a half-life of 2.1 years, was present on soil after the Chernobyl accident. The content of Sr-90 in soil increased insignificantly.

Exposure dose rate. The increase of air dose rate in the region attributed to fallout of man-made radionuclides is also connected with the Chernobyl accident. At stations of constant monitoring in southern Finland, the dose rate increased on the day of the fallout from the background level of about 0.1 µSv/h up to 1–4 µSv/h. This value decreased by an order of magnitude during the first month because of decay of short-lived radionuclides and was stabilized by the end of summer 1986 (16). In the Leningrad region, according to the data by Golikov (17, 18), on 29 April 1986, the dose rate in the area contaminated with Cs-137, at 37 kBq/m², was over 2 µSv/h, which is close to the Finnish data. Beginning in the autumn of 1986, the dose rate above soil has been determined by the presence of the "Chernobyl mixture" of Cs-137 and Cs-134, and it has relatively slowly decreased with time

because of the decay and deepening in soil. In settlements, the dose rate is lower than in open locations because of the absorption of gamma radiation by buildings and the accelerated removal of radionuclides from areas of human activity. Currently, the air dose rate from Chernobyl Cs-137 is on an order of magnitude lower than its value in the autumn of 1986.

Agricultural food products. Both in the period of global fallout and after the Chernobyl accident, the surface of agricultural plants was subjected to direct contamination by radionuclides from the air. In the first case, the most significant nuclides were Cs-137 and Sr-90, and in the second it was I-131, Cs-137 and Cs-134. The initial concentration of radionuclides in plants was determined not only by the level of contamination of the area, but by the season and the intensity of precipitation during the fallout. During several weeks after the contamination, the concentration of radionuclides in plants decreased because of natural processes of the surface decontamination and decay with the typical half-lives of 5 days for I-131 and 10–15 days for the rest of the indicated nuclides. In Finland in the 1960s, the country-average concentration of Sr-90 in milk, which gives the greatest contribution of radionuclide intake with food, reached 1–1.5 Bq/l, and that of Cs-137 was 10–15 Bq/l. By the beginning of 1986, these values had decreased to 0.1–0.2 Bq/l, and in June of the same year had increased up to 25 Bq/l (19). In Estonia, Latvia, and Lithuania, the average concentration of Cs-137 and Sr-90 in milk in the mid-1960s was about 3 and 1 Bq/l, respectively (4).

Due to the cool Baltic climate, dairy cattle were still in stalls at the end of April in 1986 and were taken out to pasture 1–2 weeks after the fallout in the Leningrad region and 1 month after the fallout in Finland. In the latter case, this was a specially recommended countermeasure which appeared to be efficient. Due to these circumstances, the intake of radioiodine and thyroid exposure by inhabitants was considerably decreased as compared with that in temperate latitudes. In contaminated areas of the Leningrad region, the I-131 concentration in milk reached only 10 Bq/l in the middle of May 1986 (figure 5) (18), and in Finland it averaged 14 Bq/l (20).

After contamination of the agricultural ecosystem, the concentration of long-lived radionuclides in vegetable and animal products is determined by transfer from soil through root systems and strongly depends on soil composition. According to our data, the transfer factors decrease with the half-life of 1–2 years during the first 5 years (21). Later, this process gradually decelerates and continues with a half-life of 10–20 years. The first process was weakly noticeable during the period of global fallout because of its prolonged character, but it was explicitly displayed after the Chernobyl accident. Currently, the transfer factors for "Chernobyl" Cs-137 and

Sr-90 from soil to plants and further along the food chain approached the levels typical for the beginning of the 1980s, and concentrations of Cs-137 and Sr-90 in milk in contaminated areas of Finland and the Leningrad region during the pasturing period are mainly within the limits of 0.1—10 Bq/l and 0.03—0.3 Bq/l, respectively. The content of nuclides in potatoes of local production is lower by 2—20 times (17, 18, 22).

Natural food products. Cesium radionuclides are retained for a long time by layers of virgin soil rich with organic substances (eg, meadow turf and forest litter). More considerable and prolonged contamination is observed for vegetable and animal natural food products (mushrooms, forest berries and game). In particular, the Cs-137 concentration in some species of mushrooms decreased in 10 years after the Chernobyl accident only by the value corresponding to its radioactive decay (23—25). Both the level of radionuclide accumulation and its dynamics depend on species and on soil and climatic conditions. This phenomenon has been studied in most detail after the Chernobyl accident because the environmental contamination levels with Cs-137 in many European countries are much higher than after global fallout, and in several countries, including those in the east Baltic region, their inhabitants traditionally consume natural food.

The Cs-137 content in mushrooms that grow and are gathered in spots of Chernobyl contamination in Finland and the Leningrad region still range from 10 to 1000 Bq/kg after 10 years (22, 26). The concentration of Cs-137 in moose in contaminated regions of Sweden did not show a noticeable decrease during 1986—1991 (25). Comparison with the content of Cs-137 in agricultural products shows a difference of 2—3 orders of magnitude.

Freshwater environment

Freshwater concentrations. During wide-scale contamination of areas with global or Chernobyl fallouts, radionuclides entered the surface water reservoirs both by direct deposition from the atmosphere, and as a result of gradual wash-off by precipitation from the water-collecting basin. The Cs-137 concentration in river waters of the Leningrad region and southern Finland in the 1960s was in the range of 10—100 Bq/m³, it decreased by an order of magnitude during the next 20 years due to transport to the sea and deposition on the river bottom, and by the beginning of 1986 it was 2—4 Bq/m³ (27, 28). With depositions from the Chernobyl cloud in May 1986, this value in the southern Finland rivers of Kymijoki and Kokemäenjoki reached 20 000 to 50 000 Bq/m³ (ie, increased, on the average, by 1000 times) (28). However, already by August 1986, the Cs-137 concentration in the water of these rivers had considerably decreased, to 100—1000 Bq/m³, and it further smoothly decreased with time with a half-life of 1—2 years (29).

According to data from the Radium Institute, for the mouths of the Narva and Luga rivers that flow into the Gulf of Finland from the south, the Cs-137 concentration in water at the end of May 1986 was 250—280 Bq/m³, and in 1—1.5 years it was 12—51 Bq/m³. In the end of July and beginning of August 1986 at the mouths of the Baltic rivers Venta, Neman, Daugava, and such, the same specialists measured 10—76 Bq/m³. These findings indicate insignificant contamination of corresponding water-collecting basins. Lake Ladoga and its surroundings were not significantly contaminated. Therefore, the Cs-137 concentration in the Neva river had increased only up to 22 Bq/m³ in December 1986 (27).

The deposition of Sr-90 was not as significant as the deposition of cesium radionuclides, and its preaccidental average concentration in rivers of the region, close to 10 Bq/m³, increased in 1986 only by 2—3 times and further gradually decreased (29).

Freshwater fish. The accumulation of radioactive cesium and strontium in tissues of freshwater fish has a prolonged character and depends not only on the concentrations in water, but on concentrations of stable isotopic and nonisotopic carriers, primarily potassium and calcium, and on the species and age of fish and other water-life. Thus, in Karelian lakes (Onega, Rugozero, Vygozero) with low mineralization, the Cs-137 content in mixed fish samples by the end of the 1960s was at the level of 60 Bq/kg, and in the Chudskoye and Ilmen lakes located to the south, with a higher concentration of potassium in the water, the concentration was 10—15 Bq/kg in the same period (4). In 1982 in Finland, the Cs-137-specific activity in different species of lake fish varied from 2 to 64 Bq/kg (30).

This parameter grew in May-June of 1986, first in vegetable-eating and benthos-eating fish and later in

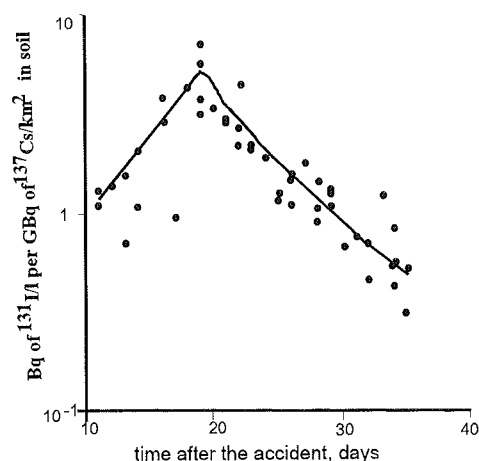


Figure 5. Concentration of iodine-131 (I-131) in milk produced in the Kingisepp district of the Leningrad region standardized by cesium-137 soil-contamination density (18).

predatory ones. In the first group, the maximum concentration, 1500 Bq/kg on the average over Finland, was reached in 1986, and in the group of predators it was at about 4000 Bq/kg in 1988 (29). The Cs-137 concentration in fish strongly varied depending on the previously mentioned factors, reaching the highest levels in fish inhabiting oligotrophic lakes. In 1987, 1% of the analyzed fish had a Cs-137 concentration of over 10 000 Bq/kg, and for 0.4% the concentration was over 20 000 Bq/kg. Furthermore, the decontamination of the freshwater ecosystem from radionuclides of the Chernobyl origin resulted in a decrease of the Cs-137 content in fish, with an average half-life of about 2 years (29).

Marine environment

Marine water. In the period before the Chernobyl accident, the concentration of long-lived radionuclides in waters of the Baltic Sea was formed by global fallout and the transfer of releases from western European reprocessing plants. In 1983, the Cs-137 concentration in the surface waters of the central part of the sea was 10–15 Bq/m³, and in the Gulf of Finland it was below 5 Bq/m³. After the Chernobyl accident, the highest depositions were on the surface of the gulfs of Bothnia and Finland. In the summer of 1986, the Cs-137 concentration in the surface waters was 50–100 Bq/m³ in the gulf of Riga, 40–580 Bq/m³ in the Gulf of Bothnia, and 110–1100 Bq/m³ in the Gulf of Finland. From the surface, the radioactive depositions penetrated the depth of water, mixed with the water and partially settled on the bottom. In the summer of 1989, the Cs-137 concentration in the surface waters of the Gulf of Finland was 110–130 Bq/m³, and the Sr-90 concentration was 15–22 Bq/m³, and it gradually decreased with time (1, 31, 32).

Seafood. Due to the high concentration of cations in sea water, the radionuclide concentration in tissues of fish and other species of the biota is lower than in fresh water. In 1984–1985, the specific activity of soft tissues of cod, herring and ling, and of mussels from different parts of the Baltic Sea, did not exceed, as a rule, 5 Bq/kg for Cs-137. Beginning in 1986, this parameter rapidly increased, especially in catches from the gulfs of

Bothnia and Finland: up to 120–270 Bq/kg in cod, 30–270 Bq/kg in sprat, and 60–280 Bq/kg in ling (1). Due to the vertical and horizontal mixing of sea water, the content of radionuclides in fish inhabiting different parts of the sea gradually becomes equal. Thus, in 1989–1990, the Cs-137 content in Baltic herring was in the range of 17–32 Bq/kg (32).

Population exposure

Considering human exposure in low doses, not causing deterministic effects, but increasing the risk of stochastic effects — carcinogenic and genetic — the commonly assumed measure of radiation impact is the effective dose measured in sieverts (Sv, mSv). However, the absorbed dose (Gy, mGy) has been indicated in the discussion of the carcinogenic effects of local exposure of human thyroid from the incorporation of I-131. In an assessment of the levels of population exposure in the east Baltic region, it should be pointed out that the ICRP assumes an effective dose of 1 mSv from controlled artificial sources, and in separate years up to 5 mSv can be used as permissible for the annual exposure of humans (2).

External exposure

Natural radiation. The dose rate of space radiation at the surface of the ground mainly depends on the height above sea level, and it is assumed to be equal in all organs of the body. According to reference 3, the average annual dose of cosmic radiation in inhabitants at altitudes up to 200 m above sea level — including the east Baltic region — is about 0.30 mSv.

The dose rate of the external gamma radiation of K-40 and of radionuclides of thorium and uranium series is attributed to their content in soil, bedrock, and building materials. Inhabitants of developed countries spend about 80% of their time indoors and 20% outdoors. The average annual external dose from natural radionuclides in inhabitants of the Soviet Union is 0.32 mSv, in Finland it is 0.49 mSv, and in Sweden it is 0.65 mSv (3). The differences between these assessments is attributed mainly to the different content of the radionuclides in building materials. In particular, one reason for a higher dose in the Swedish population is the wide-scale application of alum shale for building. According to the presented data, the annual dose for inhabitants of the east Baltic region can be estimated to be about 0.40 mSv.

Man-made radionuclides. Among gamma-emitting radionuclides deposited on the surface of the Earth from the stratosphere after nuclear weapons tests, Cs-137 has a leading role, and it is responsible for the formation of over 50% of the long-term effective dose of the

Table 2. Average external dose of inhabitants of the East Baltic region due to the Chernobyl accident standardized by soil contamination density with cesium-137 in 1986, $\mu\text{Sv}/(\text{kBq}/\text{m}^2)$.

Country	1 year after the accident	1986— 1995	1996— 2056	1986— 2056
Russia (Leningrad region)				
Rural population	15	40	31	71
Urban population	11	24	16	40
Finland				
Total population	10	-	-	-

population. The average committed effective dose of external exposure for the population of the east Baltic region attributed to this nuclide is estimated at about 0.37 mSv, and for other gamma-emitters (Zr-95, Nb-95, Ru-106, etc) it is about 0.35 mSv. Thus the total dose of external exposure from all deposited radionuclides equals 0.72 mSv (5).

The external dose in the region after the Chernobyl accident was studied by specialists of STUK in Helsinki (13, 33) and the Institute of Radiation Hygiene (IRH) in St Petersburg (17, 18, 34). In connection with the nonuniformity of area contamination, the dose assessments are given for the unit density (1 kBq/m²) of soil contamination with Cs-137, see table 2. At the same time, it should be noted that short-lived radionuclides (in particular, I-131 and I-132) added a significant contribution to the external dose during the first weeks. The contribution of ruthenium-103 and ruthenium-106 was considerable during several months. However, Cs-137 and Cs-134 played the leading role beginning in the summer of 1986.

The doses during the first year in the population of Russia and Finland agree well. According to data from the IRH, inhabitants received about 24% of their life-time dose during the year after the accident, about 60% during 10 years, and 40% more are expected during the subsequent decades. Along with normalized dose assessments, actual doses received by inhabitants of different areas are of interest. Thus, during the 10 years after the accident, inhabitants of the Kingisepp and Volosovo districts of the Leningrad region, where the soil surface contamination exceeds 37 kBq/m², received from 1 to 3 mSv including 0.6–1.4 mSv during the first year. In the future, 1–2.5 mSv more is expected (17, 18). In southern Finland, inhabitants of the most contaminated areas received 0.3–0.4 mSv during the first year after the accident (13).

Internal exposure

Radon inhalation. Among natural sources, radon in the air inside buildings is the most important. Its contribution to the world average dose is about 50% (3). Since the content of radon varies widely in the air inside buildings, this radionuclide determines the levels of population exposure in different regions. In separate buildings, the exposure levels can exceed the average regional exposure by several orders of magnitude. Therefore, radon is the main natural factor determining the radiation situation in a country, region, settlement, and a separate building.

The average effective dose of radon exposure to the population of the former Soviet Union is estimated to be 1.3 mSv (35), which is equal to the world average dose (3). This value in Finland is 1.5 times higher, about 2 mSv/year, varying from 0.2 to 350 mSv (9). Owing to this fact, the total dose from all natural sources to the

population of Finland somewhat exceeds the world average (3).

Other natural radionuclides. The internal human dose is attributed to K-40 and radionuclides of uranium and thorium series intake with water and food. The potassium content in body tissues is controlled by homeostatic mechanisms; therefore, the annual dose from K-40 (0.17 mSv) is practically uncontrollable. On the other hand, the dose from radionuclides of uranium and thorium series, with an important role for lead-210 and polonium-210, noticeably varies with their content in diet. The world average annual dose from the intake of these radionuclides is estimated to be 0.06 mSv (3). Southern Finland, with its increased amount of U-238 and the like, in underground waters is an example of increased intake. There, the annual dose for inhabitants reaches 0.55 mSv (3).

Whole-body content of cesium-137. The intake of Cs-137 from the environment, predominantly with food, began during the period of nuclear tests in the end of the 1950s and reached its maximum in the middle of the 1960s. The average Cs-137 content in the body of adult inhabitants of the east Baltic region was at 1000 Bq in 1964–1965, and it later decreased by an order of magnitude during 10 years, and by 2–5 times more by 1985. Beginning in May 1986, this value increased until the summer of 1987, when equilibrium was reached between intake and excretion from the body, and later decreased with a half-time of 1–2 years. This occurrence is well illustrated by data from STUK, in which the subjects included a group of inhabitants of Helsinki and other regions of the country; see figure 6 (36).

The Cs-137 content in the inhabitants of St Petersburg and the Leningrad region followed the same pattern. According to data from the IRH, the average content in men in St Petersburg in 1965 was about 900 Bq,

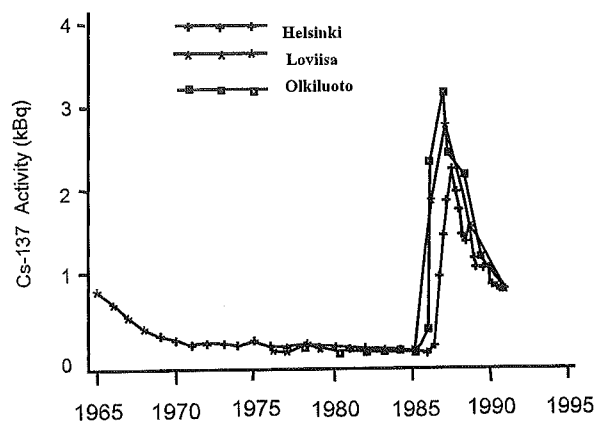


Figure 6. Body burdens of cesium-137 (Cs-137) in the Helsinki, Loviisa, and Olkiluoto reference groups from 1965 to 1990 (36).

and in women it was 550 Bq (37). In 1987 it was 1500 Bq, and it had decreased to 700 Bq by 1990. In contaminated areas of the Leningrad region, the Cs-137 content in inhabitants after the Chernobyl accident was measured in the autumn of 1988 and 1993 (22, 38). In 1988, the average Cs-137 content in adult inhabitants of the village of Taraika, located in a forest of the Kingisepp district with soil surface contamination of 50 kBq/m², was 3200 Bq. In 1993, this value in villages with soil surface contamination of 20–50 kBq/m² decreased to 300–1000 Bq.

Dose from global cesium-137 and strontium-90 and other nuclides. The life-time internal dose to inhabitants of the east Baltic region from global fallout of products of nuclear tests performed in the atmosphere before 1980 was assessed by 2 independent methods: from the intake of radionuclides with food and from the Cs-137 content measured by whole-body counters before 1986. A further decrease in both intake and content with the effective half-time of 10 years was assumed for the calculation of the dose expected after 1986 (5). The average Cs-137 exposure dose was assessed from local data to be about 0.4 mSv, which is somewhat higher than in the latitude zone 40–50° North, assessed by UNSCEAR as 0.28 mSv. The effective dose from Sr-90 and other nuclides, besides carbon-14, is assumed to be 0.7 times lower than in the zone of 40–50°, resulting from a smaller density of fallout, namely, 0.22 mSv. Thus the life-time internal dose from all nuclides to the present generation of regional inhabitants, besides C-14, is 0.62 mSv. The radiation from C-14 will be leading for future generations, with an expected dose of 2.6 mSv during millennia (3).

Thyroid exposure due to Chernobyl iodine-131. Due to the northern location of the region under consideration, wide-scale pasturing of dairy cattle did not begin during the period of radionuclide fallout in the end of April 1986. Moreover, in Finland, after the detection of the contamination, the beginning of pasturing was delayed by a month. This delay practically prevented an intake of I-131 from milk. The average I-131 intake from milk to the body was estimated to be 230 Bq, and the corresponding absorbed dose in the thyroid was 0.1 mGy (19).

In the areas of the former Soviet Union subject to the highest contamination after the accident, dosimetry of the thyroid was based on the data of measurements of I-131 content in this organ in 350 000 inhabitants. However, such measurements were not performed in the Baltic region with lower contamination, and dose assessments are based on the results of monitoring of food products, mainly milk. Thus, in the Leningrad region, where cows were taken to pasture during May (figure 5), the average thyroid doses in the inhabitants of the 70 most

contaminated villages were from 8 to 19 mGy for children and from 5 to 13 mGy for adults. The difference in dose for the children and adults is less than in the zone of the accident, where it reached an order of magnitude, because I-131 inhalation made a considerable contribution to the dose in the Leningrad region (17).

In the most contaminated areas of Lithuania, the average dose for children is estimated to be 73 mGy, and for adults it is 9 mGy. In areas with lower contamination the corresponding values are 10 and 1 mGy, respectively (39).

Dose from Chernobyl cesium-134 and cesium-137. In Finland, the internal dose from radioactive cesium was determined both from the intake of contaminated products and from the regularly measured content in the body of inhabitants of different areas. The average effective dose for the country during the first year after the accident was assessed by means of 2 methods in the narrow range of 0.15–0.2 mSv, 0.13–0.3 mSv in 1987 and 0.09–0.04 mSv per year in 1988–1990 (19, 40). In the most contaminated areas of southern Finland, the doses were 1.5–2 times higher than the country-average dose.

For inhabitants of the Leningrad region, the internal dose was determined on the basis of results of annual measurements of radionuclides in products included in the typical diet of the inhabitants. The results were compared with data of sporadic measurements of the content of cesium radionuclides in the body of inhabitants as presented earlier in this report, and they showed satisfactory agreement between the methods. For the inhabitants of the 70 most contaminated villages of the region, the average effective dose was from 1 to 3 mSv during the first year after the accident and from 1.5 to 4 mSv during 1986–1995. An insignificant addition within the limits of 0.1 to 1 mSv is predicted for the period 1996–2056 (up to 70 years after the accident) (18). The committed internal dose to inhabitants of St Petersburg due to consumption of food products imported from areas with different levels of Chernobyl contamination can be estimated to be about 0.5 mSv on the basis of measurements performed with whole-body counters.

Health risk associated with radiation

Lung cancer from radon inhalation

Inhalation causes a deposit of radon progeny on the bronchial and bronchiolar epithelium, and this deposit will generate irradiation by alpha particles. Basal and secretory cells in the bronchial epithelium are the most critical cell types.

Experimental animals. In experimental animals an exposure down to a dose of about 60 Bq/m³, corresponding

to the lifetime exposure of a man (70 years), induced lung cancer (41). Tobacco was observed to potentiate the effect of radon in animals. The generation of alpha particles can induce mutations, and an important target can be assumed to be oncogenes and tumor suppressor genes, which will impair cellular growth control. Cancer induction by radon in rats fit a model based on an inactivation of both alleles of tumor suppressor genes. In humans there are indications of radon-induced lung cancer being associated with point mutations in the p53 tumor suppressor gene.

Lung cancer in miners. Underground miners are often exposed to high concentrations of radon, and they provide strong evidence that the inhalation of radon progeny is associated with an increased risk for lung cancer (41). The data suggest that low-exposure rates, a long duration of exposure, or both may be especially hazardous. The background of this "inverse dose rate effect" is not clear, nor is it known whether it applies to the range of exposures found in most homes. Most cases among miners have been smokers, but there is also an increased risk for nonsmokers. The interaction between radon and smoking is more than additive, sometimes approaching a multiplicative model.

Residential exposure. Considering the high exposure connected with some dwellings, it can be suspected that there should be an increased risk for lung cancer. Several epidemiologic investigations have been performed with this in mind. However, much of the data suffer from imprecise exposure estimates and poor control of confounding factors. The most relevant data emanates from epidemiologic investigations using individual data of exposure and health effects — mainly case-referent data. Eight such studies have been performed, 3 in Sweden, 1 in Finland, 2 in the United States, 1 in Canada, and 1 in China (41). A summary of these results, with indications of relative risks, is given in table 3. Most of the

epidemiologic investigations of residential radon exposure have not had sufficient statistical power to detect an increase of lung cancer of the size expected from mining populations and other radiation data. The largest and most conclusive data come from the nationwide investigation in Sweden (50). The results from that investigation showed a clear association between radon exposure and lung cancer risk, and they were also consistent with a linear dose-response. The data were in accordance with expectation from the data on occupational exposure to radon, and they were also in general agreement with other investigations on residential exposure to radon. The interaction between radon and tobacco smoking was closer to multiplicity than additivity. It should be added that imprecision in the exposure estimation may have led to an underestimation of the risk by a factor of up to 2 (51).

On the basis of epidemiologic data on residential exposure to radon, it has been calculated that about 600–1500 cancer deaths occur per million people exposed to an average of 25 Bq m⁻³ during the lifetime (41). The corresponding data for an average exposure to 100 Bq/m³ is 2700–5100 deaths. The risk for lung cancer from a radon concentration of 1000 Bq/m³ is at least doubled.

A correlation between radon exposure and cancers of sites other than the lungs has been recorded in some investigations. However no clear pattern has been discerned and the situation remains unclear.

Thyroid cancer from iodine isotopes

Persons exposed as children to I-131 exhibited a highly significant increase of thyroid cancer. Up to the end of 1995, 800 cases of thyroid cancers in children under 15 years of age in Belarus, the Ukraine, and Russia have been recorded, and this rate is far beyond normal. Very young children were at the highest risk. The morbidity appears to be in reasonable agreement with the level of thyroid exposure with incorporated I-131 (6). The minimum latency period was 4 years, which is somewhat shorter than expected. The pathology did not exhibit any

Table 3. Relative risk (RR) of lung cancer related to the time-weighted-average residential radon level comparing highest and lowest exposure categories in different studies. (95% CI = 95% confidence interval)

Reference	Country	Lung cancer cases (N)	Referents (N)	Exposed category (Bq/m ³)	Reference category (Bq/m ³)	RR	95% CI
Axelsson et al, 1988 (42)	Sweden	142	264	>150 a)	<50 ^a	1.4	0.6–3.2
Pershagen et al, 1992 (43)	Sweden	210	400	>150	<75	1.7	1.0–2.9
Pershagen et al, 1994 (44)	Sweden	1360	2847	140–400 >400	<50	1.3 1.8	1.1–1.6 1.1–2.9
Ruosteenoja, 1991 (45)	Finland	238	434	>265	<109	1.2	0.7–2.1
Schoenberg et al, 1990 (46)	New Jersey, USA	433	402	>148	<37	4.2	1.0–17.5
Alavanja et al, 1994 (47)	Missouri, USA	538	1183	> 91	<29	1.2	0.9–1.7
Letourneau et al, 1994 (48)	Winnipeg, Canada	738	738	^b	^b	1.0	0.7–1.5
Blot et al, 1990 (49)	Shenyang, China	308	356	>296	<70	0.7	..

^a Radon daughter level (only rural areas included).

^b Comparing cumulative exposure categories > 7200 and < 1800 Bq/m³ years during 5–30 years prior to the study.

significant differences as compared with spontaneous cases of thyroid cancers. As therapeutic procedures are effective, only 3 of the children have died as a result of the cancer. It can be expected that this increase in the prevalence of thyroid cancers in the Chernobyl area will continue and probably persist for several decades (6).

Other carcinogenic effects

A reported increase in other types of cancer after the Chernobyl accident has not shown any consistent trend, and it can be due to differences in the follow-up of exposed populations. However, due to the limited amount of material, the power of the investigations is probably too low to demonstrate a risk. Among the 7.1 million residents of contaminated territories, it could empirically be expected that the accident would give rise to 6600 cases of fatal cancer during the next 85 years against the number of spontaneous cases of 870 000. It would be difficult to discern the induced cases under these circumstances.

Particular attention ought to be given to leukemias, which have a short latency period. On the bases of the survivor material for the Japanese atomic bombs, few cases of radiation-induced leukemia could be expected. Thus 470 cases among the 7.1 million residents in the contaminated area could be expected. This rate would be rather difficult to distinguish from spontaneous fatalities of 25 000. Among the 200 000 "liquidators", who received particularly high doses, 200 induced cases and 800 spontaneous ones could be anticipated, 150 of these 200 cases could be expected to occur during the first 10 years.

It can be concluded from the available data on malignancies that, except for thyroid cancers, no observable increase has occurred in the population as a result of the Chernobyl accident.

Inherited health effects

Induced hereditary health effects have not been unequivocally demonstrated in humans. This statement applies also to the survivors of the atom bombs in Japan. Although hereditary changes should be induced in humans, just as in experimental organisms, the material available for observation in human situations has been too limited to allow mutations to be discerned, at least with conventional genetic systems. In recent years highly sensitive mutation systems have become available for the analysis of induced genetic alterations, such as fingerprinting with minisatellite DNA, although the association with health effects is not clear (52). Such a monitoring system has been successfully applied in Chernobyl (53). By comparing exposed parents with their children, it was observed that the frequency of minisatellite mutations transmitted to children was twice as high as in a reference population. The interpretation of this observation has been questioned, primarily because the reference pop-

ulation came from another geographic area (54). However, it could also be demonstrated that the mutation frequency in Chernobyl correlated with the level of surface contamination with Cs-137. Further evidence was provided of radiation-induced minisatellite mutations from the Chernobyl accident by an extended material with new family analyses, new minisatellites, and improved dosimetry (55).

Discussion

Numerous nuclear installations have been operating in the Baltic region for more than 20 years. During this time none of them appeared to be the source of significant radioactive contamination of the region under consideration. Nevertheless, both the Baltic Sea and its coast are extensively contaminated with man-made radionuclides. The history of development of nuclear technologies since World War II explains this apparent contradiction. Since the early 1950s and during the whole decade, extensive testing of nuclear weapons has been performed in the atmosphere, and this testing led to global radioactive contamination predominantly of the northern hemisphere with long-lived radionuclides Sr-90 and Cs-137. Waters of the Baltic Sea were additionally contaminated during the 1970s and early 1980s with radionuclides discharged in the marine environment from west European plants reprocessing fuel for both military and civilian purposes. Eventually, in 1986, the severe accident at the Chernobyl plant contaminated large areas of the east Baltic region to the levels of an order of magnitude higher than the past global fallout. Thus regional environment including biota was contaminated from global and distant, but not from local, radionuclide sources.

It is important to emphasize that sources of environmental contamination and human exposure do not coincide in some cases. People are mainly affected by external and internal irradiation via terrestrial and, to a less extent, via freshwater ecosystems but very seldom via marine ones. Radionuclides that have entered the marine ecosystem are usually diluted in enormous volumes of sea water. Another reason for most radionuclides not being significantly concentrated in marine biota is the high ionic concentration in the sea water and high competition of stable and radioactive isotopes for transfer in biota.

Table 4 presents contributions of major sources and pathways in the environmental exposure of the east Baltic population. Doses from medical exposure numbering from 0.5 to 1 mSv annually are not included in this table. Two recently studied areas are presented for consideration and comparison, 2 districts of the Leningrad region (Russia) and southern Finland, both contaminated

after the Chernobyl accident. The mean effective annual doses received by the population in 1986 (year of the Chernobyl accident) and recently in 1996 are presented. The dose calculations are based on data presented in this paper. The Chernobyl radioactive products increased human doses significantly only during the first year after the accident, by 60% in the most contaminated districts of the Leningrad region and by 12% in southern Finland. During the next few years the influence of the Chernobyl accident decreased gradually, and at present its contribution to the human exposure in the region under consideration is almost negligible. The principal present environmental consequence of the Chernobyl accident is long-term contamination of natural ecosystems with Cs-137 — terrestrial such as forests and their products, for example, game, mushrooms and berries and freshwater such as oligotrophic lakes and their fish. Continuing human exposure from the Chernobyl accident is to a large part associated with this pathway.

From table 4 the level of health risk associated with population environmental irradiation can be roughly evaluated. Contemporary estimates given by the ICRP for the risk coefficient for radiation inducing fatal cancer in the population is $5 \cdot 10^{-2}$ person-Sv⁻¹ (2). Taking into account the dominating hypothesis of a linear dependence of the health risk on effective dose, multiplication of the current annual dose of 2 to 3 mSv by the risk coefficient gives an estimation of health risk of 1 to $1.5 \cdot 10^{-4}$, which is rather high for a single environmental factor.

Opportunities to decrease the dose and the associated risk exist for some components of population exposure, but they are not feasible for others. It is practically impossible to decrease the level of cosmic radiation, as well as the content of K-40 in the human body and the intake of other natural radionuclides with food. However, methods for decreasing radon air concentrations indoors are well known (hermetization of the ground floor and ventilation of cellars). Standardization and control of the natural radionuclide concentrations in drinking water and building materials has been implemented in some countries. With regard to the Chernobyl component of human exposure in the contaminated regions, the Cs-137 intake with highly contaminated natural food deserves attention. This pathway accounts for about 50% or more of the internal dose (21, 24, 25). Simple boiling of mushrooms and the removal of the boiling water provides an intake decrease by a factor of 2 to 3. It should be emphasized that any countermeasure application has to be based on optimization analysis. The countermeasure considered should bring more good than harm, including both economic cost and social consequences.

The location of numerous nuclear installations in the Baltic region — at least 8 nuclear power plants with 22 reactors, 4 research reactors, nuclear shipyards, storage sites for spent fuel and radioactive wastes — creates po-

Table 4. Mean effective dose of inhabitants of the East Baltic region^a caused by environmental sources in 1986 (nominator) and 1996 (denominator), mSv.

Source origin	Leningrad region ^b		Southern Finland	
	External	Internal	External	Internal
Natural				
Cosmic radiation	0.3	-	0.3	-
Potassium-40		0.17		0.17
Uran and thorium chains	0.3	0.1	0.5	0.1—0.6
Radon and its daughters	-	1.3	-	2.0
Subtotal	0.6	1.6	0.8	2.3—2.8
Anthropogenic				
Global fallouts	0.003/0.002	0.002	0.003/0.002	0.002
Chernobyl accident	0.5/0.05	1/0.03	0.3/0.03	0.2/0.02
Sellafield	< 0.01	< 0.01	< 0.01	< 0.01
Regional installations	< 0.01	< 0.01	< 0.01	< 0.01
Subtotal	0.5/0.05	1/0.03	0.3/0.03	0.2/0.02
Total	3.7/2.3		3.7/3.3	

^a Dose from medical exposure (0.5 to 1 mSv) not included.

^b Kingisepp and Volosovo districts.

tential threats of severe nuclear or radiation accidents in the region. Such an accident may have more serious radiological consequences than past releases. STUK experts have modeled radiological consequences of a major accident at the nuclear power plant in Sosnovy Bor near St Petersburg (56). For typical meteorological conditions it may result in significant exposure to the inhabitants of Helsinki (220 km from the source), Kotka (140 km from the source), and other Finnish, as well as Russian, cities. The effective dose for Helsinki and Kotka inhabitants was estimated to be 40 to 60 mSv per 2 days, which will not result in acute radiation injury. But thyroid dose estimates (300 to 470 mGy) may cause an increase of thyroid cancer morbidity, as is the case now in the area near Chernobyl. For unfavorable meteorological conditions exposure doses may become higher by a factor of 2 to 5.

As conditions of regional economic development determine long-term application of nuclear energy and technologies, protective measures should be concentrated both on the prevention of major accidents and their immediate detection and on appropriate radiation protection of the population if an accident occurs.

Concluding remarks

1. The east Baltic region has been contaminated with long-life man-made radionuclides since the 1960s. The principal radionuclides contributing to long-term contamination of the region are Cs-137 and Sr-90. Neither in the past nor recently have local nuclear installations constituted sources of significant radioactive contamination of the east Baltic region. Principal sources of large-scale radioactive contamination were global fallout of radioactive products of nuclear weapons tests in the 1950s and early 1960s, the Chernobyl

accident in 1986, and the marine transport of radionuclides discharged from western European nuclear fuel reprocessing plants (Sellafield and La Hague).

2. The population of the east Baltic region is affected by current mean annual exposure doses from environmental radioactivity of 2 to 3 mSv. The main contributors to these doses are natural sources, primarily radon and its decay products inhaled with air in buildings.
3. In 1986, due to the Chernobyl accident, inhabitants of the east Baltic region obtained an additional exposure dose of about 0.5 mSv in southern Finland and about 1.4 mSv in most of the contaminated districts of the Leningrad region. Both external and internal exposure via the consumption of contaminated food contributed to this exposure. In subsequent years the population exposure of the Chernobyl origin decreased gradually and is now insignificant when compared with exposure from natural sources.
4. Currently, significant long-term radiological consequences of the Chernobyl accident in the east Baltic region include persistent radioactive contamination of natural terrestrial (forest) and freshwater (oligotrophic lakes) ecosystems and food products collected in these ecosystems (mushrooms, berries, game and fish). This contamination is expected to persist for decades.
5. A present health risk caused for the population by environmental radioactivity and estimated from modern data of human radiobiology on the basis of radiation protection methodology is of the order of 100–150 cases of cancer per year in 1 million people. This level of risk is primarily determined by the level of natural radiation (radon and its daughter products in dwellings and other sites). The contribution of man-made radiation was significant (10–60%) in the early 1960s during the period of intense nuclear tests and in 1986–1988 after the Chernobyl accident.
6. The radiological threats to human health and environmental contamination in the east Baltic region are associated with a potential for nuclear accidents and failures in the containment of the large storage sites of high-level radioactive substances. Potential sources for these events include (i) accidents at nuclear reactors operating in regional nuclear power plants and, to a less extent, in nuclear-powered vessels and research centers, (ii) accidents in military operations with nuclear weapons, (iii) accidents at large regional storage sites of high-level radioactive waste and spent nuclear fuel, and (iv) the transboundary transfer of radionuclides released in large-scale accidents in adjacent regions.

Recommendations

Contemporary international guidelines on radiation protection, nuclear safety, radioactive waste management,

and emergency preparedness should be rigorously adhered to by all east Baltic states to minimize the probabilities and consequences of major accidents.

More authoritative and comprehensive evaluations should be made of the risks posed to human health and the environment by accidents. Assessments of the risks of releases of radionuclides and the radiological consequences for humans and the environment should be performed for all existing nuclear installations in, and near, the east Baltic region.

Early-warning monitoring systems should be developed in countries of the former Soviet Union and radiological emergency preparatory actions should be maintained permanently in all countries of the east Baltic region.

Countermeasures to decrease population exposure from regulated sources should be optimized and implemented, including the prevention of radon inhalation in dwellings with an elevated radon concentration in air and a decrease of Cs-137 consumption from certain natural food.

References

1. Helsinki Commission, Baltic Marine Environment Protection Commission. Radioactivity in the Baltic Sea: Baltic sea environment proceedings, 1995. no 61.
2. International Commission on Radiological Protection (ICRP). Recommendations of the International Commission on Radiological Protection: ICRP publication 60. Ann ICRP, 1990;21:1–3.
3. The United Nations Scientific Committee on the Effects of Atomic Radiation. Sources and effects of ionizing radiation. New York (NY): United Nations, 1993.
4. Marey AN, Barkhudarov RM, Knizhnikov VA, Borisov BK, Petukhova EV, Novikova NJa. Global fallouts of nuclear blast products as factor of human exposure [in Russian]. Moscow: Atomizdat, 1980..
5. The United Nations Scientific Committee on the Effects of Atomic Radiation. Sources and biological effects. New York (NY): United Nations, 1982. Publ no E.82.9.8.
6. One decade after Chernobyl: summing up the consequences of the accident. In: IAEA. Proceedings of an International Conference, Vienna, 8–12 April 1996. Vienna: IAEA, 1996.
7. Cort MD, Tsaturov YuS, editors. Atlas on caesium contamination of Europe after the Chernobyl nuclear plant accident. Luxembourg: Office for Official Publications of the European Community, 1996. Joint study project no 6. Final report no EUR 16542 EN.
8. Nordlinder S, Aggeryd I, Bergström U, Sundblad B. Environmental risk assessment for the Sillamae waste repository. In: Environmental impact of radioactive releases: proceedings of a symposium, Vienna, 8–12 May 1995. Vienna: IAEA, 1995;221–8.
9. Arvela H. Residential radon in Finland: sources, variation, modelling and dose comparisons. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1995. STUK-A124,

10. Pahapill L, Rulkov A, Astri G. Radon in Estonian buildings: establishment of a measurement system and obtained results. Stockholm: Swedish Radiation Protection Institute 1996. SSI rapport, no 96:13.
11. Tishkov VP, Gritchenko ZG, Khvostova IYu. Radioactive fallouts: results of observations in the region of St Petersburg in 1995 [in Russian]. St Petersburg: Khlopin Radium Institute, 1996.
12. Arvela H, Markkanen M, Lemmela H, Blomqvist L. Environmental gamma radiation and fallout measurements in Finland, 1986—87. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1989. Supplement 2 of the annual report STUK-A74: STUK-A76.
13. Sinkko K, Aaltonen H, Mustonen R, Taipale TK, Juutilainen J. Airborne radioactivity in Finland after the Chernobyl accident in 1986. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1987. Supplement 1 of the annual report STUK-A55: STUK-A56.
14. Saxen R, Aaltonen H, Ikäheimonen TK. Airborne and deposited radionuclides in Finland in 1989—90. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1994. Supplement 11 of the annual report STUK-A89: STUK-A109.
15. The radiobiological impact of hot beta-particles from the Chernobyl fallout: risk assessment. Presented at the research co-ordination meeting within the IAEA Co-ordinated Research Programme. Kiev, 26—30 August 1991. Vienna: IAEA, 1992.
16. Arvela H, Blomqvist L, Lemmela H, Savolainen AL, Sarkkula S. Environmental gamma radiation measurements in Finland and the influence of meteorological conditions after the Chernobyl accident in 1986. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1987. Supplement 10 of the annual report STUK-A55: STUK-A65.
17. Zvonova IA, Balonov MI, Barishkov NK, Golikov V, Shutov V, Bruk G, et al. Reconstruction and prognosis of exposure doses for population of the Volosovo district of Leningrad region due to the accident at Chernobyl NPP (1986—2056) [in Russian]. St Petersburg: Institute of Radiation Hygiene, 1995.
18. Balonov MI, Zvonova IA, Barishkov NK, Golikov V, Shutov V, Bruk G, et al. Reconstruction of exposure doses for population of the Kingisepp district of Leningrad region due to the accident at Chernobyl NPP (1986—1994) [in Russian]. St Petersburg: Institute of Radiation Hygiene, 1995.
19. Finnish Centre for Radiation and Nuclear Safety (STUK). Studies on environmental radioactivity in Finland in 1986. In: Finnish Centre for Radiation and Nuclear Safety (STUK), Annual report: STUK-A55. Helsinki: STUK, 1987.
20. Rantavaara A, Haukka S. Radioactivity of milk, meat, cereals and other agricultural products in Finland after the Chernobyl accident in 1986. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1987. Supplement 3 of the annual report STUK-A55: STUK-A58.
21. Shutov VN, Bruk GY, Balonov MI, Parkhomenko V, Pavlov IY. Cesium and strontium radionuclide migration in the agricultural ecosystem and estimation of internal doses to the population. In: Merwin SE, Balonov MI, editors. The Chernobyl papers, vol 1 (Doses to the Soviet population and early health effects studies). Richland (WA): Research Enterprises, 1993;167—220.
22. Malevanny IN, Balonov MI, Sigalov VM, et al. Complex program of the radiation investigation of settlements of the Leningrad region located in the zone of radioactive contamination due to the accident at the Chernobyl NPP [in Russian]. St Petersburg: 1993. Report for administration of the Leningrad region.
23. Shutov VN, Bruk GJ, Basalaeva LN, Vasilevitskiy VA, Ivanova NP, Kaplun IS. The role of mushrooms and berries in the formation of internal exposure doses to the population of Russia after the Chernobyl accident. Radiat Prot Dosim 1996; 67:55—64.
24. Rantavaara A. Introduction to terrestrial seminatural ecosystems. In: Dahlgaard H, editor. Nordic radioecology: the transfer of radionuclides through nordic ecosystems to man. Amsterdam-Lausanne-New York- Oxford-Shannon-Tokyo: Elsevier, 1994;263—4.
25. Johanson KJ. Radiocaesium in game animals in the nordic countries. In: Dahlgaard H, editor. Nordic radioecology: the transfer of radionuclides through nordic ecosystems to man. Amsterdam-Lausanne-New York- Oxford-Shannon-Tokyo: Elsevier, 1994;287—302.
26. Rantavaara A. Radioactivity of vegetables and mushrooms in Finland after the Chernobyl accident in 1986. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1987. Supplement 4 of the annual report STUK-A55: STUK-A59.
27. Gavrilov VM, Gritchenko ZG, Ivanova LM. Strontium-90, cesium-134 and cesium-137 in water bodies of the Baltic region of the Soviet Union (1986—1988) [in Russian]. Radiokhimiya 1990;32;3:171—9.
28. Saxén R, Aaltonen H. Radioactivity of surface water in Finland after the Chernobyl accident in 1986. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1987. Supplement 5 of the annual report STUK-A55: STUK-A60.
29. Saxén R, Koskelainen U. Radioactivity of surface water and freshwater fish in Finland in 1988—1990. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1992. Supplement 6 of the annual report STUK-A89: STUK-A94.
30. Saxén R, Rantavaara A. Radioactivity of fresh water fish in Finland after Chernobyl accident in 1986. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1987. Supplement 6 of the annual report STUK-A55: STUK-A61.
31. Lazarev LN, Kuznetsov YuV, Gedeonov LI, et al. The investigation of water contamination of the Baltic Sea in 1986 [in Russian]. Radiokhimiya 1988;30(3):395—401.
32. Ilus E, Sjöblom K-L, Ikäheimonen TK, Saxén R, Klemola S. Monitoring of radionuclides in the Baltic Sea in 1989—90. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1993. Supplement 10 of the annual report STUK-A89: STUK-A103.
33. Suomela M, Blomqvist L, Rahola T, Rantavaara A. Studies on environmental radioactivity in Finland in 1987. In: Finnish Centre for Radiation and Nuclear Safety (STUK). Annual report: STUK-A74. Helsinki: STUK, 1991.
34. Balonov M, Jacob P, Likhtarev I, Minenko V. Pathways, levels and trends of population exposure after the Chernobyl accident. In: Karaoglou A, Desmet G, Kelly GN, Menzel HG, editors. The radiological consequences of the Chernobyl accident. Brussels-Luxembourg: Office for Official Publications of the European Community, 1996;235—49. EUR 16544 EN.
35. Krisyuk EM, Konstantinov YuO, Nikitin VV, Ponikarova TM, Prokofjev ON. Doses of population exposure [in Russian]. Gig Sanit 1984;5:63—6.
36. Rahola T, Suomela M, Illukka E, Puhakainen M, Pusa S. Radioactivity of people in Finland in 1989—90. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1993. Supplement 8 of the annual report STUK-A89: STUK-A96.
37. Lebedev OV, Yakovlev VA. The correlation between ^{137}Cs

- half-time and age, body mass and height in individuals contaminated from the Chernobyl accident. In: Merwin SE, Balonov MI, editors. *The Chernobyl papers, vol 1: doses to the Soviet population and early health effects studies*. Richland (WA): Research Enterprises, 1993:221—46.
38. Balonov MI, Shutov VN, Bruk Gya, Golikov V. Report on the radiation situation in the Kingisepp district of Leningrad region and adjacent territories [in Russian]. St Petersburg: Institute of Radiation Hygiene, 1989.
39. Nedveckaite T, Filistowicz W. Estimates of thyroid equivalent dose in Lithuania following the Chernobyl accident. *Health Phys* 1995;69:2:265—8.
40. Rahola T, Suomela M, Illukka E, Pusa S. Radioactivity of people in Finland in 1988. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1991. Supplement 2 of the annual report STUK-A89: STUK-A91.
41. Pershagen G. Radon. In: World Health Organization (WHO). *Air quality guidelines for Europe*. 2nd ed. Copenhagen: Regional Office for Europe. In press.
42. Axelson O, Andersson K, Desai G, Fagerlund I, Jansson B, Karlsson C, et al. Indoor radon exposure and active and passive smoking in relation to the occurrence of lung cancer. *Scand J Work Environ Health* 1988;14:286—92.
43. Pershagen G, Liang Z-H, Hrubec Z, Svensson S, Boice JD. Residential radon exposure and lung cancer in Swedish women. *Health Phys* 1992;63:179—86.
44. Pershagen G, Åkerblom G, Axelson O, Clavensjö B, Damber L, Desai G, et al. Residential radon exposure and lung cancer in Sweden. *N Engl J Med* 1994;330:159—64.
45. Ruosteenoja E. Indoor radon and risk of lung cancer: a epidemiological study in Finland [thesis]. Helsinki: Finnish Centre for Radiation and Nuclear Safety, 1991.
46. Schoenberg JB, Klotz JB, Wilcox HB, Nicholls GP, Gil-del Real MT, et al. Case-control study of residential radon and lung cancer among New Jersey women. *Cancer Res* 1990;50:6520—4.
47. Alavanja MCR, Brownson RC, Lubin JH, Berger E, Chang J, Boice JD Jr. Residential radon exposure and lung cancer among nonsmoking women. *J Natl Cancer Inst* 1994;86:1829—37.
48. Létourneau EG, Krewski D, Choi NW, Goddard MJ, McGregor RG, Zielinski JM, et al. Case-control study of residential radon and lung cancer in Winnipeg, Manitoba, Canada. *Am J Epidemiol* 1994;140:310—22.
49. Blot WJ, Xu Z-Y, Boice JD Jr, Zhao D-Z, Stone BJ, Sun J, et al. Indoor radon and lung cancer in China. *J Natl Cancer Inst* 1990;82:1025—30.
50. Pershagen G, Åkerblom G, Axelson O, Clavensjö, Damber L, Desai G, et al. Residential radon exposure and lung cancer in Sweden. *N Engl J Med* 1994;330:159—64.
51. Lagarde F, Pershagen G, Åkerblom G, Axelson O, Bäverfjord U, Damber L et al. Residential radon and lung cancer in Sweden: risk analysis accounting for radon error in the exposure assessment. *Health Phys* 1997;72:269—76.
52. Jeffreys AJ, Wilson V, Thein SL. Hypervariable “minisatellite” regions in human DNA. *Nature* 1985;316:67—73.
53. Dubrova YE, Nesterov VN, Krouchinsky NG, Ostapenko VA, Neuman R, Neil DL, et al. Human minisatellite mutation rate after the Chernobyl accident. *Nature* 1996;380:683—6.
54. Sato C, Kodaira M. Effects of radiation on children. *Nature* 1996;383:226.
55. Dubrova YE, Nesterov VN, Krouchinsky NG, Ostapenko VA, Vergnaud G, Giraudeau F, et al. Further evidence for elevated human satellite mutations rate in Belarus eight years after the Chernobyl accident. *Mutat Res* 1997;381:267—78.
56. Lathinen J, Toivonen H, Pöllänen R, Nordlund G. A hypothetical severe reactor accident in Sosnovy Bor, Russia: short-term radiological consequences in Southern Finland. Helsinki: Finnish Centre for Radiation and Nuclear Safety (STUK), 1993. STUK-A113.