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Environmental Consequences of the Chernobyl Accident and Their Remediation: Twenty Years of Experience

Report of the UN Chernobyl Forum Expert Group "Environment" (EGE)

August 2005

FOREWORD

The explosion on 26 April 1986 at the Chernobyl Nuclear Power Plant located just 100 km from the city of Kyiv in what was then the Soviet Union and now is Ukraine, and consequent ten days' reactor fire resulted in an unprecedented release of radiation and unpredicted adverse consequences both for the public and the environment. Indeed, the IAEA has characterized the event as the "foremost nuclear catastrophe in human history" and the "largest regional release of radionuclides into the atmosphere".

Massive radioactive contamination forced the evacuation of more than 100,000 people from the affected region during 1986, and the relocation, after 1986, of another 200,000 from Belarus, the Russian Federation and Ukraine. Some five million people continue to live in areas contaminated by the accident and have to deal with its environmental, health, social and economic consequences. The national governments of the three affected countries, supported by international organizations, have undertaken costly efforts to remedy contamination, provide medical services and restore the region's social and economic well-being.

The accident's consequences were not limited to the territories of Belarus, Russia and Ukraine but resulted in substantial transboundary atmospheric transfer and subsequent contamination of numerous European countries that also encountered problems of radiation protection of their populations, although to less extent than the three more affected countries.

Although the accident occurred nearly two decades ago, controversy still surrounds the impact of the nuclear disaster. Therefore the IAEA, in cooperation with FAO, UNDP, UNEP, UN-OCHA, UNSCEAR, WHO and The World Bank, as well as the competent authorities of Belarus, the Russian Federation and Ukraine, established the Chernobyl Forum in 2003. The mission of the Forum was — through a series of managerial and expert meetings — to generate "authoritative consensual statements" on the environmental consequences and health effects attributable to radiation exposure arising from the accident as well as to provide advice on environmental remediation and special health care programmes, and to suggest areas where further research is required. The Forum was created as a contribution to the United Nations' ten years strategy for Chernobyl, launched in 2002 with the publication of *Human Consequences of the Chernobyl Nuclear Accident – A Strategy for Recovery*.

In 2003-2004, two groups of experts from twelve countries, including Belarus, Russia and Ukraine, and from relevant international organizations have assessed the accident'senvironmental and health consequences. In early 2005, the group "Environment," coordinated by the IAEA, and the group "Health," coordinated by the WHO, have presented their reports for Forum consideration. Both reports were considered and approved by the Forum at its meeting on 18-20 April 2005. This meeting also decided, *inter alia*, 'to consider the approved reports … as a common position of the Forum members, i.e., of the eight United Nations organizations and the three more affected countries, regarding environmental and health consequences of the Chernobyl accident, as well as recommended future actions, i.e., as a consensus within the United Nations system'.

This report presents the findings and recommendations of the Chernobyl Forum concerning environmental effects of the Chernobyl accident. The Forum's report considering health effects is in process of publication under WHO responsibility. The environmental group of experts was chaired by Dr. Lynn Anspaugh from the University of Utah, USA; the scientific secretary of this group and of the whole Chernobyl Forum activity was Dr. Mikhail Balonov of the Division of Radiation, Transport and Waste Safety, IAEA. In all cases the scientists from the UN organisations, the international community, and the three more affected countries have been able to reach consensus in the preparation of their respective documents. After approval by the members of the Forum, this report is the result of that process.

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INTRODUCTION

The purpose of this report is to provide an up-to-date evaluation of the environmental effects of the 26 April 1986 accident at the Chernobyl Nuclear Power Plant. Even though it is now nearly 20 years after the accident and substantial monies have been spent on such evaluations, there are still many conflicting reports and rumours. This joint report has been developed with the full cooperation of the United Nations (UN) family of relevant organisations and with political representatives from the three more affected countries: Ukraine, Belarus, and the Russian Federation. In addition, recognised scientific experts from the three countries and additional international experts provided the basis for the preparation of reports for review by the actual members of the Chernobyl Forum.

The "Chernobyl Forum" is a high-level political forum whose suggestion for existence was initiated by the International Atomic Energy Agency (IAEA) in cooperation with the Food and Agriculture Organisation (FAO), the United Nations Office for Coordination of Humanitarian Affairs (OCHA), the United Nations Development Programme (UNDP), the United Nations Environment Programme (UNEP), the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), the World Health Organisation (WHO), and the World Bank, as well as the competent authorities of Belarus, the Russian Federation, and Ukraine. The organisational meeting of the Chernobyl Forum was held on 3–5 February 2003, at which time the decision was reached to establish the Forum as an ongoing entity of the above named organisations.

Thus, the organisational meeting of the Forum decided to establish the Chernobyl Forum as a series of managerial, expert and public meetings in order to generate authoritative consensual statements on the health effects attributable to radiation exposure arising from the accident and the environmental consequences induced by the released radioactive materials, to provide advice on remediation and special health-care programmes, and to suggest areas where further research is required; and to accept the following Terms of Reference (TOR) of the Forum.

The objectives of the Chernobyl Forum were defined as follows:

— To explore and refine the current scientific assessments on the long-term health and environmental consequences of the Chernobyl accident, with a view to producing authoritative consensus statements focusing on:
  - the health effects attributable to radiation exposure caused by the accident,
  - the environmental consequences induced by the radioactive materials released due to the accident, e.g., contamination of foodstuffs, and additionally
  - to address the consequences attributable to the accident although not directly related to the radiation exposure or radioactive contamination;

— To identify gaps in scientific research relevant to the radiation-induced or radioactive contamination-induced health and environmental impacts of the accident, and suggest areas where further work is required based on an assessment of the work done in the past, and bearing in mind ongoing work and projects;

— To provide advice on, and to facilitate implementation of scientifically sound programmes on mitigation of the accident consequences, including possible joint actions of the organizations participating in the Forum, such as:

- remediation of contaminated land with the aim of making it suitable for normal agricultural, economic and social life under safe conditions,
- special health care of the affected population,
- monitoring of the long-term human exposure to radiation, and
- addressing the environmental issues pertaining to the decommissioning of the Shelter and management of radioactive waste originating from the Chernobyl accident.

The Chernobyl Forum itself continued as a high-level organisation of senior officials from UN agencies and the three more affected countries. The actual work has been accomplished by two expert groups: Expert Group “Environment” (EGE) and Expert Group “Health” (EGH). Members of each of these two groups consisted of recognised international scientists, including those from the three more affected countries. Within these two groups and their sub-working groups the draft documents to support this report were prepared for consideration by the members of the Forum. The EGE was coordinated by the IAEA and the EGH was coordinated by the WHO.

In all cases the scientists from the UN organisations, the international community, and the three affected countries have been able to reach consensus in the preparation of their respective draft documents. After approval by the members of the Forum itself and final editing of the draft documents, this report is the result of that process regarding the environmental consequences of the Chernobyl accident. A similar report on the health effects attributable to radiation exposure arising from the accident has been prepared for Forum consideration and approval by the Expert Group “Health.”

RADIOACTIVE CONTAMINATION OF THE ENVIRONMENT

The significance of the Chernobyl accident as the foremost nuclear catastrophe in human history is determined by its being the largest regional release of radionuclides into the atmosphere and the subsequent radioactive contamination of the environment. A number of European countries were subjected to different levels of radioactive contamination, amongst them the three former USSR republics, now Belarus, Russia and Ukraine, all in the vicinity of whose common border the Chernobyl NPP is located. The deposited radionuclides gradually decayed and moved inside the environments – atmospheric, aquatic, terrestrial and urban – and among the environments. These processes that determine patterns and regularities of radiation effects, both on humans and non-human species, are presented in this section.

Conclusions

Radionuclide release and deposition

Major releases from Unit 4 continued for ten days, and included radioactive gases, condensed aerosols and a large amount of fuel particles. The total release of radioactive substances was about 14 EBq[1] (as of 26 April 1986) which included 1.8 EBq of ^{131}I, 0.085 EBq of ^{137}Cs, other Cs-isotopes, 0.01 EBq of ^{90}Sr and 0.003 EBq of Pu radioisotopes. The noble gases contributed about 50% of the total release.

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[1] 1 EBq = 10^{18} Bq (Becquerel).

Large areas of Europe were affected to some degree by the Chernobyl releases. A total of more than 200 thousand km^2 in Europe were contaminated with radiocaesium (above 0.04 MBq of ^{137}Cs per sq. m) of which 71% is in the three more affected countries, Belarus, Russia and Ukraine. The deposition was highly heterogeneous; it was strongly influenced by where it was raining when the contaminated air masses passed. In the mapping of the deposition, ^{137}Cs was chosen because it is easy to measure and of radiological significance. Most of the strontium and plutonium radioisotopes were deposited close (less than 100 km) to the reactor due to their being contained within larger particle sizes.

Many of the more important radionuclides in the releases had short physical half lives, whilst the long-lived radionuclides were released in smaller amounts. Thus, most of the radionuclides released by the accident have since long decayed away. The releases of radioactive iodines caused concern immediately after the accident. Due to the emergency situation and the short half life of ^{131}I, there were few reliable measurements on the spatial distribution of deposited radioiodine which is important in determining doses to the thyroid. Current measurement of ^{129}I may assist in estimating ^{131}I deposition better and thereby improving thyroid-dose reconstruction.

After that and for the decades to come ^{137}Cs will continue to be of greatest importance, with secondary attention to ^{90}Sr. For the first years ^{134}Cs was also important. Over the longer term (100s to 1000s of years) the only radionuclides anticipated to be of interest are the plutonium isotopes and ^{241}Am.

Urban environment

In urban areas open surfaces such as lawns, parks, streets, roads, squares, building roofs and walls became contaminated with radionuclides. Under dry conditions, trees, bushes, lawns and roofs became more contaminated, under wet conditions horizontal surfaces received the highest contamination, such as soil plots, lawns, etc. Particularly high ^{137}Cs-activity concentrations were found around houses where the rain had transported the radioactivity from the roofs to the ground. The deposition in urban areas in the nearest city of Pripyat and surrounding settlements could have initially given rise to substantial external dose, which was partially averted by the evacuation of the people. The deposition of radioactive material in other urban areas has given rise to various contributions to dose in the subsequent years and continues to do so.

Due to wind and rain and human activities, including traffic, street washing and cleanup, surface contamination by radioactive material has been reduced significantly in inhabited and recreational areas during 1986 and afterwards. One of the consequences of these processes has been secondary contamination of sewage systems and sludge storage.

At present, in most of the settlements subjected to radioactive contamination the air-dose rate above solid surfaces has returned to the pre-accidental background level. The elevated air-dose rate remains mainly over undisturbed soil in gardens, kitchen-gardens and parks.

Agricultural environment

In the early phase direct surface deposition of many different radionuclides dominated contamination of agricultural plants and animals consuming them. The release and deposition of radioiodine isotopes caused the most immediate concern, but the problem was confined to the first 2 months, because of the short physical half-life of 8 days of the most important isotope, ^{131}I. The radioiodine was rapidly transferred to milk at a high rate in Russia, Ukraine and Belarus leading to significant thyroid doses to those consuming milk, especially children. In the rest of Europe the consequences of the accident varied depending on the season and increased levels of radioiodine in milk were observed in some contaminated southern areas, where dairy animals were already outdoors.

Different crop types, in particular, green leafy vegetables, were also contaminated with radionuclide mixture to various degrees depending on the deposition levels, and time of the growing season. The direct deposition onto plant surfaces was of concern for about two months.

After the early phase of direct contamination, uptake of radionuclides through plant roots from soil became increasingly important and showed strong time dependence. Radioisotopes of cesium (^{137}Cs and ^{134}Cs) were the nuclides which led to the largest problems, and after decay of ^{134}Cs, ^{137}Cs still causes problems in some Belarusian, Russian and Ukrainian areas. In addition, ^{90}Sr could cause problems in the near field but at longer distances from the reactor its deposition levels were too low. Other radionuclides such as plutonium isotopes and ^{241}Am were either in so low deposition levels, or not very available for root uptake, to cause real problems in agriculture.

In general, there has been an initial substantial reduction in transfer of radionuclides to vegetation and animals as would be expected due to weathering, physical decay, migration of radionuclides down the soil and reductions in bioavailability in soil. Particularly in contaminated intensive agricultural systems, mostly in the former Soviet Union, there has been substantial reduction in the transfer of ^{137}Cs to plants and animals, especially in the first few years. However, in the last decade there has been little further obvious decline and long-term effective half lives have been difficult to quantify with precision.

The radiocaesium-activity concentrations in foodstuffs after the early phase were influenced not only by deposition levels but also by soil types, management practices and type of ecosystem. The major and persistent problems in the affected areas occur in extensive agricultural systems with soils with a high organic content and animals grazing in unimproved pastures which are not ploughed or fertilized. This particularly affects rural residents in the former Soviet Union who are commonly subsistence farmers with privately owned dairy cows.

In the long term ^{137}Cs in meat and milk and to less extent ^{137}Cs in the vegetable foods remain the most important contributors to human internal dose. As its activity concentration both in vegetable and animal foods has been decreasing during the last decade very slowly, 3 to 7% per year, contribution of ^{137}Cs to dose will continue to dominate for years and decades to come. Importance of other long-lived radionuclides, ^{90}Sr, plutonium isotopes and ^{241}Am, in terms of human dose will remain insignificant.

Forest environment

Following the Chernobyl accident vegetation and animals in forests and mountain areas have shown particularly high uptake of radiocaesium, with the highest recorded ^{137}Cs-activity concentrations found in forest products due to the persistent recycling of radiocaesium in forest ecosystems. Particularly high ^{137}Cs activity-concentrations have been found in mushrooms, berries and game, and these high levels have persisted for many years. Thus, whilst there has been a general decline in the magnitude of exposures through agricultural products there has been continued high levels of contamination of forest-food products which still exceed intervention limits in many countries. This can be expected to continue for several decades to come. Therefore, the relative importance of forests in contributing to radiological exposures of the populations of several affected countries has increased with time. It will primarily be the combination of downward migration in the soil and the physical decay of ^{137}Cs which contributes to any further slow long-term reduction in contamination of forest food products.

The high transfer of radiocaesium in the pathway lichen – reindeer meat - humans has been demonstrated again after the Chernobyl accident in the Arctic and sub-Arctic areas of Europe. The Chernobyl accident led to considerable contamination of reindeer meat in Finland, Norway, Russia and Sweden and caused significant problems for the Sami people.

The use of timber and associated products only makes a small contribution to exposure of the general public, although wood ash can contain high amounts of ^{137}Cs and could potentially give rise to a higher doses than other uses of wood. ^{137}Cs in timber is of minor importance, although doses in the wood-pulp industry have to be considered.

Forest fires have increased air-activity concentrations in 1992, but not to a high extent. The possible radiological consequences of forest fires have been much discussed, but these are not expected to cause any problems of radionuclide transfer from contaminated forests, except possibly in the nearest surroundings of the fire.

Aquatic environment

Radioactivity from Chernobyl contaminated surface-water systems not only in areas close to the site but also in many other parts of Europe. The initial contamination of water was due primarily to direct deposition of radionuclides on the surface of rivers and lakes, dominated by short-lived radionuclides (most importantly, ^{131}I). In the first few weeks after the accident, activity concentrations in drinking water from the Kyiv Reservoir were a particular concern.

The contamination of water bodies decreased rapidly during the weeks after fallout through dilution, physical decay and absorption of radioactivity to catchment soils. For lakes and reservoirs, settling of suspended particles to the bed sediments also played an important role in reducing levels in water. Bed sediments are an important long-term sink for radioactivity.

Initial uptake of radioiodine to fish was rapid, but activity concentrations declined quickly, due primarily to physical decay. Bioaccumulation of radiocaesium in the aquatic food chain led to significant concentrations in fish, in the most affected areas, and in some lakes as far away as Scandinavia and Germany. Because of generally lower fallout, and lower bioaccumulation, ^{90}Sr-activity concentrations in fish were not significant for human doses in comparison to radiocaesium, particularly since ^{90}Sr is accumulated in bone rather than in edible muscle.

In the long term, secondary contamination by washoff of long lived ^{137}Cs and ^{90}Sr from contaminated soils and remobilisation from bed sediments continues (at a much lower level) to the present day. Catchments with high organic content (peat soils) release much more radiocaesium to surface waters than those with mostly mineral soils. At the present time, surface-water-activity concentrations are low. Therefore, irrigation with surface water is not considered to be a problem.

Fuel particles deposited in the sediments of rivers and lakes close to the Chernobyl NPP show significantly lower weathering rates than the same particles in terrestrial soils. The half-life of these particles is roughly the same as the physical half time of the radionuclides ^{90}Sr and ^{137}Cs.

While ^{137}Cs- and ^{90}Sr-activity concentrations in water and fish of rivers, open lakes and reservoirs are currently low, the most contaminated lakes are those few lakes with limited inflowing and outflowing streams ("closed" lakes) in Ukraine, Belarus and Russia which have a poor mineral-nutrient status. Activity concentrations of ^{137}Cs in fish in some of these lakes will remain contaminated for a significant time into the future. In a population living next to a "closed" lake system (e.g., Kozhanovskoe Lake, Russia), consumption of fish has dominated the total ^{137}Cs ingestion for some people.

Owing to the large distance of the Black and Baltic Seas from Chernobyl, and the dilution in these systems, activity concentrations in sea water were much lower than in freshwater. The low water radionuclide concentrations combined with low bioaccumulation of radiocaesium in marine biota has led to activity concentrations in marine fish which were not of concern.

Recommendations for future research and monitoring

General recommendations

Various ecosystems considered in the present report have been intensively monitored and studied during the years after Chernobyl and transfers and bioaccumulation of the most important long term contaminants, ^{137}Cs and ^{90}Sr, are now generally well understood. There is, therefore, little urgent need for major new research programmes on radioactivity; there is, however, a requirement for continued but more limited target monitoring of the environments, and for further research in some specific areas, as detailed below.

Long-term monitoring of radionuclides (especially, ^{137}Cs and ^{90}Sr) in various environmental compartments is required to meet the following general practical and scientific needs:

Practical:

— To assess current and predict future levels of human exposure and contamination of foods in order to justify remedial actions and long-term countermeasures;

— To inform the general public in affected areas about the persistence of radioactive contamination in food products and its seasonal and annual variability in natural food products gathered by themselves (such as mushrooms, game, freshwater fish from closed lakes, berries, etc.) as well as give dietary advice and food preparation possibilities to reduce radionuclide intake in humans.

— To inform the general public in affected areas about changing radiological conditions in order to relieve public concerns.

Scientific:

— To determine parameters of long-term transfer of radionuclides in various ecosystems and different natural conditions in order to specify predictive models both for the Chernobyl-affected areas and for potential future radioactive releases;

— To determine mechanisms of radionuclide behaviour in less studied ecosystems (e.g., role of fungi in the forest) in order to clarify persistence of radionuclides and explore remediation possibilities with special attention to processes of importance for contribution to human and biota doses.

As activity concentrations in environmental compartments are now in quasi-equilibrium and change slowly, the number and frequency of sampling and measurements performed in the frame of monitoring and research programmes can be substantially reduced compared with the early years after the Chernobyl accident.

The deposition of ^{137}Cs and a number of other long-lived radionuclides in the 30-km zone should be used for radioecological studies in various ecosystems located in this highly contaminated area. Such studies are, except for very small scale experiments, not possible or difficult to perform otherwise.

Specific recommendations

Updated mapping of ^{137}Cs deposition in Albania, Bulgaria and Georgia should be performed in order to complete the study of post-Chernobyl contamination of Europe.

Improved mapping of ^{131}I deposition based both on historical environmental measurements carried out in 1986 and on recent measurements of ^{129}I in soil samples in areas where elevated thyroid-cancer incidence has been detected after the Chernobyl accident would reduce uncertainty of thyroid-dose reconstruction needed for determination of radiation risks.

Long-term monitoring of ^{137}Cs- and ^{90}Sr-activity concentrations in agricultural plant and animal products produced in areas with various soil and climate conditions and different agricultural practices should be performed for decades to come in the frame of limited target research programmes on selected sites with the aim as presented above.

Study of distribution of ^{137}Cs and plutonium radionuclides in the urban environment (Pripyat, Chernobyl and some other contaminated towns) in remote time periods would improve modelling of human external exposure and inhalation of radionuclides in case of a nuclear or radiological accident or malevolent action.

Continued long-term monitoring of specific forest products such as mushrooms, berries and game needs to be carried out in those areas in which forests were significantly contaminated and where the public consumes wild foods. Results from such monitoring are being used by the relevant authorities in affected countries to provide advice to the general public on their continued use of forests for recreation and gathering of wild foods.

In addition to the general monitoring of forest products required for radiation protection, more detailed, scientifically based, long-term monitoring of specific forest sites is required to provide an ongoing and improved understanding of the mechanisms, long-term dynamics and persistence of radiocaesium contamination and its variability. It is desirable to explore further the key organisms, for example fungi, and their role in radiocaesium mobility and long term behaviour in forest ecosystems. Such monitoring programmes are being carried out in the more severely affected countries such as Belarus and Russia, and it is important that these continue into the foreseeable future if current uncertainties on long-term forecasts are to be reduced.

Aquatic systems have been intensively monitored and studied during the years after the Chernobyl accident and transfers and bioaccumulation of the most important long term contaminants, ^{90}Sr and ^{137}Cs, are now well understood. There is, however, a requirement for continued (but perhaps more limited) monitoring of the aquatic environment, and for further research in some specific areas, as detailed below.

Although there is currently no need for major new research programmes on radioactivity in aquatic systems, predictions of future contamination of aquatic systems by ^{90}Sr and ^{137}Cs would be improved by continued monitoring of radioactivity in key systems (the Pripyat-Dnieper system, the seas, and selected rivers and lakes in the most affected areas and in Western Europe). This would continue the excellent time-series measurements of activity concentrations in water, sediments and fish and refine predictive models for these radionuclides.

Although they are currently of minor radiological importance in comparison to ^{90}Sr and ^{137}Cs, further studies of transuranic elements in the Chernobyl zone would improve predictions of environmental contamination in the very long term (100's-1000's of years). Further empirical studies of transuranic radionuclides and ^{99}Tc are unlikely to have direct implications for radiological protection in the Chernobyl-affected areas, but would further add to our knowledge of the environmental behaviour of these very long-lived radionuclides.

Future plans to reduce the water level of the Chernobyl Cooling Pond will have significant implications for its ecology and the behaviour of radionuclides/fuel particles in newly exposed sediments. Specific studies on the Cooling Pond should continue. In particular, further study of fuel-particle dissolution rates in aquatic systems such as the Cooling Pond would improve knowledge of their behaviour.

ENVIRONMENTAL COUNTERMEASURES AND REMEDIATION

The Chernobyl accident led to an extensive set of actions from the USSR authorities by introducing a range of short- and long-term environmental countermeasures to reduce the negative consequences. The countermeasures involved a great amount of human, economic and scientific resources. Unfortunately, there was not always openness and transparency for the public, and information was withheld. This can in part explain some of the problems experienced later on in communication with the public and the mistrust of the competent authorities. This sort of an information crisis also led, in many other countries outside Russia, Belarus and Ukraine, to a distrust in authority which in many countries has led to investigation on how to deal with such major accidents in an open and transparent way and how the affected people can be part of decision-making processes.

Unique experience of countermeasure application after the Chernobyl accident has already been widely used both at national and international levels in order to improve preparedness for future nuclear and radiological emergencies; however, more should be done in order to account comprehensively for both positive and negative Chernobyl lessons.

The environmental countermeasures addressed both the reduction of external exposure and exposure from ingestion as well as, in some cases, exposure due to inhalation. A major initial countermeasure was the evacuation of people from a zone around the reactor.

Conclusions

Radiological criteria

At the moment of the Chernobyl accident, well developed international and national guidance on general radiation protection of the public and specific guidance applicable to major nuclear emergencies was in place. The basic methodology of guidance used in the former USSR was different from the international system, but the values of radiation-safety standards were similar. The then available international and national standards were widely applied for protection of populations of European countries affected by the accident.

The unprecedented scale and long-term consequences of the Chernobyl accident required development of some additional radiation-safety standards, which had been developed following changes of radiation conditions, and promoted harmonisation of standards worldwide.

Urban countermeasures

Decontamination of settlements was widely applied in radiocontaminated regions of the USSR during the first years after the Chernobyl accident as a means of reducing external exposure of the public and against inhalation of resuspended radioactive substances.

Decontamination was cost-effective with regard to external dose reduction when its planning and implementation was preceded by a remediation assessment based on cost-benefit techniques and external dosimetry data.

Once the areas had been cleaned up, secondary contamination with radionuclides of cleaned up plots in the long term has not been observed.

The decontamination of urban environments has produced a considerable amount of low-level radioactive waste which in turn creates a problem of disposal.

Numerous experimental studies conducted after the Chernobyl accident and appropriate modelling have been used as a scientific basis for development of improved recommendations for decontamination of the urban environment in case of future large-scale radioactive contamination.

Agricultural countermeasures

Countermeasures applied in the early phase of the Chernobyl accident were only partially effective in reducing radioiodine intake via milk, because of the lack of timely information about the accident and recommended actions, particularly for private farmers. This led to significant radioiodine exposure of some people in the affected countries.

The most effective countermeasures in the early phase were exclusion of contaminated pasture grasses from animals' diet and rejection of milk (with further processing) based on radiation-monitoring data. Feeding animals with "clean" fodder was effectively performed in some affected countries; however, this countermeasure was not widely applied in the USSR due to a lack of uncontaminated feeds. The slaughtering of cattle was unjustified from a radiological point of view and had great hygienic, practical and economic problems.

Several months after the accident long-term countermeasures against radiocaesium and radiostrontium were effectively implemented in all the contaminated regions both in plant and animal growing, including feeding of animals with "clean" fodder and obligatory milk processing. This enabled most farming practices to continue in affected areas and resulted in a large dose reduction. The most important precondition was radiation monitoring of agricultural lands, feeds and foodstuff, including *in vivo* monitoring of caesium-activity concentrations in muscle of cattle.

The greatest long term problem has been radiocaesium contamination of milk and meat. In the USSR and later on in the independent countries, this has been addressed by the treatment of land used for fodder crops, clean feeding and application of caesium binders to animals. Clean feeding is one of the more important and effective measures used extensively in countries where animal products have ^{137}Cs-activity concentrations exceeding action levels. While in the long term environmental radiation conditions change slowly, the efficiency of environmental countermeasures is also assumed to remain stable.

Application of agricultural countermeasures in the three more affected countries has substantially decreased since the middle of the 1990s, because of economic problems. In a short time, this resulted in an increase of radionuclide content in plant and animal agricultural products.

There are still agricultural areas in the three countries which have been taken out of use. However this land can be used after appropriate remediation, for which at present technologies are available, but at the moment legal, economic, and social constraint may make this difficult.

When social and economic factors along with the radiological ones are taken into account during planning and application of countermeasures, better acceptability of them by the public has been achieved.

In Western Europe, because of the high and prolonged uptake of radiocaesium in the affected extensive systems, a range of countermeasures are still being used for animal products from uplands and forests.

For the first time in the world practical, long-term agricultural countermeasures were developed, tested and implemented on a large scale, including radical improvement of meadows, pre-slaughter clean feeding, application of caesium binders, and soil treatment and cultivation. Their implementation on more than 3 billion hectares of agricultural land made it possible to minimise the amount of products with radionuclide-activity concentrations above action levels in all three countries.

Forest countermeasures

The principal forest-related countermeasures applied after the Chernobyl accident involved the management based ones (restrictions of various activities normally carried out in forests) and technologically based ones.

Restrictions widely applied in the three more affected countries and partially in Scandinavia included the following actions that have reduced human exposure due to residence in radioactively contaminated forests and use of forest products:

— Restrictions on public and forest-worker access as a countermeasure against external exposure;

— Restricted harvesting of food products, such as game, berries and mushrooms, by the public that contributed to reduction of internal doses. In the three more affected countries mushrooms are readily consumed and, therefore, this restriction has been particularly important;

— Restricted collection of firewood by the public in order to prevent exposures in the home and garden, when the wood is burned and the ash is disposed of or used as a fertilizer;

— Alteration of hunting practices aiming to avoid consumption of meat with high seasonal levels of radiocaesium; and

— Fire prevention, especially in areas with large-scale radionuclide deposition, aiming to avoid secondary contamination of the environment.

However, the experience in the three more affected countries has shown that such restrictions can also result in significant negative social consequences, and advice from the authorities to the general public may be ignored as a result. This situation can be offset by the provision of suitable educational programmes targeted at the local scale to emphasise the relevance of suggested changes in use of some forest areas.

It is unlikely that any technologically based forest countermeasures, i.e., the use of machinery and/or chemical treatments to alter the distribution or transfer of radiocaesium in the forest, will be practicable on a large scale.

Aquatic countermeasures

Numerous countermeasures were put in place in the months and years after the accident to protect water systems from transfers of radioactivity from contaminated soils. In general, these measures were ineffective and expensive and led to relatively high exposures to workers implementing the countermeasures.

The most effective countermeasure was the early restriction of drinking-water abstraction and changing to alternative supplies. Restrictions on consumption of freshwater fish have also proved effective in Scandinavia and Germany, though in Belarus, Russia and Ukraine such restrictions may not always have been adhered to.

It is unlikely that any future countermeasures to protect surface waters will be justifiable in terms of economic cost per unit of dose reduction. It is expected that restrictions on consumption of fish will remain, in a few cases (in so-called closed lakes), for several more decades.

Future efforts in this area should be focused on public information, because there are still public misconceptions concerning perceived health risks due to radioactively contaminated waters and fish.

Recommendations

Countries affected by the Chernobyl accident

All kinds of long-term remediation measures and regular countermeasures in the areas contaminated with radionuclides should be applied if they are radiologically justified and optimised. In optimising countermeasures, social and economic factors should be taken into account, along with formal cost-benefit analysis, aiming to achieve acceptability of countermeasures by the public.

Members of the general public, along with the authorities, should be particularly informed on the existing radiation-risk factors and technological possibilities to reduce them in the long term via remediation and regular countermeasures, and involved in discussion and decision-making.

In the long term after the Chernobyl accident, remediation measures and regular countermeasures remain efficient and justified mainly in the agricultural areas with poor (sandy and peaty) soils, where high radionuclide transfer from soil to plants can occur.

Particular attention must be given to the production of subsistence farming of several hundred settlements and about 50 intensive farms in Belarus, Russia and Ukraine, where radionuclide concentrations in milk still exceed national action levels.

Among long-term remediation measures, radical improvement of pastures and grasslands as well as draining of wet peaty areas is of high efficiency. The most efficient regular agricultural countermeasures are pre-slaughter clean feeding of animals accompanied by *in-vivo* monitoring, application of Prussian Blue to cattle and enhanced application of mineral fertilisers in plant growing.

Restricting harvesting of wild food products, such as game, berries, mushrooms and fish from ‘closed lakes,’ by the public still may be needed in areas where their activity concentrations exceed national action levels.

Advice on diet aiming to reduce consumption of highly contaminated wild food products and on simple cooking procedures to remove radioactive caesium still is an important countermeasure in reducing internal exposure.

It is necessary to identify sustainable ways to make use of the most affected areas that reflect the radiation hazard, but also revive the economic potential for the benefit of the community.

Worldwide

The unique experience of countermeasure application after the Chernobyl accident should be carefully documented and used for preparation of international and national guidance for authorities and experts responsible for radiation protection of the public and the environment.

Practically all long-term agricultural countermeasures implemented in a large scale on contaminated lands of the three more affected countries can be recommended for use in case of future accidents. However, the effectiveness of soil-based countermeasures varies at each site. Therefore, analysis of soil properties and agricultural practice before the application of countermeasures is of great importance.

Recommendations for decontamination of the urban environment in case of large-scale radioactive contamination should be distributed to management of nuclear facilities having the potential of substantial accidental radioactive release (NPPs and reprocessing plants) and to authorities of adjacent regions.

Research

Generally, physical and chemical processes involved in environmental countermeasures and remediation technologies, both of mechanical nature (radionuclide removal, mixing with soil, etc.), chemical nature (soil liming, fertilisation, etc.) or their combinations, are understood well enough in order to be modeled and applied in similar circumstances worldwide. Much less understood are the biological processes that could be used in environmental remediation, e.g., re-profiling of agricultural production, bioremediation, etc. These processes require more research.

An important issue that requires more sociological research is perception by the public of the introduction, performance and withdrawal of countermeasures in case of emergency, as well as development of social measures aiming at involvement of the public in these processes at all stages beginning with the decision making.

There still is substantial diversity in international and national radiological criteria and safety standards applicable to remediation of affected areas in case of environmental contamination with radionuclides. Experience of radiological protection of the public after the Chernobyl accident has clearly shown the need in further international harmonisation of appropriate radiological criteria and safety standards.

HUMAN EXPOSURE

Following the Chernobyl accident, both the workers and the general public were affected by radiation that resulted or still can result in adverse health effects. In this document we consider primarily exposure patterns of the members of the general public exposed from radionuclides released to the environment. Information on doses received by members of general public, both evacuated from the accident area and those who permanently live in radiocontaminated areas, is required for the following health-related purposes:

- Substantiation of countermeasures and remediation programmes;
- Forecast of expected adverse health effects and justification of corresponding health-protection measures;
- Information of the public and of the authorities; and
- Epidemiological and other medical studies of radiation-caused adverse health effects.

The results of post-accidental environmental monitoring indicate that the more affected countries are Belarus, Ukraine, and Russia. Much of the information on doses from the Chernobyl accident have been focused on the three primarily contaminated countries.

There were four mechanisms of delivering dose to the public: external dose from cloud passage, internal dose from inhalation of the cloud and resuspended materials, external dose from radioactive materials deposited upon soil and other surfaces, and internal dose from the ingestion of food products and water. Except for unusual circumstances, the latter two pathways were the more important. External dose and internal dose tended to be approximately equally important, although this general conclusion was subject to large variation due to shielding afforded by types of buildings and the soil from which crops were grown.

Estimates of doses to individual members of population groups were based on literally millions of measurements of concentrations of radioactive materials in air, soil, foods, water, human thyroids, and the whole body contents of humans. In addition, there were many measurements of external gamma-exposure rate over undisturbed and disturbed fields, and external doses to humans were measured with the use of individual thermoluminescent dosimeters. Thus, the results of estimated doses are firmly based upon measurements and tend to be realistic rather than conservative.

As the major health effect of Chernobyl for the general public was an elevated thyroid-cancer incidence in children and adolescents, much attention was paid to dosimetry of the thyroid gland. The assessment of thyroid doses resulting from the intake of ^{131}I is based on the results of 350,000 human measurements and a few thousands of ^{131}I in milk measurements performed in Belarus, Russia and Ukraine within a few weeks following the accident.

Doses to humans were reduced significantly by a number of countermeasures. Official countermeasures included evacuations and relocations of persons, the blockage of contaminated food supplies, the removal of contaminated soil, the treatment of agricultural fields to reduce the uptake of radionuclides, the substitution of foods, and the prohibition of usage of “wild” foods. Unofficial countermeasures included the self-initiated avoidance of foods judged to be contaminated.

Conclusions

The collective effective dose (not including dose to the thyroid) received by about five million residents living in the areas of Belarus, Russia and Ukraine contaminated from the Chernobyl accident (^{137}Cs deposition on soil of >37 kBq m^{-2}) was approximately 40,000 man-Sv during the period of 1986–1995. The groups of exposed persons within each country received an approximately equal collective dose. The additional amount of collective effective dose projected to be received during 1996 to 2006 is about 9000 man Sv.

The collective dose to the thyroid was nearly 2 million man-Gy with nearly half received by persons exposed in Ukraine.

The main pathways leading to human exposure were external exposure from radionuclides deposited on the ground and by the ingestion of contaminated terrestrial food products. Inhalation and ingestion of drinking water, fish, and products contaminated with irrigation water were generally minor pathways.

The range in thyroid dose in different settlements and in all age-gender groups is large, between less than 0.1 Gy and more than 10 Gy. In some groups, and especially in younger children, doses were high enough to cause both short-term functional thyroid changes and remote thyroid-cancer effects in some individuals.

The internal thyroid dose from intake of ^{131}I was mainly due to the consumption of fresh cow’s milk and, to less extent, of green vegetables; children on average received a dose that was much greater than that received by adults, because of their small thyroid mass and a consumption rate of fresh cow’s milk that was similar to that of adults.

For populations permanently residing in contaminated areas and exposed predominantly via ingestion, the contribution of short-lived radioiodines, i.e., ^{132}I, ^{133}I, and ^{135}I, to thyroid dose was minor, i.e., about 1% of the ^{131}I-thyroid dose, as during transport of radioiodines along food chains short-lived radioiodines decayed. The highest relative contribution (20 to 50%) to the thyroid doses of the public from short-lived radionuclides was received by the residents of Pripyat, who were evacuated before they could consume contaminated food.

According to both measurement and modelling data, the urban population has been exposed to a lower external dose by a factor of 1.5–2 compared to the rural population living in areas with similar levels of radioactive contamination. This arises because of the better shielding features of urban buildings and different occupational habits. Also, as the urban population depends less on local agricultural products and wild food than the rural population, both effective and thyroid internal doses caused predominantly by ingestion are by a factor of two to three lower in urban than in rural populations

The initial high rates of exposure declined rapidly due to the decay of short-lived radionuclides and to the movement of radiocaesiums into the soil profile. The latter caused a decrease in the rate of external dose due to increased shielding. In addition, as caesium moves into the soil column it binds to soil particles, and this reduced the availability of caesium to plants.

The great majority of dose from the accident has already been accumulated.

Persons who received effective doses (not including dose to the thyroid) larger than average by a factor of two to three were those who lived in rural areas in single story homes and who ate large amounts of “wild” foods, such as game meats, mushrooms, and berries.

The long-term internal doses to residents of rural settlements strongly depend on soil properties. Contributions due to internal and external exposure are comparable in areas with light sandy soils, and the contribution of internal exposure to the total (external and internal) dose does not exceed 10% in areas with dominantly black soils. The contribution of ^{90}Sr to the internal dose regardless of natural conditions is usually less than 5%.

Long-term internal doses of children caused by ingestion of food containing caesium radionuclides are usually lower by a factor of about 1.1 to 1.5 than those in adults and adolescents.

Both accumulated and predicted mean doses in settlement residents vary in the range of two orders of magnitude depending on radioactive contamination of the area, dominating soil type and settlement type. In 1986-2000 the accumulated dose range comprised from 2 mSv in towns located in black soil areas up to 300 mSv in villages located in areas with podzol sandy soils. The doses expected in 2001-2056 are substantially lower than already received ones, i.e., in the range of 1 to 100 mSv.

If countermeasures had not been applied, the population of some more contaminated villages could have received life-time (70 years) effective doses up to 400 mSv. Intensive application of countermeasures, such as settlement decontamination and agricultural countermeasures, has substantially reduced dose. For comparison, a worldwide average life time dose from natural background radiation is about 170 mSv with the typical range of 70 to 700 mSv in various regions.

The vast majority of about five million population residing in contaminated areas of Belarus, Russia and Ukraine currently, i.e., in the early 2000s, receive annual effective dose less than 1 mSv (equal to national action levels in the three countries). For comparison, a worldwide average annual dose from natural background radiation is about 2.4 mSv with the typical range of 1 to 10 mSv in various regions.

The number of residents of the contaminated areas in the three more affected countries that currently receive more than 1 mSv annually can be broadly estimated to be about 100,000 persons. As future reduction of both external dose rate and radionuclide (mainly ^{137}Cs) activity concentrations in food is rather slow, reduction of human-exposure levels is expected to be slow as well, i.e., of about 3 to 5% per year for current countermeasures.

Based upon available information it does not appear that doses associated with “hot particles” have been significant.

The assessment of the experts working on the Chernobyl Forum agrees with that of the UNSCEAR (2000) experts in terms of the dose received by the populations of the three more affected countries: Belarus, Ukraine, and the Russian Federation.

Recommendations

Large-scale monitoring of foodstuffs, whole-body counting of individuals, and provision of thermoluminescent detectors to members of the general population are no longer necessary. Critical groups in areas of high contamination and/or high transfer of radiocaesium to foods are known. Representative members of these critical groups should be monitored by dosimeters for external dose and by whole body counting for internal dose.

Sentinel individuals in more highly contaminated areas not scheduled for further remediation might be identified with the goal of continued periodic whole body counts and monitoring for external dose. The goal would be to follow the expected continued decrease in external and internal dose and to determine whether such decreases are due to radioactive decay alone or to further ecological elimination.

RADIATION-INDUCED EFFECTS ON PLANTS AND ANIMALS

The biological effects of radiation on plants and animals have long been of interest to scientists; in fact, much of the information on effects on humans has evolved from experimental studies on plants and animals. Additional effects research followed the development of nuclear energy and concerns about the possible impacts of radioactive releases into the terrestrial and aquatic environments. By the mid 1970s sufficient information had been accrued on the effects of ionising radiation on plants and animals, including the data from accidental releases of radionuclides from nuclear weapon production and test sites.

However, the Chernobyl nuclear accident that happened in Spring 1986 not in a desert or ocean but at the territory with temperate climate and flourishing flora and fauna resulted in numerous adverse effects on non-human biota species. Both the acute radiation effects (radiation death of plants and animals, loss of reproduction, etc.) and long-term effects (change of biodiversity, cytogenetic anomalies, etc.) have been revealed in the affected areas. By natural reasons, biota located in the area nearest to the radioactive release source, the so-called 30-km zone or Chernobyl Exclusion Zone that was rapidly cleared from people because of radiation danger, was most affected. As a result, in this area population and ecosystem effects on biota caused, on the one hand by high radiation levels, and on the other hand by plant succession and animal migration due to intraspecific and interspecific competition, have occurred.

The plant and animal conditions in the 30-km zone around Chernobyl NPP were changing rapidly during the first months and years after the accident and later on arrived to a quasi-stationery equilibrium. Presently, traces of adverse radiation effects on biota can be hardly found in the near vicinity of the radiation source (a few kilometers from the damaged reactor), and on the rest of the territory both wild plants and animals are flourishing because of removal of the major natural stressor, the human being.

Conclusions

Irradiation from radionuclides released from the Chernobyl accident caused numerous acute adverse effects in the biota located in areas of highest exposure, i.e., up to a distance of few tens of kilometres from the release point. Beyond the exclusion zone, no acute radiation-induced effects on biota have been reported.

The environmental response to the Chernobyl accident was a complex interaction among radiation dose, dose rate and its temporal and spatial variations, as well as the radiosensitivities of the different taxons. Both individual and population effects caused by radiation-induced cell death have been observed in plants and animals as follows:

— Increased mortality of coniferous plants, soil invertebrates and mammals;

— Reproductive losses in plants and animals; and

— Chronic radiation syndrome of animals (mammals, birds, etc.).

No adverse radiation-induced effect has been reported in plants and animals exposed to a cumulative dose of less than 0.3 Gy during the first month after the radionuclide fallout.

Following the natural reduction of exposure levels due to radionuclide decay and migration, populations have been recovering from acute radiation effects. By the next growing season population viability of plants and animals substantially recovered as a result of the combined effects of reproduction and immigration. A few years were needed for recovery from major radiation-induced adverse effects in plants and animals.

The acute radiobiological effects observed in the Chernobyl accident area are consistent with radiobiological data obtained in experimental studies or observed in natural conditions in other areas affected by ionising radiation. Thus, rapidly developing cell systems, such as meristems of plants and insect larva, were predominantly affected by radiation. At the organism level, the young plants and animals have been found to be the most sensitive to acute effects of radiation.

Genetic effects of radiation, in both somatic and germ cells, have been observed in plants and animals of the exclusion zone during the first few years after the Chernobyl accident. Both in the exclusion zone, and beyond, different cytogenetic anomalies attributable to radiation continue to be reported from experimental studies performed on plants and animals. Whether the observed cytogenetic anomalies have any detrimental biological significance is not known.

The recovery of affected biota in the exclusion zone has been confounded by the overriding response to the removal of human activities, e.g., termination of agricultural and industrial activities accompanied with the environmental pollution in the most affected area. As a result, populations of many plants and animals have eventually expanded, and the present environmental conditions have had positive impact on the biota in the exclusion zone.

Recommendations for future research

In order to develop further the system of environmental protection against radiation, the long-term impact of radiation on plant and animal populations should be further investigated in the exclusion zone of the Chernobyl accident; this is a globally unique area for radioecological and radiobiological research in an otherwise natural setting.

In particular, multigenerational studies of the recent radiobiological phenomena of genome instability and of the radiation effect on the genetic structure of plant and animal populations might bring fundamentally new scientific information.

There is a need to develop standardised methods for biota-dose reconstruction, e.g., in the form of a unified dosimetric protocol.

Recommendations for countermeasures and remediation

Protective actions for farm animals in case of a nuclear or radiological emergency should be developed based on modern radiobiological data, including the experience gained in the Chernobyl Exclusion Zone, and internationally harmonised.

It is unlikely that any technologically based remediation actions aiming to improve the radiological conditions for plants and animals in the Exclusion Zone of the Chernobyl NPP would not have adverse impacts to biota.

ENVIRONMENTAL ASPECTS OF DISMANTLEMENT OF THE SHELTER AND RADIOACTIVE WASTE MANAGEMENT

Conclusions

The accidental destruction of the Unit 4 reactor at the Chernobyl NPP resulted in the generation of extensive radioactive contamination and large amounts of radioactive waste in the Unit, the ChNPP site and surrounding area (Exclusion Zone). Construction of the Shelter between May and November 1986, aimed at environmental containment of the damaged reactor, reduced radiation levels on-site and prevented further release of radionuclides off-site.

While the Shelter was erected in an extremely short period of time under conditions of severe radiation exposure to personnel, measures taken to save time and cost during the construction, and the high dose rates inside the structure, led to imperfection of the newly constructed Shelter as well as to lack of comprehensive data on the stability of the damaged Unit 4 structures. In addition to uncertainties in stability at the time of its construction, structural elements of the Shelter have degraded as a result of moisture-induced corrosion during the nearly two decades that have passed since the Shelter was erected. The main potential hazard of the Shelter is a possible collapse of its top structures and release of radioactive dust into the environment.

In order to avoid potential collapse of the Shelter in the future, measures are planned to strengthen unstable structures of the Shelter. In addition a New Safe Confinement (NSC) with more than 100-years service life is planned to be built as a cover over the existing Shelter as a longer-term solution. The construction of the NSC is expected to allow for the dismantlement of the current Shelter, removal of highly radioactive Fuel Containing Mass (FCM) from Unit 4, and eventual decommissioning of the damaged reactor.

In the course of remediation activities both at the Chernobyl NPP site and in its vicinity large volumes of radioactive waste were generated after the accident as a result of the clean up of contaminated areas and placed in temporary near surface waste-storage and disposal facilities. Facilities of trench and landfill type were created from 1986 to 1987 in the Exclusion Zone at distances 0.5 to 15 km from the NPP site with the intention to avoid dust spread, reduce the radiation levels, and enable better working conditions at Unit 4 and in its surrounding. These facilities were established without proper design documentation, engineered barriers, or hydrogeological investigations and do not meet contemporary waste-safety requirements.

During the years following the accident large resources were expanded to provide a systematic analysis and an acceptable strategy for management of existing radioactive waste. However, to date a broadly accepted strategy for radioactive waste management at the ChNPP site and the Exclusion Zone, and especially for high-level and long-lived waste, has not been developed yet. Some of the reasons are the large number and areas covered by the radioactive waste-storage and disposal facilities of which only half are well studied and inventoried. This results in large uncertainties of radioactive waste inventories.

More radioactive waste is potentially expected in the years to come to be generated during NSC construction, possible Shelter dismantling, FCM removal and decommissioning of Unit 4. This waste belonging to different categories should be properly disposed of.

According to the Ukrainian National Programme on radioactive waste management, there are different options for different waste categories. The planned options for low radioactivity waste are to sort the waste according to its physical characteristics (e.g., soil, concrete, metal) and possibly decontaminate and/or condition it for beneficial reuse (reuse of soil for NSC foundations, melting of metal pieces), or send it for disposal.

The long-lived waste is planned to be placed into interim storage. Different storage options are being considered, and a decision has not yet been made. After construction of the NSC and decommissioning of the Shelter facilities, it is envisaged that Shelter dismantlement and further removal of FCM will occur. High-level radioactive waste is planned to be partially processed in place and then stored at a temporary storage site until a deep geological disposal site is ready for final disposal.

Such a strategic approach is foreseen by the Comprehensive Programme on radioactive waste management that was approved by the Ukrainian Government in 1996 and confirmed in 2004. According to this concept, it is considered reasonable to begin a specific investigation for exploring the most appropriate geological site in this area in 2006. Following such planning, the construction of a deep geological disposal facility might be completed before 2035-2040.

The future development of the Exclusion Zone as an industrial site or natural reserve depends on the future strategy for conversion of Unit 4 into an ecologically safe system, i.e., the development of a New Safe Confinement, the dismantlement of the current Shelter, removal of fuel-containing material, and eventual decommissioning of the Unit 4 reactor site. Currently Units 1, 2, and 3 (1000 MW RBMK reactors) are shutdown with a view to be decommissioned, and two additional reactors (Units 5 and 6) that had been near completion were abandoned in 1986 following the accident.

There are uncertainties related to the current radioactive material inventory at the Shelter, and also at the waste-storage and disposal sites within the Exclusion Zone. This situation impacts not only the safety assessments and environmental analyses, but also the design of remediation actions and criteria for new facilities.

Recommendations for future actions

Recognising the ongoing effort on improving safety and addressing the aforementioned uncertainties in the existing input data, the following main recommendations are made regarding the dismantlement of the Shelter and management of radioactive waste generated as a result of the accident:

Because individual safety and environmental assessments have been performed only for individual facilities at and around the ChNPP, a comprehensive safety and environmental impact assessment according to the international standards and recommendations that encompasses all activities inside the entire Exclusion Zone should be performed.

During the preparation and construction of the NSC and soil removal special monitoring wells are expected to be destroyed. Therefore, taking into account the changing hydrogeological conditions at the Shelter and potential exposure pathways (e.g., through groundwater), it is important to maintain and improve environmental monitoring strategies, methods, equipment and staff qualification needed for the adequate performance of monitoring the conditions at the ChNPP site and the Exclusion Zone.

Dismantlement of the Shelter after a long period of time (about 50 years) does not seem to be a viable option, due to the need for long-term maintenance of structure stability and integrity, resources and knowledge. The long-term strategy raises concerns related to the potential loss of the most experienced personnel at the Chernobyl NPP and maintaining stable manpower-loading necessary for the safe operation of the NSC. It is reasonable to begin retrieving FCM soon after dismantling unstable structures of the Shelter rather than waiting for the availability of a geological disposal facility. However, in optimization of the Shelter dismantlement, special attention should be paid to doses that will be received by the workers during different time periods.

Development of an integrated radioactive waste-management programme for the Shelter, the ChNPP site and the Exclusion Zone is needed to assure application of consistent management approaches, and sufficient facility capacity for all waste types. Specific emphasis needs to be paid to the characterisation and classification of waste (in particular waste with transuranic elements) from all remediation and decommissioning activities, as well as the establishment of a sufficient infrastructure for safe long-term management of long-lived and high-level waste. Therefore development of a necessary waste-management infrastructure is needed in order to assure sufficient waste capacity, which at present limits the rate and continuity of remediation activities at the Chernobyl NPP site and in the Exclusion Zone.

A coherent and comprehensive strategy for rehabilitation of the Exclusion Zone is needed with particular focus on improving safety of the existing waste-storage and disposal facilities. This will require development of a prioritisation method for remediation of the sites, based on safety-assessment results, aiming at decisions on which sites from which waste will be retrieved and disposed, and at which sites the waste will be allowed to decay *in situ*.

1. INTRODUCTION

1.1. Purpose

The purpose of this report is to provide an up-to-date evaluation of the environmental effects of the 26 April 1986 accident at the Chernobyl Nuclear Power Plant. Even though it is now nearly 20 years after the accident and substantial monies have been spent on such evaluations, there are still many conflicting reports and rumours. This joint report has been developed with the full cooperation of the United Nations (UN) family of organisations and with political representatives from the three more affected countries: Ukraine, Belarus, and the Russian Federation. In addition, recognised scientific experts from the three countries and additional international experts provided the basis for the preparation of reports for review by the members of the Chernobyl Forum.

1.2. Background

The “Chernobyl Forum” is a high-level political forum whose suggestion for existence was initiated by the International Atomic Energy Agency (IAEA) in cooperation with the Food and Agriculture Organisation (FAO), the UN Office for the Coordination of Humanitarian Affairs (OCHA), the United Nations Development Programme (UNDP), the United Nations Environment Program (UNEP), the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), the World Health Organisation (WHO), and the World Bank, as well as the competent authorities of Belarus, the Russian Federation, and Ukraine. The organisational meeting of the Chernobyl Forum was held on 3–5 February 2003, at which time the decision was reached to establish the Forum as an ongoing entity of the above named organisations.

The background for the establishment of the Forum dates back to the year 2000, when the UNSCEAR published its 2000 report to the UN General Assembly (UNSCEAR 2000). In essence this report stated that apart from the very early deaths due to extreme overexposure, the only clearly indicated health effect on the population that could be attributed to radiation exposure was an increased rate in the diagnosis of thyroid cancer among persons who were young children at time of exposure. The political representatives of the Russian Federation, Belarus, and Ukraine had strong reservations regarding the report. These reservations appear to have had two bases: (1) the statement on health effects was widely divergent from what was being reported in the popular press and even by some other members of the UN family, and (2) the political representatives felt that the views of scientists from the three affected countries had not been considered by members of the UNSCEAR.

Subsequently, during his visit to Belarus and meetings with the Belarusian authorities and scientific community, the Director General of the IAEA, Mr Elbaradei, noticed that ‘a lack of trust still prevails among the people of the region, however, due in part to the contradictory data and reports - on the precise environmental and health impacts of the accident - among national authorities, as well as among the relevant international organizations.’ This was in general agreement with the communicated views of the political authorities of the three countries that they still desired some stronger opportunity for an exchange of views and an opportunity to discuss trans scientific issues, such as optimisation of activities related to remediation of contaminated land and provision of health care to those affected by the accident. During visits with these representatives the Director General of the IAEA indicated his support for the concept of the Chernobyl Forum as a joint activity of the UN family and the three affected countries.

1.3. Objectives

Thus, the organisational meeting of the Forum was held with the following conclusions:

— To establish the Chernobyl Forum as a series of managerial, expert and public meetings in order to generate authoritative consensual statements on the health effects attributable to radiation exposure arising from the accident and the environmental consequences induced by the released radioactive materials, to provide advice on remediation and special health care programmes, and to suggest areas where further research is required; and

— To accept the Terms of Reference (TOR) of the Chernobyl Forum.

The objectives of the TOR were as follows:

— To explore and refine the current scientific assessments on the long-term health and environmental consequences of the Chernobyl accident, with a view to producing authoritative consensus statements focusing on:

- the health effects attributable to radiation exposure caused by the accident,
- the environmental consequences induced by the radioactive materials released due to the accident, e.g., contamination of foodstuffs, and additionally
- to address the consequences attributable to the accident although not directly related to the radiation exposure or radioactive contamination;

— To identify gaps in scientific research relevant to the radiation-induced or radioactive contamination induced health and environmental impacts of the accident, and suggest areas where further work is required based on an assessment of the work done in the past, and bearing in mind the ongoing work and projects;

— To provide advice on, and to facilitate implementation of scientifically sound programmes on mitigation of the accident consequences, including possible joint actions of the organizations participating in the Forum, such as:

- remediation of contaminated land with the aim of making it suitable for normal agricultural, economic and social life under safe conditions,
- special health care of the affected population,
- monitoring of the long-term human exposure to radiation,
- addressing the environmental issues pertaining to the decommissioning of the Shelter and management of radioactive waste originating from the Chernobyl accident.

1.4. Method of operation

The Chernobyl Forum itself continued as a high level organisation of senior officials from UN agencies and the three affected countries. The actual work has been accomplished by two expert groups: Expert Group Environment (EGE) and Expert Group Health (EGH). Members of each of these two groups consisted of recognised international scientists, including those from the three affected countries. Within these two groups and their sub-working groups the draft documents to support this report were prepared for consideration by the members of the Forum. The EGE was coordinated by the IAEA and the EGH was coordinated by the WHO.

The method of preparation of the draft documents was to assemble a group of experts on one or more topics and to break up into one or more sub-working groups to consider in detail the data available in the literature and unpublished data from scientists from the three affected countries. At the end of each meeting one or more draft documents were prepared and/or redrafted for further consideration and editing at future meetings.

In all cases the scientists from the UN organisations, the international community, and the three affected countries have been able to reach consensus in the preparation of their respective draft documents. After approval by the members of the Forum itself and final editing of the draft documents, this report is the result of that process in the environmental area. The Forum's report considering health effects is in process of publication under WHO responsibility (WHO 2005).

1.5. Structure of the Report

The report includes six sections. Following the Introduction, Section 2 describes processes and patterns of radioactive contamination of the urban, agricultural, forest and aquatic environments affected by deposition of the Chernobyl radioactive release. Section 3 identifies the major environmental countermeasures and remediation applied to the aforementioned four environments in order to mitigate accident consequences and specifically to reduce human exposure. Section 4 deals with an assessment of human exposure to radiation within the affected areas and in all of Europe based on data on environmental radioactive contamination and countermeasures presented in Sections 2 and 3. Section 5 presents an overview of experimental data on radiation-induced effects on plants and animals observed predominantly in the near zone of radioactive contamination. Finally, Section 6 discusses environmental aspects of dismantlement of the Shelter facility and radioactive waste management in the Chernobyl Exclusion Zone.

Each section is completed with relevant conclusions and recommendations for future environmental remediation actions, monitoring and research. The entire report is preceded with its Executive Summary.

SECTION 1 REFERENCES

UNITED NATIONS, Sources and Effects of Ionizing Radiation (2000 Report to the General Assembly, with Scientific Annexes), Scientific Committee on the Effects of Atomic Radiation, UN, New York (2000) Volume II 451–566.

WORLD HEALTH ORGANIZATION (2005) Health Effects of the Chernobyl Accident and Special Health Care Programmes, Report of the Chernobyl Forum Expert Group “Health” (EGH), WHO, Geneva.

2. RADIOACTIVE CONTAMINATION OF THE ENVIRONMENT

The accident at the Chernobyl Nuclear Power Plant (ChNNP) ranks among the larger technogenic accidents, due to the large release of radionuclides to the atmosphere and subsequent contamination of the environment. A number of European countries were subjected to different levels of radioactive contamination and among the more affected were three former USSR republics, now Belarus, Russia and Ukraine, in the vicinity of whose common border the Chernobyl NPP is located. The deposited radionuclides gradually decayed and moved within the environments – atmospheric, aquatic, terrestrial and urban – and among the environments. Those processes that determined patterns and regularities of radioactive contamination in those environments are presented in this section.

The focus in this Section is mainly on radioactive contamination of the off-site environment. Significant attention is given to the Chernobyl NPP site, the 30-km Exclusion Zone, and the Shelter constructed around it in 1986 in Section 6 of this report.

2.1. Radionuclide release and deposition

2.1.1. Radionuclide source term

The accident at Reactor Number 4 of the Chernobyl Nuclear Power Plant took place shortly after midnight on 26 April 26 1986. Prior to the accident, the reactor had been operated for many hours in non-design configurations in preparation for an experiment on recovering the energy in the turbine in the event of an unplanned shutdown. The cause of the accident was rather complicated, but can be considered as a runaway surge in the power level that caused the water coolant to vaporize inside the reactor. This, in turn, caused a further increase in the power level with a resulting steam explosion that tore the reactor apart. After the initial explosion the graphite in the reactor caught fire. Despite heroic efforts to control this fire, the graphite burned for many days and releases of radioactive materials continued until May 6. The reconstructed time course of the release of radioactive materials is shown in Fig. 2.1 (INSAG 1986, Izrael et al. 1990, Izrael 2002).

The occurrence of the accident was not immediately announced by authorities of the then Soviet Union. However, the releases were so large that the presence of fresh fission products was soon detected in Scandinavian countries, and retrospective calculations of possible trajectories indicated that the accident had occurred in the former Soviet Union. Further details of the accident and its immediate consequences are available from reports by the International Nuclear Safety Advisory Group (INSAG 1986), the International Advisory Committee (IAC 1991) and the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR 1988, 2000).

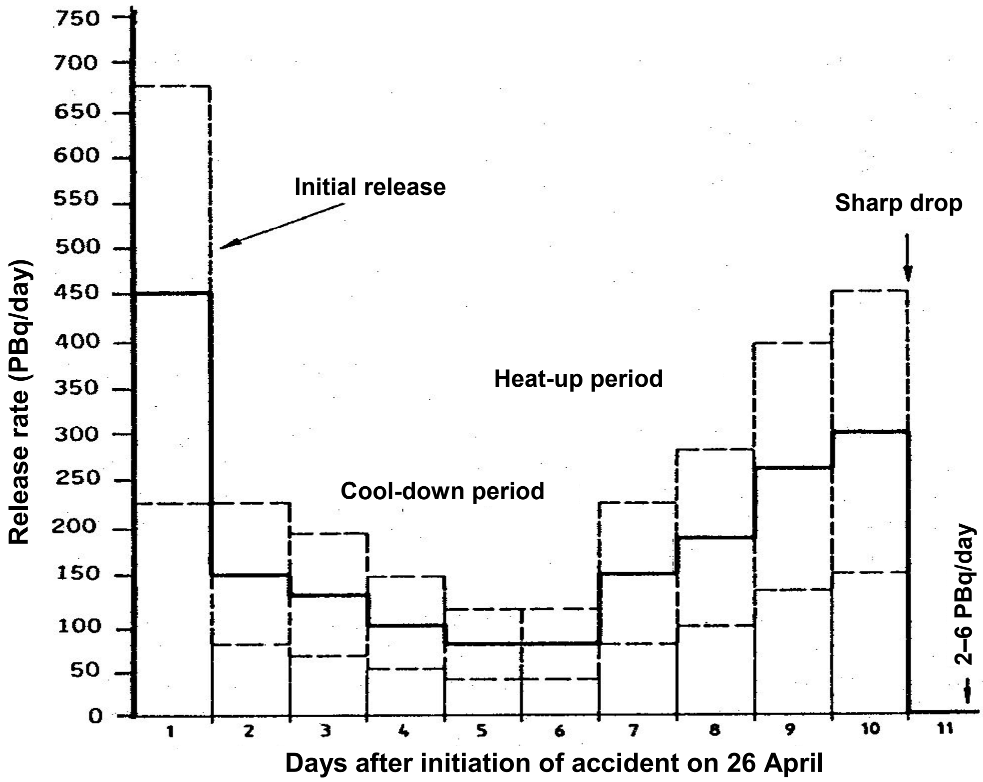


FIG. 2.1. Daily release rate to the atmosphere of radioactive materials, excluding noble gases, during the Chernobyl accident. The values are decay-corrected to 6 May 1986 and are uncertain by ±50% (INSAG 1986).

One of the earlier responses to the accident was airborne radiometric measurement of the contaminated parts of the former Soviet Union. An early estimate, based upon these data, of the amount of ^{137}Cs released by the accident and deposited in the former Soviet Union was about forty PBq (one million curies). Estimates of the releases have been refined over the years and the current estimate of the total deposited in the former Soviet Union is about twice the earlier estimate, i.e., eighty PBq. Current estimates of the amounts of the more important radionuclides released are shown in Table 2.1. Many of the radionuclides for which there were large releases have short physical half lives, and many radionuclides with long half lives were released in small amounts. During the early period after the accident the radionuclide of most concern was ^{131}I; later, most concern has been on ^{137}Cs.

Most of the radionuclides released by the accident have already decayed. Interest over the next few decades will continue to be on ^{137}Cs, with secondary attention on ^{90}Sr; the latter remains more important in the near zone of the Chernobyl NPP. Over the longer term (100s to 1000s of years) the only radionuclides anticipated to be of continuing interest are plutonium isotopes. The only radionuclide expected to increase in the coming years is ^{241}Am which arises from the decay of ^{241}Pu, and it takes about 100 years for the maximum amount of ^{241}Am to form from ^{241}Pu.

TABLE 2.1. REVISED ESTIMATES OF THE PRINCIPAL RADIONUCLIDES RELEASED DURING THE COURSE OF THE CHERNOBYL ACCIDENT*, DECAY-CORRECTED TO 26 APRIL 1986

Radionuclide	Half life	Activity released, PBq
	Inert gases	
^{85}Kr	10.72 a	33
^{133}Xe	5.25 d	6,500
	Volatile elements	
^{129m}Te	33.6 d	240
^{132}Te	3.26 d	~1,150
^{131}I	8.04 d	~1,760
^{133}I	20.8 h	2,500
^{134}Cs	2.06 a	~47**
^{136}Cs	13.1 d	36
^{137}Cs	30.0 a	~85
	Elements with intermediate volatility	
^{89}Sr	50.5 d	~115
^{90}Sr	29.12 a	~10
^{103}Ru	39.3 d	>168
^{106}Ru	368 d	>73
^{140}Ba	12.7 d	240
	Refractory elements (including fuel particles)***	
^{95}Zr	64.0 d	84
^{99}Mo	2.75 d	> 72
^{141}Ce	32.5 d	84
^{144}Ce	284 d	~ 50
^{239}Np	2.35 d	400
^{238}Pu	87.74 a	0.015
^{239}Pu	24,065 a	0.013
^{240}Pu	6,537 a	0.018
^{241}Pu	14.4 a	~2.6
^{242}Pu	376,000 a	0.00004
^{242}Cm	18.1 a	~0.4

*Most of the data are from UNSCEAR (2000) or Dreicer et al. (1996).
** Based on ^{134}Cs/^{137}Cs ratio 0.55 as of 26 April 1986 (Mück et al. 2002).
*** Based on fuel particle release of 1.5% (Kashparov et al. 2003).

2.1.2. Physical and chemical forms of released materials; hot particles

Radionuclides in the releases from the stricken reactor were in the form of gases, condensed particles and fuel particles. The presence of the latter was an important characteristic of the accident. The oxidation of nuclear fuel was the basic mechanism of fuel-particle formation. Less oxidized fuel particles were formed as a result of the initial explosion and were released primarily toward the western direction. More oxidized and soluble particles prevailed in the remaining fallout which was deposited in many other areas.

During oxidation and dispersal of the nuclear fuel, volatilisation of some radionuclides took place. After the initial cloud cooled, the more volatile of the released radionuclides remained in the gas phase, whilst the less volatile condensed on particles of construction materials, soot and dust. Thus, the chemical and physical forms of radionuclides in the Chernobyl release were determined by the volatility of their compounds and the conditions inside the reactor. Radioactive compounds with relatively high vapour pressure (primarily isotopes of inert gases and iodine in different chemical forms) were transported in the atmosphere in the gas phase. Isotopes of refractory elements (e.g., cerium, zirconium, niobium, and plutonium) were released into the atmosphere primarily in the form of fuel particles. Other radionuclides (isotopes of caesium, tellurium, antimony, etc.) were found in both fuel and condensed particles. The relative contributions of condensed and fuel components in deposition at a given site can be estimated from the activity ratios of radionuclides of different volatility classes.

Fuel particles made up the most important part of the fallout in the vicinity of the release source. Radionuclides such as ^{95}Zr, ^{95}Nb, ^{99}Mo, ^{141,144}Ce, ^{154,155}Eu, ^{237,239}Np, $^{238\text{-}242}$Pu, ^{241,243}Am and ^{242,244}Cm were released in a matrix of fuel particles only. More than 90% of ^{89,90}Sr and ^{103,106}Ru activities were also released in fuel particles. The release fraction of ^{90}Sr, ^{154}Eu, ^{238}Pu, $^{239+240}$Pu and ^{241}Am, and, therefore, of the nuclear fuel itself, deposited outside the ChNPP industrial site has been recently estimated to be only 1.5±0.5 % (Kashparov et al. 2003), which is half that of earlier estimates (INSAG 1986).

The chemical and radionuclide composition of fuel particles was close to that of irradiated nuclear fuel, but with a lower fraction of volatile radionuclides, a higher oxidation state of uranium, and the presence of various admixtures, especially in the surface layer. In contrast, the chemical and radionuclide composition of condensed particles varied widely. The specific activity of radionuclides in these particles was determined by the duration of the condensation process and the process temperature, as well as particle characteristics. The radionuclide content of some of the particles was dominated by just one or two nuclides, e.g., ^{103,106}Ru or ^{140}Ba/^{140}La (Sandalls et al. 1993).

The form of a radionuclide in the release determined the distance of its atmospheric transport. Even the smallest fuel particles consisting of a single grain of nuclear fuel crystallite had a relatively large size (up to 10 μm) and high density (8-10 g cm^{-3}). Because of their size, they were transported only a few tens of kilometers. Larger aggregates of particles were found only within distances of several kilometers from the power plant. For this reason, the deposition of refractory radionuclides strongly decreased with distance from the damaged reactor and only traces of refractory elements could be found outside the industrial site of the power plant. In contrast, significant deposition of gaseous radionuclides and sub-micron condensed particles took place at thousands of kilometers from Chernobyl. Ruthenium particles, for example, were found throughout Europe (Pollanen et al. 1997). At distances of hundreds of kilometers from Chernobyl the deposition of ^{137}Cs was as high as 1 MBq m^{-2} (Izrael 1998; De Cort et al. 1998).

Another important characteristic of fallout is solubility in aqueous solutions. This determines the mobility and bioavailability of deposited radionuclides in soils and surface waters during the initial period after deposition. In fallout sampled at the Chernobyl meteorological station from 26 April to 5 May 1986 with a 24-hour sampling period, the water-soluble and exchangeable (extractable with 1M CH_3COONH_4) forms of ^{137}Cs varied from 5 to more than 30% (Bobovnikova et al. 1991). The water-soluble and exchangeable forms of ^{90}Sr in deposition on 26 April accounted for only about 1% and this value increased to 5–10% in subsequent days.

The low solubility of deposited ^{137}Cs and ^{90}Sr near the NPP indicates that fuel particles were the major part of fallout, even 20 km from the source. At smaller distances, the portion of water-soluble and exchangeable forms of ^{137}Cs and ^{90}Sr was, obviously, lower due to the presence of larger particles; at farther distances the fraction of soluble condensed particles increased. As one example, almost all ^{137}Cs deposited in 1986 in the United Kingdom was water-soluble and exchangeable (Hilton et al. 1992).

2.1.3. Meteorological conditions during the course of the accident

At the moment of the accident the weather in most of Europe was dominated by a vast anticyclone. At the 700–800 m and 1,500 m altitudes, the area of the Chernobyl Nuclear Power Plant was at the southwest periphery of the high atmospheric pressure zone with air masses moving northwest with a velocity of 5–10 m s^{-1} (Izrael 1998).

At daybreak, the altitude of the air-mixing layer was about 2,500 m. This resulted in rapid mixing of the airborne debris throughout the mixing layer and dispersion of the cloud at different layers of the mixing height. Further dissemination of the particles originating from the time of the accident within the 700–1,500 m layer occurred as the air mass moved towards the northeast with a subsequent turn to the north; this plume was detected in Scandinavian countries.

Ground-level air on 26 April was transported to the west and northwest and reached Poland and Scandinavian countries by 27–29 April. In southern and western Ukraine, Moldavia, Romania, Slovakia, and Poland the weather was influenced by a low-gradient pressure field. In the following days the cyclone moved slowly southeast and the low-gradient pressure field with several poorly defined pressure areas dispersed over the major part of the European sector of the former Soviet Union. One of the pressure areas was a small near-surface cyclone located south of Gomel on the morning of 27 April.

Later, the releases from the reactor were carried predominantly in southwest and southern directions until 7–8 May. During the first five days after the accident commenced, the wind pattern had changed through all directions of the compass (Izrael 1998).

Within a few days after the accident, measurements of radiation levels in air over Europe, Japan and the U.S.A. showed the presence of radionuclides at altitudes of up to 7,000 m. The force of the explosion, rapid mixing of air layers due to thunderstorms near the CNPP, and the presence of warm frontal air masses between the ChNPP and the Baltic Sea all contributed to the transport of radionuclides to such heights.

To understand the complex meteorological situation better, Borsilov and Klepikova (1993) carried out calculations with assumed input pulses of unit activity at various times of the accident. The height of the source was selected to be 1000 m until 14:00 (GMT) on 28 April and later 500 m. The results of calculations are presented in Fig. 2.2 for six time periods (GMT time) with differing long-range transport conditions as follows:

(1) From the start of the accident to 12:00 (GMT) on 26 April—toward Belarus, Lithuania, Kaliningrad Oblast (of Russia), Sweden and Finland;

(2) From 12:00 on 26 April to 12:00 27 April—to Polesye then Poland and then south-west;

(3) From 12:00 on 27 April to 29 April—to Gomel (Belarus) Oblast, Bryansk (Russia) Oblast, and to the east;

(4) 29 April to 30 April—to the Sumy and Poltava Oblasts (Ukraine) and toward Romania;

(5) 1–3 May—to southern Ukraine and across the Black Sea to Turkey; and

(6) 4–5 May—to western Ukraine and Romania, and then a turn to Belarus.

Atmospheric precipitation plays an important role in determining whether an area might receive heavy contamination or not, as the processes of rainout (entrainment in a storm system) and washout (rain falling through a contaminated air mass) are important mechanisms in bringing released materials to the ground. In particular, significant heterogeneity of the deposition of radioactive material is related to the presence or absence of precipitation during passage of the cloud. Also, there are differences in behaviour regarding how effectively different radionuclides, or chemical forms of the same radionuclide, are rained or washed out.

There were many precipitation events during the course of the accident, and these events produced some areas of high ground deposition at distances rather far from the reactor. An example of the complex precipitation situation during the accident is shown in Fig. 2.3, which is a map of average daily precipitation intensity on 29 April for the parts of Belarus, Ukraine, and Russia most heavily affected by the accident.

In the case of dry deposition, the contamination levels were lower, but the radionuclide mixture intercepted by vegetation was substantially enriched with radioiodine isotopes; in the case of wet deposition the radionuclide content in fallout was similar to that in the radioactive cloud. As a result, both the levels and radionuclide ratios in areas with various deposition types were different.

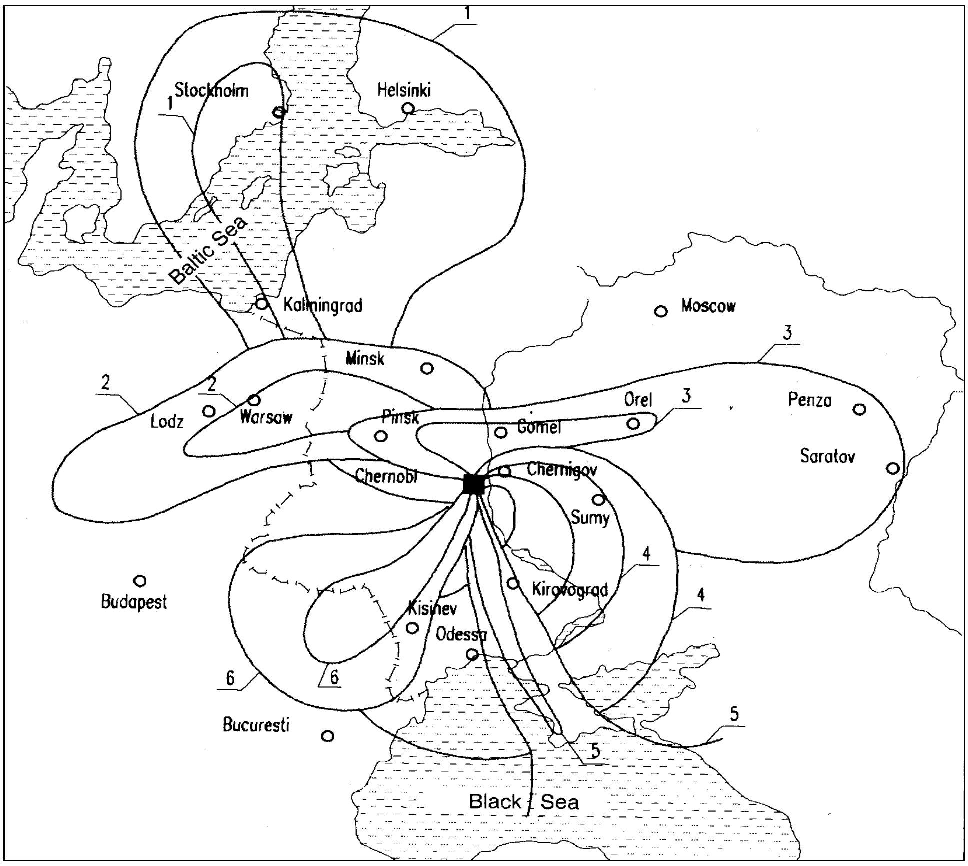


FIG. 2.2. Calculated plume formation according to meteorological conditions for instantaneous releases on the following dates and times (GMT): (1) 26 April, 00:00; (2) 27 April, 00:00; (3) 27 April, 12:00; (4) 29 April, 00:00; (5) 2 May, 00:00; and (6) 4 May, 12:00 (Borsilov and Klepikova 1993).

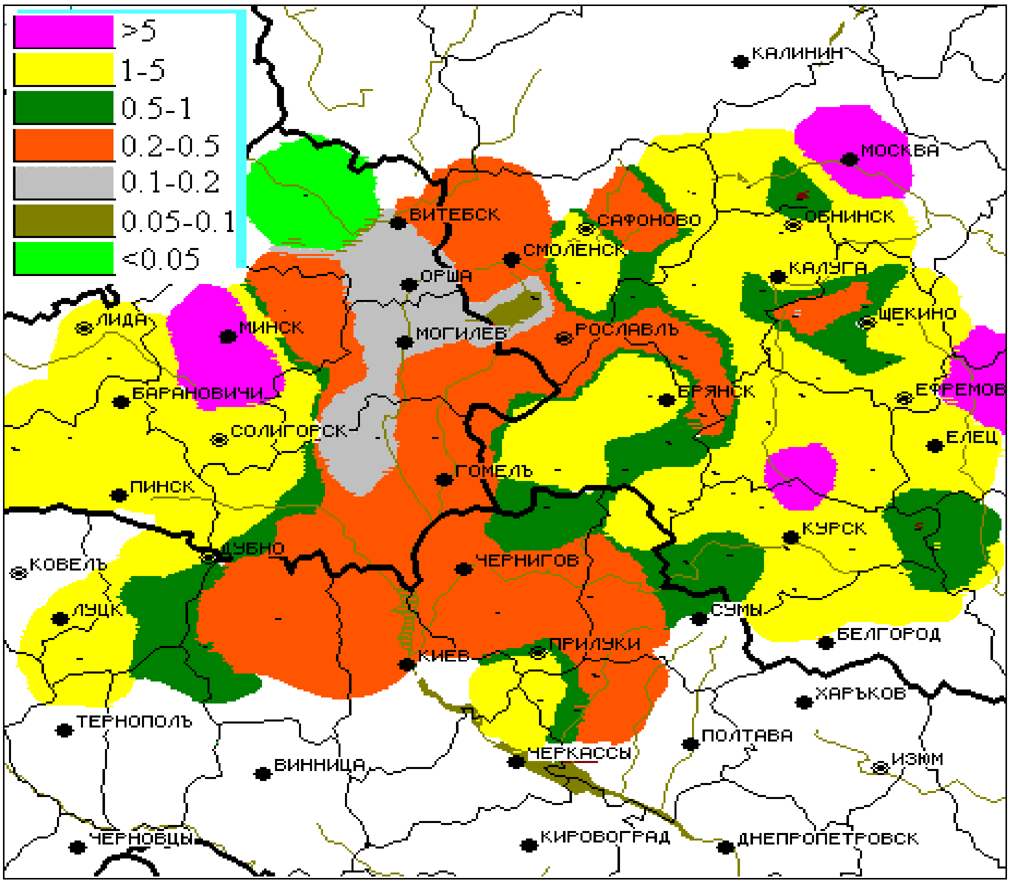


FIG. 2.3. Map of average daily precipitation intensity (mm h^{-1}) on 29 April 1986, in the area nearby the Chernobyl Nuclear Power Plant (Izrael 1998). Replace with English version.

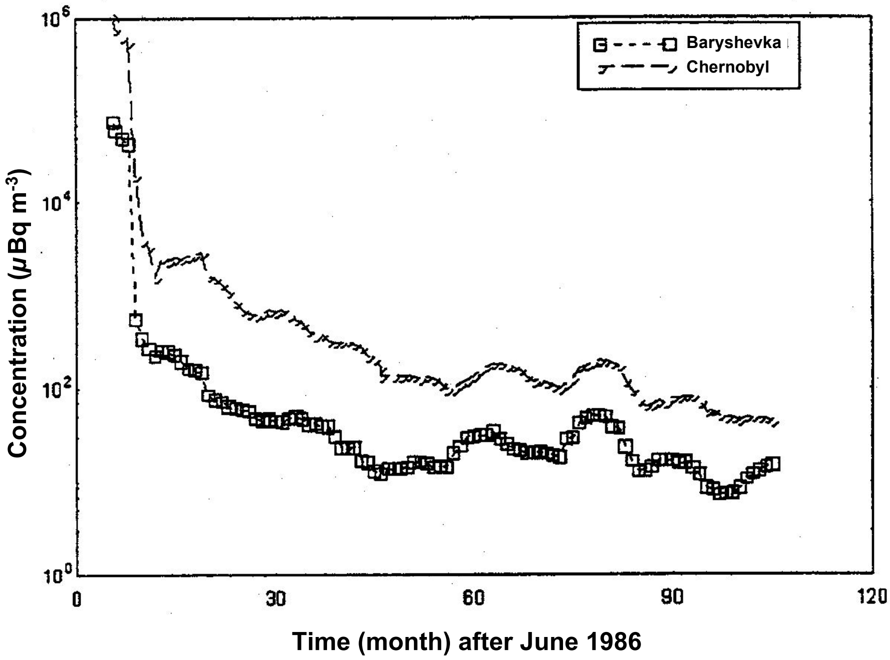


FIG. 2.4. Rolling seven-month mean atmospheric concentration of ^{137}Cs at Baryshevka and Chernobyl (June 1986 to August 1994) (Holländer and Garger 1996).

2.1.4. *Concentration of radionuclides in air*

The activity concentration of radioactive materials in air was measured at many locations in the former Soviet Union and throughout the world. Examples of such activity concentrations in air are shown in Fig. 2.4 for two locations: Chernobyl and Baryshevka, Ukraine. The location of the Chernobyl sampler was at the meteorological station in the City of Chernobyl, which is more than 15 km southeast of the ChNPP. The initial concentrations of airborne materials were very high, but dropped in two phases. There was a rapid fall over a few months, and then a more gradual decrease over several years. Over the long term, the sampler at Chernobyl records consistently higher activity concentrations than the sampler at Baryshevka (about 150 km southeast of the ChNPP), presumably due to resuspension (Holländer and Garger 1996).

Even with the data smoothed by a rolling average, there are some notable features in the data collected over the long term. The clearly discernible peak that occurred during the summer of 1992 (month 78) was due to widespread forest fires in Ukraine and Belarus.

2.1.5. *Deposition of radionuclides on the soil surface*

As already mentioned, surveys by airborne spectrometers over large areas were undertaken soon after the accident to measure the deposition of ^{137}Cs (and other radionuclides) on the soil surface in several countries. In the mapping of the deposition, ^{137}Cs was chosen because it is easy to measure and is of radiological significance. Soil deposition of ^{137}Cs equal to 37 kBq m^{-2} (1 Ci km^{-2}) was chosen as a provisional minimum contamination level, because (1) this level was about ten times higher than the ^{137}Cs deposition in Europe from global fallout, and (2) at this level, the human dose during the 1st year after the accident was about 1 mSv and was considered to be radiologically important. Knowledge of the extent and spatial variation in deposition is critical in (1) defining the magnitude of the accident, (2) predicting future levels of external and internal dose, and (3) determining what radiation-protection measures are necessary. In addition, many soil samples were collected and analysed at radiological laboratories.

Thus, massive amounts of data were collected and subsequently published in the form of an Atlas that covers essentially all of Europe (De Cort et al. 1998). Another Atlas produced in Russia (Izrael 1998) covers the European part of the former Soviet Union. Two examples of these maps are shown in Figs. 2.5 and 2.6.

It is clear from Figs. 2.5 and 2.6 and Table 2.2 that the three countries most heavily impacted by the accident were Ukraine, Belarus, and Russia. From the total ^{137}Cs activity of about 64 TBq (1.7 MCi) deposited on European territory in 1986, Belarus received 23%, Russia 30% and Ukraine 18%. However, due to the wet deposition processes discussed above, there were also major contaminated areas in Austria, Finland, Germany, Norway, Romania, and Sweden. A more detailed view of the nearby heavily contaminated areas in shown in Fig. 2.7 (International Advisory Committee 1991).

Water and wind erosion of soils may lead to ^{137}Cs transfer and redistribution on a local scale, at relatively small distances. Wind erosion may also lead to ^{137}Cs transfer with soil particles on a regional scale.

Soon after the accident, a 30-km-radius exclusion zone was established around the reactor. Further re-locations took place in subsequent months and years in Ukraine, Belarus, and Russia; eventually a total of about 116,000 persons were evacuated or re-located.

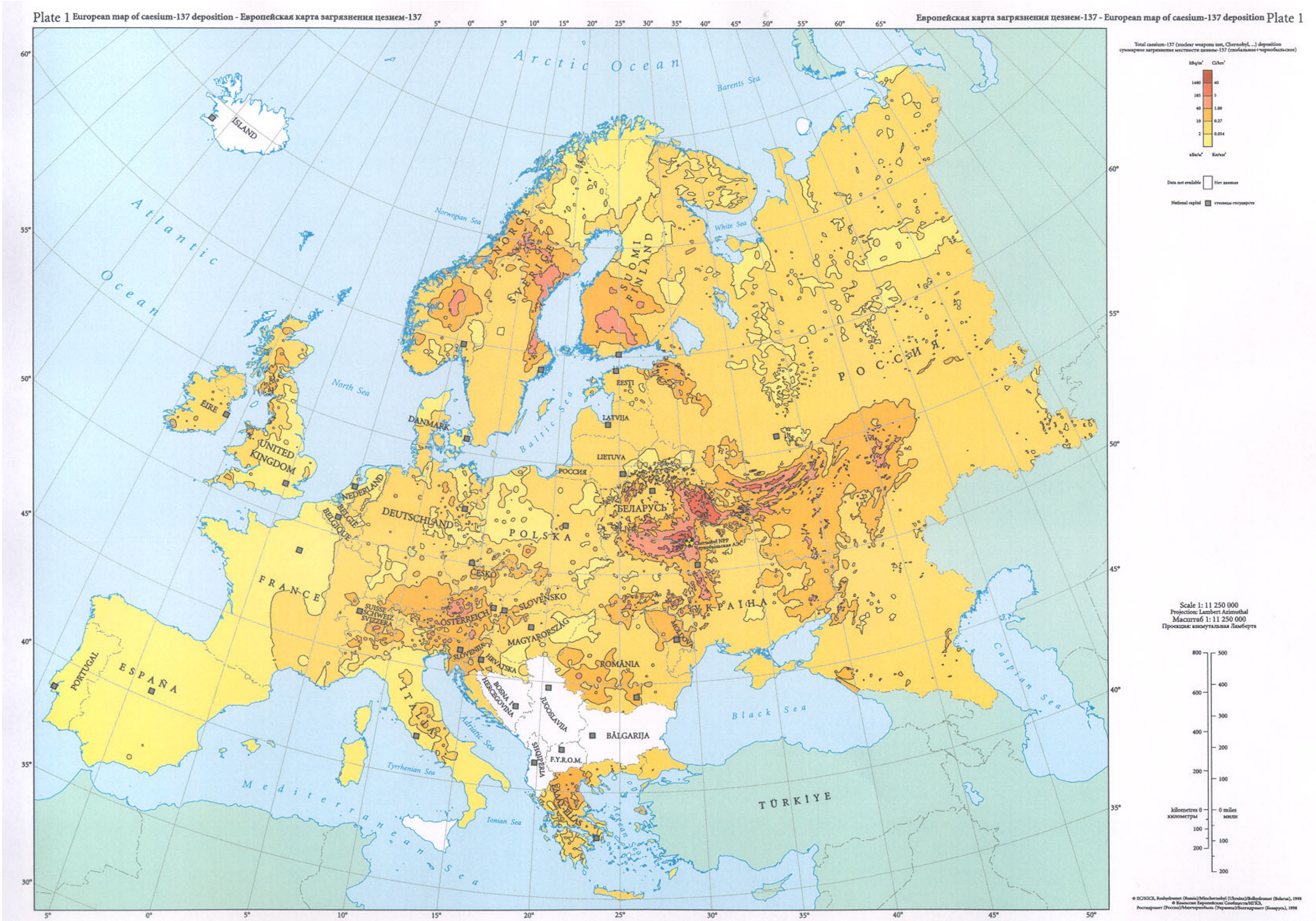


FIG. 2.5. Surface-ground deposition of ^{137}Cs throughout Europe as a result of the Chernobyl accident (De Cort et al. 1998).

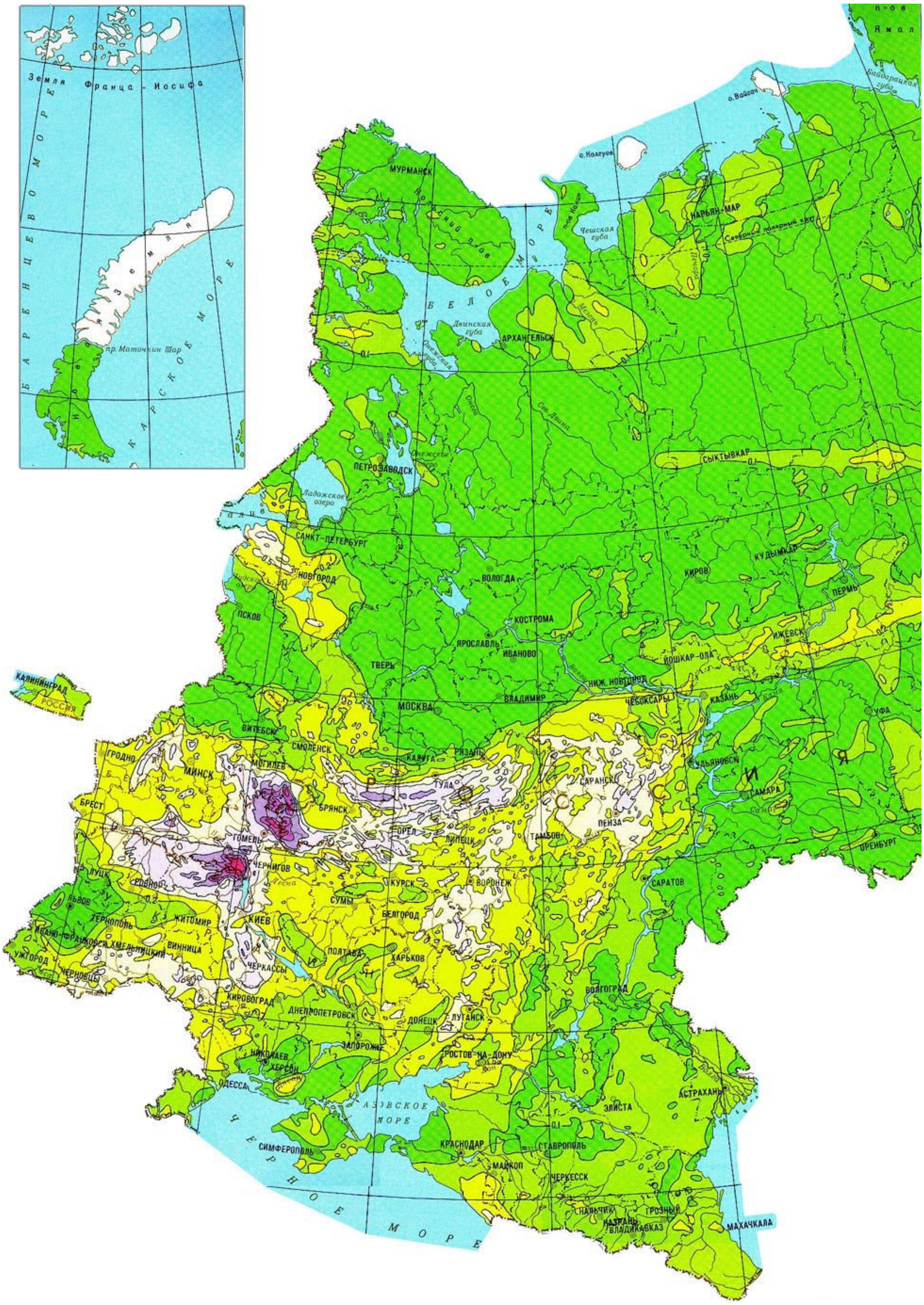


FIG. 2.6. Radioactive contamination of the European part of the former Soviet Union after the Chernobyl accident (Izrael 1998).-Replace with English version!.

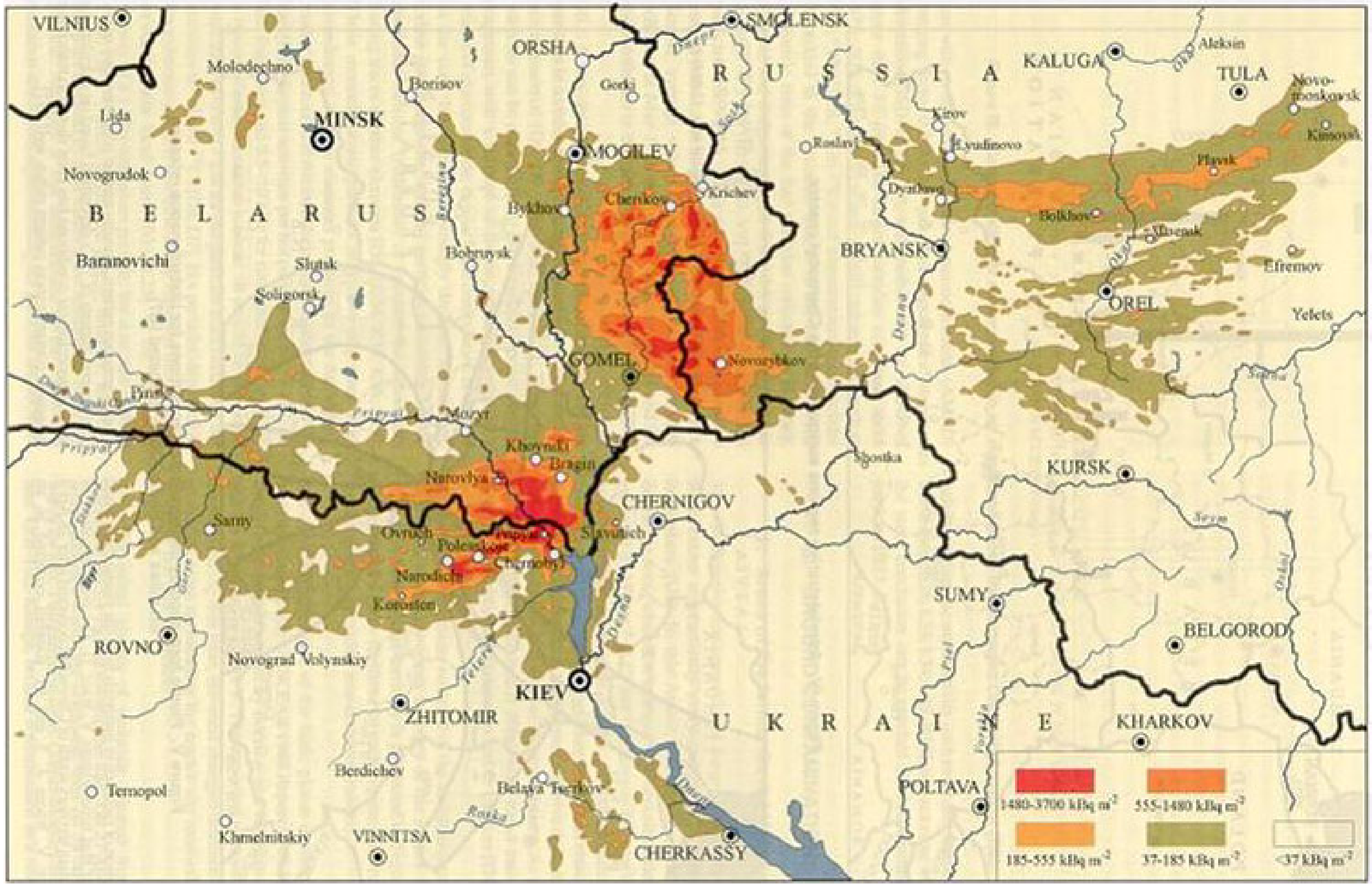


FIG.2.7. Surface-ground deposition of ^{137}Cs in areas of Ukraine, Belarus, and Russia nearby the accident. (International Advisory Committee 1991).

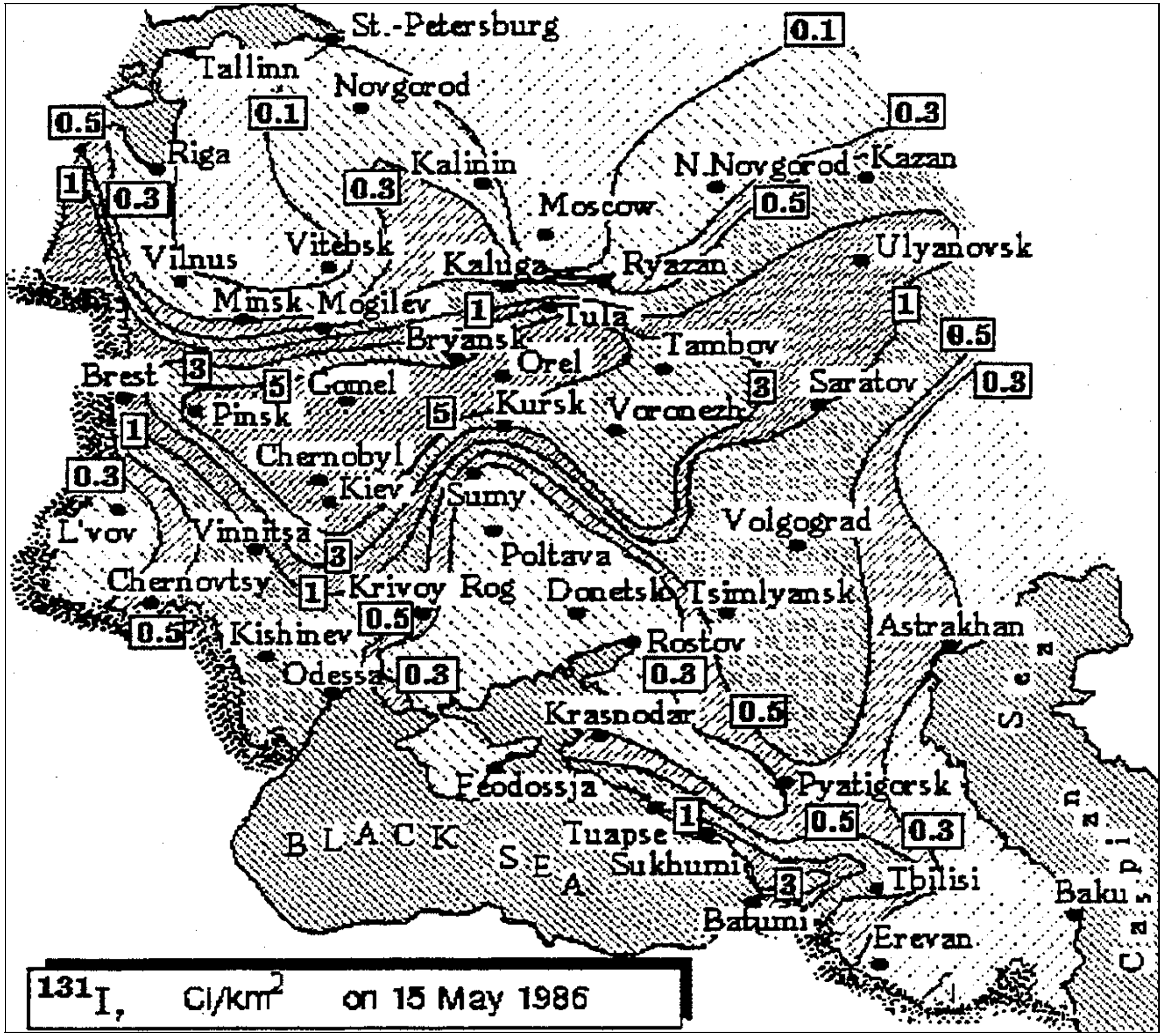


FIG. 2.8. Surface ground deposition of ^{131}I (Makhonko et al. 1996).

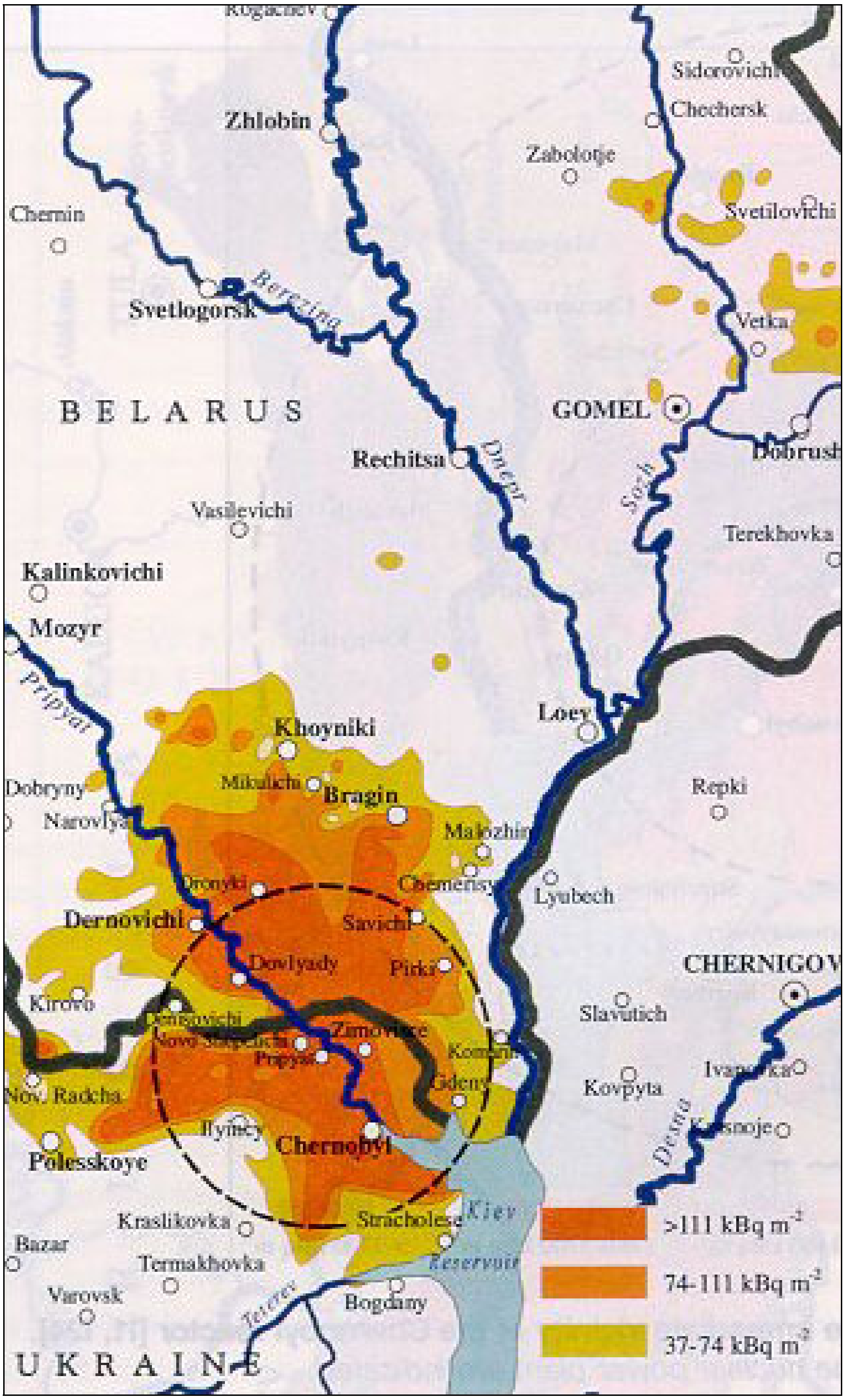


FIG. 2.9. Surface ground deposition of ^{90}Sr (International Advisory Committee 1991).

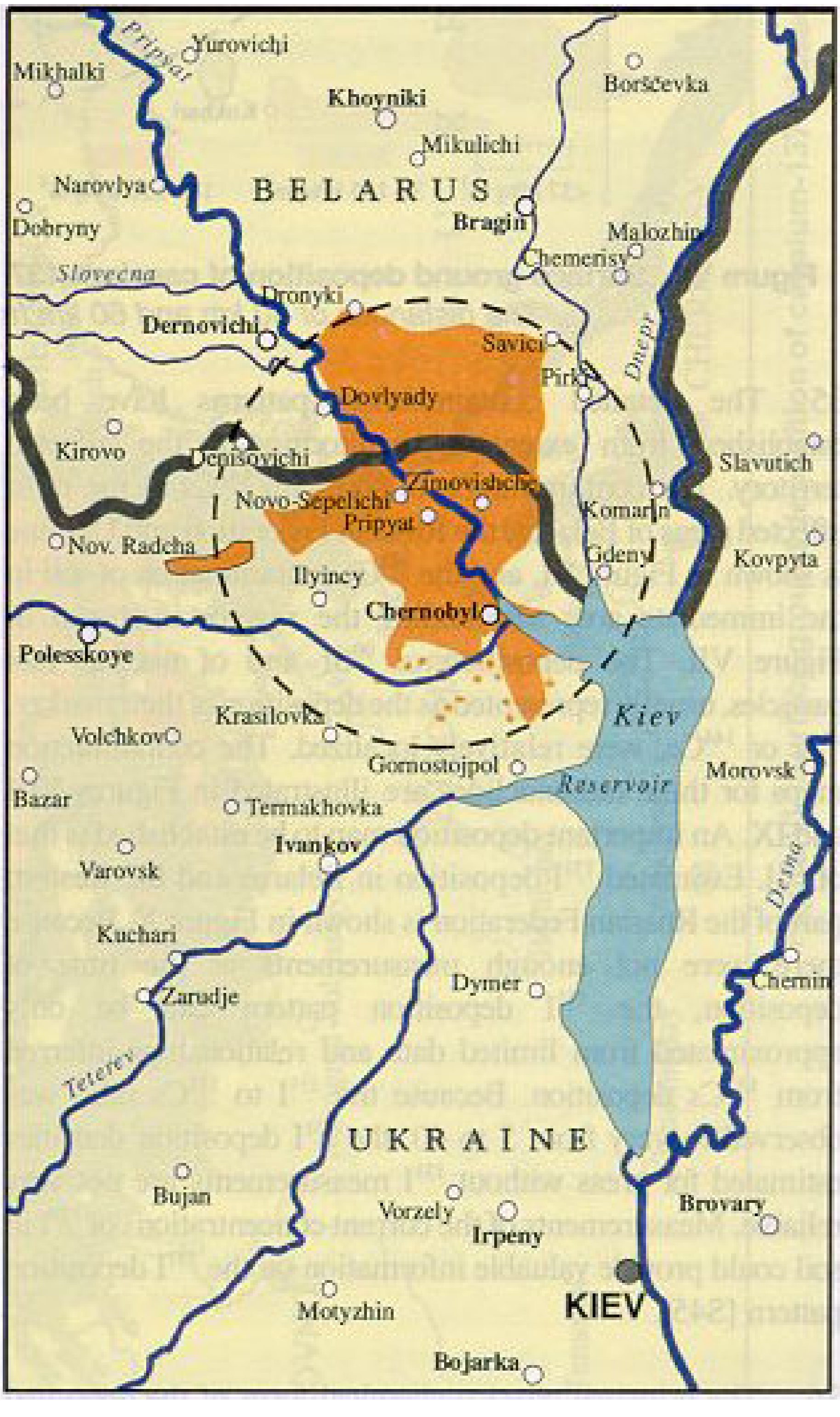


FIG. 2. 10. Areas (orange) where the surface-ground deposition of $^{239+240}Pu$ exceeds 3.7 kBq m^{-2} (International Advisory Committee 1991).

TABLE 2.2. CONTAMINATED AREAS IN EUROPE FROM CHERNOBYL FALLOUT IN 1986 (UNSCEAR 2000, EC 2001)

Country	Area with ^{137}Cs deposition density range (km^2)			
	37-185 kBq m^{-2}	185-555 kBq m^{-2}	555-1480 kBq m^{-2}	> 1480 kBq m^{-2}
Russian Federation	49.800	5700	2100	300
Belarus	29,900	10,200	4200	2200
Ukraine	37,200	3200	900	600
Sweden	12,000			
Finland	11,500			
Austria	8600			
Norway	5200			
Bulgaria	4800			
Switzerland	1300			
Greece	1200			
Slovenia	300			
Italy	300			
Republic of Moldova	60			

The total area with ^{137}Cs-soil deposition of 0.6 MBq m^{-2} (15 Ci km^{-2}) and above in 1986 was 10,300 km^2, including 6,400 km^2 in Belarus, 2,400 km^2 in Russia and 1,500 km^2 in Ukraine. In total, 640 settlements with about 230,000 inhabitants were located on these contaminated territories. The relevant laws on social protection of the three most impacted countries considered that areas with ^{137}Cs depositions of more than 1 Ci km^{-2} (37 kBq m^{-2}) are classified as radioactively contaminated. The number of people who were living in such contaminated areas in 1995 is shown in Table 2.3.

Immediately after the accident most concern was focused on contamination of food with ^{131}I. The broad pattern of the deposition of ^{131}I is shown in Figure 2.8. Unfortunately, due to the rapid decay of ^{131}I after its deposition, there was not enough time to collect a large number of samples for detailed analysis. At first, it was assumed that a strong correlation could be assumed between depositions of ^{131}I and ^{137}Cs. However, this has not consistently been found to be correct. More recently, soil samples have been collected and analysed for ^{129}I, which has a physical half life of 16 million years and can only be measured at very low levels with accelerator-mass spectrometry. Straume et al. (1996) have reported successful analysis of samples taken in Belarus from which they have established that at the time of the accident there were 15±3 atoms of ^{129}I for each atom of ^{131}I. This estimated ratio enables better estimates of the deposition of ^{131}I for the purpose of reconstructing doses.

TABLE 2.3. DISTRIBUTION OF INHABITANTS LIVING IN AREAS CONSIDERED TO BE RADIOACTIVELY CONTAMINATED IN UKRAINE, BELARUS, AND RUSSIA IN 1995

^{137}Cs-deposition density, kBq m^{-2}	Thousands of inhabitants[a]			
	Belarus	Russia	Ukraine	Total
37–185	1,543	1,654	1,189	4,386
185–555	239	234	107	580
555–1480	98	95	0.3	193
Total	1,880	1,983	1,296	5,159

[a] For social and economic reasons, some people living in areas of contamination less than 37 kBq m^{-2} are also included.

Similar maps could be drawn for the other radionuclides of interest shown in Table 2.1. A map of the deposition of ^{90}Sr is shown in Fig. 2.9. In comparison with ^{137}Cs there was less ^{90}Sr released from the reactor and strontium is less volatile than caesium. Thus, the spatial extent of ^{90}Sr deposition was much more confined to areas close to the Chernobyl NPP than that of ^{137}Cs. Amounts of plutonium deposited on the soil have also been measured, and a map is shown in Fig. 2.10. Nearly all areas with plutonium deposition above 3.7 kBq m^{-2} (0.1 Ci km^{-2}) are within the 30-km exclusion area.

2.1.6. Isotopic composition of deposition

The most extensive measurements of activity per unit area have been performed for ^{137}Cs. Activities of other radionuclides, especially of ^{134}Cs, ^{136}Cs, ^{131}I, ^{133}I, ^{140}Ba+^{140}La, ^{95}Zr+^{95}Nb, ^{103}Ru, ^{106}Ru, ^{132}Te, ^{125}Sb, and ^{144}Ce, have been expressed as ratios to the reference radionuclide ^{137}Cs. These ratios depend on the location, because of (1) the different deposition behaviour of fuel particles, aerosols and gaseous radionuclides and (2) the variation in radionuclide composition with time of release. However, the ratio of radionuclides of different elements is not necessarily constant with time. According to the time of release and the corresponding release characteristics (e.g., temperature of the core) significant variations in the release ratios were observed after the Chernobyl accident (Izrael 1990; Buzulukov and Dobrynin 1993).

The first plume, which moved to the west, carried the release during the explosive phase, when the exposed core was not as hot as in the later phases. The second plume, which moved north to north-east, carried releases from a core of increasing temperature, while the third plume, moving mainly south, was characterized by releases from a core heated to temperatures above 2000 °C; at such temperatures less-volatile radionuclides such as molybdenum, strontium, zirconium, ruthenium and barium are readily released. During this phase, the iodine releases also increased.

Caesium "hot spots" occurred in the far zone of Belarus and Russia and in the Kaluga, Tula, and Orel Oblasts. The composition of the deposited radionuclides in each of these highly contaminated areas was relatively similar. The ratios of different radionuclides to ^{137}Cs as observed in ground depositions in the different release vectors are given in Table 2. 4.

TABLE 2.4. ESTIMATION OF RELATIVE ACTIVITY PER UNIT AREA IN THE NEAR-FIELD ZONE AND "CAESIUM HOT SPOTS" AFTER THE RELEASE FROM THE CHERNOBYL NUCLEAR POWER PLANT (ON 26 APRIL 1986) (IZRAEL ET AL.1990)

Nuclide	Half-life	Activity deposition relative to ^{137}Cs			
		Western plume	Northern plume	Southern plume	Caesium hot spots (far zone)
^{90}Sr	28.5 a	0.5	0.13	1.5	0.014
^{95}Zr	64.0 d	5	3	10	0.06
^{99}Mo	66.0 h	8	3	25	0.11
^{103}Ru	39.35 d	4	2.7	12	1.9
^{132}Te	78.0 h	15	17	13	13
$^{131}I_{aer}$	8.02 d	18	17	30	10
^{137}Cs	30.0 a	1.0	1.0	1.0	1.0
^{140}Ba	12.79 d	7	3	20	0.7
^{144}Ce	284.8 d	3	2.3	6	0.07
^{239}Np	2.355 d	25	7	140	0.6
^{239}Pu	24,400 a	0.0015	0.0015	–	–

The activity ratios for the western plume and the northern plume were similar and in many cases identical, in contrast with the ratios for the southern plume. All activity ratios show, with the exception of $^{132}Te/^{137}Cs$, a decrease with increasing distance from the nuclear power plant. The decrease is less profound for ^{95}Zr and ^{144}Ce (about a factor of three) than with ^{99}Mo and ^{140}Ba (two orders of magnitude) or ^{90}Sr and ^{103}Ru (one order of magnitude). For the ratio $^{131}I/^{137}Cs$ there was only a slight decrease by about a factor of 4 observed over a 1000-km distance. Within the first 200 km virtually no variation of the ratio was observed.

2.2. Urban environment

2.2.1. Deposition patterns

Radioactive fallout on thousands of settlements, both in the USSR and some other European countries, resulted in long-term contamination with radionuclides and corresponding irradiation of their inhabitants due to both external gamma radiation and internal exposure due to consumption of contaminated food. Near the Chernobyl NPP, the towns of Pripyat and Chernobyl and some other smaller settlements were subjected to substantial contamination from an "undiluted" radioactive cloud under dry meteorological conditions, whereas many more distant settlements were significantly affected because of precipitation at the time of the cloud passage.

When radioactive fallout was deposited on settlements, the exposed surfaces such as lawns, parks, streets, building roofs and walls became contaminated with radionuclides. Both the level and elemental composition of the radioactive fallout was significantly influenced by the type of deposition mechanism, namely wet deposition with precipitation or dry deposition influenced by atmospheric mixing, diffusion and chemical adsorption. Under dry conditions, trees, bushes, lawns and roofs became more contaminated than when there is precipitation. Under wet conditions, horizontal surfaces received the highest contamination, including soil plots and lawns – see Fig. 2.11 for ^{137}Cs-deposition pattern (Roed and Andersson 2002). Particularly high ^{137}Cs-activity concentrations were found around houses, where rain had transported radioactivity from roofs to the ground.

2.2.2. Migration of radionuclides in the urban environment

Due to natural weathering processes such as rain and snow melting and human activities, including traffic, street washing and cleanup, radionuclides have been detached from surfaces and transported within settlements, some being removed within litter and sewage. Thus, contaminated leaves and needles were removed from cities after seasonal defoliation; radionuclides deposited on asphalt and concrete pavements have been eroded or washed-off and removed through sewage systems. These normally occurring natural processes and human activities significantly reduced dose rates in inhabited and recreational areas during 1986 and thereafter (Los and Likhtarev 1993).

In general, vertical surfaces of houses were not subjected to the same degree of weathering through rain as are horizontal surfaces such as roofs. The loss of contamination on walls has been typically 50-70 % of the initial deposit after 14 years. Roof contamination levels in Denmark naturally decreased by 60-95% of that originally present after 14 years - Fig. 2.12 (Andersson et al. 2002).

In contrast to walls, the level of radiocaesium on asphalt surfaces has decreased so much that generally less than 10 % of the initially retained radiocaesium is now left. Only a small fraction of the radiocaesium contamination is associated with the bitumen fraction of the asphalt; most is associated with a thin layer of street dust, which will eventually be weathered off.

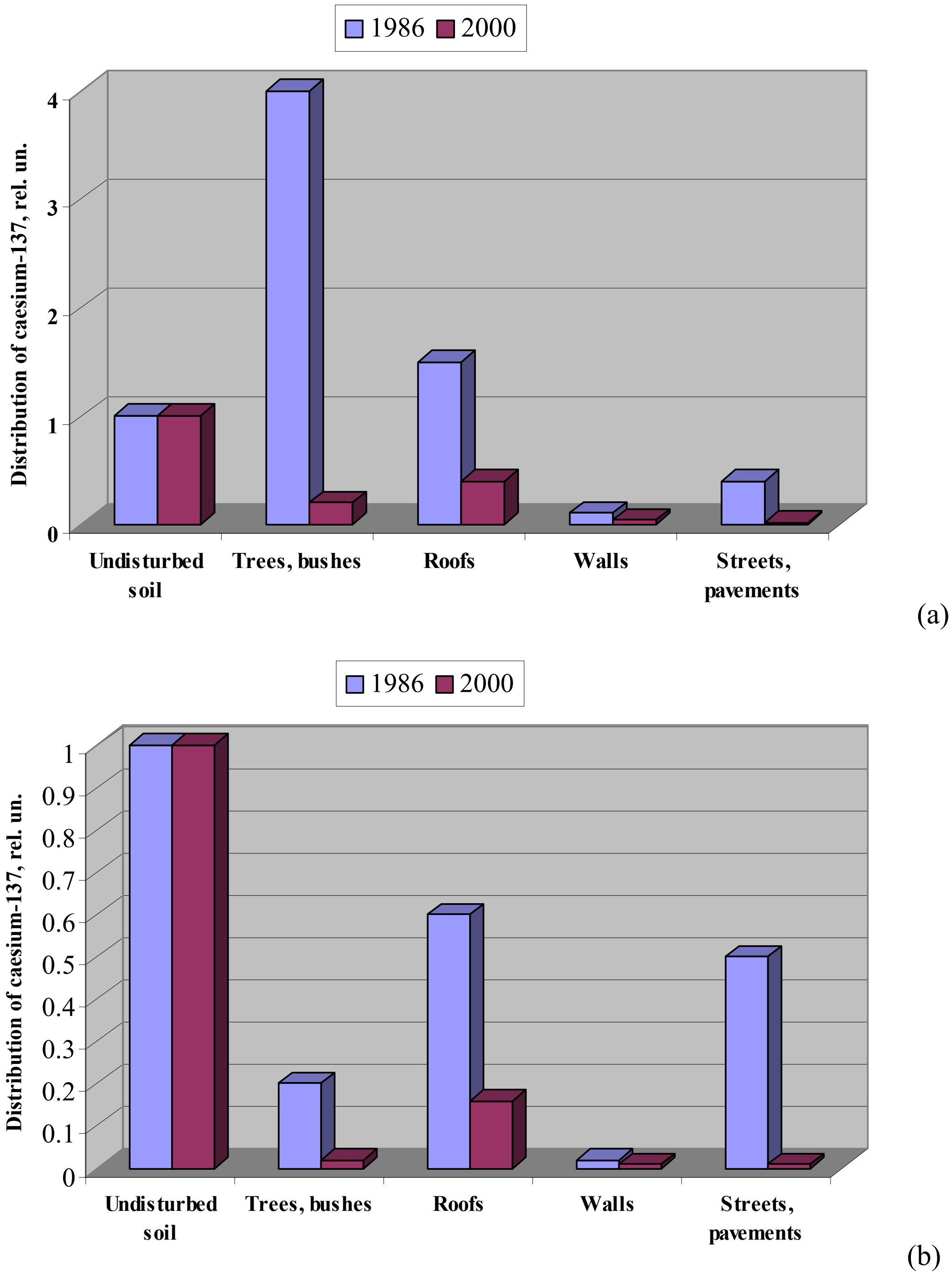


FIG. 2.11. Typical distribution of ^{137}*Cs on different surfaces within settlements in 1986 and 14 years after deposition of the Chernobyl fallout (a – dry deposition; b – wet deposition) (Roed and Andersson 2002).*

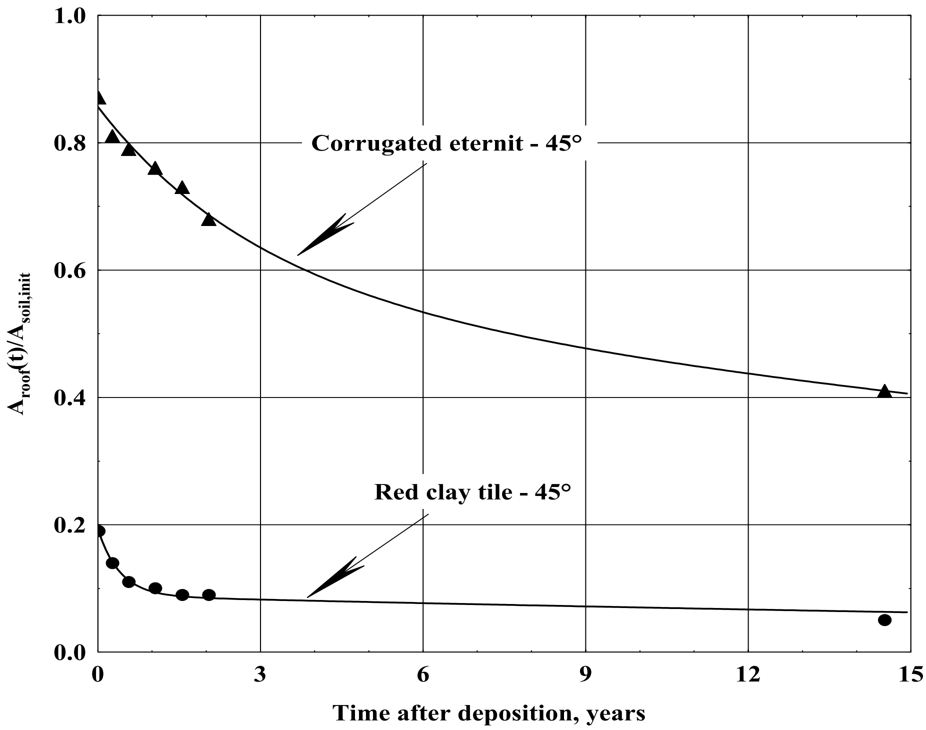


FIG. 2.12. Measured ^{137}Cs-contamination levels (relative to the initial soil contamination) on two types of roofs at Risø, Denmark (Andersson et al. 2002).

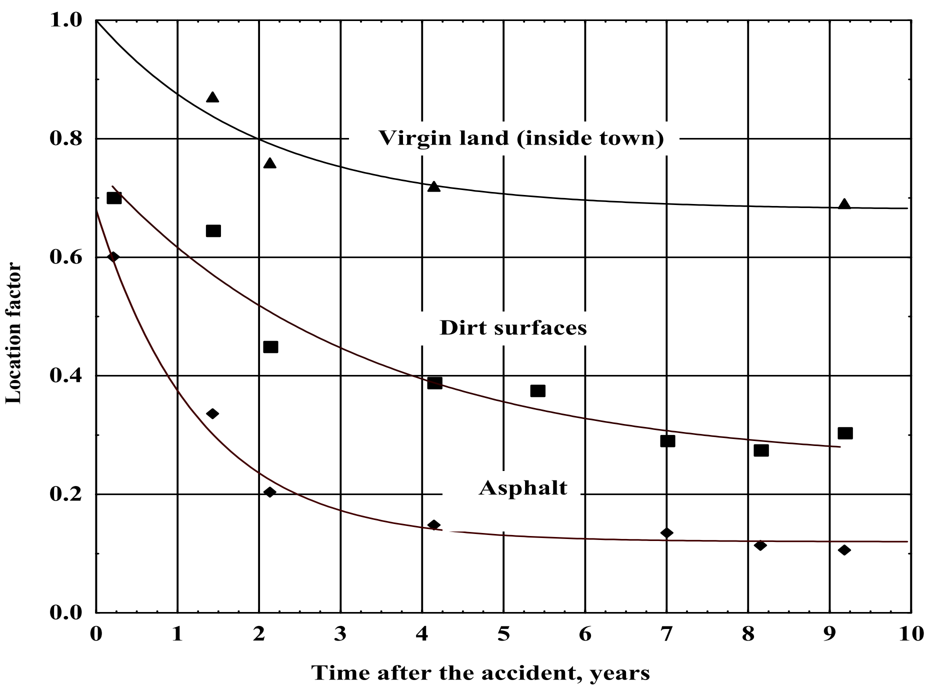


FIG. 2.13. Ratio of dose rate above particular surfaces to that in open field in the town of Novozybkov, Russia, after the Chernobyl accident (Golikov et al. 2002).

Measurements made in 1993 in the city of Pripyat near the Chernobyl NPP showed high residual levels of radiocaesium on roads. However, this town was evacuated in the early accident phase, and traffic after the contamination has been limited. Some. 5-10 % of the initially deposited radiocaesium seems to be firmly fixed to concrete-paved surfaces and no significant decrease has been recorded over the last few years. The weathering on horizontal hard surfaces was, as expected, generally faster in the areas with more traffic.

One of the consequences of these processes has been secondary contamination of sewage systems and sludge storage, which necessitated special cleanup measures. Generally, radionuclides have not been transferred to other urban areas from soil within cities but have migrated down into the soil due to natural processes or mixing during digging up of gardens, kitchen-gardens and parks.

2.2.3. Dynamics of exposure rate

Gamma radiation from radionuclides deposited in the urban environment has contributed to air-dose rate and human external exposure. Compared to the dose rate in open fields (see Sub-section 4.2.2 below), the dose rate within a settlement has been always significantly lower because of photon absorption in building structures, especially those made of brick and concrete. The lower dose rates have been observed inside buildings, especially on the upper floors of multi-story buildings. Due to radioactive decay of the initial radionuclide mixture, wash-off from solid surfaces and soil migration, air-dose rates have been gradually decreasing with time.

Another relevant parameter is the time dependence of the ratio of air-dose rate at an urban location compared to that in an open field (so-called location factor) due to radionuclide-migration processes. The dependence of urban location factors on time after the Chernobyl accident is shown in Fig. 2.13 as derived from measurements performed in the town of Novozybkov, Russia (Golikov et al. 2002). Whereas for virgin sites like parks or grassy plots the location factors are relatively constant, values for hard surfaces such as asphalt decrease considerably with time. Similar time dependence has been found in other countries (Andersson et al. 1995; Jacob and Meckbach 2000).

At present, in most of the settlements subjected to radioactive contamination after the Chernobyl accident the air-dose rate above solid surfaces has returned to the pre-accident background level. Elevated air-dose rates remain mainly over undisturbed soil. The highest level of radioactive contamination of the urban environment remains in Pripyat, which is three kilometres from the Chernobyl NPP; its inhabitants were resettled to non-contaminated areas within 1.5 days after the accident.

2.3. Agricultural environment

2.3.1. Radionuclide transfer in the terrestrial environment

Various radionuclides behave differently in the environment. Some radionuclides, such as radiocaesium, radioiodine and radiostrontium are environmentally mobile and transfer readily, under certain environmental conditions, to foodstuffs. In contrast, radionuclides with low solubility such as the actinides are relatively immobile and largely remain in the soil. The main routes of cycling of radionuclides and possible exposure pathways to man are shown in Fig. 2.14.

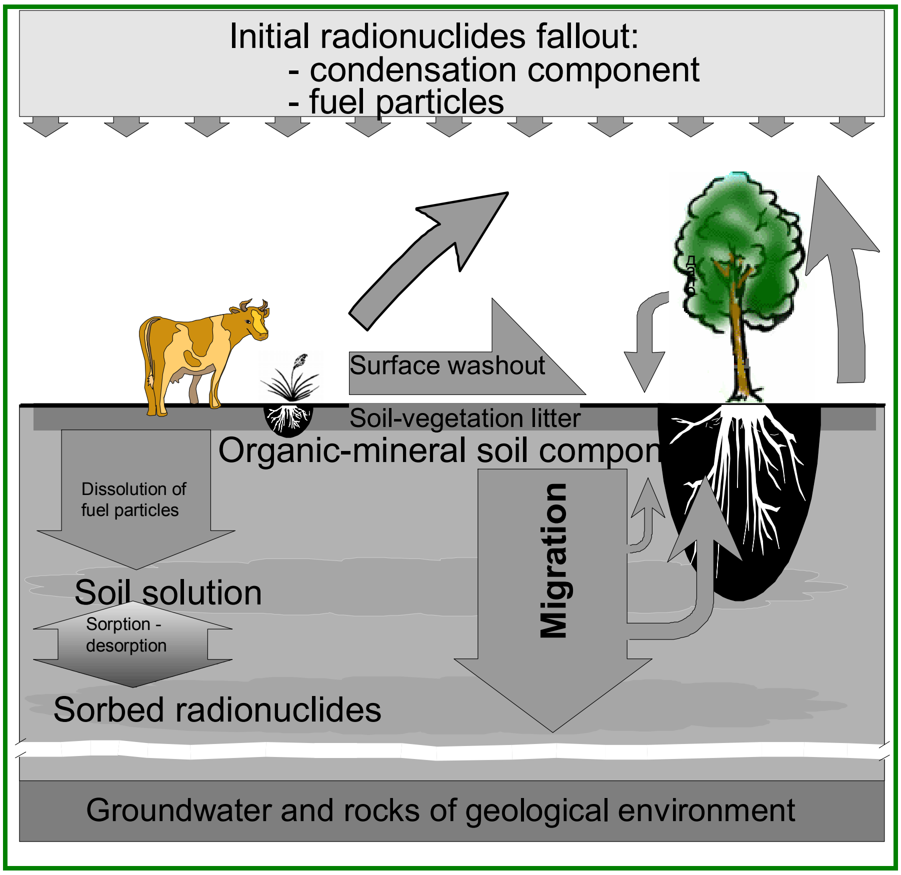


FIG. 2.14. Main terrestrial environmental pathways of radionuclides (adopted from Shestopalov et al. 2003)- to be replaced

Many factors influence the extent to which radionuclides are transferred through ecological pathways. If transfer is high in a particular environment then it is referred to as radioecologically sensitive, because such transfer can lead to relatively high radiological exposure (Howard 2000).

Of the radionuclides deposited after the Chernobyl accident, during the short initial phase (0 to 2 months) radioiodine was the most important with regard to human exposure via agricultural food chains, and in the longer term radiocaesium has been the most important (and to a much lesser extent radiostrontium).

Radioecological sensitivity to radiocaesium in semi-natural ecosystems is generally higher than in agricultural ecosystems, sometimes by a few orders of magnitude (Howard et al. 2002). This difference is caused by a number of factors, the more important being differing physico-chemical behaviour in soils, the lack of competition between Cs and K resulting in higher transfer rates of radiocaesium in nutrient-poor ecosystems, and by the presence of specific food-chain pathways leading to highly contaminated produce from semi-natural ecosystems. Also, forest soils are fundamentally different from agricultural soils with a clear multilayered vertical structure characterised mainly by a clay-poor mineral layer, which supports an organic layer rich in organic matter. In contrast, agricultural soils generally contain less organic matter and higher amounts of clay.

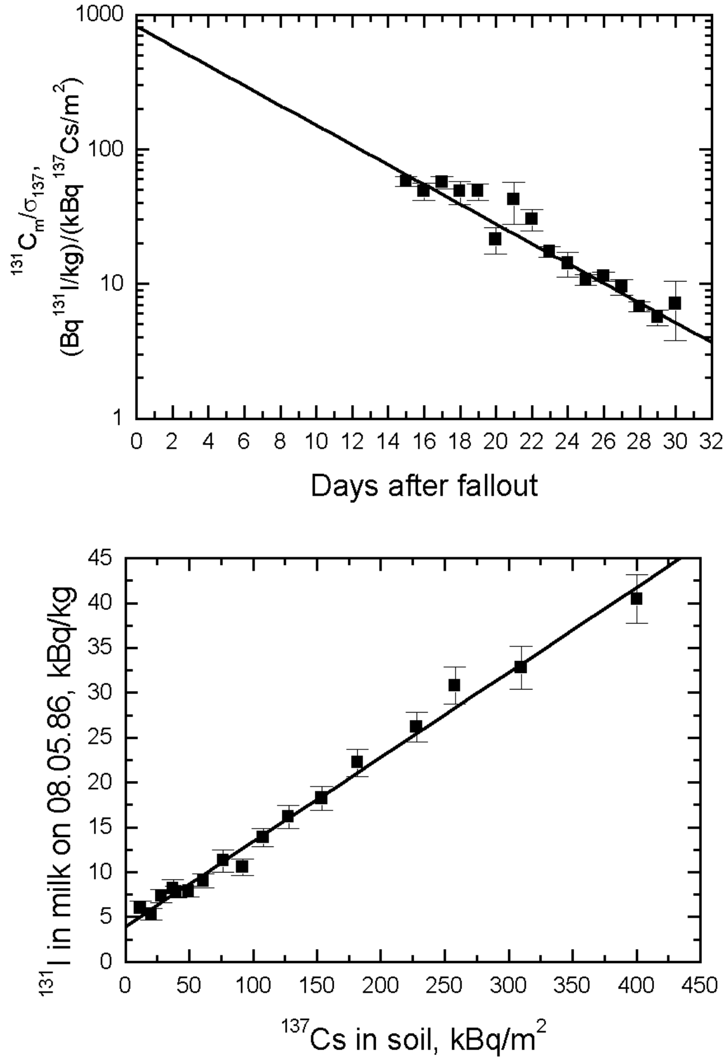


FIG. 2.15. Variation in ^{131}I transfer to milk in the Tula Oblast, Russia (a) with time after deposition standardized by ^{137}Cs-soil deposition and (b) with on ^{137}Cs soil deposition time corrected to 08.05.1986 (Balonov et al. 2000).

2.3.2. Food-production systems affected by the accident

The Chernobyl fallout contaminated large areas of the terrestrial environment with a major impact both on agricultural and natural ecosystems not only within the former Soviet Union but also in many other countries in Europe.

In the former Soviet Union countries, the prevailing food-production system at the time of the accident could be divided into two groups: large collective farms and small private farms. Collective farms routinely use land rotation combined with ploughing and fertilisation to improve productivity. Traditional small private farms, in contrast, seldom apply artificial fertilisers and often use manure for improving yield. They have one or a few cows, and produce milk mainly for personal consumption. The grazing regime of private farms was initially limited to utilisation of marginal land not used by the collective farms, but nowadays includes some better quality pasture.

In Western Europe, poor soils are used for extensive agriculture, mainly for grazing of ruminants (e.g., sheep, goats, reindeer, and cattle). Such areas include alpine meadows and upland regions in Western and Northern Europe with organic soils.

2.3.3. Early phase

At the time of the Chernobyl NPP accident, vegetation was at different growth stages depending on latitude and elevation. Initially, interception on plant leaves of dry deposition and atmospheric washout with precipitation were the main pathways of contamination. In the medium and long-term, root uptake predominated. The highest activity concentrations of radionuclides in most foodstuffs occurred in 1986.

In the initial phase, ^{131}I was the radionuclide of most concern and milk was the main contributor to internal dose. This is because radioiodine was released in large amounts and intercepted by plant surfaces that were then grazed by dairy cows. The ingested radioiodine was completely absorbed in the gut (Beresford et al. 2000) and then rapidly transferred to the animals' thyroid and milk (within about 1 day). Thus, peak values occurred rapidly after deposition in late April or early May 1986 depending on when deposition occurred in different countries. During this period, in the former Soviet Union and some other European countries ^{131}I-activity concentrations in milk exceeded national and regional (EU) action levels of a few hundred to a few thousand Becquerel per litre (see Sub-section 3.1 below).

There are no time-trend data available for ^{131}I-activity concentrations in milk in the first few days after the accident in the heavily affected areas of the USSR for the obvious and understandable reason that the authorities were dealing with other immediate accident-response priorities. Nevertheless, data are available after 2 weeks from the Tula Oblast in Russia and the data in Fig. 2.15a show an exponential decline in ^{131}I-activity concentrations normalised to ^{137}Cs deposition, which can be extrapolated back to the first days to estimate the initial ^{131}I-activity concentration. Furthermore, a direct comparison of ^{131}I activity in early May with ^{137}Cs deposition shows the contribution of dry deposition to ^{131}I in milk, because the linear relationship line shown does not go through the zero-deposition point (Fig. 2.15b). In the early spring in Northern Europe dairy cows and goats were not yet on pasture, therefore there was very little milk contamination. In contrast, in Southern regions of the USSR, as well as in Germany, France and Southern Europe, dairy animals were already grazing outdoors and some contamination of cow, goat and sheep milk occurred. The ^{131}I-activity concentration in milk decreased with an effective half-life of 4 to 5 days (Pröhl and Hoffman 1996) due to its short physical half life and the reduction in iodine-activity concentrations on plants due to atmospheric removal processes from leaf surfaces (Fig. 2.16). This removal occurred with a mean weathering half life on grass of 9 days for radioiodine and 11 days for radiocaesium (Kirchner 1994). Leafy vegetables were another food product which was contaminated on the surface and which contributed dose via the foodchain ((Fig. 2.16).

Along with radioiodine contamination, both plants and animals were contaminated with radiocaesium and, to lesser extent, radiostrontium. From June 1986, radiocaesium dominated in most environmental samples (except for the 30-km zone) and in food products. As shown in Fig. 2.17, the contamination of milk with radiocaesium decreased during Spring 1986 with an effective half-life of about two weeks due to weathering, biomass growth and other natural processes. However, radiocaesium-activity concentrations increased again during Winter 1986/87 due to the feeding of cows with contaminated hay harvested in Spring/Summer 1986. This phenomenon was observed in the winter time in many countries after the accident.

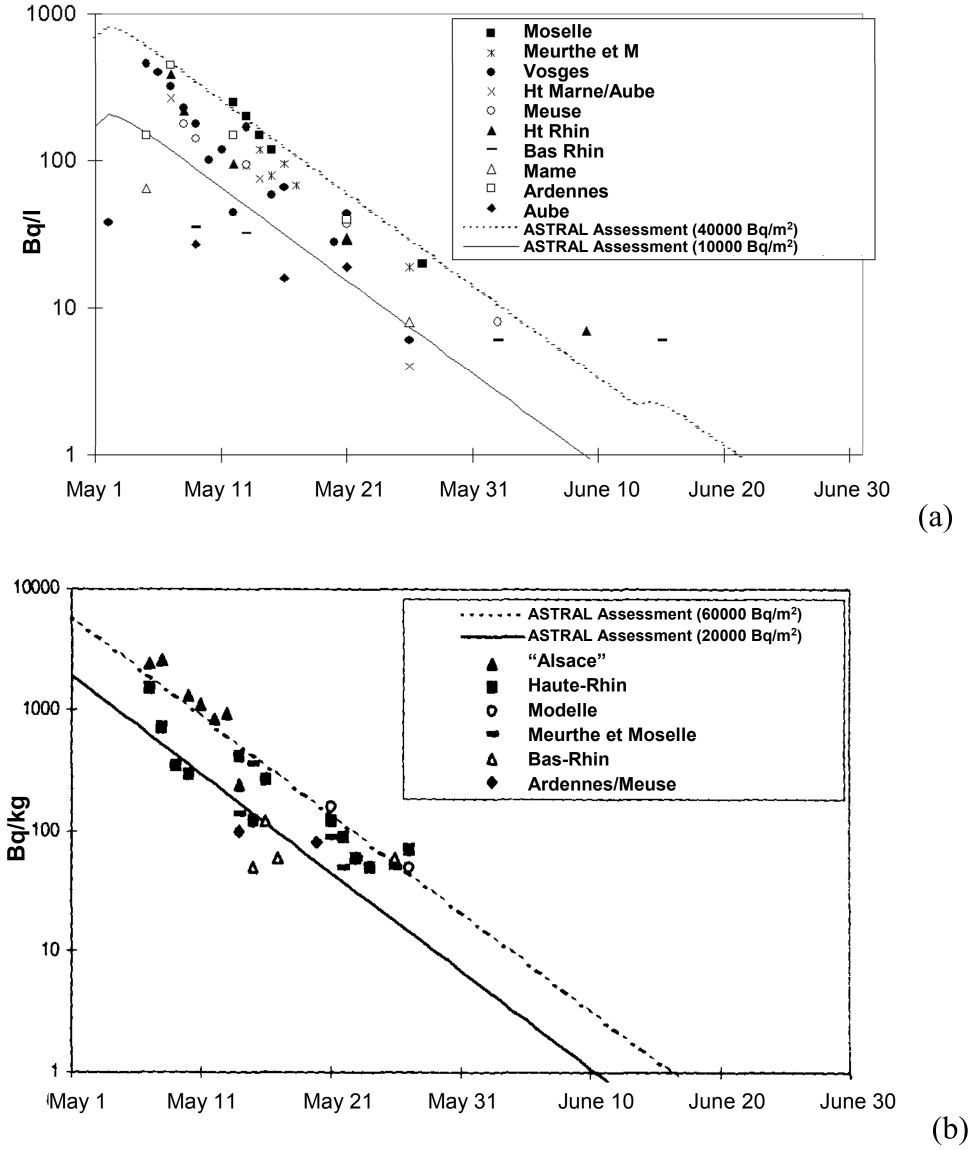


FIG. 2.16. Changes with time in ^{131}I-activity concentrations in (a) cow milk and (b) vegetables or leaves in different regions in France, May-June 1986 (Renaud et al. 1999).

The transfer to milk of many of the other radionuclides present in the terrestrial environment during the early phase of the accident was low. This was because of low inherent transfer of the elements in the gut, compounded by low bioavailability due to their association within the matrix of fuel particles (Howard and Beresford 2000). Nevertheless, some high transfers occurred, notably that of ^{110m}Ag to liver of ruminants (Beresford 1989).

2.3.4. Long-term phase

Since 1987, the radionuclide content in both plants and animals has been largely determined by interaction between radionuclides and different soil components, as soil is the main reservoir of long-lived radionuclides deposited on terrestrial ecosystems. This process controls radionuclide bioavailability (Alexakhin and Korneev 1991; Desmet et al. 1991) for uptake into plants and animals and also influences radionuclide migration down the soil column.

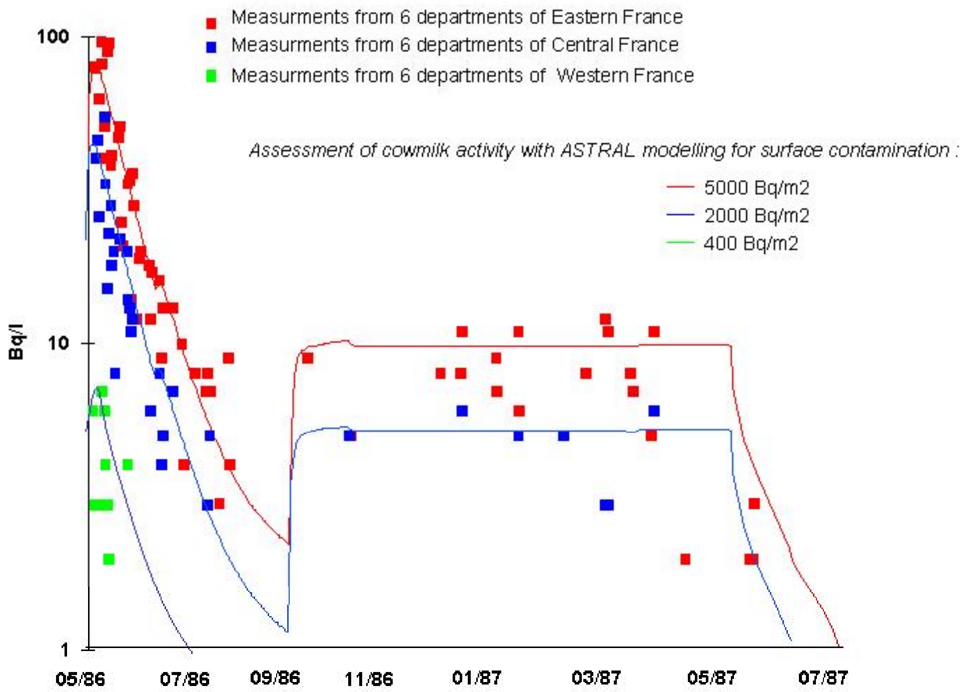


FIG. 2.17. Changes with time in ^{137}Cs-activity concentrations in cow milk in France in 1986-87 as observed and simulated by the ASTRAL model (Renaud et al. 1999).

2.3.4.1. Physico-chemistry of radionuclides in the soil plant system

Plants take up nutrients and pollutants from the soil solution. The activity concentration of radionuclides in soil solution is the result of physico-chemical interactions with the soil matrix of which competitive ion exchange is the dominant mechanism. The concentration and composition of major and competitive elements present in the soil are thus of prime importance for determining radionuclide distribution between soil and soil solution. Many data after the Chernobyl accident demonstrate that the amount and nature of clay minerals present in soils are key factors in determining radioecological sensitivity with regard to radiocaesium. These features are crucially important for understanding radiocaesium behaviour especially in areas distant from the Chernobyl NPP where ^{137}Cs was initially deposited mainly in condensed, water soluble, forms.

Close to the NPP, radionuclides were deposited in a matrix of fuel particles which have slowly dissolved with time; this process is not completed today. The more significant factors influencing the fuel-particle dissolution rate in soil are the acidity of the soil solution and the physico-chemical properties of the particles (notably the degree of oxidization) – see Fig. 2.18. In a low pH of 4, the time taken for 50% dissolution of particles was about 1 year, whereas for a higher pH of 7 as much as 14 years were needed (Kashparov et al. 2004, Salbu et al. 2001; Fesenko et al. 1996). Thus, in acid soils most of the fuel particles have already dissolved. In neutral soils, the amount of mobile ^{90}Sr released from the fuel particles is now increasing, and this will continue during the next 10-20 years.

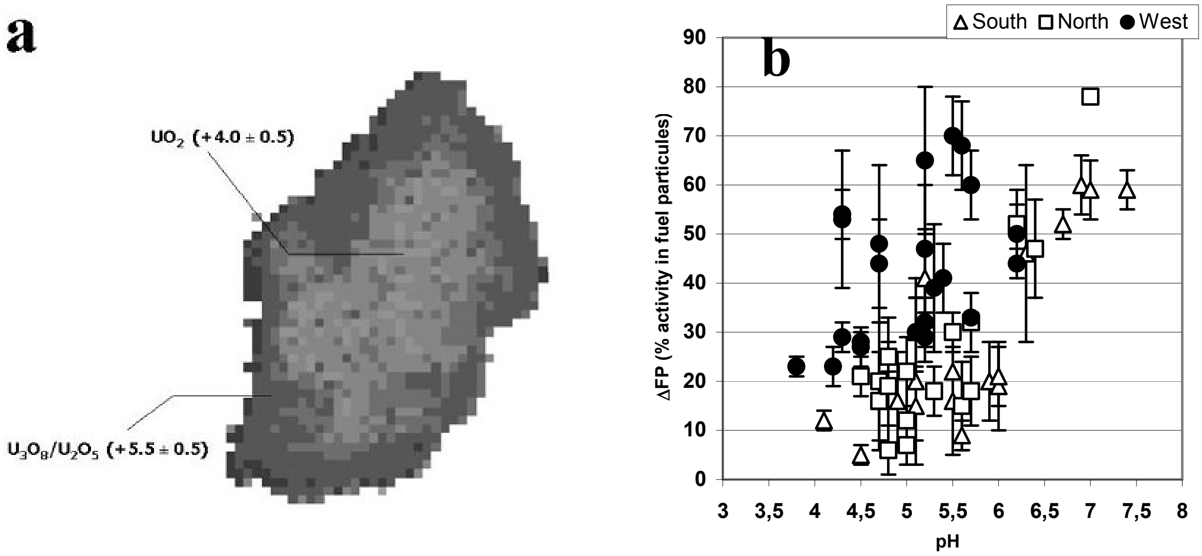


FIG. 2.18: (a) – Variation in oxidaton within a Chernobyl fuel particle (Salbu et al. 2001); (b) Fraction of ^{90}Sr present in fuel particles (ΔFP) 10 years after the Chernobyl accident as a function of soil acidity (Kashparov et al. 2004).

In addition to soil minerals, microorganisms can significantly influence the fate of radionuclides in soils (Gadd 1996; Tamponnet et al. 2001). They can interact with minerals and organic matter and consequently affect the bioavailability of radionuclides. In the specific case of mycorrhizal fungi, soil microorganisms may even act as a carrier transporting radionuclides from the soil solution to the associated plant.

A traditional approach of characterising mobility and bioavailability of a radioactive contaminant in soils is by applying sequential extraction techniques. A number of experimental protocols have been developed, which use a sequence of progressively aggressive chemicals, each of which is assumed to selectively leach a fraction of the contaminant bound to a specific soil constituent. An example of results available from this procedure is presented in Fig. 2.19 showing that a much higher proportion of radiocaesium was fixed in the soil than that of radiostrontium. The selectivity and reproducibility of chemical extraction procedures varies and therefore often should not be considered to give quantitative estimates of bioavailability, only qualitative.

With use of sequential extraction techniques, the fraction of exchangeable ^{137}Cs was found to decrease by a factor of 3-5 within a decade after 1986 (Sanzharova et al. 1994; Fesenko et al. 1997). This time trend, which resulted in reduction of plant contamination, may be due to progressive fixation of radiocaesium in interlayer positions of clay minerals and of its slow diffusion and binding to frayed edge sites of clay minerals. This process reduces the exchangeability of radiocaesium so that is not then available to enter the soil solution from which plants take up most of the radiocaesium via the roots. For ^{90}Sr, an increase with time of the exchangeable fraction has been observed, which is attributed to the leaching of the fuel particles (Kashparov et al. 2004).

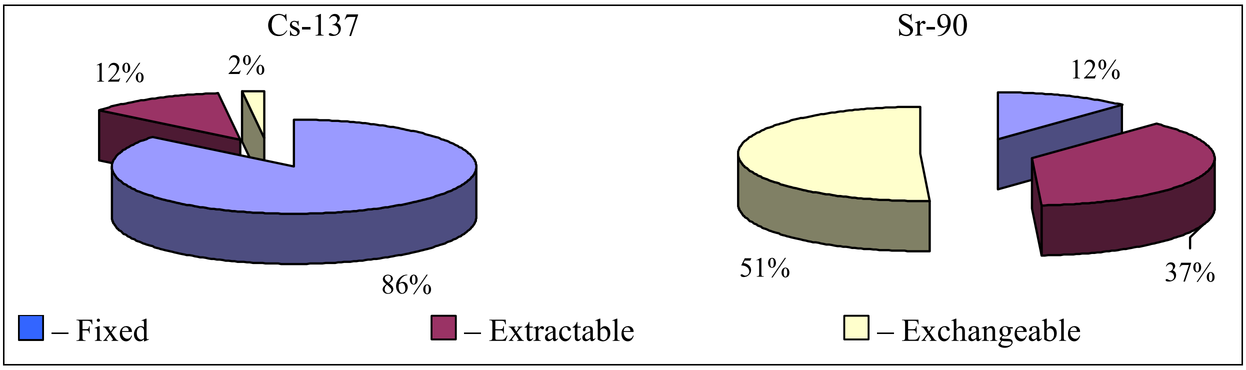


FIG. 2.19. Forms of presence of radionuclides in sod-podzolic loamy sand soil of the Gomel Oblast of Belarus in 1998 (Shevchouk and Gourachevsky 2001).

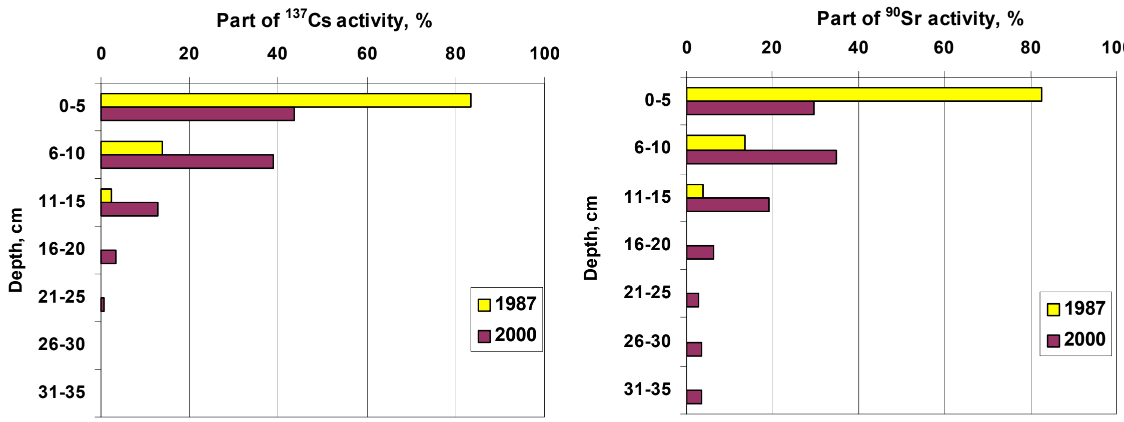


FIG. 2.20. Depth distributions of ^{137}Cs and ^{90}Sr measured in 1987 and 2000 in a soddy gley sandy soil (in % of total content) (Shevchouk and Gourachevsky 2001).

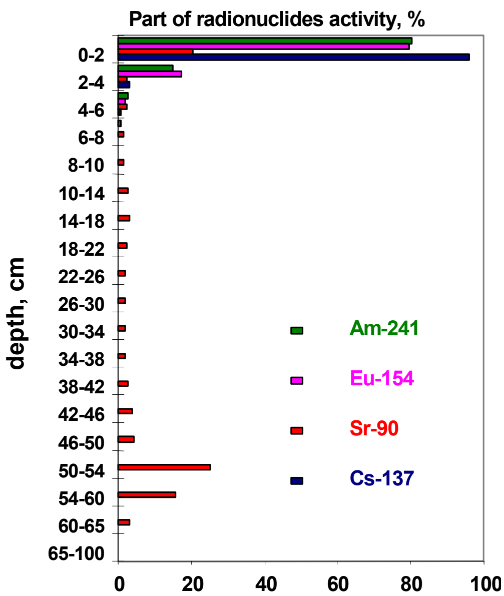


FIG. 2.21. Depth distributions of radionuclides measured in 1996 in low-humified sandy soil uncovered with sod (in % of total content) (Kashparov et al. 1999).

2.3.4.2. Migration of radionuclides in soil

Vertical migration of radionuclides down the soil column could arise from various transport mechanisms including convection, dispersion, diffusion and biological mixing. High root uptake of radionuclides in plants is correlated with high vertical migration, because in both processes the radionuclides are relatively mobile. Typically, the rate of movement of radionuclides will thus vary with soil type and physico-chemical form. As an example, Fig. 2.20 shows the change with time in the depth distributions of ^{90}Sr and ^{137}Cs measured in the Gomel Oblast of Belarus. Although there has been a significant downward migration of both radionuclides, much of the radionuclide activity has remained within the rooting zone of plants. At such sites where contamination occurred through atmospheric deposition, there is a low risk of radionuclide migration to groundwater.

The rate of migration down different types of soils varies for radiocaesium and radiostrontium. The lower rates of ^{90}Sr vertical migration are observed in peat soils, whereas ^{137}Cs migrates at the highest rate in these highly organic soils, but moves much more slowly in soddy-podzolic sandy soils. In dry meadows, migration of ^{137}Cs down from the root containing zone (0-10 cm) was hardly detectable in 10 years after the fallout. Thus, contribution of vertical migration to the decrease of ^{137}Cs-activity concentrations in the root-containing zone of mineral soils is negligible. On the contrary, in wet meadow and in peatland downward migration can be an important factor that reduces availability of ^{137}Cs for plants (Sanzharova et.al. 1996).

The higher rates of ^{90}Sr vertical migration are observed in low-humified sandy soil uncovered with sod (Fig.2.21), soddy-podzolic sandy and sandy loam soil with a low organic content of < 1% (Shestopalov et al. 2003). Generally, the highest rate of ^{90}Sr vertical migration is characteristic of non-equilibrium soil conditions. This occurs in flood-plains of rivers where soil isn't fully structurally formed (light humified sands), arable lands in a non equilibrium state, soils where the organic layers have been removed, for instance in forest fires and sites with deposited sand with a low content of organic matter (<1%). In such conditions there is a high rate of radiostrontium vertical migration to groundwater with convective moisture flow, and high activity in localised soil zones can occur. Thus, the spatial distribution of ^{90}Sr can be particularly heterogeneous in soils where there have been changes in sorption properties.

Agricultural practices have a major impact on radionuclide behaviour. Depending on the type of soil tillage and on the tools used, a mechanical redistribution of radionuclides in the soil may occur. On arable soils, radionuclides are distributed fairly uniformly along the whole depth of the tilled layer.

Lateral redistribution of radionuclides in catchments, which can be caused by both water and wind erosion, is significantly less than their vertical migration into a soil layer and geological environment (Shestopalov et al. 2003). The type and density of plant cover may significantly affect erosion rates. Depending on the intensity of erosive processes, the content of radionuclides in the arable layer on flat land with small slopes may increase by 75 % (Bogdevitch 2002).

2.3.4.3. Radionuclide transfer from soil to crops

The uptake of radionuclides, as well as of other trace elements by plant roots, is a competitive plant-physiological process (Ehlken and Kirchner 2002). For radiocaesium and radiostrontium, the main competing elements are potassium and calcium, respectively. The major processes influencing radionuclide-transport processes within the rooting zone are schematically represented in Fig. 2.22, although the relative importance of each component varies with radionuclide and soil type.

The fraction of deposited radionuclides taken up by plant roots differs by orders of magnitude depending primarily on soil type. For radiocaesium and radiostrontium, the radioecological sensitivity of soils can be broadly divided into the categories listed in Table 2.5. For all soil and plant species, root uptake of plutonium is negligible compared to direct contamination of leaves via rain splash or resuspension.

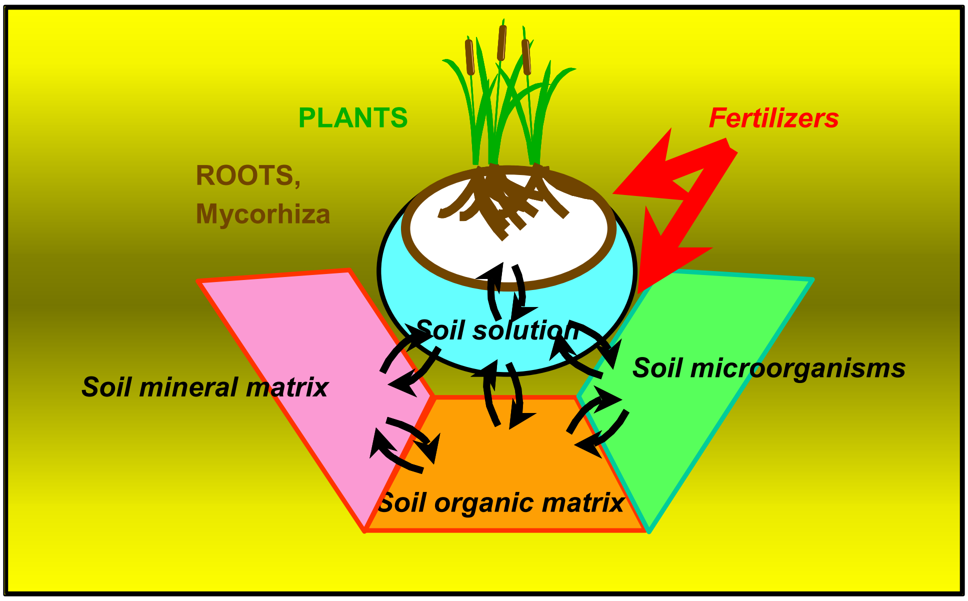


FIG. 2.22. Diagram of radionuclide pathways from soil to plants with consideration of biotic and abiotic processes (adapted from Tamponnet et al. 2001).

TABLE 2.5. CLASSIFICATION OF RADIOECOLOGICAL SENSITIVITY FOR SOIL-PLANT TRANSFER OF RADIOCAESIUM AND RADIOSTRONTIUM

For radiocaesium			
Sensitivity	Characteristics	Mechanism	Example
High	– low nutrient content – absence of clay minerals – high organic content	– little competition with potassium and ammonium in root uptake	Peat soils
Medium	– poor nutrient status, consisting of minerals including some clays	– limited competition with potassium and ammonium in root uptake	Podzol, other sandy soils
Low	– high nutrient status – considerable fraction of clay minerals	– radiocaesium strongly held to soil matrix (clay minerals) – strong competition with potassium and ammonium in root uptake	Chernozems and Pozlluvisol, Clay and loam soils (used for intensive agriculture)
For radiostrontium			
High	– low nutrient status – low organic matter content	– limited competition with calcium in root uptake	Podzol sandy soils
Low	– high nutrient status – medium to high organic matter content	– strong competition with calcium in root uptake	Umbric gley soils, Peaty soils

Transfer from soil to plants is commonly quantified using either the Transfer Factor (TF, dimensionless, equal to plant activity concentration, Bq kg^{-1}, divided by soil activity concentration, Bq kg^{-1}) or the aggregated transfer coefficient (T_{ag}, m^2 kg^{-1}, equal to plant activity concentration, Bq kg^{-1}, divided by soil deposition, Bq m^{-2}).

The highest ^{137}Cs uptake by roots from soil to plants occurs in peaty, boggy soils, and is one to two orders of magnitude higher than in sandy soils; this uptake often exceeds that of plants grown on fertile agricultural soils by more than three orders of magnitude. The high radiocaesium uptake from peaty soil became important after the Chernobyl accident, because in many European countries such soils are vegetated by natural unmanaged grassland used for the grazing of ruminants and the production of hay.

The amount of radiocaesium in agricultural products in the mid-to-long term depends not only on the density of contamination, but also on soil type, moisture regime, texture, agrochemical properties and the plant species. Agricultural activity often reduces transfer of radionuclides from soils to plant by physical dilution (e.g., ploughing) or by adding competitive elements (e.g., fertilizing). There are differences in radionuclide uptake with plant species. Although among-species variations in transfer between soil and plant may exceed one or more order of magnitude for radiocaesium, the impact of differing radioecological sensitivities of soils is often more important in explaining spatial variation in transfer in agricultural systems.

The influence of other factors, which have been reported to influence plant root uptake of radionuclides (e.g. soil moisture), is less clear or may be explained by the basic mechanisms discussed above. For example, accumulation of radiocaesium into crops and pastures is related to soil texture. On sandy soils, uptake of radiocaesium by plants is approximately twice as high as on loam, but this effect is mainly due to the lower concentrations of its main competing element, potassium, in sand.

The main process controlling plant-root uptake of radiocaesium, the main long-living constituent of the Chernobyl fallout, is the interaction between soil matrix and solution which depends primarily on the cation-exchange capacity of the soil. For mineral soils, this is influenced by the concentrations and types of clay minerals and the concentrations of competitive major cations, especially potassium and ammonium. Examples of these relationships are given in Fig. 2.23 for both radiocaesium and radiostrontium. Modelling of soil-solution physico-chemistry, which takes into account these major factors enables the prediction of plant-root uptake of both radionuclides (Sauras Yera et al. 1999; Konoplev et al. 2000).

Thus, differences in radioecological sensitivities of soils explain why in some areas of low deposition high concentrations of radiocaesium are found in plants and mushrooms harvested from semi-natural ecosystems and conversely why areas of high deposition can show only low to moderate concentrations of radiocaesium in plants. This is illustrated in Fig. 2.24, which shows the variability in activity concentrations of radiocaesium and radiostrontium in plants observed in the former Soviet Union for a normalised concentration in soil.

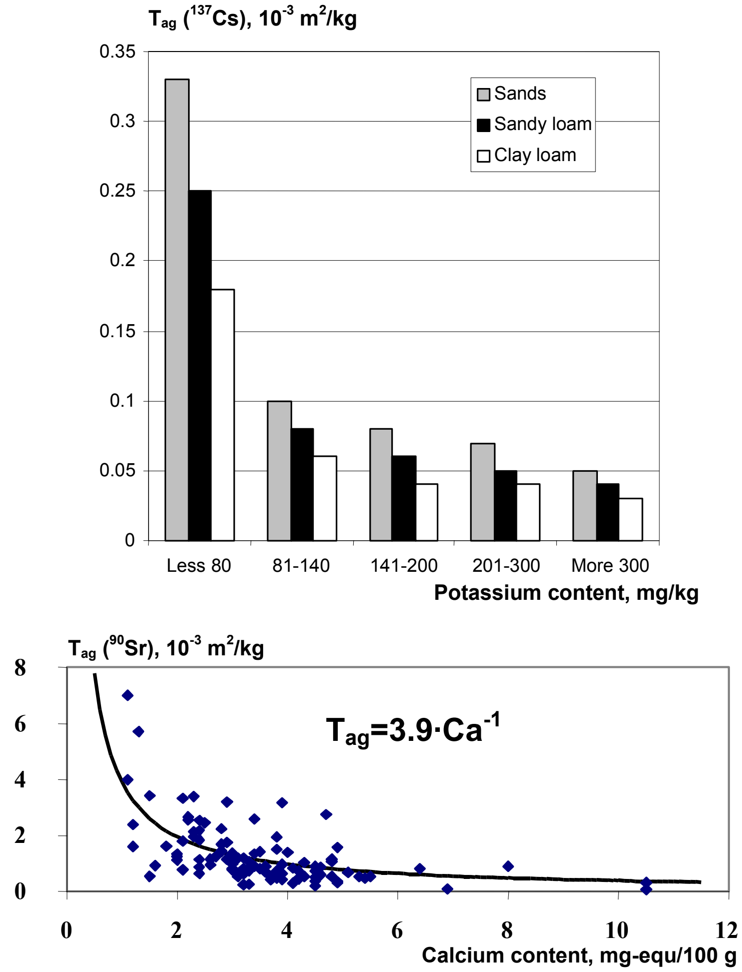


FIG. 2.23. Relationship between Cs and K and Sr and Ca in plant uptake (a) transfer of ^{137}Cs in oat grain in sod-podzolic soils of various textures with varying potassium content in soil (Bogdevitch 1999) and (b) ^{90}Sr transfer factor into seeds of winter rye with varying exchangeable calcium in different soils (Kashparov et al. 2001). [The latter reference is not in the list.]

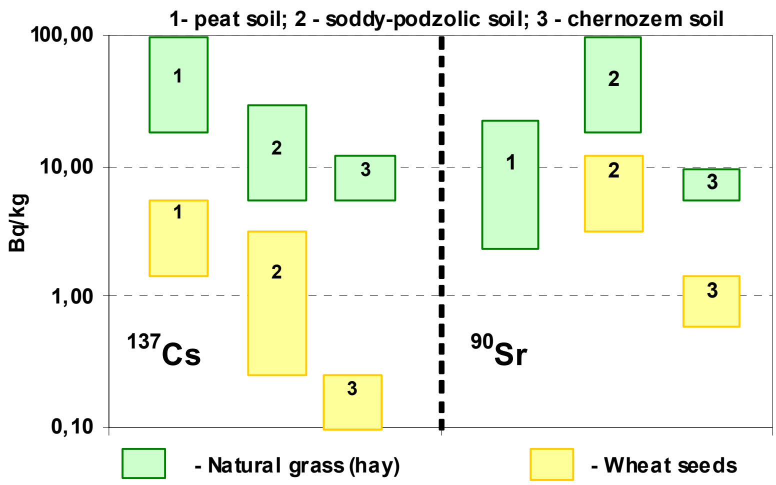


FIG. 2.24. Variability of the concentrations of ^{137}Cs and ^{90}Sr in two plant specimens with various soil types after 1996; for comparison, data refer to soil deposition of 1 kBq.m^{-2} (Deville-Cavelin et al. 2004).

2.3.4.4. Dynamics of radionuclide transfer to crops

In 1986, the ^{137}Cs content in plants was primarily determined by aerial contamination and reached its maximum value. During the first post-accident year (1987), the ^{137}Cs content in plants dropped by a factor of three to one hundred as root uptake from different soil types became the dominant contamination route.

For meadow plants in the first years after deposition, ^{137}Cs behaviour has been considerably influenced by radionuclide distribution between soil and mat. In this period, ^{137}Cs uptake from mat exceeded significantly (up to 8 times) that from soil. Further, as a result of mat decomposition and radionuclide transfer to soil, the contribution of mat decreased rapidly, and in the 5th year after the deposition it did not exceed 6% for automorphous soils and 11% for hydromorphous ones (Fesenko et al. 1996).

In most soils, the transfer rate of ^{137}Cs to plants has continued to decrease since 1987, although the rate of decrease has slowed, as can be seen from Fig 2.25 (Fesenko et al. 2004). A decrease with time similar to that shown in Fig. 2.25 has been observed in many studies of plant-root uptake with different crops, as can be seen in Figs. 2.26 and 2.27 for cereals and natural grasses, respectively, growing in two different soil types (Shutov et al. 2004). On Fig. 2.26, two more time-distant experimental points for chernozem soil (18 and 20 y) have been obtained from the measurements made in 1980-1985, i.e., after ^{137}Cs global fallout and before the Chernobyl accident. Values of ^{137}Cs-transfer factors for cereals as well as for potato and cow milk obtained about 20 years following global fallout do not significantly differ from these observed 8-9 years and later after Chernobyl fallout in remote areas with dominant sandy, sandy loam and chernozem soils (Bruk et al. 1998, Shutov et al. 2004). Difference between T_{ag} values relevant to cereals grown on fertilised soil is much lower than the difference for natural grasses.

For soil-to-plant transfer of radiocaesium, a decrease with time is likely to reflect (1) physical decay (2) the downward migration of the radionuclide out of the rooting zone and (3) physico-chemical interactions with the soil matrix that result in decreasing bioavailability. In many soils, ecological half-lives of plant-root uptake of radiocaesium could be characterised by two components: 1) Relatively fast decrease with a half life between 0.7 and 1.8 years, dominating for the first 4-6 years which led to reduction of concentrations in plants by about an order of magnitude compared with 1987; 2) a slower decrease with ecological half-life between 7 and 60 years (Fesenko et al. 1997; Bruk et al. 1998; Prister et al. 2003; Fesenko et al. 2004). The dynamics of the decrease of ^{137}Cs availability in the soil-plant system is considerably influenced by soil properties, and the rates of decreasing ^{137}Cs uptake by plants can differ by a factor of 3-5, being dependent on soil characteristics (Fesenko et al. 1996).

Some caution should be used, however, in generalising these observations, because some data show almost no decrease of root uptake of radiocaesium with time beyond the first 4-6 years, which suggests no reduction in bioavailability in soil within the time period of observation. Furthermore, quantifying ecological half lives that exceed the period of observation is highly uncertain. The successful application of countermeasures aimed at reducing the concentrations of radiocaesium in plants will also modify the ecological half-life.

Compared to radiocaesium, plant uptake of ^{90}Sr often has not shown such a marked decrease with time. In the areas close to the Chernobyl NPP gradual dissolution of fuel particles has enhanced the bioavailability of ^{90}Sr, and, therefore, there was an increase with time in ^{90}Sr uptake by plants, Fig. 2.28 (Kashparov et al. 2004). The difference in $^{90}Sr/^{137}Cs$ ratios was affected by plant species, rooting depth, soil pH, and calcium content in soil.

In remote areas, where strontium radionuclides were predominantly deposited in condensed form, and in lesser amounts as fine dispersed fuel particles, the dynamics of long-term transfer of ^{90}Sr to plants was similar to that of radiocaesium, however, with different ecological half-lives of plant-root uptake and their contributions. This difference reflected various mechanisms of soil transfer of these two elements. Regarding radiostrontium, its fixation by soil components less depends on clay content of soil, which is not the case for radiocaesium (see Table 2.4 above). More generally, values of ^{90}Sr-transfer parameters from soil to plants depend less on soil properties than for radiocaesium (Alexakhin and Korneev 1991). An example of the time dependence of ^{90}Sr uptake by plants is given in Fig. 2.29 (Shutov et al. 2004).

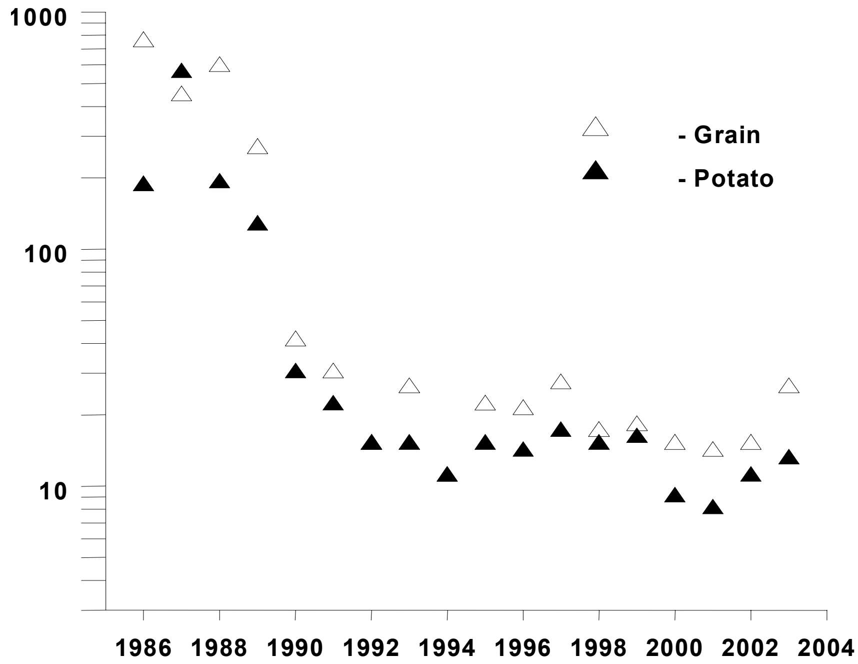


FIG. 2.25. Changes with time of ^{137}Cs concentrations in grain and potato, produced in contaminated districts of the Bryansk Oblast, Bq kg^{-1} (Fesenko et al. 2004).

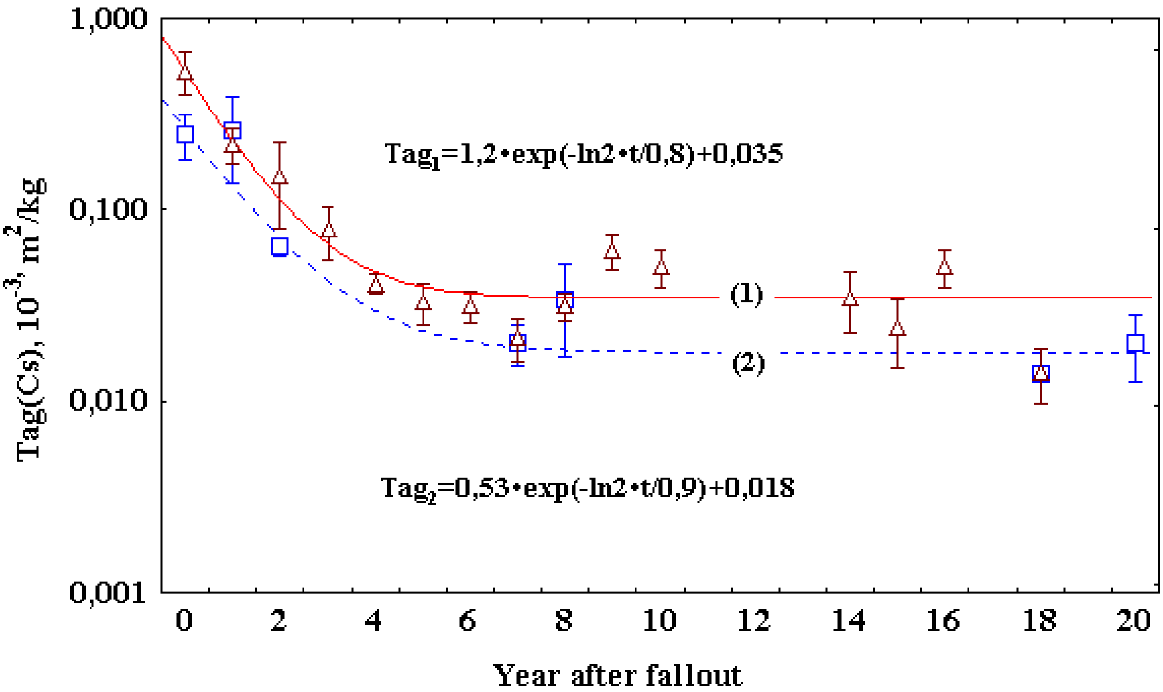


FIG. 2.26. Dynamics of [137]Cs aggregated transfer coefficients for cereals (1 - sandy and sandy loam soils, Bryansk Oblast; 2 – chernozem soil, Tula and Orel Oblasts of Russia) (Shutov et al. 2004).

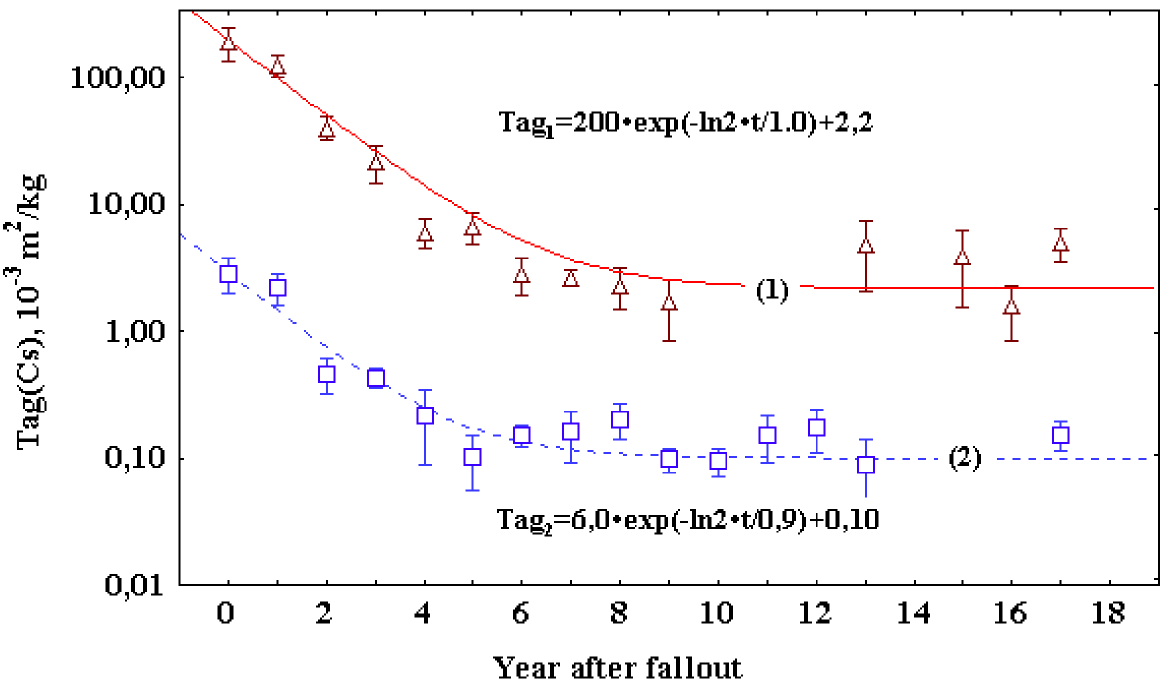


FIG. 2.27. Dynamics of [137]Cs aggregated transfer coefficients (dry weight) for natural grasses (1 - sandy and sandy loam soils, Bryansk Oblast; 2 – chernozem soil, Tula and Orel Oblasts of Russia) (Shutov et al. 2004).

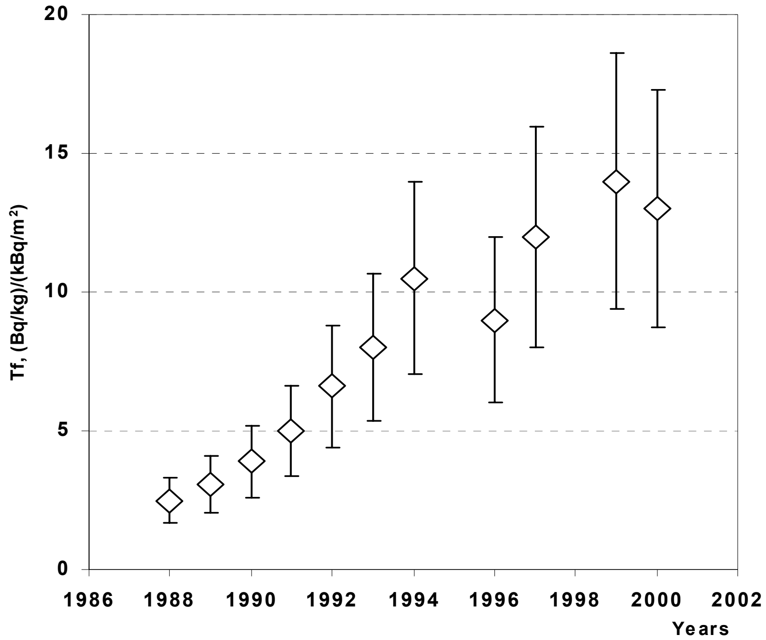


FIG. 2.28. Dynamics of ^{90}Sr-transfer factor into natural grass from soddy-podzolic soil in the 30-km zone of the ChNPP (Kashparov et al. 2004).

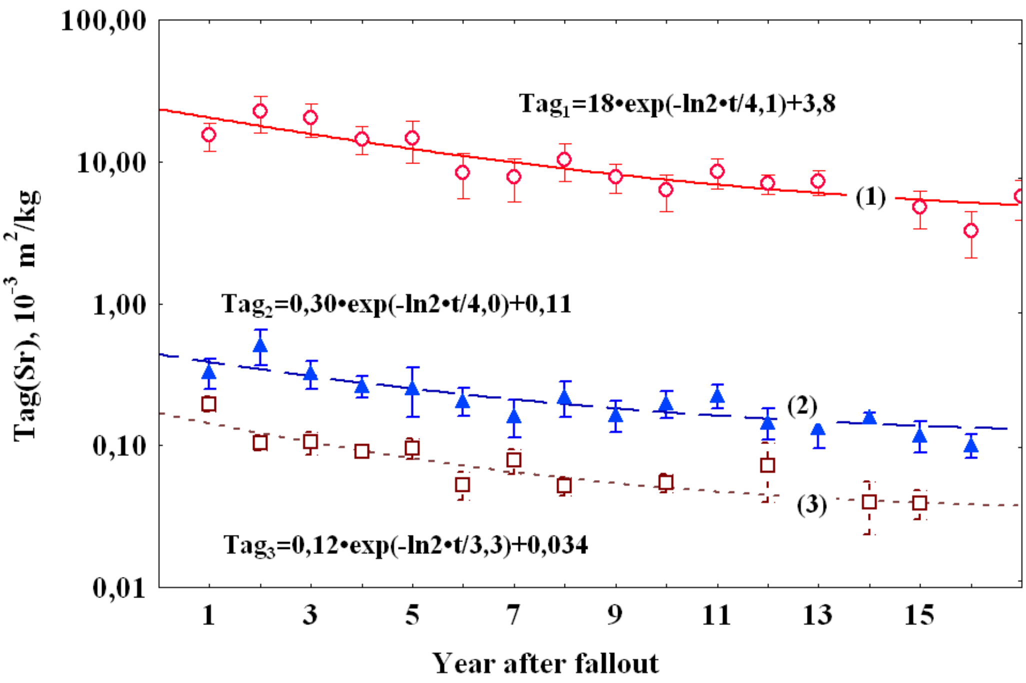


FIG. 2.29. Dynamics of ^{90}Sr aggregated transfer factor for natural grasses (1-sandy and sandy loam soils, Bryansk Oblast) and cow milk (2 - sandy and sandy loam soils, Bryansk region; 3 – chernozem soil, Tula and Orel Oblasts of Russia) (Shutov et al. 2004).

2.3.4.5. *Radionuclide transfer to animals*

Animals take up radionuclides through contaminated forage and direct soil ingestion. Milk and meat were major sources of human internal radiation dose after the Chernobyl accident both in the short term for ^{131}I and in the long-term for radiocaesium. In intensively managed agricultural ecosystems, high levels of contamination in animal-food products will often occur only for a few weeks, or at most a few months, after a pulse of fallout. In these circumstances, the extent of interception and retention on plant surfaces largely determines both the duration and level of contamination. The exception to a relatively short lived peak in contamination levels in animal products is where very high deposition occurs or where plant uptake is high and sustained, both of which occurred in some areas after the Chernobyl accident.

Contamination levels of radiocaesium in animal-food products from extensive ecosystems can be high and persist for a long time, even though the original deposition may not have been very high. This is because: (1) the soils often allow significant uptake of radiocaesium; (2) some species accumulate relatively high levels of radiocaesium, e.g., ericaceous species and fungi; and (3) these areas are often grazed by small ruminants that accumulate higher caesium-activity concentrations than do larger ruminants (Howard et al. 1991).

Radionuclide contamination of animal products depends on the behaviour of the radionuclide in the plant-soil system, absorption rate, metabolic fate in the animal and the rate of loss (principally in urine, faeces and milk). Although absorption can occur through the skin and lungs, oral ingestion of radionuclides in feed, and subsequent absorption through the gut, is the major route of entry of most radionuclides. Absorption of most nutrients takes place in the rumen or the small intestine at rates that vary from almost negligible, in the case of actinides, to 100% for radioiodine and varying from 60-100% for radiocaesium depending on the form (Beresford et al. 2000).

After absorption, radionuclides circulate in the blood. Some accumulate in specific organs, for instance, radioiodine accumulates in the thyroid and many metal ions including ^{144}Ce, ^{106}Ru, and ^{110m}Ag accumulate in the liver. Actinides and especially radiostrontium tend to be deposited in the bone, whereas radiocaesium is distributed throughout the soft tissues (Alexakhin and Korneev 1991; Beresford 1989; Beresford et al 1998; Crout et al. 2004).

The transfer of radionuclides to animal products is often described by transfer coefficients defined as the equilibrium ratio between the radionuclide activity concentration in milk, meat or eggs divided by the daily dietary radionuclide intake. Transfer coefficients of radioiodine and radiocaesium to milk, and of radiocaesium to meat, are generally lower for large animals, such as cattle, than for small animals, such as sheep, goats and hens. Transfer of radiocaesium to meat is higher than that to milk.

The long-term time-trend of radiocaesium contamination levels in meat and milk, an example of which is displayed in Fig. 2.30, follows that for vegetation and can be divided into two time periods (Bruk et al. 1998; Prister et al. 2003; Fesenko et al. 2004). For the first 4-6 years after deposition of radiocaesium there was an initial fast decrease with an ecological half-life between 0.8 and 1.2 years. For later times only a small decrease has been observed (Fesenko et al. 2004; Shutov et al. 2004).

There are differing rates of ^{137}Cs transfer to milk in areas with different soil types, which is demonstrated over nearly two decades after the accident in Fig. 2.31 in milk from the Bryansk and Tula Oblasts of Russia, where few countermeasures have been used. The transfer of ^{137}Cs to milk is shown quantified as the aggregated transfer coefficient (T_{ag}), which normalizes the data for different levels of soil contamination; this makes comparison among soil types easier. The transfer to milk declines in the order peat bog > sandy and sandy loam > chernozem and grey forest soils. Both the dynamics of ^{137}Cs-activity concentration in milk and its dependence on soil type are similar to those in natural grasses (see Fig. 2.27) sampled in areas where cattle graze.

Similar long-term data are available for comparing transfer of ^{137}Cs to beef in Russia for different soil types and also show higher transfer in areas with sandy/sandy loam soils compared with chernozem soils (Fig. 2.32); there has been little decline in ^{137}Cs transfer over the last decade.

Typical long-term dynamics of ^{90}Sr in cow milk sampled in Russian areas both with dominant soddy-podzolic and chernozem soils (see Fig. 2.29) are different from that of ^{137}Cs. The graphs for ^{90}Sr in milk do not contain the initial decreasing portion with an ecological half-life of about 1 y, as shown in graphs for ^{137}Cs; the latter is presumed to reflect fixation of caesium in soil matrix. In contrast, ^{90}Sr-activity concentration in cow milk gradually decreases with an ecological half-life of 3 to 4 years; the second component (if any) still has not been identified. The physical and chemical processes responsible for this time dynamics obviously include diffusion and convection with vertical transfer of ^{90}Sr into soil, as well as its radioactive decay. However, chemical interaction with the soil components might significantly differ from those known for caesium.

By combining information on radionuclide transfer with spatially varying information in geographic information systems (GIS), it is possible to identify zones in which a specified average-activity concentration of milk is likely to be exceeded. An example is shown in Fig. 2.33.

Extensive production in the former Soviet Union is largely confined to the grazing of privately owned cows on poor, unimproved meadows. Because of the poor productivity of these areas, radiocaesium uptake is relatively high compared to that on land used by collective farms. As an example of the difference between farming systems, changes in ^{137}Cs-activity concentrations in milk from private and collective farms in the Rovno Oblast, Ukraine, are shown in Fig. 2.34. The activity concentrations in private farms exceeded action levels until 1991, when countermeasures resulting in radical improvement were implemented.

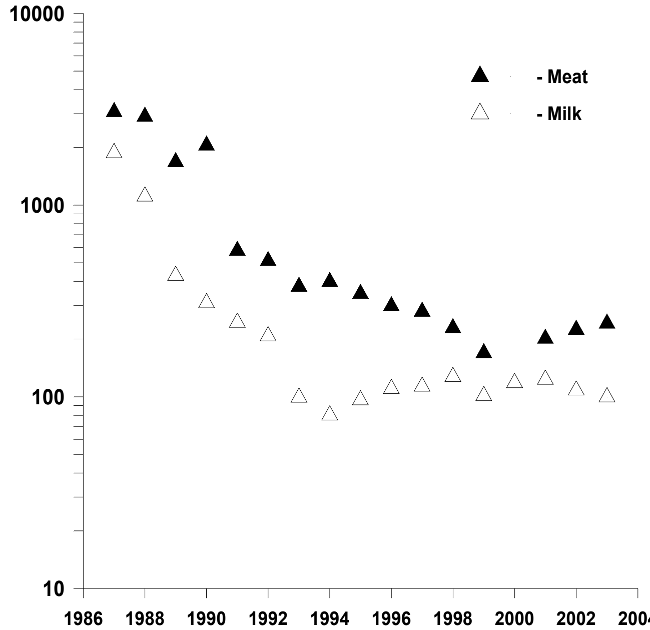


FIG. 2.30. Changes with time in mean ^{137}Cs-activity concentrations in meat and milk produced in contaminated districts of the Bryansk Oblast, Russia, Bq kg^{-1} (Fesenko et al. 2004).

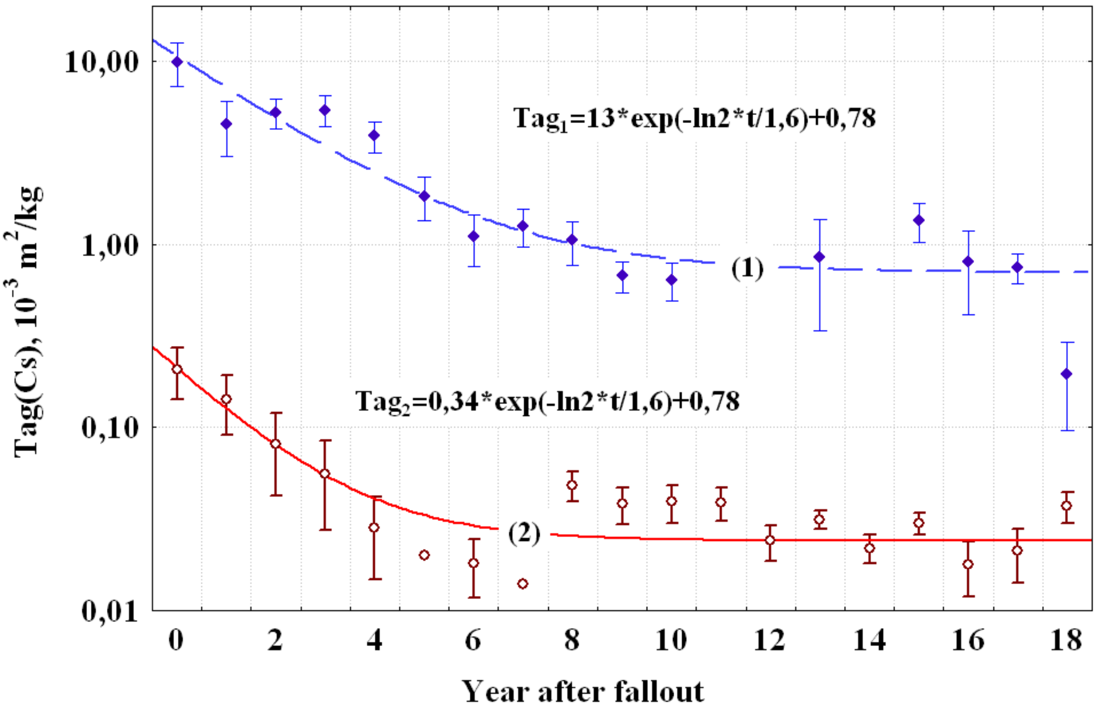


FIG. 2.31. (a). Dynamics of ^{137}Cs aggregated transfer factor for cow milk (1 – peat-bog soils, Bryansk Oblast; 2 – chernozem soil, Tula and Orel Oblasts of Russia) (Shutov et al. 2004).

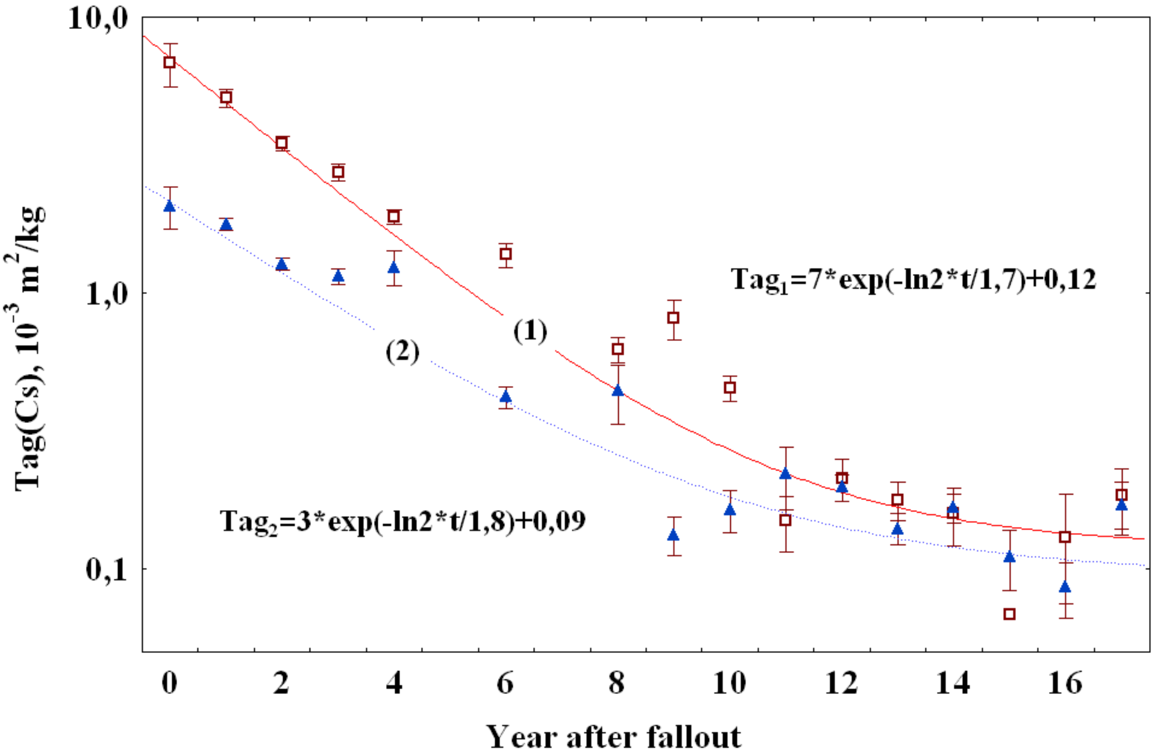


FIG. 2.31(b). Dynamics of ^{137}Cs aggregated transfer factor for cow milk (sandy and sandy-loam soils, Bryansk Oblast of Russia). 1 - ^{137}Cs soil deposition <370 kBq/m^2; 2 -^{137}Cs soil deposition >370 kBq/m^2 (Shutov et al. 2004).

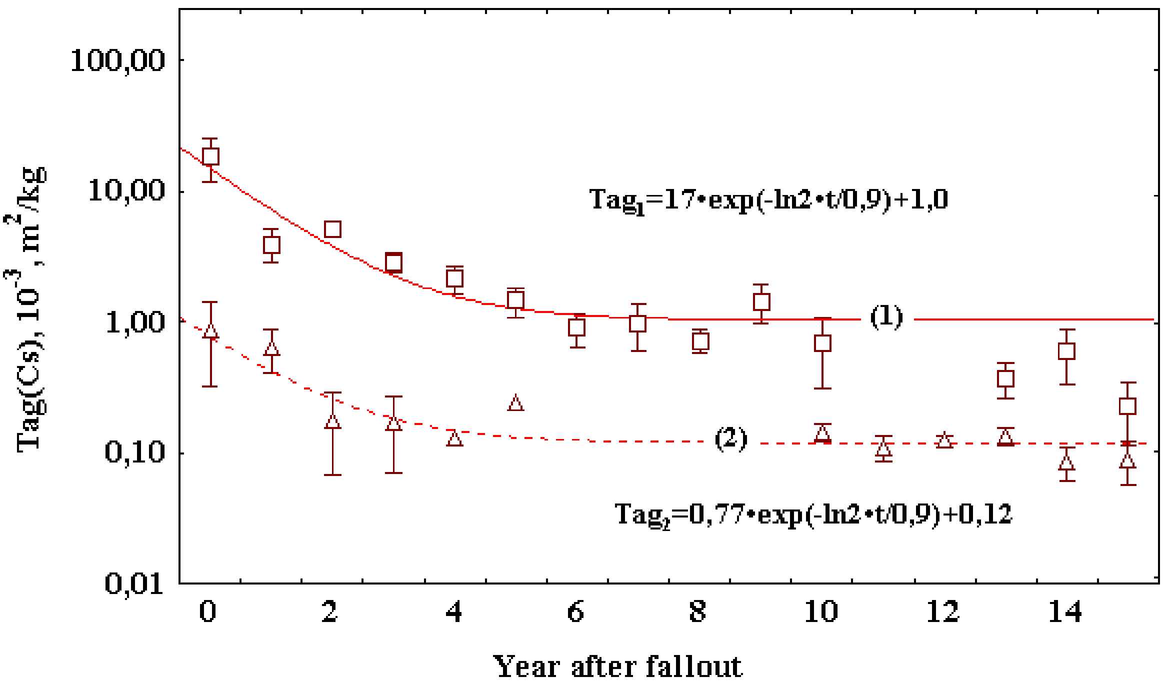


FIG. 2.32. Dynamics of ^{137}Cs aggregated transfer factors for beef (1 - sandy and sandy loam soils; 2 – chernozem soil) (Shutov et al. 2004).

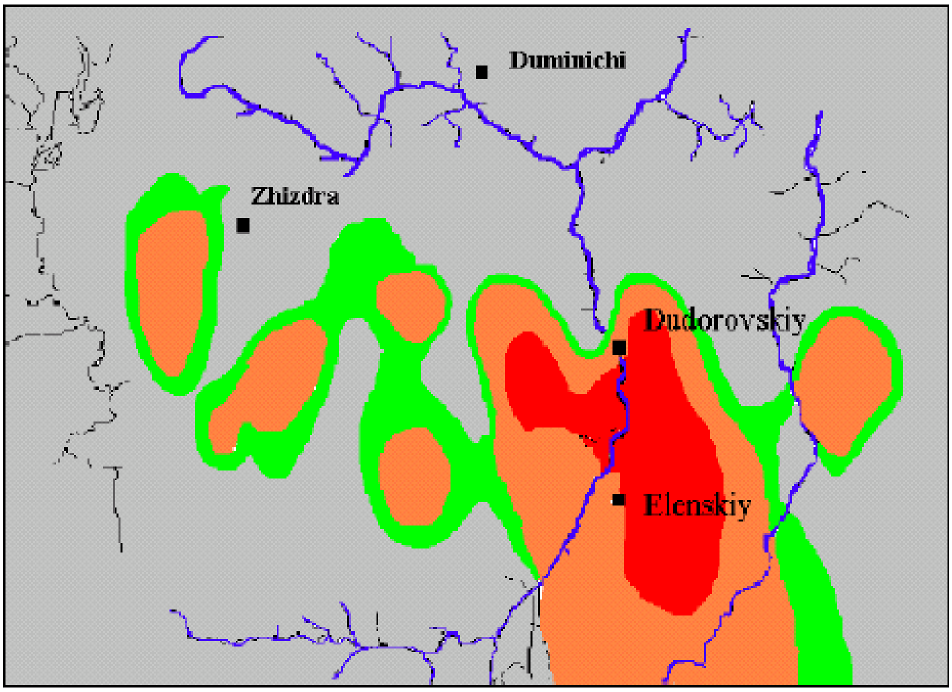


FIG. 2.33. A chart of isolines of different levels of probability for exceeding 100 BqL^{-1} of ^{137}Cs in milk (for 1991) in the Kaluga Oblast (Deville-Cavelin et al. 2004). Red colour means high probability, pink and green colours mean medium and low probabilities, correspondingly.

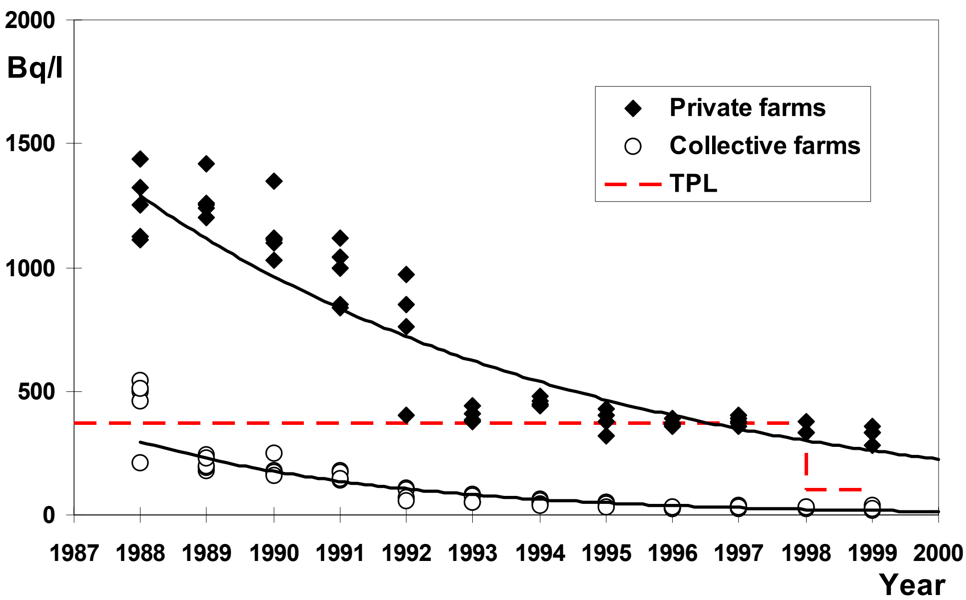


FIG. 2.34. Typical dynamics of ^{137}Cs-activity concentration in milk produced on private and collective farms of the Rovno Oblast of Ukraine with a comparison to the temporary permissible level (TPL) (Recommendations 1998).

2.3.5. *Current contamination of foodstuffs and expected future trends*

Summarised data of measured current (2000-2003) activity concentrations of radiocaesium in grain, potato, milk and meat produced in highly and less highly contaminated areas covering many different types of soil with widely differing radioecological sensitivities are given in Table 2.6 for Belarus, Russia and Ukraine. Cs-137 activity concentrations are consistently higher in animal products than in plant products.

TABLE 2.6. MEAN AND RANGE OF CURRENT ^{137}Cs-ACTIVITY CONCENTRATIONS IN AGRICULTURAL PRODUCTS ACROSS CONTAMINATED AREAS OF BELARUS (BOGDEVICH 2002), RUSSIA (FESENKO et al. 2004) AND UKRAINE (BEBESHKO et al. 2001); DATA ARE IN Bq kg^{-1} FRESH WEIGHT FOR GRAIN, POTATO AND MEAT AND IN Bq L^{-1} FOR MILK

^{137}Cs soil deposition range	Grain	Potato	Milk	Meat
Belarus				
>185 kBq m^{-2}, (contaminated districts of Gomel Oblast)	30 (8-80)	10 (6-20)	80 (40 – 220)	220 (80- 550)
37-185 kBq m^{-2}, (contaminated districts of Mogilev Oblast)	10 (4-30)	6 (3-12)	30 (10-110)	100 (40-300)
Russia				
>185 kBq m^{-2}, (contaminated districts of Bryansk Oblast)	26 (11-45)	13 (9-19)	110 (70-150)	240 (110-300)
37-185 kBq m^{-2}, (contaminated districts of Kaluga, Tula and Orel Oblasts)	12 (8-19)	9 (5-14)	20 (4- 40)	42 (12-78)
Ukraine				
>185 kBq m^{-2}, (contaminated districts of Zhitomir and Rovno Oblast)	32 (12-75)	14 (10-28)	160 (45-350)	400 (100-700)
37-185 kBq m^{-2}, (contaminated districts of Zhitomir and Rovno Oblasts)	14 (9-24)	8 (4-18)	90 (15-240)	200 (40-500)

Currently, due to natural processes and agricultural countermeasures, radiocaesium-activity concentrations in agricultural food products produced in areas affected by the Chernobyl fallout are generally below national, regional (EU) and international action levels (CAC 1989, IAEA 1996). However, in some limited areas with high radionuclide contamination (parts of Gomel and Mogilev Oblasts in Belarus and Bryansk Oblast in Russia) or poor organic soils (Zhitomir and Rovno Oblasts in Ukraine) radiocaesium-activity concentrations in food products, especially milk, still exceed national action levels of about one hundred Bq kg^{-1}. In these areas remediation may still be warranted – see Section 3 of this report.

Contaminated milk from private cows with ^{137}Cs-activity concentrations exceeding 100 Bq L^{-1} (the current permissible level for milk) was being produced in more than 400, 200 and 100 Ukrainian, Belarussian and Russian settlements, respectively, fifteen years after the accident. Levels of milk contamination higher than 500 Bq L^{-1} occur in 6 Ukrainian, 5 Belarussian and 5 Russian settlements (e.g., in 2001).

The concentrations and transfer coefficients shown in the above figures and tables show that there only has been a slow decrease in radiocaesium-activity concentrations in most plant and animal foodstuffs during the last decade. This indicates that radionuclides must be close to equilibrium within the agricultural ecosystems, although continued reductions with time would be expected due to continuing migration down the soil profile and radioactive decay, even if there was an equilibrium established between ^{137}Cs in the labile and non labile pools of soil. Given the slow current declines, and the difficulties in quantifying long-term effective half lives for currently available data because of high uncertainties, it is not possible to conclude that there will be any further substantial decrease over the next decades, except due to additional radioactive decay of both ^{137}Cs and ^{90}Sr with a half-lives of about 30 y.

Radionuclide-activity concentrations in foodstuffs can increase through fuel-particle dissolution, changes in the water table as a consequence of change of management of the currently abandoned land, or the cessation of the application of countermeasures.

2.4. Forest environment

2.4.1. Radionuclides in European forests

Forest ecosystems were one of the major semi-natural ecosystems contaminated as a result of fallout from the Chernobyl plume. The primary concern from a radiological perspective is the long-term contamination of the forest environment and its products with ^{137}Cs due to its 30-year half-life. Also, in the years immediately following contamination the shorter lived ^{134}Cs isotope was also significant. In forests, other radionuclides such as ^{90}Sr and the plutonium isotopes are of limited significance for humans, except in relatively small areas in and around the Chernobyl exclusion zone. As a result, most of the available environmental data concern ^{137}Cs behaviour and associated radiation doses. The emphasis of this sub-section is on the distribution of ^{137}Cs and relevant exposure pathways.

Forests are extensive natural resources, which provide an economic, nutritional, and recreational resource in many countries. Figure 2.35 shows the wide distribution of forests across the European continent. Following the Chernobyl accident substantial radioactive contamination of forests occurred in Ukraine, Belarus and Russia, and also in countries beyond the borders of the former Soviet Union, notably Finland, Sweden and Austria (see Figs. 2.5 to 2.7). The degree of forest contamination with ^{137}Cs in these countries ranged from >10 MBq m^{-2} in some locations to between 10 and 50 kBq m^{-2}, the latter range being typical of ^{137}Cs deposition in several countries of western Europe. In each of these countries, not only do forests provide an economic resource of major importance, but they are also at the heart of many social and cultural activities which, in some cases, have been curtailed due to contamination with ^{137}Cs from Chernobyl.

Since the Chernobyl accident, it has become apparent that natural decontamination of forests is proceeding extremely slowly. Net export of ^{137}Cs from forest ecosystems was less than 1% per year (Tikhomirov and Shcheglov 1994, Nylén, 1996), so it is likely that, without artificial intervention, it is the physical decay rate of ^{137}Cs that will largely influence the duration over which forests continue to be affected by the Chernobyl legacy. Despite the fact that the absolute natural losses of ^{137}Cs from the forest are small, recycling of radiocaesium within the forest is a dynamic process in which reciprocal transfers occur on a seasonal, or longer-term, basis between biotic and abiotic components of the ecosystem. To facilitate appropriate long-term management of forests a reliable understanding of these exchange processes is required. Much information on such processes has been obtained from experiments and field measurements, and many of these data have been used to develop predictive mathematical models (IAEA 2002).

This sub-section summarises the findings of studies conducted on the cycling of radiocaesium in forests in both the short- and long-term and the radiological exposure pathways to man.

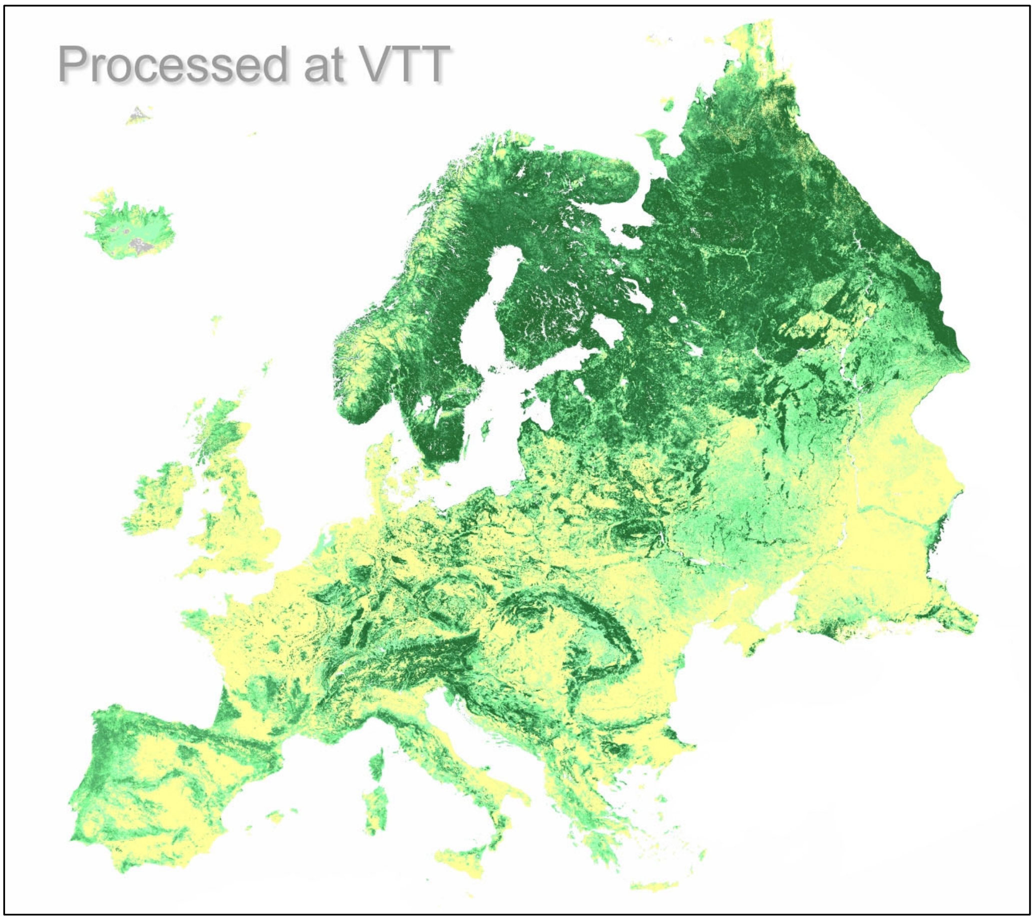


FIG. 2.35. Forest map of Europe. The darkest colour green indicates a proportion of 88% forest in the area while yellow indicates less than 10% (Shuck et al. 2003).

2.4.2. The dynamics of contamination during the early phase

Forests in the USSR located along the trajectory of the first radioactive plume were contaminated primarily as a result of dry deposition whilst further afield, in countries such as Sweden and Austria, wet deposition occurred and resulted in significant 'hot spots' of contamination. Other areas in the USSR, such as the Mogilev Oblast in Belarus, Bryansk and some other oblasts in Russia, were also contaminated by deposition with rain.

Tree canopies, particularly at forest edges, are efficient filters of atmospheric pollutants of all kinds. The primary mechanism of tree contamination after the Chernobyl accident was direct interception of radiocaesium by the tree canopy which intercepted between 60 and 90% of the initial deposition (Tikhomirov and Shcheglov 1994). Within a 7-km radius of the reactor this led to very high levels of contamination on the canopies of pine trees which, as a consequence, received lethal doses of radiation from the complex mixture of short and long-lived radionuclides released by the accident. Gamma-dose rates in the days and weeks immediately following the accident were in excess of 5 mGy h^{-1} in the area close to the reactor. The calculated absorbed gamma dose amounted to 80 to 100 Gy in the needles of pine trees. This small area of forest became known as the Red Forest as the trees died and became a reddish-brown colour, which was the most readily observable effect of radiation damage on organisms in the area, see Section 5.

Contamination of tree canopies reduced rapidly over a period of weeks to months due to wash-off by rainwater and the natural process of leaf/needle fall (Fig. 2.36). Absorption of radiocaesium by leaf surfaces also occurred, although this was difficult to measure directly. By the end of Summer 1986, approximately 15% of the initial radiocaesium burden in tree canopies remained, and by the summer of 1987 this had been further reduced to approximately 5%. Within this roughly one-year period, therefore, the bulk of radiocaesium was transferred from the tree canopy to the underlying soil.

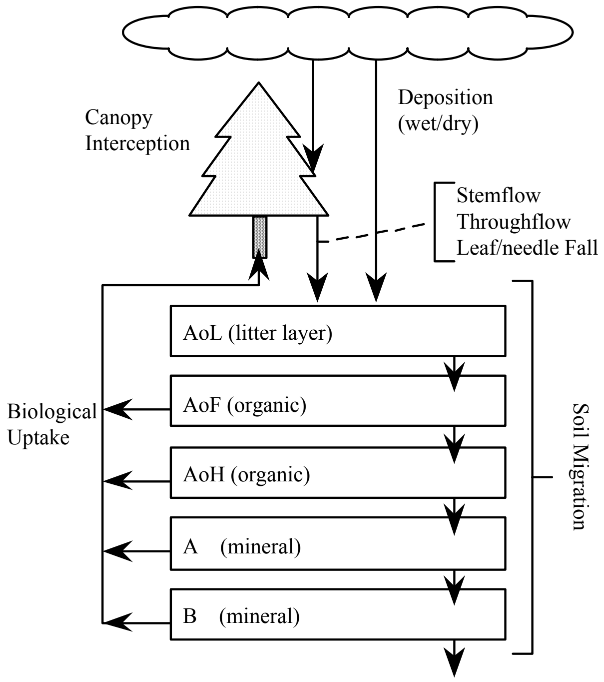


FIG. 2.36. Major storages and fluxes in radionuclides in contaminated forest ecosystems (Shaw et al. 2002).

During Summer 1986 radiocaesium contamination in natural products, such as mushrooms and berries, increased and this led to increasing body burdens in forest animals, such as deer and moose. In Sweden activity concentrations of ^{137}Cs in moose exceeded 2 kBq kg^{-1} fresh weight and those in roe deer were even higher (Johansson 1994).

2.4.3. The long-term dynamics of radiocaesium in forests

Within approximately one year after the initial deposition the soil became the major repository of radiocaesium contamination within the forest. This was followed by root uptake by trees and understorey plants which has prevailed over the longer term as radiocaesium has migrated into the soil profile. Just as in the case of its nutrient analogue, potassium, the rate of radiocaesium cycling within forests is rapid and a quasi-equilibrium of its distribution is reached a few years after atmospheric fallout (Shcheglov et al. 2001). The upper, organic-rich, soil layers act as a long-term sink but also as a general source of radiocaesium for contamination of forest vegetation, although individual plant species differ greatly in their ability to accumulate radiocaesium from this organic soil (Fig. 2.37).

Output from the system via drainage water is generally limited due to radiocaesium fixation on micaceous clay minerals (Nylén 1996). An important role of forest vegetation in the recycling of radiocaesium is the partial and transient storage of radiocaesium, particularly in perennial woody components such as tree trunks and branches which can have a large biomass. A portion of radiocaesium taken up by vegetation from the soil, however, is recycled annually through leaching and needle/leaf fall, resulting in the long-lasting biological availability of radiocaesium in surface soil. Storage of radiocaesium in the standing biomass of the forest amounts to approximately 5% of the total activity in the temperate forest ecosystem with the bulk of this activity residing in trees.

Due to biological recycling and storage of radiocaesium, migration within forest soils is limited and the bulk of contamination in the long-term resides in the upper organic horizons (Fig. 2.38). Slow downward migration of radiocaesium continues to take place, however, although the rate of migration varies considerably according to the soil type and climate.

The hydrological regime of forest soils is an important factor governing radionuclide transfer in forest ecosystems (Belli and Tikhomirov 1996). Depending on the hydrological regime the radiocaesium T_{ag} for trees, mushrooms, berries and shrubs can vary over a range of more than three orders of magnitude. The minimal T_{ag} values were found for automorphic (dry) forests and soils developed on even slopes under free surface-runoff conditions. The maximal T_{ag} values are related to hydromorphic ones developed under prolonged stagnation of surface waters. Among other factors influencing radionuclide transfer in forests, distribution of root systems (mycelia) in the soil profile and the capacity of different plants for radiocaesium accumulation is of importance (Fesenko et al. 2001).

The vertical distribution of radiocaesium within the soil has an important influence on the dynamics of uptake by herbaceous plants, trees and mushrooms. Another major consequence is a reduction in external gamma-dose rate with time, as the upper soil layers provide a shield as the peak of contamination migrates into the sub-surface (Fig. 2.39). The most rapid downward vertical transfer was observed for hydromorphic forests (Belli and Tikhomirov 1996).

Once forests become contaminated with radiocaesium, any further redistribution is limited. Processes of small scale redistribution include resuspension (Ould-Dada and Baghini 2001), fire (Amiro and Dvornik 1999) and erosion/runoff, although none of these processes is likely to result in any significant further migration of radiocaesium beyond the location of initial deposition.

2.4.4. Uptake into edible products

Edible products obtained from the forest include mushrooms, fruits and game animals: each of these products has become contaminated in forests affected by the Chernobyl deposition. The highest levels of contamination with radiocaesium have been observed in mushrooms, which is due to their great capacity to accumulate some mineral nutrients as well as radiocaesium. Mushrooms provide a common and significant food source in many of the affected countries, particularly in the countries of the former Soviet Union. Changes with time in contamination of mushrooms reflect the bioavailability of ^{137}Cs in the various relevant nutrient sources utilised by different species.

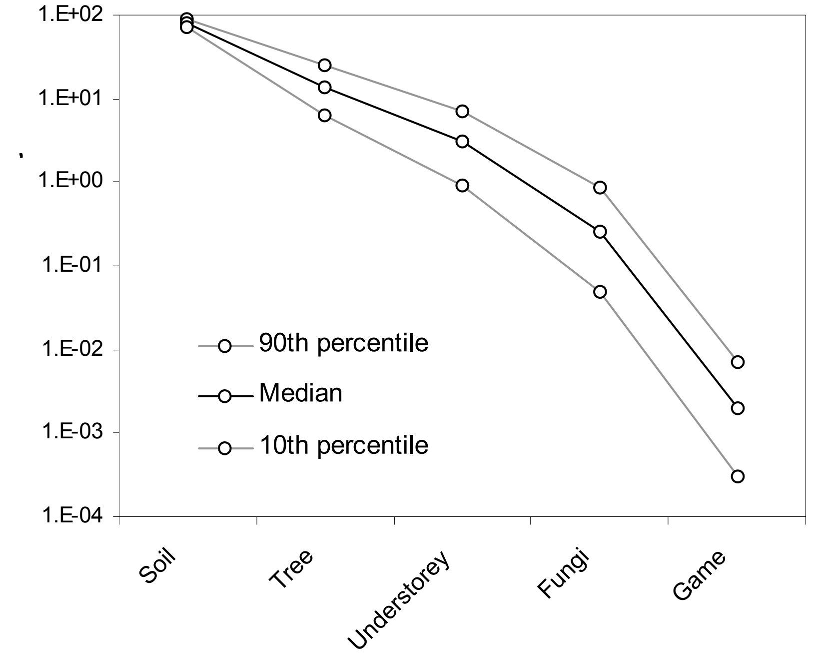


FIG. 2.37. Calculated percentage distributions of radiocaesium in specified components of coniferous forest ecosystems (Shaw 2004).

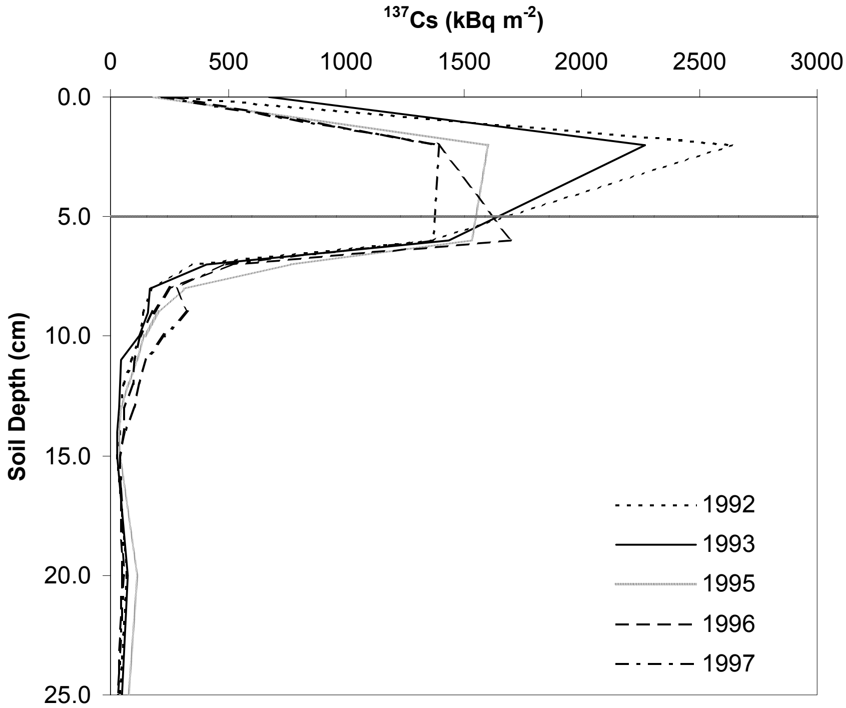


FIG. 2.38. Soil profiles of radiocaesium in a Scots Pine forest near Gomel in Belarus, 1992 to 1997 (Dvornik and Zhuchenko 2004). The horizontal line indicates the boundary between organic and mineral soil layers.

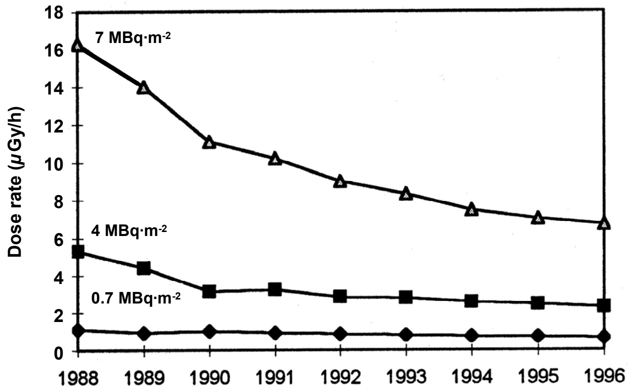


FIG. 2.39. Gamma-dose rates in air at three forest locations with different ^{137}Cs-soil deposition in the Bryansk Oblast of Russia, 150 km northeast of Chernobyl (Panfilov, 1999).

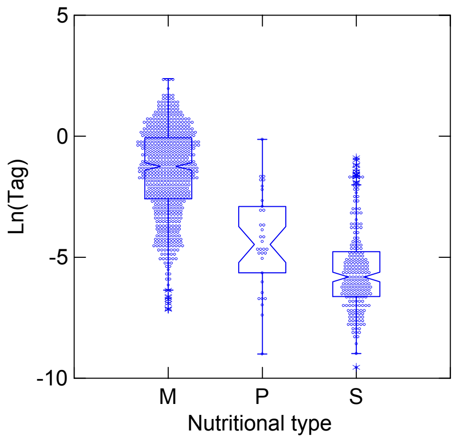


FIG. 2.40. Variation in log of the aggregated transfer coefficient to different nutritional types of mushrooms derived from the data presented in Barnett et al. (2001). M is mycorrhizal, P is parasitic, S is saproptrophic nutritional type.

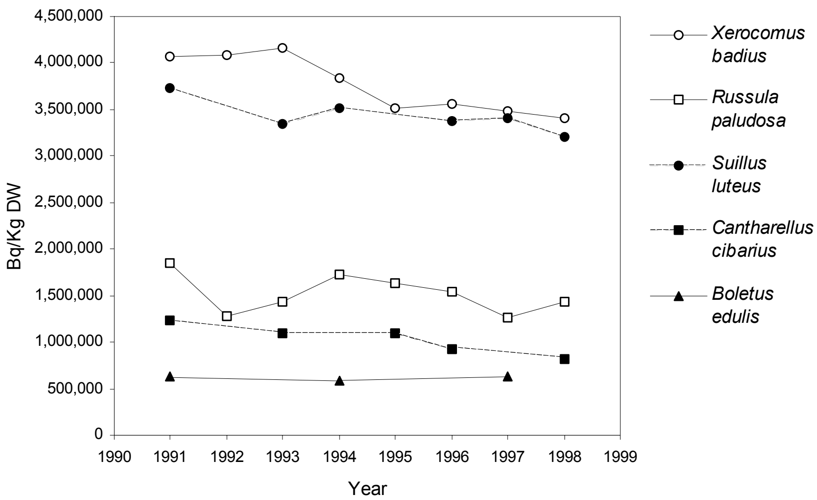


FIG. 2.41. ^{137}Cs-activity concentrations (Bq kg^{-1} dry weight) in selected mushroom species harvested in a pine forest in the Zhytomir Oblast of Ukraine, approximately 130 km southwest of Chernobyl. The soil deposition of ^{137}Cs at this site in 1986 was 555 kBq m^{-2} (data provided by A. Orlov, published in IAEA, 2002).

Some mushroom species exploit specific soil layers for their nutrition, and the dynamics of contamination of such species has been related to the contamination of these specific layers (Rühm et al. 1997). The high levels of contamination in mushroom species are reflected in generally high soil-mushroom transfer coefficients. However, these transfer coefficients (T_{ag}) are also subject to considerable variability and can range from 0.003 to 7 m^2 kg^{-1}, i.e., by a factor of approximately 2000 (IAEA 1994). Significant differences in accumulation of radiocaesium occur among species of mushrooms, which generally reflects the ecological niche that the individual species occupies (Fig. 2.40; Barnett et al 2001). In general, the saprotrophs and wood degrading fungi such as the honey fungus (*Armillaria mellea*) have a low degree of contamination while those fungi forming symbioses with tree roots (mycorrhizal fungi such as *Xerocomus* and *Lactarius*) have a high degree of uptake. The degree of variability of mushroom contamination is illustrated by Fig. 2.41, which also indicates the tendency for a slow decrease in contamination during the 1990s.

Contamination of mushrooms in forests is often much higher than that of forest fruits such as bilberries. This is reflected in the aggregated transfer coefficients for forest berries, which range from 0.02 to 0.2 m^2 kg^{-1} (IAEA 1994). Due to the generally lower radiocaesium levels and the masses consumed, forest berries pose a smaller radiological hazard to man than do mushrooms. However, both products contribute significantly to the diet of grazing animals and, therefore, provide a second route of exposure to humans via game. Animals grazing in forests and other semi-natural ecosystems often produce meat with high radiocaesium-activity concentrations. Such animals include wild boar, roe deer, moose and reindeer, but also domestic animals such as cows and sheep, which may graze marginal areas of forests.

Most data on contamination of game animals such as deer and moose have been obtained from western European countries in which the hunting and eating of game is commonplace. Significant seasonal variations occur in the body burden of radiocaesium in these animals due to the seasonal availability of foods such as mushrooms and lichens, the latter being particularly important as a component of the reindeer's diet. Particularly good time-series measurements have been obtained from the Nordic countries and Germany. Fig. 2.42 shows a complete time series of annual average radiocaesium activities for moose from 1986 to 2002 for one hunting area in Sweden, and Fig. 2.43 shows individual measurements of ^{137}Cs-activity concentrations in the muscle of roe deer in southern Germany. A major factor for the contamination of game, and roe deer in particular, is the high concentrations of radiocaesium in mushrooms. Aggregated transfer coefficients for moose range from 0.006 to 0.03 m^2 kg^{-1} (IAEA 1994). The mean aggregated transfer coefficient for moose in Sweden has been falling since the period of high initial contamination, indicating that the ecological half-life for radiocaesium in moose is less than 30 years, the physical half-life of ^{137}Cs.

2.4.5. Contamination of wood

Most forests affected by the Chernobyl accident in Europe and the former Soviet Union are planted and managed for the production of timber. Potentially, one of the major impacts of contamination of forests with radiocaesium is on the timber crop. Export of contaminated timber, and its subsequent processing and use, gives rise to radiological exposure pathways outside of the forest, including doses to people who would not normally be exposed in the forest itself. Uptake of radiocaesium from forest soils into wood is rather low; aggregated transfer factors range from 0.0003 to 0.003 m^2 kg^{-1}. Hence, wood used for making furniture or the walls and floors of houses is unlikely to give rise to significant radiological exposure of people using these products (IAEA 2003a). However, manufacture of consumer goods such as paper involves the production of both liquid and solid wastes that can become significantly contaminated with radiocaesium. Handling of these wastes by workers in paper-pulp factories can give rise to radiological doses within the industry (Ravila and Holm 1994).

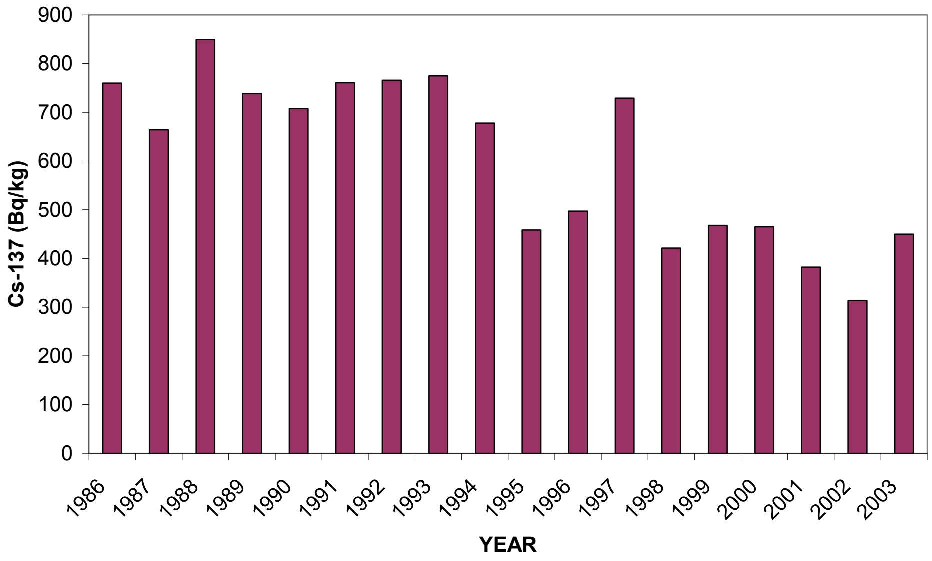


FIG. 2.42. The average concentration of ^{137}Cs in moose in one hunting area in Sweden based on approximately 100 animals per year (Johanson 2003).

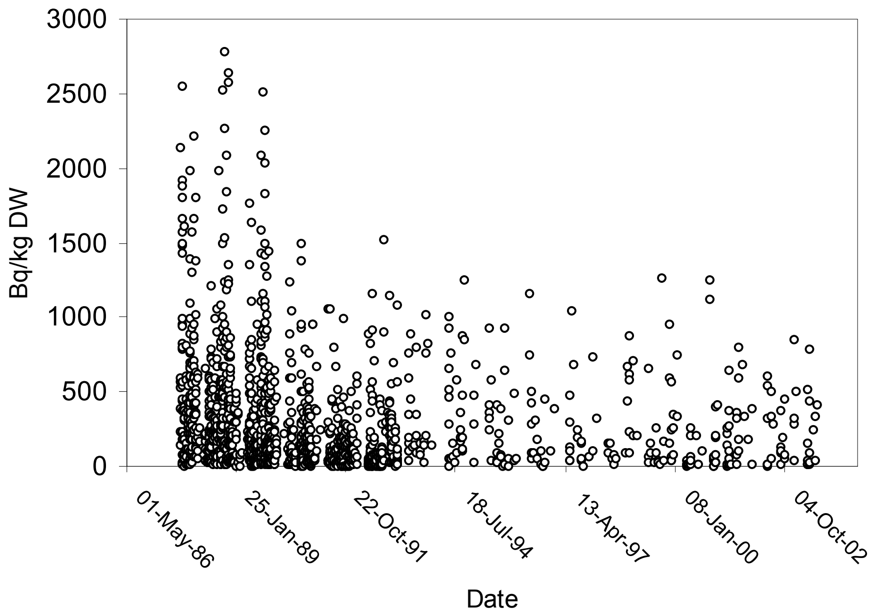


FIG. 2.43. ^{137}Cs-activity concentrations in the muscle of roe deer (Capreolus capreolus) harvested in a forest close to Bad Waldsee, southern Germany. The total deposition of ^{137}Cs at this site in 1986 was 27 kBq m^{-2} (Zibold 2004).

Use of other parts of trees such as needles, bark and branches for combustion may give rise to the problem of disposal of radioactive wood ash. This practice has increased in recent years due to the upsurge in biofuel technology in the Nordic countries and the problem of radiocaesium in wood ash is particularly notable because the radiocaesium-activity concentration in ash is a factor of 50-100 times greater than in the original wood. For domestic users of firewood in contaminated regions a build up of ash in the home and/or garden may give rise to external exposure to gamma radiation from radiocaesium (IAEA 2003a).

2.4.6. Expected Future Trends

Much effort has been put into developing mathematical models which make use of the large array of measurements of radiocaesium contamination in forests since 1986 (IAEA 2002). These models are useful in helping to improve our understanding of the way the Chernobyl contamination behaves in forest ecosystems. Furthermore, they can also be used to provide forecasts of future trends of contamination which can assist when making decisions about the future management of contaminated regions.

Predictive models of radiocaesium behaviour in forests aim to quantify the fluxes and distributions in the ecosystem over time. Forecasts can be made for specific ecological compartments such as the wood of trees and edible products such as mushrooms. Figures 2.44 and 2.45 show examples of such forecasts obtained using a variety of models. Fig. 2.44 shows predictions of the evolution of radiocaesium activity in wood for two distinct types of forest ecosystem with two age classes of trees. This illustrates the importance of both soil conditions and the stage of tree development at the time of deposition in controlling the contamination of harvestable wood. Fig. 2.45 shows a summary of 50-year forecasts for a pine forest in the Zhytomir Oblast of Ukraine, approximately 130 km southwest of Chernobyl. This illustrates the degree of variability among the predictions made with use of 11 different models and also the inherent variability within data collected from a single forest site. The uncertainty in both monitoring data and among models makes the task of forecasting future trends of forest contamination rather difficult. However, computer models help scientists to understand the ecological processes of radiocaesium transfer and to make indicative predictions.

2.4.7. Radiation Exposure Pathways Associated With Forests and Forest Products

Contaminated forests can give rise to radiation exposures of both workers in the forest and in associated industries, as well as members of the general public. Forest workers receive direct radiation exposure during their working hours due to the retention of radiocaesium in the tree canopy and the upper soil layers. Similarly, members of the public can receive external exposures from wood products, e.g., furniture or wooden floors, but in addition they may be exposed as a result of the consumption of game, wild mushrooms and berries containing radiocaesium. Forest margins may also be used to graze domestic animals such as cows and sheep; this can lead to their milk becoming contaminated and consequent dietary exposure. A further exposure pathway is the collection and use of firewood for domestic purposes which can give rise to exposures both in the home and in the garden if wood ash is used as a domestic fertilizer. Also, industrial use of forest products for energy production can give rise to exposure both to workers and to members of the public. Quantitative information on human doses associated with forest and forest products is given in (IAEA 2003a) and in Section 5 of this report.

Another set of important exposure pathways results from the harvesting, processing and use of timber and wood products from contaminated forest areas. Timber and wood products become sources of potential exposure once they are exported from the forest, often over considerable distances and sometimes across national borders. The relative importance of these exposure pathways has been evaluated and quantified (IAEA 2003a).

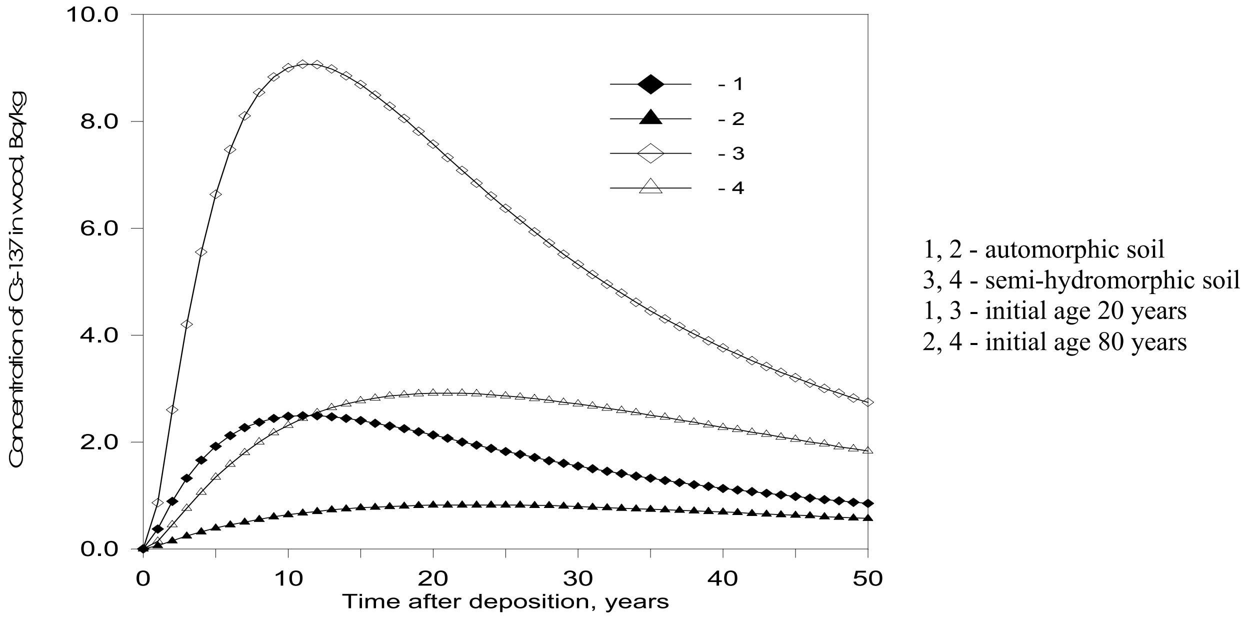


FIG. 2.44. Predicted ^{137}Cs-activity concentration in wood for different types of forest soil and ages of trees calculated using a computer model, FORESTLAND, for a deposition 1 kBq m^{-2} (Avila et al. 1999).

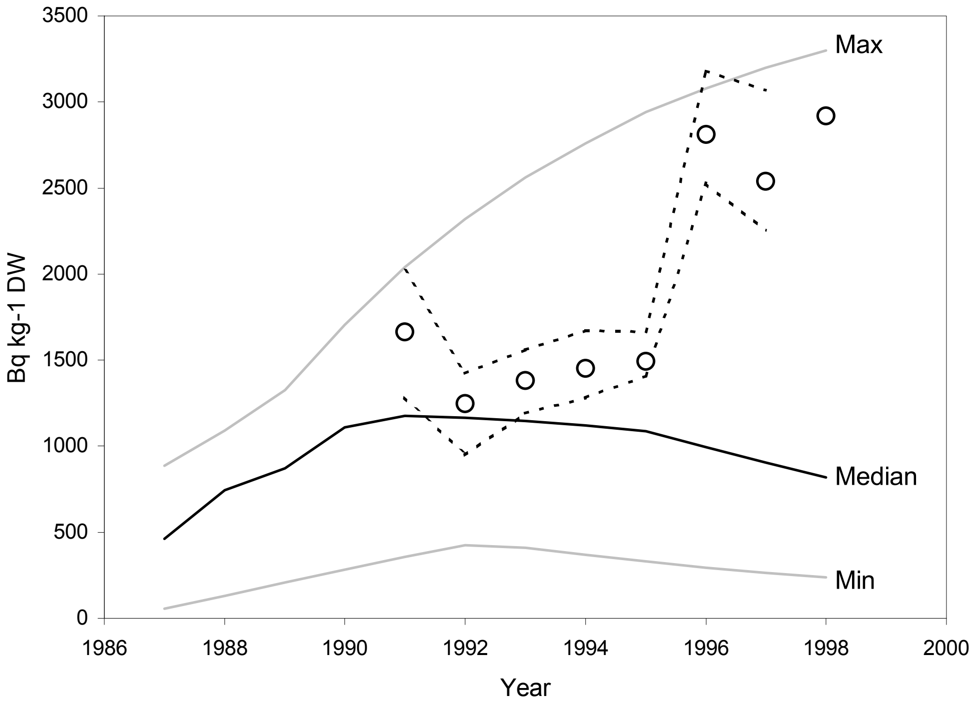


FIG. 2.45. Summary of predictions of pine wood contamination with ^{137}Cs in the Zhytomir Oblast of Ukraine made by use of 11 models within the IAEA's BIOMASS programme. ^{137}Cs-soil deposition was about 555 kBq m^{-2}. Max, Median and Min indicate maximum, median and minimum values of pooled model prediction, respectively. The points show means of measured values and the broken lines indicate the maximum and minimum values of measurements. (Shaw et al. 2004).

2.5. Radioactivity in aquatic systems

2.5.1. Introduction

Radioactivity from Chernobyl affected surface-water systems in many parts of Europe. The majority of radioactive fallout, however, was deposited in the catchment of the river Pripyat, which forms an important component of the Dnieper River-Reservoir system, one of the larger surface-water systems in Europe (De Cort et al. 1998). After the accident, therefore, there was particular concern over contamination of the water supply for the area along the Dnieper cascade of reservoirs covering a distance of approximately 1000 km to the Black Sea (see Figs 2.6 to 2.10). Other large river systems in Europe, such as the Rhine and Danube, were also affected by fallout though contamination levels in the rivers were not significant (UNSCEAR 1988, 2000).

Initial radioactivity concentrations in river water in parts of Ukraine, Russia and Belarus were relatively high, compared both to other European rivers and standards for radionuclides in drinking water, due to direct fallout onto river surfaces and runoff of contamination from the catchment area. During the first few weeks after the accident, however, activity concentrations in river waters rapidly declined, because of physical decay of short-lived isotopes and as radionuclide deposits became absorbed to catchment soils and bottom sediments. In the longer term, the long-lived ^{137}Cs and ^{90}Sr formed the major component of contamination of aquatic ecosystems. Though long-term levels of these radionuclides in rivers were low, temporary increases in activity concentrations during flooding of the Pripyat caused serious concern in areas using water from the Dnieper cascade.

Lakes and reservoirs were contaminated by fallout to the water surface and transfers of radionuclides from the surrounding catchment. Radionuclide concentrations in water declined rapidly in reservoirs and in those lakes with significant inflow and outflow of water ("open" lake systems). In some cases, however, activity concentrations of radiocaesium in lakes remained relatively high due to runoff from organic soils in the catchment. In addition, internal cycling of radiocaesium in "closed" lake systems (i.e., lakes with little inflow and outflow of water) led to much higher activity concentrations in their water and aquatic biota than were typically seen in open lakes and rivers.

Bio-accumulation of radionuclides (particularly radiocaesium) in fish resulted in activity concentrations (both in the most affected regions and in Western Europe) which were in some cases significantly above permissible levels for consumption (Kryshev and Ryabov 1990; Håkanson et al. 1992; Kryshev 1995; Ryabov et al. 1996; Zibold et al. 2002; Smith et al. 2000a). In some lakes in Russia, Belarus and Ukraine these problems have continued to the present day and may continue for the foreseeable future. Freshwater fish provide an important food source for many inhabitants of the contaminated regions. In the Dnieper cascade in Ukraine, commercial fisheries catch more than 20,000 tonnes of fish per year. In some parts of Western Europe, particularly parts of Scandinavia, radiocaesium-activity concentrations in fish are still relatively high (Jonsson et al. 1999).

The closest marine systems to Chernobyl are the Black Sea and the Baltic Sea, both several hundred kilometres from the site. Radioactivity in water and fish of these seas has been intensively studied since Chernobyl accident. Because average deposition to these seas was relatively low, and owing to the large dilution in marine systems, radioactivity concentrations were much lower than in freshwater systems (IAEA 2003b).

2.5.2. Radionuclides in surface waters

2.5.2.1. Distribution of radionuclides between dissolved and particulate phases

The interception of radionuclide fallout by catchment soils and river and lake sediments plays an important role in determining subsequent transport in aquatic systems. The fraction of a radionuclide which is absorbed to suspended particles, which varies considerably in surface waters, strongly influences both its transport and bioaccumulation. Most ^{90}Sr occurs in the dissolved phase (0.05 - 5% in the solid phase), but in the near zone, a significant proportion of strontium fallout was in the form of fuel particles. The soils of the Chernobyl Exclusion Zone are heavily contaminated with ^{90}Sr (see Fig. 6.7) and some is washed off during flood events when the low lying areas become inundated.

In the Pripyat River during the first decade after the accident approximately 40-60% of radiocaesium was found in the particulate phase (Voitsekhovich et al. 1997), but estimates in other systems (Comans et al. 1999) vary from 4 – 80%, depending on the composition and concentration of suspended particles, and water chemistry. Fine clay and silt particles absorb more radioactivity than larger, less reactive sand particles. Sandy river beds, even close to the reactor, were relatively uncontaminated, but fine particles could transport radioactivity relatively large distances. Settling of fine particles in the deep parts of the Kiev Reservoir led to high levels of contamination of bed sediments (Voitsekhovitch et al. 1991).

Measurements of the distribution of radionuclides between dissolved and particulate phases in Pripyat River water showed that the strength of absorption to suspended particles increases in the following order: ^{90}Sr, ^{137}Cs, transuranic elements ($^{239,240}Pu$, ^{241}Am) (Matsunaga et al. 1998). There is a possibility that natural organic colloids may determine the stability of transuranic elements in surface water and their transport from contaminated soil, while colloids have less effect on ^{90}Sr and ^{137}Cs (Matsunaga et al. 2004).

In marine systems, generally lower particle-sorption capacities and higher concentrations of competing ions (i.e., higher salinity) tend to make particle sorption of radionuclides less significant than in freshwaters. In the Baltic Sea after the Chernobyl accident, less than 10% of ^{137}Cs was bound to particles and the average particulate sorbed fraction was approximately 1%, (Carlson and Holm 1992; Knapinska-Skiba et al. 2001). For the Black Sea the particulate bound fraction of ^{137}Cs was less than 3% (IAEA 2003b).

2.5.2.2. Radioactivity in rivers

Initial radioactivity concentrations in rivers close to Chernobyl (the Pripyat, Teterev, Irpen and Dnieper) were largely due to direct deposition of radioactivity on the river surface. The highest concentrations of radionuclides were observed in the Pripyat River at Chernobyl, where the ^{131}I-activity concentration was up to 4440 Bq L^{-1} (Table 2.7). In all water bodies, the radioactivity levels declined rapidly during the first few weeks due to decay of short-lived isotopes, and absorption of nuclides to catchment soils and river bed sediments.

Over longer time periods after fallout, relatively long-lived ^{90}Sr and ^{137}Cs retained in catchment soils are slowly transferred to river water by erosion of soil particles and by desorption from soils. The rates of transfer are influenced by the extent of soil erosion, the strength of radionuclide binding to catchment soils and migration down the soil profile. An example time series of ^{90}Sr- and ^{137}Cs-activity concentrations in the water of the Pripyat River near Chernobyl is shown in Fig. 2.46.

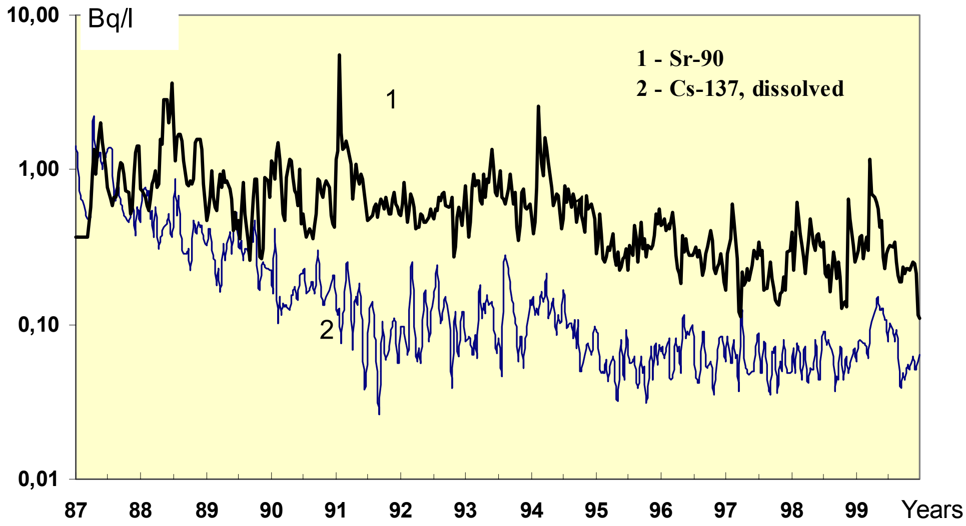


FIG. 2.46. Monthly averaged ^{90}Sr and ^{137}Cs activity concentrations in the Pripyat River (UHMI 2004).

After the Chernobyl accident, water-monitoring stations were established within the exclusion zone and along the major rivers to determine the concentration of radionuclides and their total fluxes. Measurements from these stations allow estimates to be made of radionuclide fluxes of ^{90}Sr and ^{137}Cs into and out of the exclusion zone. The migration of ^{137}Cs has decreased markedly with time and shows relatively little change from upstream to downstream of the exclusion zone (see Fig. 2.47).

In contrast, the transboundary migration of ^{90}Sr has fluctuated yearly depending on the extent of annual flooding along the banks of the Prypiat River (see lower chart in Fig. 2.47). There is also a significant flux from the exclusion zone - fluxes downstream of the zone are much higher than those upstream. Note, however, that the extent of wash out of radionuclides by the river system is only a very small percentage of the total inventory contained in the catchment area.

Declines in ^{90}Sr- and ^{137}Cs-activity concentrations occurred at a similar rate for different rivers in the vicinity of Chernobyl and in rivers in Western Europe (Monte 1995). Measurements of ^{137}Cs-activity concentration in different European rivers (Fig. 2.48) show a range of approximately a factor of 30, even when differences in fallout have been accounted for. In small catchments (Hilton et al. 1993; Nylén, 1996; Kudelsky et al. 1996) highly organic soils (particularly saturated peat soils) released up to an order of magnitude more radiocaesium to surface waters than some mineral soils. Thus, rivers in Finland with large areas of wet organic soils in the catchment have higher radiocaesium concentrations (per unit of radioactive fallout) than rivers with predominantly mineral catchments (Saxén and Illus 2001; Smith et al. 2004).

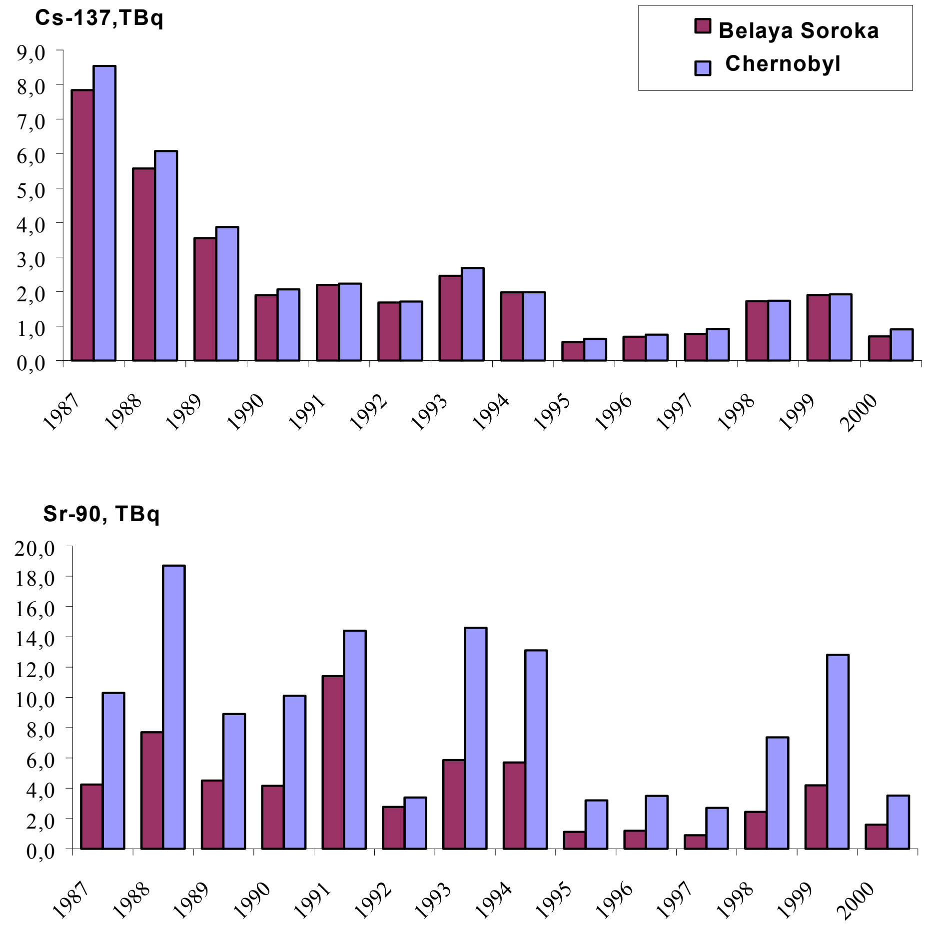


FIG. 2.47. Annual fluxes of ^{137}Cs and ^{90}Sr in the Prypiat River at Belaya Soroka, near the Belarus-Ukraine border (inlet to the Chernobyl Exclusion Zone) and downstream of Chernobyl (outlet of the Chernobyl Exclusion Zone) (IAEA 2005).

TABLE 2.7. MAXIMUM RADIONUCLIDE-ACTIVITY CONCENTRATIONS (DISSOLVED PHASE) MEASURED IN THE PRIPYAT RIVER AT CHERNOBYL (VAKULOVSKY ET AL. 1990; 1994; KRYSHEV 1995)

Radionuclide	Maximum conc. in Pripyat River, Bq L^{-1}	Radionuclide	Maximum conc. in Pripyat River, Bq L^{-1}
^{137}Cs	1591	^{106}Ru	271
^{134}Cs	827	^{144}Ce	380
^{131}I	4440	^{141}Ce	400
^{90}Sr	30	^{95}Zr	1554
^{140}Ba	1400	^{95}Nb	420
^{99}Mo	670	^{241}Pu	33
^{103}Ru	814	$^{239+240}Pu$	0.4

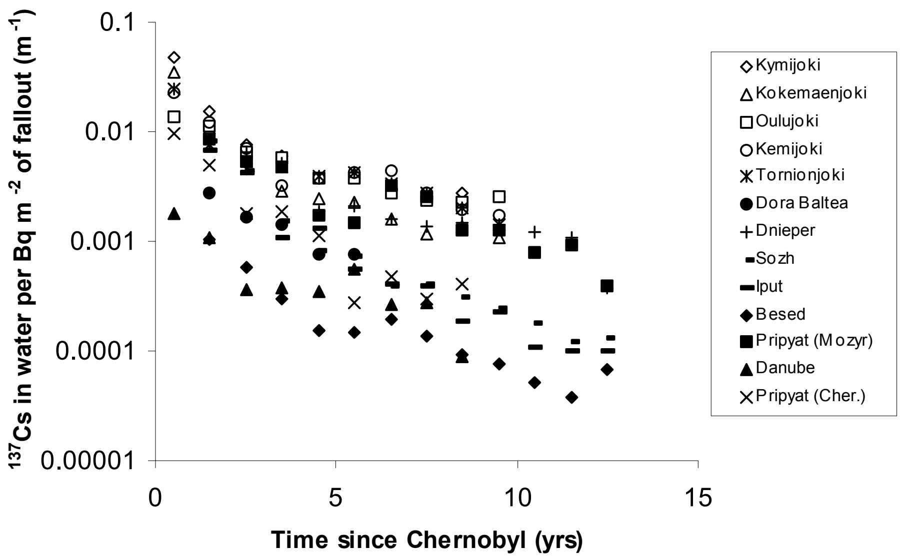


FIG. 2.48. ^{137}Cs-activity concentration in different rivers per unit of deposition (Saxén and Ilus 2001; Kudelsky et al. 1998; Smith et al. 2004).

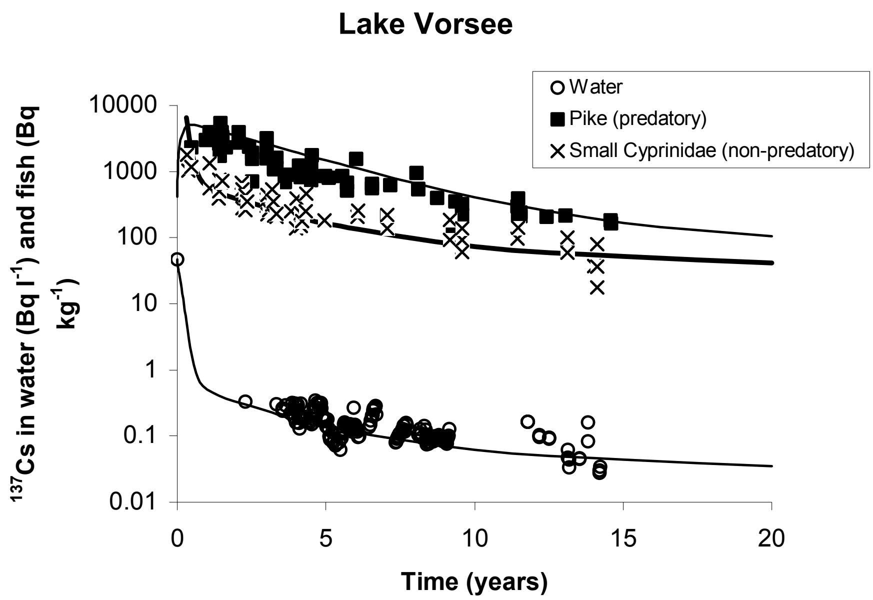


FIG. 2.49. Time series of ^{137}Cs in Lake Vorsee, (Germany) (Zibold et al. 2002).

2.5.2.3. *Radioactivity in lakes and reservoirs*

In the affected areas of Ukraine, Russia and Belarus, there were many lakes which were significantly contaminated by radionuclides. In most lakes, radionuclides were well mixed throughout the lake water during the first days to weeks after fallout occurred. In deep lakes such as Lake Zurich (mean depth 143 m), however, it took several months for full vertical mixing to take place (Santschi et al. 1990). In some areas of Northern Europe, lakes were covered in ice at the time of the accident, so maximum activity concentrations in lake waters were only observed after the ice melted.

Radionuclides deposited to a lake or reservoir are removed through water outflow, and by transfers to the lake-bed sediments. As in rivers, radiocaesium-activity concentrations in lakes declined initially relatively rapidly in lakes during the first weeks to months after fallout. This was followed by slower declines over a period of years as radiocaesium became more strongly absorbed to catchment soils and lake sediments, as well as migrating to deeper layers in the soil and sediments. Fig. 2.49 illustrates the temporal change in ^{137}Cs-activity concentration in lakes using measurements from Lake Vorsee, a small shallow lake in Germany.

Inputs to the lake also result from transport of radionuclides from contaminated catchment soils. In the longer term ("secondary phase"), ^{137}Cs-activity concentrations in Vorsee remained much higher than in most other lakes due to inputs of ^{137}Cs from organic soils in the catchment and remobilisation from bed sediments (Fig.2.49; Zibold et al. 2002). In Devoke Water (UK), radioactivity flowing from organic catchment soils maintained activity concentrations in the water which were approximately an order of magnitude higher than in nearby lakes with mineral catchments (Hilton et al. 1993). In some cases, lakes in Western Europe with organic catchments had activity concentrations in water and fish similar to those in some lakes in the more highly contaminated areas in Ukraine and Belarus.

Long-term contamination can also be caused by remobilisation of radionuclides from bed sediments (Comans et al. 1989). In some shallow lakes where there is no significant surface inflow and outflow of water, the bed sediments play a major role in controlling radionuclide-activity concentration in the water. Such lakes have been termed "closed" lakes (Vakulovsky et al. 1994; Bulgakov et al. 2002). The more highly contaminated water bodies in the Chernobyl affected areas are the closed lakes of the Pripyat flood plain within the 30-km zone. During 1991, ^{137}Cs-activity concentrations in these lakes were up to 74 Bq L^{-1} (Lake Glubokoye) and ^{90}Sr-activity concentrations were between 100 and 370 Bq L^{-1} in 6 of the17 studied water bodies (Vakulovsky et al. 1994). Seventeen years after the accident, there were still relatively high activity concentrations in the closed lakes in the Chernobyl exclusion zone (Kuzmenko et al. 2001) and at quite large distances from the reactor.

For example, during 1996 lakes Kozhanovskoe and Svyatoe in the Bryansk Oblast of Russia (approximately 200 km from Chernobyl) contained 0.6-1.5 Bq L^{-1} of ^{90}Sr and 10-20 Bq L^{-1} of ^{137}Cs (Fig. 2.50). Activity concentrations in water were higher than in many lakes close to Chernobyl, because of remobilisation from sediments in these "closed" lakes (Bulgakov et al. 2002). The Russian intervention level (IL) for drinking water of 11 Bq L^{-1} for ^{137}Cs (Ministry of Health 1999) is shown for comparison.

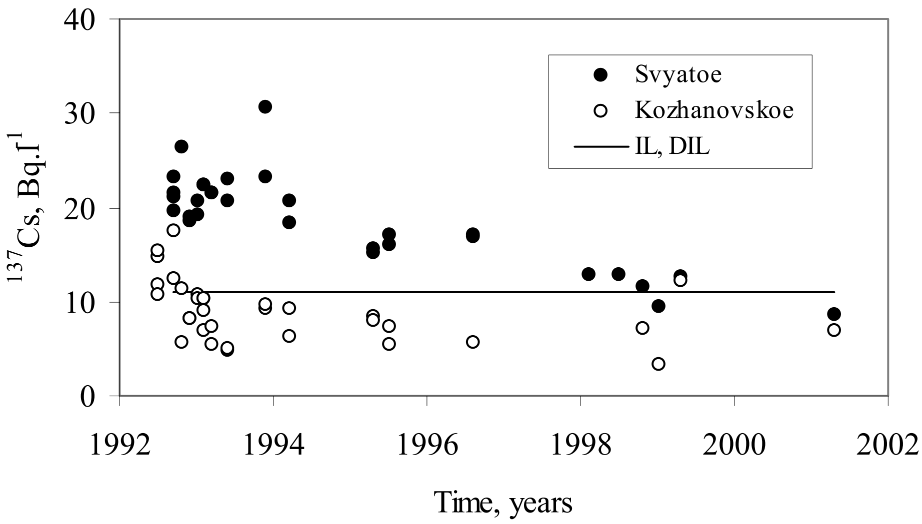


FIG. 2.50. Dynamics of ^{137}Cs-activity concentration in the water of Lakes Svyatoe and Kozhanovskoe (Russia), approximately 200 km from Chernobyl (Konoplev et al. 1998).

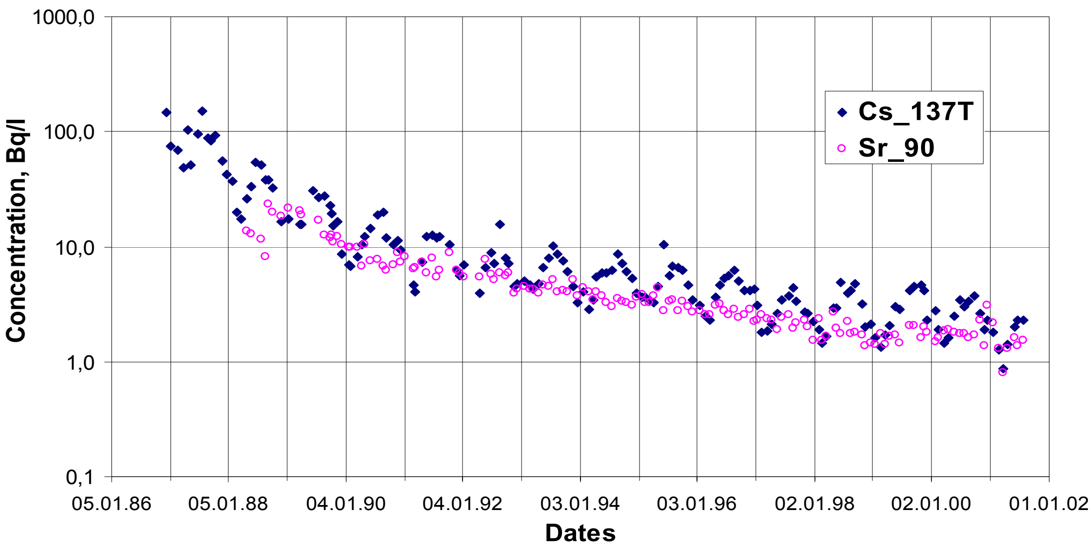


FIG. 2.51. Monthly averaged ^{137}Cs and ^{90}Sr activity concentrations in the water of the Chernobyl Cooling Pond (Nasvit 2002).

Chernobyl Cooling Pond

The Chernobyl Cooling Pond covers an area of approximately 23 km^2 and contains approximately 149 million m^3 of water. It is located between the former ChNPP and the Prypiat River. The total inventory of radionuclides in the pond is in excess of 200 TBq (about 80% is ^{137}Cs, 10% ^{90}Sr, 10% ^{241}Pu and < 0.5% of each of ^{238}Pu, ^{239}Pu, ^{240}Pu and ^{241}Am) with the deep sediments containing most of the radioactivity. The ^{90}Sr annual flux to the Prypiat River from the Cooling Pond via ground water was estimated in a recent study as 0.37 TBq (Buckley et al. 2002). This is a factor of 10-30 less than the total annual ^{90}Sr fluxes via the Prypyat River during recent years. Thus, the Cooling Pond is not a significant source of ^{90}Sr contamination of the Pripyat River. Radionuclide-activity concentrations in the Cooling Pond water (Fig. 2.51) are currently low, at 1-2 Bq L^{-1}. Seasonal variations of ^{137}Cs concentration are caused by changes in algae and phytoplankton biomass (Nasvit 2002).

Reservoirs of the Dnieper cascade

The Dnieper cascade reservoirs were significantly affected due to both atmospheric fallout and riverine inputs from the contaminated zones (See Figs. 2.6 and 2.7). The different affinities of ^{137}Cs and ^{90}Sr for suspended matter influenced their transport through the Dnieper system. Caesium-137 tends to become fixed onto clay sediments which are deposited in the deeper sections of the reservoirs, particularly in the Kiev Reservoir (Fig. 2.52). Because of this process, very little ^{137}Cs flows through the cascade of reservoirs, and consequently the present concentration entering the Black Sea is indistinguishable from background levels.

On the other hand, although ^{90}Sr-activity concentration decreases with distance from the source (mainly due to dilution), about 40 to 60% passes through the cascade and reaches the Black Sea. Figure 2.53 shows the trend in average annual ^{90}Sr-activity concentration in the Dnieper reservoirs since the accident. As ^{137}Cs is trapped by sediments in the reservoir system, activity concentrations in the lower part (Novaya Kahovka) of the system are orders of magnitude lower than in the Kiev Reservoir (Vishgorod). In contrast, ^{90}Sr is not strongly bound by sediments, so activity concentrations in the lower part of the river-reservoir system are similar to those measured in the Kiev Reservoir.

The peaks in ^{90}Sr-activity concentration in the reservoirs of the Dnieper cascade (Fig. 2.53) were caused by flooding of the most contaminated floodplains in the Chernobyl Exclusion zone. For instance, flooding of the River Pripyat, caused by blockages of the river by ice in Winter 1991 led to temporary significant increases in ^{90}Sr in this system, but did not significantly affect ^{137}Cs. Activity concentrations of ^{90}Sr in the river water increased from ca. 1 Bq L^{-1} to 8 Bq L^{-1} for a 5-10 day period (Vakulovsky et al. 1994). Similar events took place during the winter flood of 1994, during summer rainfall in July 1993, and during the high spring flood in 1999 (Voitsekhovitch et al. 1997).

Radionuclide run-off from catchment soils

Small amounts of radionuclides are eroded from soils and transferred to rivers, lakes, and eventually the marine system. Such transfers can take place through erosion of surface-soil particles and by runoff in the dissolved phase. Studies of nuclear weapons test and Chernobyl ^{90}Sr in rivers in (Saxén and Ilus, 2001; Konoplev et al. 1999; Kudelsky et al. 1998; Smith et al. 1999) suggest long-term loss rates of about 1-2% per year or less from the terrestrial environment to rivers. Thus, in the long term, run-off of radioactivity does not significantly reduce the amount of radioactivity in the terrestrial system, though it does result in continuing (low level) contamination of river and lake systems.

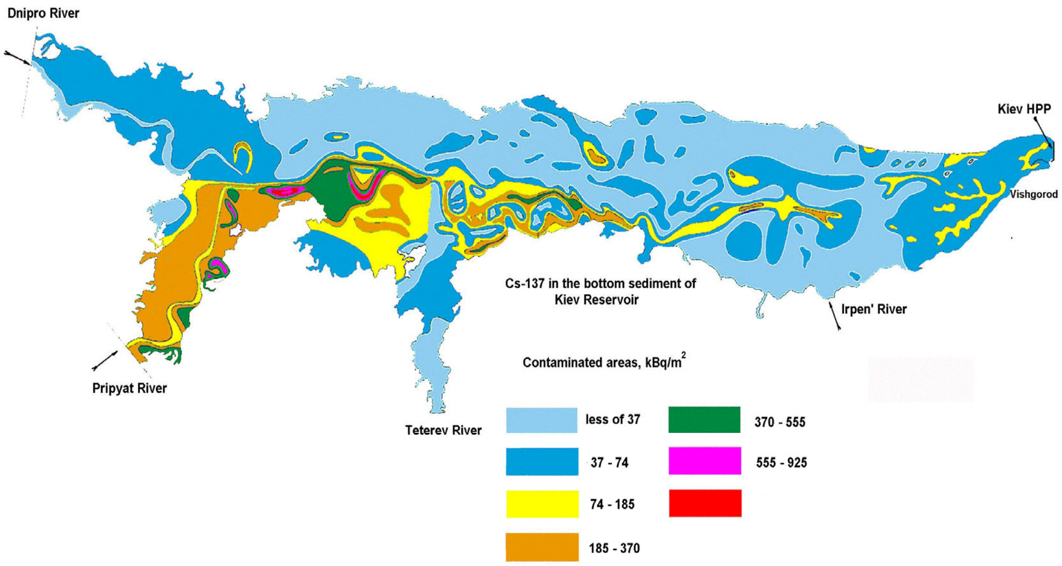


FIG. 2.52. Cs-137 in the bottom sediment of the Kiev reservoir (Voitsekhovitch and Kanivets 1997).

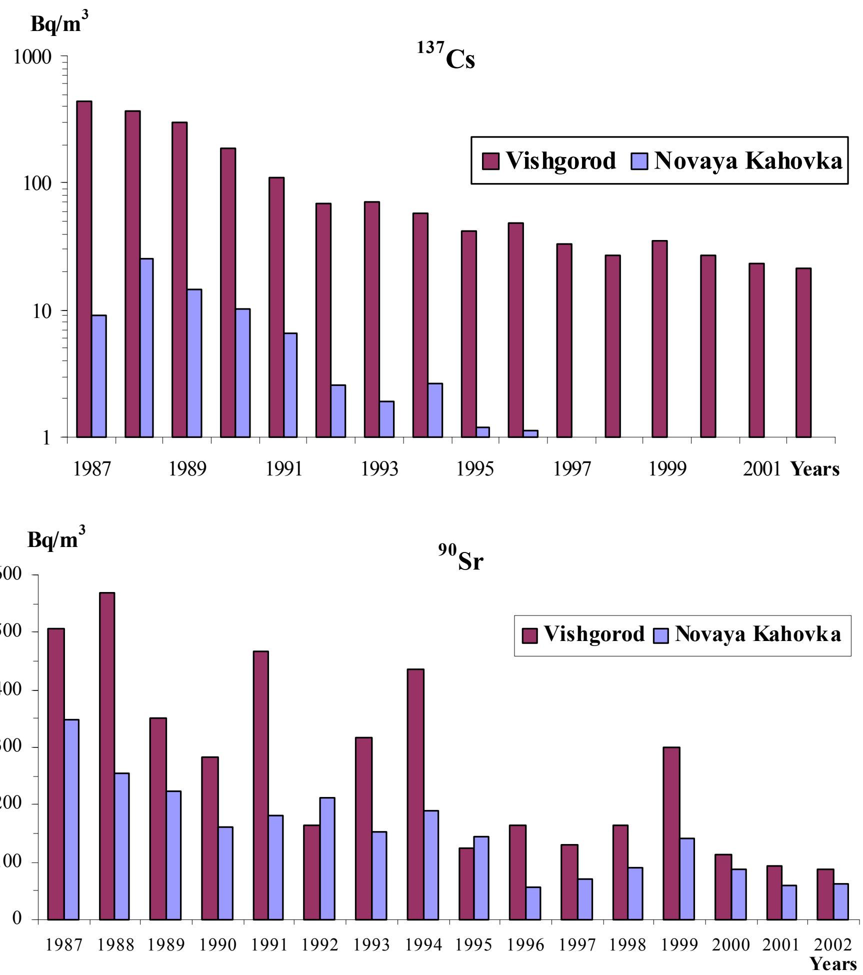


FIG. 2.53. The annually averaged ^{137}Cs- and ^{90}Sr-activity concentrations in the water of Kyiv Reservoir (Vishgorod), near the dam and Kahovka Reservoir of the Dnieper cascade (IAEA 2005).

2.5.2.4. *Freshwater sediments*

Bed sediments are an important long-term sink for radionuclides. Radionuclides can attach to suspended particles in lakes which then fall and settle on bed sediments. Radioactivity in lake water can also diffuse into bed sediments. These processes of radionuclide removal from lake water have been termed "self-cleaning" of the lake or reservoir (Santschi et al. 1990).

In the Chernobyl Cooling Pond approximately one month after the accident, most of the radioactivity was found in bed sediments (Kryshev 1995; Voitsekhovich and Kanivets, 1997). In the long term, approximately 99% of the radiocaesium in a lake is typically found in the bed sediment. From measurements in Lake Svyatoe (Kostyukovichy district, Belarus), during 1997, approximately 3×10^9 Bq of ^{137}Cs were in the water and 2.5×10^{11} Bq in sediments (Smith et al. 2001). In Lake Kozhanovskoe, Russia, approximately 90% of the radiostrontium was found in the bed sediments during 1993-94 (Sansone and Voitsekhovitch 1996).

In the rapidly accumulating sediments of the Kiev Reservoir, the layer of maximum radioactivity concentration is now buried several tens of cm below the sediment surface (Fig. 2.54). In more slowly accumulating sediments, however, the peak in radiocaesium-activity concentration remains near the sediment surface. Peaks in the sediment-layer contamination in 1988 and 1993 reflect the consequences of high summer rainfall floods and soil-erosion events.

Close to Chernobyl, a high proportion of radioactivity was deposited in the form of fuel particles (See Sub-section 2.1). Radionuclides deposited as fuel particles are generally less mobile than those deposited in dissolved form. In sediments of Glubokoye Lake in 1993, most fuel particles remained in the surface 5 cm of sediment (Sansone and Voitsekhovitch 1996). Fuel-particle breakdown was at a much lower rate in lake sediments than in soils. Studies in the Cooling Pond have shown that the half life of fuel particles in sediments is approximately 30 years, so by 2056 (70 years after Chernobyl) one quarter of the radioactivity deposited as fuel particles in the Cooling Pond will still remain in fuel-particle form (Kashparov et al. 2004; Konoplev, personal communication).

2.5.3. Uptake of radionuclides to freshwater fish

Consumption of freshwater fish is an important part of the aquatic dose pathway for transfers of radionuclides to humans. Although transfer of radioactivity to fish has been studied in many countries, most attention here will be focused on Ukraine, Belarus and Russia, because of the relatively significantly higher contamination of water bodies in these areas.

2.5.3.1. *^{131}I in freshwater fish*

There are limited data on ^{131}I in fish. Iodine-131 was rapidly absorbed by fish in the Kiev Reservoir, with maximum concentrations in fish being observed in early May 1986 (Kryshev 1995). Activity concentrations in fish muscle declined from around 6000 Bq kg^{-1} fresh weight on 1 May 1986 to 50 Bq kg^{-1} fresh weight on 20 June 1986. This represents a rate of decline similar to that of the physical decay of ^{131}I. Because of the rapid physical decay, ^{131}I-activity concentrations in fish became insignificant a few months after the accident.

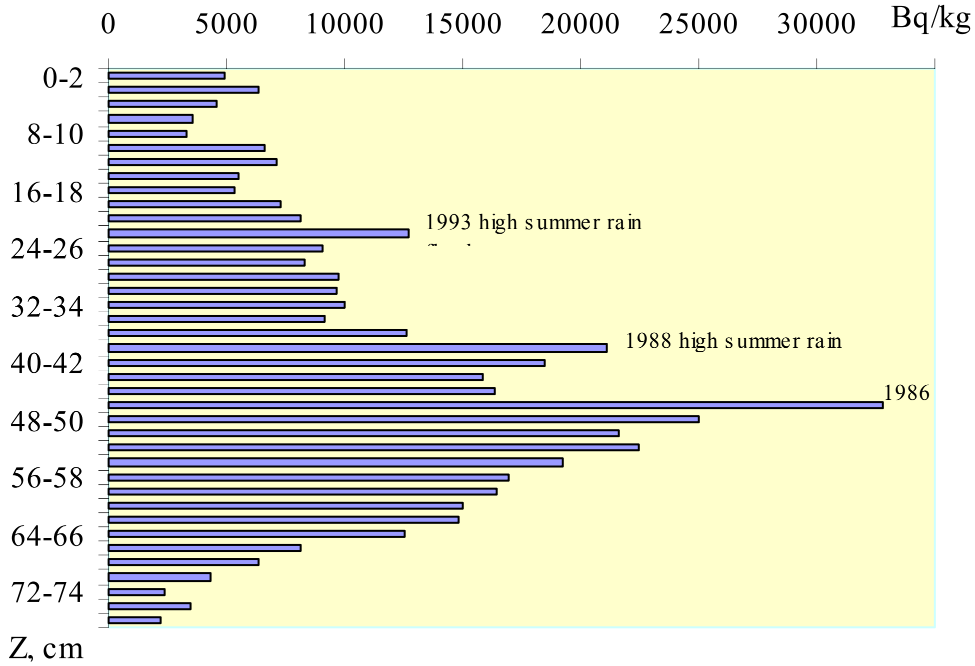


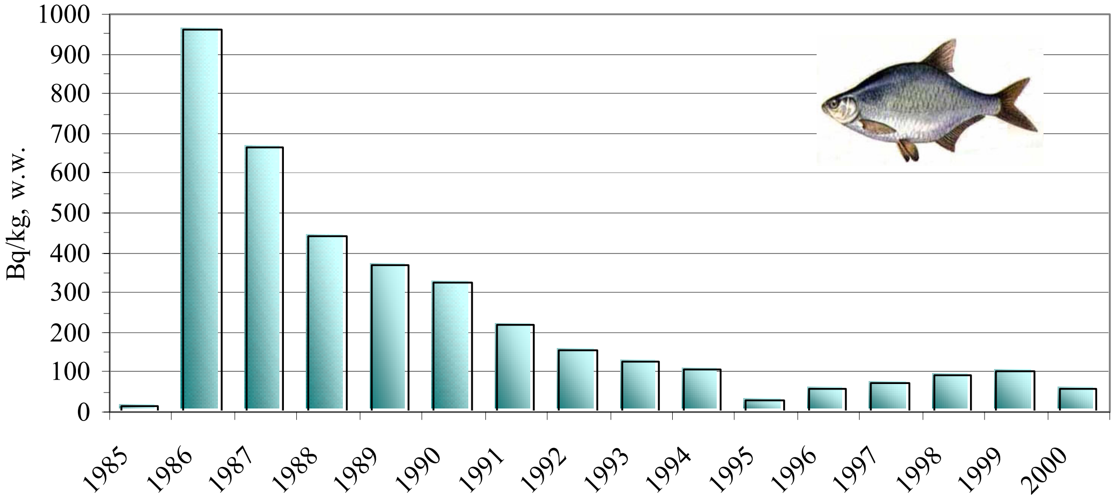
FIG. 2.54. Activity concentrations of ^{137}Cs in the deep silt deposits in the upper part of Kiev Reservoir, 1998 (UHMI 2004).

2.5.3.2. ^{137}Cs in freshwater fish and other aquatic biota

During the years following the Chernobyl accident there have been many studies of the levels of radiocaesium contamination of freshwater fish. As a result of high radiocaesium-bioaccumulation factors, fish have remained contaminated in some areas, despite low radiocaesium levels in water. Uptake of radiocaesium in small fish was relatively rapid, the maximum concentration being observed during the first weeks after the accident (Jonsson et al., 1999; Zibold et al., 2002). Due to the slow uptake rates of radiocaesium in large predatory fish (pike, eel), maximum activity concentrations, were not observed until 6-12 months after fallout (Elliot, et al. 1992; Zibold et al. 2002 and see Fig. 2.49).

In the Chernobyl Cooling Pond, ^{137}Cs-activity concentrations in carp, silver bream, perch and pike were about 100 kBq kg^{-1} fresh weight in 1986, declining to a few tens of kBq kg^{-1} in 1990 (Kryshev and Ryabov 1990; Kryshev 1995) and 2-6 kBq kg^{-1} in 2001. In some closed lakes in the vicinity of the Chernobyl NPP (Nasvit 2002) the ^{137}Cs-activity concentration in predatory fish 15 years after the accident was 10-27 kBq kg^{-1} fresh weight. Typical changes with time in ^{137}Cs in two fish species over 16 years after the accident are illustrated in Fig. 2.55.

In the Kyiv Reservoir, ^{137}Cs-activity concentrations in fish were 0.6-1.6 kBq kg^{-1} fresh weight (in 1987), 0.2-0.8 kBq kg^{-1} fresh weight (for 1990-1995) and declined to 0.2 kBq kg^{-1} or less for adult non-predatory fish in 2002. Values for predatory fish species were1-7 kBq kg^{-1} during 1987 and 0.2-1.2 kBq kg^{-1} from 1990-1995 (Ryabov et al. 1996).



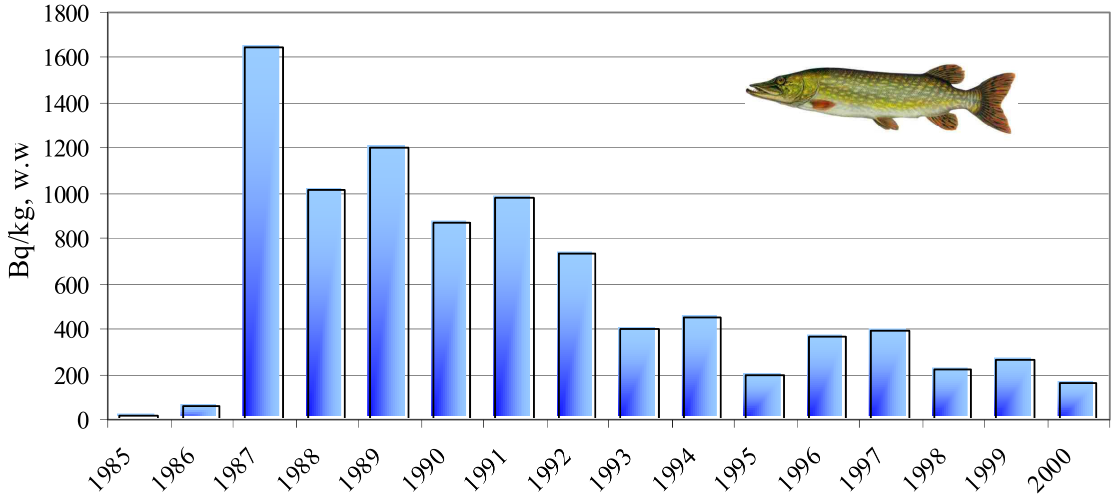


FIG. 2.55. Averaged ^{137}Cs-activity concentrations in non-predatory (bream, upper histogram) and predatory (pike, lower histogram); fish are from Kyiv reservoirs (UHMI 2004).

In lakes of the Bryansk Oblast of Russia, approximately 200 km from Chernobyl, ^{137}Cs-activity concentrations in a number of fish species varied within the range 0.2-19 kBq.kg^{-1} fresh weight during the period 1990-92 (Fleishman et al. 1994; Sansone and Voitsekhovitch, 1996). In shallow closed lakes such as Kozhanovskoe (Bryansk Oblast, Russia) and Svyatoe (Kostiukovichy Raion, Belarus), ^{137}Cs-activity concentrations in fish have declined slowly in comparison with fish in rivers and open lake systems, due to the slow decline in ^{137}Cs-activity concentrations in water of the lakes (Bulgakov et al. 2002).

In Western Europe, lakes in some parts of Norway, Sweden and Finland were particularly heavily contaminated. About 14,000 lakes in Sweden had fish with ^{137}Cs-activity concentrations above 1500 Bq kg^{-1} fresh weight (the Swedish guideline value) in 1987 (Håkanson et al. 1992). In some alpine lakes in Germany, ^{137}Cs-activity concentrations in pike were up to 5000 Bq kg^{-1} fresh weight shortly after the Chernobyl accident (Zibold et al. 2002). In Devoke Water in the English Lake District, perch and brown trout contained around 1000 Bq kg^{-1} fresh weight in 1988 declining slowly to a few hundreds of Bq kg^{-1} in 1993 (Camplin et al. 1989 and personal communication).

Bioaccumulation of radiocaesium in fish is dependent on a number of factors. Because of its chemical similarity to caesium, the potassium concentration of lake or river water influences the rate of accumulation of radiocaesium in fish (Blaylock 1982). Strong inverse relationships were observed between lake water potassium concentration and ^{137}Cs-activity concentration in fish following nuclear weapons testing (Fleishman 1973; Blaylock 1982) and the Chernobyl accident (Smith et al. 2000a). In the long term, activity concentrations in predatory fish were significantly higher than non predatory fish and large fish tended to have higher activity concentrations than small. The increase in activity concentration in large fish is termed the "size effect" (Elliot et al. 1992; Hadderingh et al. 1997) and is due to metabolic and dietary differences. In addition, older, larger fish were exposed to higher levels of ^{137}Cs in the water than younger, smaller fish.

The differences in bioaccumulation of radiocaesium in different fish species can be significant. For example, in Lake Svyatoe, Belarus, radioactivity in large pike and perch (predatory fish) was 5-10 times higher than in non-predatory fish such as roach. Similarly, bioaccumulation factors in lakes of low potassium concentration can be one order of magnitude higher than in lakes of high potassium concentration. Thus, it was observed (Smith et al. 2000a) that fish from lakes in agricultural areas of Belarus (where runoff of potassium fertiliser is significant) had lower bioaccumulation factors than fish from lakes in semi-natural areas.

2.5.3.3. ^{90}Sr in freshwater fish

Strontium behaves, chemically and biologically, in a similar way to calcium. Strontium is most strongly bioaccumulated in low calcium (“soft”) waters. Relatively low fish-water bioaccumulation factors for ^{90}Sr (in the order of 10^2 L kg^{-1}) and lower fallout of this isotope meant that ^{90}Sr-activity concentrations in fish were typically much lower than those of ^{137}Cs. In the Chernobyl Cooling Pond, ^{90}Sr-activity concentrations were around 2 kBq kg^{-1} (whole fish) in fish during 1986, compared with around 100 kBq kg^{-1} for ^{137}Cs in 1993 (Kryshev 1995). In 2000, for the most contaminated lakes around Chernobyl, the maximum level of ^{90}Sr concentration in the muscles of predatory and non-predatory fish varied between 2 and 15 Bq kg^{-1} fresh weight. In 2002-2003 ^{90}Sr in fish in reservoirs of the Dnieper cascade was only 1-2 Bq kg^{-1}, which is close to pre-Chernobyl levels. Freshwater molluscs showed significantly stronger bioaccumulation of ^{90}Sr than fish. In the Dnieper River, molluscs had approximately ten times more ^{90}Sr in their tissues than fish muscle (Kryshev and Sazykina 1994). Similarly, bioaccumulation of ^{90}Sr in bones and skin of fish is approximately a factor of 10 times higher than in muscle (Blaylock 1982).

2.5.4. Radioactivity in marine ecosystems

Marine ecosystems were not seriously affected by fallout from Chernobyl, the nearest seas to the reactor being the Black Sea (around 520 km) and the Baltic Sea (about 750 km). The primary route of contamination of these seas was atmospheric fallout, with smaller inputs from riverine transport occurring over the years following the accident. Surface deposition of ^{137}Cs was approximately 2.8 PBq over the Black Sea (Eremeev et. al. 1995; IAEA 2003b) and 3.0 PBq over the Baltic Sea (Vakulovsky et al. 1994).

2.5.4.1. Distribution of radionuclides in the sea

Radioactive fallout onto the surface of the Black Sea was not uniform and mainly occurred during 1 and 3 May (Eremeev et al. 1995; Vakulovsky et al. 1994). In the Black Sea surface water concentrations of ^{137}Cs ranged from 15-500 Bq m^{-3} in June-July 1986, though by 1989 horizontal mixing of surface waters had resulted in relatively uniform concentrations in the range 41-78 Bq m^{-3} (Vakulovsky et al. 1994) and by the year 2000 had declined to between 20 and 35 Bq m^{-3} (IAEA 2003b).

In addition to the Cs isotopes, short-lived radionuclides such as ^{144}Ce and ^{106}Ru were observed. The inventory of ^{137}Cs in the water of the Black Sea due to Chernobyl deposition doubled the existing inventory from global fallout from atmospheric nuclear weapons testing to approximately 3100 TBq. The amount of ^{90}Sr increased by 19% in comparison with the pre-Chernobyl period and was estimated to be about 1760 TBq (Vakulovsky et al. 1994; IAEA 2003b). Vertical mixing of surface deposited radioactivity also reduced the maximum concentrations observed in water over the months to years after fallout. Removal of radioactivity to deeper waters steadily reduced ^{137}Cs-activity concentrations in the surface (0-50 m) layer of the Black Sea. The present situation with the Black Sea marine environment is shown in Table 2.8 (IAEA 2003b).

Note that a significant proportion of the ^{137}Cs, ^{90}Sr and $^{239,240}Pu$ in the Black Sea originated with nuclear weapons testing rather than Chernobyl. The riverine radionuclide input to the Black Sea was much less significant than direct atmospheric fallout to the sea surface. Over the period 1986-2000, riverine input for ^{137}Cs was only 4-5% of the atmospheric deposition, though ^{90}Sr inputs were more significant, being approximately 25% of the total inputs from atmospheric deposition (Kanivets et al. 1999; IAEA 2003b). For the Baltic Sea, riverine inputs were at a similar level to the Black Sea, being approximately 4% and 35% of atmospheric fallout for ^{137}Cs and ^{90}Sr, respectively (Nielsen et al. 1999). The greater relative riverine input of ^{90}Sr is due to its weaker adsorption to catchment soils and lake and river sediments and to lower ^{90}Sr atmospheric fallout (compared to ^{137}Cs) at large distances from the Chernobyl reactor site. Sedimentation processes in the marine environment, as in the freshwater environment, are an important factor in the "self-purification" of the aquatic ecosystem. The sedimentation rate for the Black Sea is relatively low (IAEA 2003b).

Data presented in Fig. 2.56 demonstrate that, for the central deep basin of the Black Sea, the Chernobyl deposition was covered by a layer of less than 1 cm of sediment formed since the accident (IAEA 2003b).

Due to dilution and sedimentation, the concentration of ^{137}Cs quickly declined, reducing the sea water contamination at the end of 1987 to 2-4 times lower than that observed in Summer 1986. The average ^{137}Cs-activity concentration in the Baltic Sea estimated in (EU 2000) for the initial period after deposition was in the range approximately 60-50 Bq m^{-3} with maximum values 2-4 times greater being observed in some areas of the Sea.

TABLE 2.8. RADIONUCLIDES IN VARIOUS SAMPLES TAKEN FROM THE BLACK SEA COAST DURING 1998-2001 (IAEA 2003b)

Environmental samples	^{137}Cs	^{90}Sr	$^{239,240}Pu$
Seawater, Bq m^{-3}	14 – 29	12 - 28	(2.4 - 28)×10^{-3}
Beach sand and shells, Bq kg^{-1}	0.9 – 8.0	0.5 – 60 (shell)	(80-140)×10^{-3}
Seaweeds, *Cystoseira barbata* (Bq kg^{-1} wet weight)	0.2 – 2.3	0.4 – 0.9	(9.0-14)×10^{-3}
Mussels *Mytilus galloprovincialis* (tissue, Bq kg^{-1} wet weight)	0.3 – 1.7	0.02-3.2	(1.5 – 2.5)×10^{-3}
Fish, *Sprattus sprattus, Trashurus* (Bq kg^{-1} wet weight)	0.2 – 6.0	0.02 – 0.7	(0.3 – 0.5) ×10^{-3}

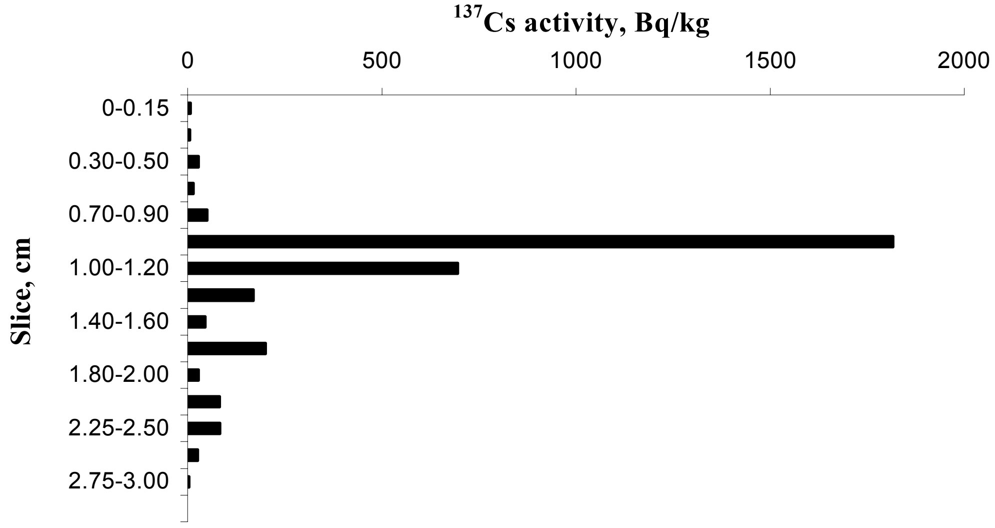


FIG. 2.56. ^{137}Cs profile in bottom sediment (Core BS-23 /2000) taken during IAEA Black Sea expedition in 2000 (IAEA 2003b).

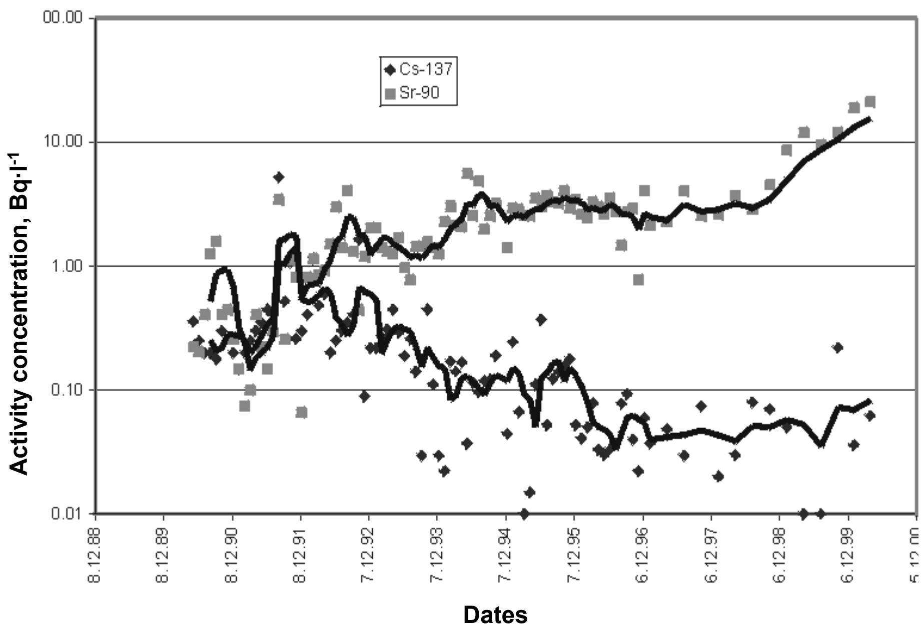


FIG. 2.57. ^{137}Cs and ^{90}Sr in shallow groundwater at the Red Forest area near the Chernobyl industrial site (Data presented by the Chernobyl Exclusion zone Ecological Centre).

2.5.4.2. Transfers of radionuclides to marine biota

Bioaccumulation of radiocaesium and radiostrontium in marine systems is generally lower than in freshwater, because of the much higher content of competing ions in saline waters. The lower bioaccumulation of ^{137}Cs and ^{90}Sr in marine systems, and the large dilution in these systems meant that activity concentrations in marine biota were relatively low.

Table 2.7.gives examples of ^{137}Cs, ^{90}Sr and $^{239,240}Pu$ in water and marine biota of the Black Sea during the period 1998-2001 (IAEA 2003b). Detailed data on Baltic Sea fish contamination during the post-Chernobyl decades is available in (EU 2000). This shows that most species of fish had a relatively low level of radiocaesium contamination, in most cases in the range 30-100 Bq kg^{-1} or less during the period to 1995.

2.5.5. Radionuclides in groundwater

2.5.5.1. Radionuclides in groundwater – Chernobyl Exclusion Zone.

Sampling of groundwater in the affected areas showed that radionuclides can be transferred from surface soils to groundwater. However, the level of the groundwater contamination in most areas (excluding locations of radioactive waste storage and the Chernobyl Shelter industrial site) is low. Furthermore, the rates of migration from the soil surface to groundwater are also very low. Some areas in the exclusion zone with relatively fast radionuclide migration to the aquifers were found by Shestopalov et. al. (2002) in areas with morphological depressions. Horizontal fluxes of radionuclides in groundwaters are also very low because of the slow flow velocity of groundwaters and high retardation of radionuclides (Bugai et al. 1996).

Short-lived radionuclides are not expected to affect groundwater supplies, because groundwater residence times are much longer than the physical decay time of short-lived nuclides. The only significant transfer of radionuclides to groundwater has occurred within the Chernobyl Exclusion Zone. In some wells during the last 10 years ^{137}Cs-activity concentration has declined whilst those of ^{90}Sr continue to increase in shallow ground waters (Fig. 2.57). Transfer of radionuclides to groundwater has occurred from radioactive waste-disposal sites in the Exclusion Zone. After the accident, fuel containing material and radioactive debris were temporarily stored at industrial sites at the power station and in areas near the floodplain of the Pripyat River. In addition, trees from the "red forest" were buried in shallow unlined trenches. At these waste-disposal sites, ^{90}Sr-activity concentrations in groundwaters are in some cases of the order of 1000 Bq L^{-1} (Voitsekhovitch et al. 1996). Health risks from groundwater consumption to hypothetical residents returning to these areas, however, were low in comparison to external radiation and internal doses from foodstuffs (Bugai et al. 1996).

Although there is a potential for off-site transfer of radionuclides from the disposal sites, Bugai and co-workers (1996) concluded that this will not be significant in comparison to wash out of surface deposited radioactivity. Studies have shown that groundwater fluxes of radionuclides are in the direction of the Pripyat River, but the rate of radionuclide migration is very low and does not present a significant risk to the Dnieper reservoir system. Off-site transport of groundwater contamination around the Shelter is also expected to be insignificant, because radioactivity in the "Shelter" is separated from groundwaters by an unsaturated zone of thickness 5-6 m, and groundwater velocities are low (Bugai et al. 1996). It is predicted that the maximum subsurface ^{90}Sr-transport rate from waste-disposal sites to surface-water bodies will occur from 33 to 145 years following the accident. Maximum cumulative annual transport from all of the sources described above is estimated to be 130 GBq in approximately 100 years, or 0.02% per year of the total inventory within the contaminated catchments. Integrated radionuclide transport for a 300-year period is estimated by Bugai et al. (1996) as 15 TBq, or 3% of the total initial inventory of radioactivity within the catchments (Fig. 2.58).

The water level of the Chernobyl Cooling Pond significantly influences groundwater flows around the Chernobyl site. Currently, the water level of the Cooling Pond is kept artificially high at 6-7 m above the average water level in the Prypiat River. However, this will change when the cooling systems at the ChNPP are finally shut down and pumping of water into the pond is terminated. As the pond dries out, the sediments will be partly exposed and subject to dispersal. Recent studies report that the best strategy for remediation of the Cooling Pond is to allow the water level to decline naturally, with some limited action to prevent secondary wind resuspension using phytoremediation techniques (Vandenhove 2002).

When the water level in the Cooling Pond declines to that of the river-water-surface level, this will lead to reduction of groundwater fluxes from the Chernobyl industrial sites toward the rivers. This will also reduce radionuclide fluxes from the main radioactive waste-disposal sites and Shelter to the Dnieper cascade. The groundwater fluxes of ^{90}Sr from the Chernobyl Shelter to the Pripyat River have been modelled by M. Zheleznyak and S. Kivva within the framework of environmental impact assessment studies of the New Safe Confinement to be erected above the "Shelter" of ChNPP (Bechtel-Batttele-EDF 2003), see Fig. 2.59. It has been shown that it would take approximately 800 years for ^{90}Sr to reach the Pripyat River. With its half-life of 29.1 years, ^{90}Sr would reduce to insignificant levels during this time. Thus, infiltration of ^{90}Sr from the Shelter will not cause harmful impacts to the Pripyat River. ^{137}Cs moves much more slowly than ^{90}Sr, and even after 2000 years, its plume is predicted to be only 200 m from the Shelter.

Because of its high adsorption to the soil matrix, ^{239}Pu migrates at a much slower rate than ^{90}Sr or ^{137}Cs, however, its half-life is much longer (24,000 years). The maximum groundwater ^{239}Pu influx from the "Shelter" into the Prypiat River is predicted to be 2 Bq s^{-1}. When this influx is fully mixed with the average Prypiat River discharge of 400 m^3 s^{-1}, the resulting ^{239}Pu concentration in the river would be only 0.005 Bq m^{-3}, as compared to the current ^{239}Pu level of 0.25 Bq m^{-3} (Bechtel-Batttele-EDF 2003). In Ukraine, the regulatory limit for ^{239}Pu in water is 1 Bq m^{-3}. Thus, infiltration of ^{239}Pu from the Shelter, even without the New Safe Confinement (NSC) will not cause any significant impact on the Pripyat River.

2.5.5.2. Radionuclides in groundwater – outside the Chernobyl Exclusion Zone

The most detailed current studies of groundwater contamination in the far zone (UNDP 2001; Shestopalov et al. 2002) have concluded that 10 years after the initial surface ground pollution, the groundwater in the upper horizons of the aquifer had ^{137}Cs and ^{90}Sr contamination of 40-50 mBq L^{-1} around Kyiv and 20-50 mBq L^{-1} in the Bryansk Oblast of Russia and the majority of contaminated areas in Belarus. In these areas far from the Chernobyl reactor (in Belarus and Russia) the ^{137}Cs-activity concentration of water in the saturated zone of soils had a significant correlation with the ^{137}Cs-soil deposition. In most of the studied area, the activity concentration in groundwater (per unit of ^{137}Cs soil deposition) was significantly lower than in most river and lake systems. All studies reported that the radionuclide concentration in the contaminated area outside the Chernobyl Exclusion Zone never exceeded the safety level for consumption of water and was usually several orders of magnitude lower than the permissible level for drinking water.

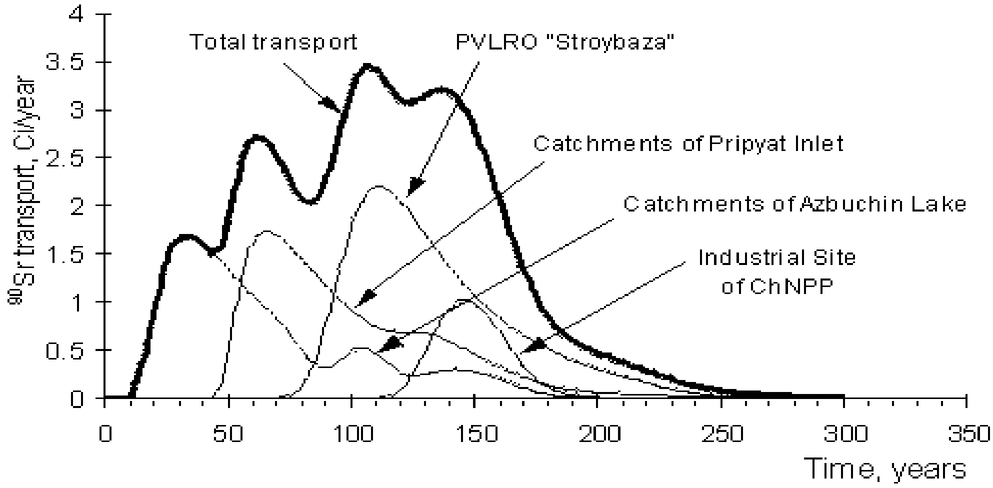


FIG. 2.58. Prediction of ^{90}Sr (1 Ci = 37GBq) transport via the groundwater pathway to the Pripyat River in the ChNPP near zone (Bugai et al. 1996).

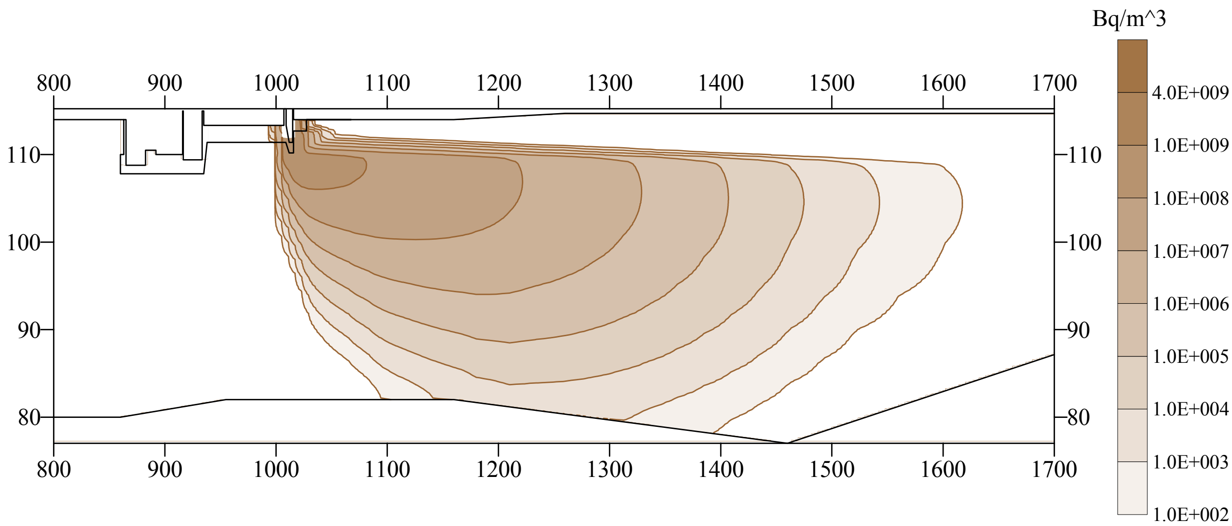


FIG. 2.59. Predicted ^{90}Sr concentrations in the aqueous phase without NSC after 100 y (Bechtel-Battelle-EDF 2003).

After fallout from nuclear weapons testing, it was observed that ^{90}Sr in Danish groundwater was approximately 10 times lower than in surface streams (Hansen and Aarkrog 1990). These authors also observed that after Chernobyl, despite measurable quantities of ^{137}Cs in streams, activity concentrations were below detection limits in groundwater.

2.5.5.3. Irrigation water

In the Dnieper basin there are more than 1.8 million ha of irrigated agricultural land. Almost 72% of this territory is irrigated with water from the Kakhovka Reservoir and other Dnieper reservoirs. Accumulation of radionuclides in plants on irrigated fields can take place via root uptake of radionuclides introduced with irrigation water and due to direct incorporation of radionuclides through leaves due to sprinkling. However, in the case of irrigated lands of southern Ukraine, radionuclides in irrigation water did not add significant radioactivity to crops in comparison with that which had been initially deposited in atmospheric fallout and subsequently taken up *in situ* from the soil (IAEA-FAO 1994).

2.5.6. Future trends

2.5.6.1. Freshwater ecosystem

For the rivers and reservoirs of the Dnieper system the intensity of runoff of radionuclides will gradually reduce. In the worst case scenario, hydrological runoff during the next 50 years (Zheleznyak et al. 1997) would have average concentrations of ^{137}Cs and ^{90}Sr approaching pre-accident levels. Contamination levels of the water and main consumer fish species in the Reservoirs of the middle and lower Dnieper will become close to the background levels (Fig. 2.60). At the same time in the isolated (closed) water bodies of the contaminated territories increased contents of ^{137}Cs both in water and aquatic biota will be maintained for several decades.

Recent data (Jonsson et al. 1999; Smith et al. 2000b) shows that at present ^{137}Cs-activity concentrations in surface water and fish are declining quite slowly. The effective ecological half-life in water and young fish has increased from 1-4 years during the first five years after the accident to 6-30 years in recent years. Future contamination levels can be estimated with use of an estimated long-term decline of radicaesium-activity concentrations in water and fish with an effective ecological half life (T_{eff}) of approximately 20 years, although there is wide variation in rates of decline (Smith et al. 2001).

Activity concentrations of radiocaesium in water are at present relatively low (1 Bq L^{-1} at most), except in the shallow closed lakes in the 30-km zone and in other highly contaminated areas. Activity concentrations are expected to continue declining slowly during the coming decades. In some lakes, however, ^{137}Cs-activity concentrations both in water and fish are expected to remain relatively high for some decades, as illustrated in Tables 2.9 and 2.10. Activity concentrations of ^{90}Sr in water were also estimated using a predicted T_{eff} of 20 years. This may again be slightly conservative, as long-term rates of decline of ^{90}Sr from weapons testing had an T_{eff} of about 10 years (Cross et al. 2002). Similar to ^{137}Cs, activity concentrations of ^{90}Sr in water are expected to decline from their current low levels during the coming decades (Table 2.11).

Fuel-particle breakdown was at a much slower rate in lake sediments than in soils (Konoplev et al. 1996). The half life of fuel particles in sediments in the Cooling Pond is approximately 30 years (Kashparov et al. 2004; Konoplev, personal communication) so radionuclides in fuel particles will remain for many years in original form.

2.5.6.2. Marine ecosystems

At present radionuclides (mainly radiocaesium) in marine systems are at much lower concentrations than observed in freshwater systems. Activity concentrations in sea water and marine biota in the Black Sea are expected to continue to decline (see Table 2.8). This is mainly due to physical decay, but continued transfers to sea-bed sediments and further dilution will also contribute to the decline.

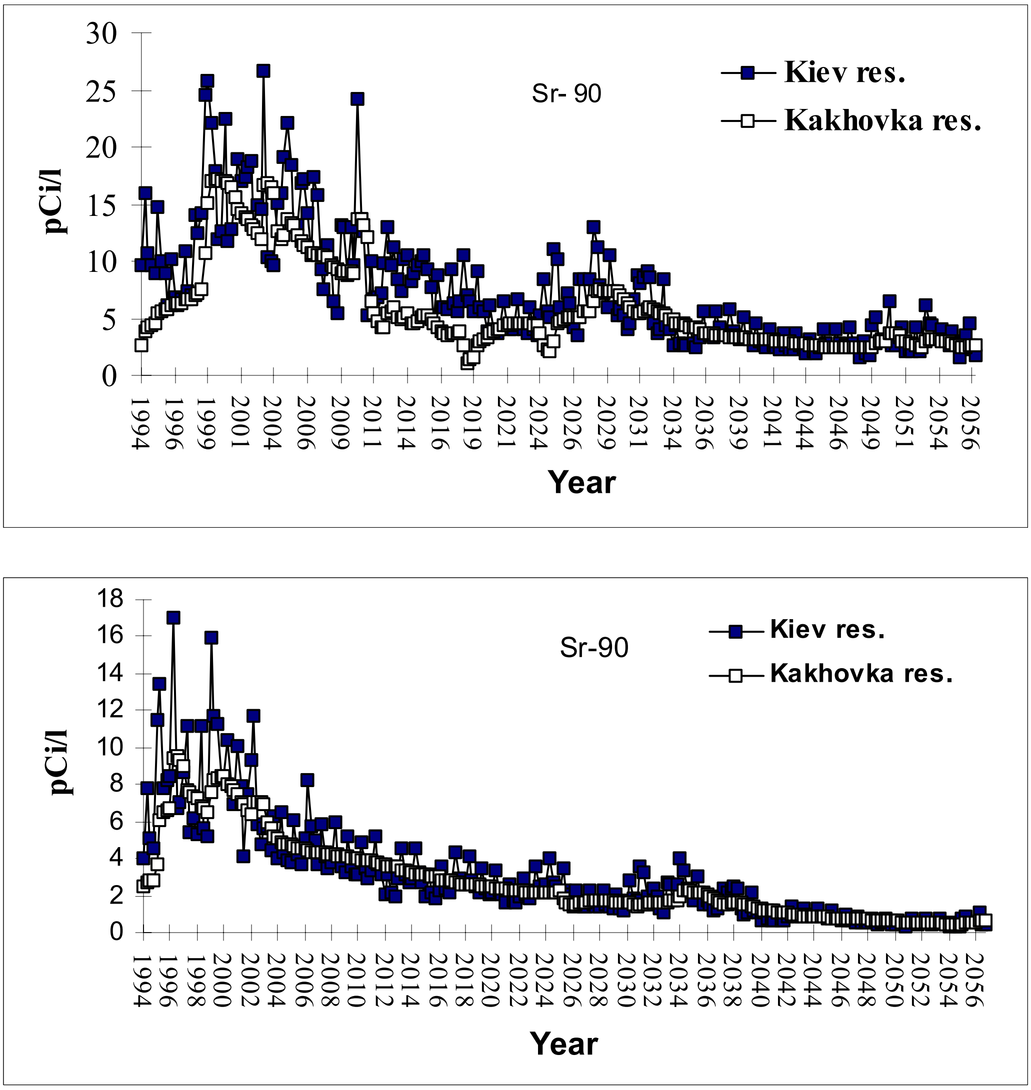


FIG. 2.60. Concentration of ^{90}Sr (1 pCi = 3.7 ×10^{-2} Bq) in water of the upper and downstream reservoirs for the worst (top) and best probabilistic hydrological scenarios to be expected for the Pripyat River basin (Zheleznyak et al. 1997).

TABLE 2.9. ^{137}Cs-ACTIVITY CONCENTRATIONS IN WATER IN VARIOUS CHERNOBYL AFFECTED LAKES AND RIVERS AROUND EUROPE AND PREDICTIONS FOR 30, 50 AND 70 YEARS AFTER THE ACCIDENT

Lake	Measured ^{137}Cs, Bq L^{-1}, (year of measurement)	Predicted ^{137}Cs, Bq L^{-1}		
		2016	*2036*	*2056*
Kozhanovskoe, Russia	7.0 (2001)	4.2	2.1	1.0
Kyiv Reservoir	0.028 (1998)	0.015	0.007	0.004
Chernobyl Cooling Pond	2.5 (2001)	1.5	0.8	0.4
Lake Svyatoe, Belarus*	4.7 (1997)	2.4	1.2	0.6
L. Vorsee, Germany	0.055 (2000)	0.032	0.016	0.008
Devoke Water, UK	0.012 (1998)	0.006	0.003	0.002

* Note that this lake had a countermeasure applied in 1998. The prediction is for levels in the absence of countermeasures.

TABLE 2.10. ^{137}CS-ACTIVITY CONCENTRATIONS (PER UNIT WET WEIGHT) IN FISH IN VARIOUS CHERNOBYL AFFECTED LAKES AROUND EUROPE AND PREDICTIONS FOR 30, 50 AND 70 YEARS AFTER THE ACCIDENT

Lake	Fish species	Measured ^{137}Cs, Bq kg^{-1}, (year of measurement)	Predicted ^{137}Cs, Bq kg^{-1}, wet weight		
			2016	*2036*	*2056*
Kozhanovskoe, Russia	Goldfish	10,000 (1997)	5200	2600	1300
Kyiv Reservoir, Ukraine	Perch	300 (1997)	160	80	40
Chernobyl Cooling Pond	Perch	18,000 (2001)	11000	5400	2700
Lake Svyatoe, Belarus*	Perch	104,000 (1997)	54000	27000	14000
L. Vorsee, Germany	Pike	174 (2000)	100	50	25
L. Høysjøen, Norway	Trout	390 (1998)	210	100	50
Devoke Water, UK	Trout	370 (1996-98)	200	100	50

* Note that this lake had a countermeasure applied in 1998. The prediction is for levels in the absence of countermeasures.

TABLE 2.11. ^{90}Sr-ACTIVITY CONCENTRATIONS IN WATER IN VARIOUS CHERNOBYL AFFECTED LAKES AND RIVERS AND PREDICTIONS FOR 30, 50 AND 70 YEARS AFTER THE ACCIDENT

Lake	Measured ^{90}Sr, Bq l^{-1}, (year of measurement)	Predicted ^{90}Sr, Bq l^{-1}		
		2016	*2036*	*2056*
Pripyat River	0.28 (1998)	0.15	0.08	0.04
Kyiv Reservoir	0.16 (1998)	0.09	0.04	0.02
Chernobyl Cooling Pond	2.0 (2001)	1.2	0.6	0.3
L. Glubokoye, 30-km Zone	120 (2004)	80-90	40-60	20-30

2.6.Conclusions

The Chernobyl accident was the largest nuclear accident in history. The higher radionuclide depositions occurred in Belarus, Russia and Ukraine, but high deposition also occurred in a number of other European countries.

Most of the strontium and plutonium radionuclides was deposited close to the reactor and was associated with fuel particles. The environmental mobility of these radionuclides was lower than fallout associated with condensed particles which predominated in other areas, although the bioavailability of ^{90}Sr has increased with time as the fuel particles have partially dissolved.

Most of the originally released radionuclides have disappeared by radioactive decay and ^{137}Cs is currently of most concern. For the long time future (more than 100 years) only plutonium isotopes and ^{241}Am will remain.

The deposition in urban areas in the nearest city of Pripyat and surrounding settlements could have initially given rise to substantial external dose which was averted by the evacuation of the public. The deposition of radioactive material in other urban areas has provided substantial contributions to dose during subsequent years after the accident up to the present.

During the first weeks to months after the accident the transfer of short lived radioiodine isotopes to milk was rapid and high, leading to substantial dose problems in the former Soviet Union. Due to the emergency situation and the short half life of ^{131}I, there are few reliable data on the spatial distribution of deposited radioiodine. Current measurement of ^{129}I may assist in estimating ^{131}I deposition better and thereby improving thyroid-dose reconstruction.

The high concentrations of radioactive substances in surface water directly after the accident reduced rapidly and drinking water as well as water used for irrigation have very low concentrations of radioactivity today.

Due to radioactive decay, rain and wind, human activities, and countermeasures, surface contamination in urban areas by radioactive material has substantially reduced. External doses from urban areas are reduced by shielding effects compared to open areas.

At present, in most of the settlements subjected to radioactive contamination the air-dose rate above solid surfaces has returned to the pre-accident background level. Elevated air-dose rates remain mainly over undisturbed soil.

From Summer 1986 onwards, ^{137}Cs in milk and meat was the dominant radionuclide of concern in agricultural products. During the first few years substantial amounts of food were discarded from human consumption. The highest activity concentrations of ^{137}Cs have been found in food products from forested areas, especially in mushrooms, berries, game and reindeer. High ^{137}Cs-activity concentrations in fish occur in lakes with slow or no turnover of water, particularly if the lake is also shallow and mineral-nutrient poor.

The relative importance of ^{90}Sr in food products is reduced by its restricted deposition and because milk is the only major animal food product to which it is transferred. Strontium accumulation in bone of agricultural animals and fish occurs but does not typically lead to doses to humans.

There have been large long-term variations in ^{137}Cs-activity concentrations in food products due not just to deposition levels, but also to differences in soil types and management practices. In many areas there are still food products, particularly from extensive agricultural production systems and forests, with ^{137}Cs-activity concentrations exceeding intervention limits. Large land areas in the Former Soviet Union are still out of agricultural production because of radiological reasons.

The major and persistent problems in the affected areas occur in extensive agricultural systems with soils with a high organic content and animals grazing in unimproved pastures. This particularly affects rural residents in the former Soviet Union, who are commonly subsistence farmers with privately owned dairy cows.

In general, there has been an initial substantial reduction in transfer of ^{137}Cs to vegetation and animals, as would be expected due to weathering, physical decay, migration of radionuclides down the soil columns and reductions in bioavailability. However, in the last decade there has been little further obvious decline and long-term effective half lives have been difficult to quantify.

There has been a particularly slow decrease since deposition in ^{137}Cs-activity concentrations in some products from the forest, and some species of mushrooms are expected to have high ^{137}Cs-activity concentrations for decades to come. Under certain weather and ecological conditions, the biomass of mushrooms in autumn can be much higher than normal leading to relatively high seasonal increases in ^{137}Cs-activity concentrations in game. Thus, it must not always be assumed that ^{137}Cs-activity concentrations in animals will remain as they are now or decline each year.

Cesium-137 in timber is of minor importance, although doses in the wood industry have to be considered. Wood ash can contain higher amounts of ^{137}Cs. Forest fires have increased air-activity concentrations in local areas but not to a high extent.

Due to dilution there was never a high concentrations of ^{137}Cs in marine fish in the Black Sea or the Baltic Sea.

2.7.Further monitoring and research needed

Updated mapping of ^{137}Cs deposition in Albania, Bulgaria and Georgia should be performed in order to complete the study of post-Chernobyl contamination of Europe.

Improved mapping of ^{131}I deposition based both on historical environmental measurements carried out in 1986 and on recent measurements of ^{129}I in soil samples in areas where elevated thyroid-cancer incidence has been detected after the Chernobyl accident would reduce uncertainty of thyroid-dose reconstruction needed for determination of radiation risks.

Long-term monitoring of ^{137}Cs- and ^{90}Sr-activity concentrations in agricultural vegetable and animal products produced in areas with various soil and climate conditions and different agricultural practices should be performed for decades to come in the framework of limited target research programmes on selected sites.

Study of the distribution of ^{137}Cs and plutonium radionuclides in the urban environment (Pripyat, Chernobyl and some other contaminated towns) in future time periods would improve modelling of human external exposure and inhalation of radionuclides in case of a nuclear or radiological accident or malevolent action.

Continued long-term monitoring of specific forest products, such as mushrooms and game, needs to be carried out in those areas in which forests were significantly contaminated. Results from such monitoring are being used by the relevant authorities in affected countries to provide advice to the general public on their continued use of forests for recreation and gathering of wild foods.

In addition to the general monitoring of forest products required for radiation protection, more detailed, scientifically based, long-term monitoring of specific forest sites is required to provide an ongoing and improved understanding of the long term dynamics and persistence of radiocaesium contamination and its variability. Such monitoring is also necessary to improve the existing predictive models. Monitoring programmes are being carried out in several of the more severely affected countries such as Belarus and Russia, and it is important that these continue into the foreseeable future if current uncertainties on long term forecasts are to be reduced.

Aquatic systems have been intensively studied and monitored during the years after the Chernobyl accident and transfers and bioaccumulation of the most important long term contaminants, ^{90}Sr and ^{137}Cs, are now well understood. There is therefore little urgent need for major new research programmes on radioactivity in aquatic systems. There is, however, a requirement for continued (but more limited) monitoring of the aquatic environment and for further research in some specific areas, as detailed below.

Predictions of future contamination of aquatic systems by ^{90}Sr and ^{137}Cs would be improved by continued monitoring of radioactivity in key systems (the Pripyat-Dnieper system, the seas, and selected rivers and lakes in the more affected areas and in Western Europe). This would continue the excellent time series measurements of activity concentrations in water, sediments and fish and refine predictive models for these radionuclides.

Although they are currently of minor radiological importance in comparison to ^{90}Sr and ^{137}Cs, further studies of transuranic elements in the Chernobyl zone would improve predictions of environmental contamination in the very long term (100's-1000's of years). Further empirical studies of transuranic elements and ^{99}Tc are unlikely to have direct implications for radiological protection in the Chernobyl affected areas, but would further add to our knowledge of the environmental behaviour of these very long-lived radionuclides.

Future plans to reduce the water level of the Chernobyl Cooling Pond (CP) will have significant implications for its ecology and the behaviour of radionuclides/fuel particles in newly exposed sediments. Specific studies on the CP should continue. In particular, the dissolution of fuel particles has been shown to be significantly slower in aquatic systems than in terrestrial. Further study of fuel-particle dissolution rates in aquatic systems such as the Cooling Pond would improve knowledge of their behaviour.

SECTION 2 REFERENCES

ALEXAKHIN, R. A., KORNEEV, N. A. (Eds.) (1991) Agricultural Radioecology. Ecology, Moscow, 397 p. (in Russian).

AMIRO, B.D and DVORNIK, A. (1999) Fire and radioactivity in contaminated forests. In: Contaminated forests, Ed. LINKOV, I. and SCHELL, W.R., NATO Science Series Vol. 58, pp. 311-324.

ANDERSSON, K.G., ROED, J., JACOB, P., MECKBACH, R. (1995) Weathering of Cs-137 on various surfaces in inhabitated areas and calculated locations factors. In: Deposition of radionuclides, their subsequent relocation in the environment and resulting implications. Report EUR 16604 EN, European Commission, Luxemburg, pp. 47-56.

ANDERSSON, K.G., ROED, J. AND FOGH, C.L. (2002) Weathering of radiocaesium contamination on urban streets, walls and roofs. *Journal of Environmental Radioactivity,* 62, pp. 49-60.

AVILA, R., MOBERG, L., HUBBARD, L, FESENKO, S., SPIRIDONOV, S., and ALEXAKHIN, R. (1999) Conceptual overview of FORESTLAND – a model to interpret and predict temporal and spatial patterns of radioactively contaminated forest landscapes. In: Contaminated forests, Ed. by LINKOV, I. and SCHELL, W.R., NATO Science Series Vol. 58, pp. 173-183

BALONOV M.I., BRUK G.YA., ZVONOVA I.A., PITKEVICH V.A., BRATILOVA A.A., JESKO T.V. and SHUTOV V.N. (2000) Methodology of internal dose reconstruction for Russian population after the Chernobyl accident. *Radiation Protection Dosimetry*, 92, 247-253.

BARNETT, C. L., BERESFORD, N. A., FRANKLAND, J. C., SELF, P. L., HOWARD, B. J. and MARRIOTT, J.V.R. (2001) Radiocaesium intake in Great Britain as a consequence of the consumption of wild fungi. Mycologist, 15, 98-104.

BEBESHKO V.G., LIKHTAREV I.A., KOVGAN L.M. et al. (2001) General dosimetric passportisation of settlements in Ukraine radioactive contaminated after Chernobyl accident. Summarised data for 1998-2000. (Collection 9), MES, Kyiv, 59p.

BECHTEL-BATELLE-EDF (2003) Environmental Impact Assessment. Conceptual project of the New Safe Confinement: ChNPP Unit 4. Bechtel-EDF-Battelle.

BELLI M, TIKHOMIROV F, editors (1996) Behaviour of radionuclides in natural and semi-natural environments. Experimental collaboration project No 5. Final report. EUR-16531 EN.

BERESFORD, N.A. (1989) The transfer of Ag-110m to sheep tissues. Science of the Total Environment, 85, 81-90.

BERESFORD, N.A., CROUT, N.M.J., MAYES, R.W., HOWARD, B.J. and LAMB, C.S. (1998) Dynamic distribution of radioisotopes of cerium, ruthenium and silver in sheep tissues. *Journal of Environmental radioactivity*, 38, 317-338.

BERESFORD, N.A., MAYES, R.W., COOKE, A.I., BARNETT, C.L., HOWARD, B.J., LAMB C.S. and NAYLOR G.P.L. (2000) The importance of source dependent bioavailability in determining the transfer of ingested radionuclides to ruminant derived food products. *Environmental Science and Technology*, 34, 4455-4462.

BLAYLOCK, B.G. (1982) Radionuclide data bases available for bioaccumulation factors for freshwater biota. Nuclear Safety 23, 427-438.

BOBOVNIKOVA, TS.I., MAKHONKO, K.P., SIVERINA, A.A., RABOTNOVA, F.A., VOLOKITIN, A.A. (1991) Physical chemical forms of radionuclides in atmospheric fallout after the Chernobyl accident and their transformation in soil, *Atomnaya Energiya* № 5, pp. 449–454.

BOGDEVITCH, I.M. (1999) Soil conditions of Belarus and efficiency of potassium fertilizers. In: Proceedings of the 16 World Congress of Soil Science, IPI, Basel, pp. 21-26.

BOGDEVITCH I.M. (Ed.) (2002) Guide for Agricultural Practice on Lands Contaminated by Radionuclides in the Republic of Belarus for 2002-2005. 74 p. (in Russian).

BORZILOV, V.A., KLEPIKOVA, N.V. (1993) Effect of meteorological conditions and release composition on radionuclide deposition after the Chernobyl accident. In: The Chernobyl Papers (MERWIN, S.E., BALONOV, M.I., Eds.) Research Enterprises, Richland, Washington pp. 47–68.

BRUK G. Ya., SHUTOV V.N., BALONOV M.I., BASALEYAVA L.N. and KISLOV M.V. (1998) Dynamics of ^{137}Cs content in agricultural food products produced in regions of Russia contaminated after the Chernobyl accident. Radiation Protection Dosimetry, 76, 169-178.

BUCKLEY, M.J., BUGAI, D., DUTTON, L., GERCHIKOV, M. KASHPAROV, V. LEDENEV, A. VOITSEKHOVITCH, O., WEISS, D., ZHELEZNYAK, M.(2002) Drawing up and Evaluating Remediation Strategies for the Chernobyl Cooling Pond. Final Report NNC Ltd. C6476/TR/001/2002

BUGAI, D.A., WATERS, R.D., DZHEPO, S.P. and SKALSKY, A.S. (1996) Risks from radionuclide migration to groundwater in the Chernobyl 30-km zone. Health Physics 71: 9-18.

BULGAKOV, A.A., KONOPLEV, A.V., SMITH, J.T., HILTON, J., COMANS, R.N.J., LAPTEV, G.V. and CHRISTYUK, B.F. (2002) Modelling the long-term dynamics of radiocaesium in a closed lake system. Journal of Environmental Radioactivity 61, 41-53.

BUZULUKOV, YU.P. and YU.L. DOBRYNIN (1993) Release of radionuclides during the Chernobyl accident. In: The Chernobyl Papers (MERWIN, S.E., BALONOV, M.I., Eds.) Research Enterprises, Richland, Washington pp. 3-21.

CAMPLIN, W.C., LEONARD, D.R.P., TIPPLE, J.R. and DUCKETT, L. (1989) Radioactivity in freshwater systems in Cumbria (UK) following the Chernobyl accident. MAFF Fisheries Research Data Report No. 18, 90 p.

CARLSON, L. and HOLM, E. (1992) Radioactivity in *Fucus vesiculosus* L. from the Baltic Sea following the Chernobyl accident. Journal of Environmental Radioactivity 15, 231-248.

COMANS, R.N.J., MIDDELBURG, J.J., ZONDERHUIS, J., WOITTIEZ, J.R.W., DE LANGE, G.J., DAS, H.A. and VAN DER WEIJDEN C.H. (1989) Mobilization of radiocaesium in pore water of lake sediments. Nature 339, 367-369.

COMANS, R.N.J., BERGEN, T., BLUST, R., VAZ CARREIRO, C., FERNANDEZ, J., HAKANSON, L., SANSONE, U. and SMITH, J.T. (1999) Modelling fluxes and bioavailability of radiocaesium and radiostrontium in freshwaters in support of a theoretical basis for chemical/hydrological countermeasures. Final report to the European Commission.

CODEX ALIMENTARIUS COMMISSION (1989) Guideline Levels for Radionuclides in Foods Following Accidental Nuclear Contamination for Use in International Trade (CAC/GL 5-1989).

CROSS MA, SMITH JT, SAXÉN R and TIMMS DN. (2002) An analysis of the time dependent environmental mobility of radiostrontium in Finnish river catchments. Journal of Environmental Radioactivity 60, 149-163.

CROUT, N.M.J., BERESFORD, N.A. DAWSON, J. SOAR, J., MAYES, R.W. (2004) The transfer of ^{73}As, ^{109}Cd and ^{203}Hg to the milk and tissues of dairy cattle. J. Agr. Sci. Camb. 142, 203-212.

DE CORT, M., DUBOIS, G., FRIDMAN, SH.D., GERMENCHUK, M.G., IZRAEL, YU.A. JANSSENS, A., JONES, A.R., KELLY, G.N., KVASNIKOVA, E.V., MATVEENKO, I.I., NAZAROV, I.M., POKUMEIKO, YU.M., SITAK, V.A., STUKIN, E.D., TABACHNY, L.YA., TSATUROV, YU.S., AVDYUSHIN, S.I. (1998) Atlas of caesium deposition on Europe after the Chernobyl accident, Rept. EUR 16733, European Commission, Luxembourg.

DESMET, G.M., VAN LOON, L.R. and HOWARD, B.J. (1991) Chemical speciation and bioavailability of elements in the environment and their relevance to radioecology. *Science of the Total Environment*, 100, 105-124.

DEVILLE-CAVELIN, G., BIESOLD, H., CHABANYUK, V., BRUN-YABA, Ch., RUTSCHKOWSKY, N. and FRIEDERICHS, H.G. (2004) Synthesis Report of the Radioecology Project of the French-German Initiative for Chernobyl – FGI Report 04-01.

DREICER, M., AARKROG, A., ALEXAKHIN, R., ANSPAUGH, L., ARKHIPOV, N.P., JOHANSSON, K.-J. (1996) Consequences of the Chernobyl accident for the natural and human environments. In: One Decade after Chernobyl: Summing up the Consequences of the Accident, IAEA, Vienna, pp. 319–361.

DVORNIK, A.M. and ZHUCHENKO, T. (2004) Personal communication to the UN Chernobyl Forum.

EHLKEN, S., KIRCHNER, G. (2002) Environmental processes affecting plant root uptake of radioactive trace elements and variability of transfer factor data: a review. Journal of Environmental Radioactivity, 58: 97-112.

ELLIOTT, J.M., HILTON, J., RIGG, E., TULLETT, P.A., SWIFT, D.J. and LEONARD D.R.P. (1992) Sources of variation in post-Chernobyl radiocaesium in fish from two Cumbrian lakes (north-west England). Journal of Applied Ecology 29, 108-119.

EREMEEV, V.N., IVANOV, L.M., KIRWAN, A.D. and MARGOLINA, T.M. (1995) Amount of ^{137}Cs and ^{134}Cs radionuclides in the Black Sea produced by the Chernobyl accident. Journal of Environmental Radioactivity 27, 49-63.

EUROPEAN UNION (2000) MARINA–BALT PROJECT REPORT, The radiological exposure of the population of the European Community to radioactivity in the Baltic Sea, REPORT EUR 19200 EN, EUROPEAN COMMISSION, LUXEMBURG.

FESENKO, S.V., SANZHAROVA, N.I., SPIRIDONOV, S.I., ALEXAKHIN, R.M. (1996) Dynamics of ^{137}Cs Bioavailability in a Soil-Plant System in Areas of the Chernobyl Nuclear Power Plant Accident Zone with a Different Physico-chemical Composition of Radioactive Fallout. J. Environmental. Radioactivity. 34/3: 287-313.

FESENKO S.V., COLGAN P.A., LISSIANSKI K.B., VAZQUEZ C., GUARDANS R. (1997) The dynamics of the transfer of caesium-137 to animal fodder in areas of Russia affected by the Chernobyl accident and doses resulting from the consumption of milk and milk products. Radiation Protection Dosimetry. 69/4, 289-299.

FESENKO S.V., SOUKHOVA N., SANZHAROVA N., AVILA R., SPIRIDONOV S., KLEIN D., LUCOT E., BADOT P. (2001) Identification of processes governing long-term accumulation of ^{137}Cs by forest trees following the Chernobyl accident. Radiation and Environmental Biophysics. 40, 105-113.

FESENKO S.V. PAKHOMOV A. YU, PASTERNAK A.D., GORYAENOV V.A., FESENKO G.A., PANOV A.V. (2004) Regularities of changing in ^{137}Cs activity concentrations in the long term after the accident at the Chernobyl NPP. Radiation biology. Radioecology, 44, 1, pp. 35-49.

FLEISHMAN, D.G. (1973) Radioecology of marine plants and animals. In: Klechkovsky, V.M., Polikarpov, G.G. and Aleksakhin, R.M. (eds) Radioecology, John Wiley & Sons, pp. 347-370.

FLEISHMAN, D.G., NIKIFOROV, V.A., SAULUS, A.A. and KOMOV, V.T. (1994) ^{137}Cs in fish of some lakes and rivers of the Bryansk region and North-West Russia in 1990-1992. Journal of Environmental Radioactivity 24, 145-158.

GADD G.M. (1996) Role of microorganisms in the environmental fate of radionuclides. Endeavour, 20:150-156.

GOLIKOV, V.YU., BALONOV, M. I., JACOB, P. (2002) External Exposure of the Population Living in Areas of Russia Contaminated due to the Chernobyl Accident. Radiation and Environmental Biophysics, 41, Nr. 10, pp. 185-193.

HADDERINGH, R.H., VAN AERSSEN, G.H.F.M, RYABOV, I.N., KOULIKOV, O.A., BELOVA, N. (1997) Contamination of fish with ^{137}Cs in Kiev reservoir and old river bed of Pripyat near Chernobyl. In: Desmet, G., Blust, R., Comans, R.N.J., Fernandez, J., Hilton, J. and de Bettencourt, A. (eds) *Freshwater and Estuarine Radioecology*. Elsevier Studies in Environmental Science 68, Amsterdam, pp. 339-351.

HÅKANSON, L., ANDERSSON, T. and NILSSON, Å (1992) Radioactive caesium in fish from Swedish lakes in 1986-1988 – general pattern related to fallout and lake characteristics. *Journal of Environmental Radioactivity* 15, 207-229.

HANSEN, H.J.M. and AARKROG, A. (1990) A different surface geology in Denmark, The Faroe Islands, and Greenland influences the radiological contamination of drinking water. *Water Research* 24, 1137-1141.

HILTON, J., CAMBRAY, R.S., GREEN, N. (1992) Fractionation of radioactive cesium in airborne particles containing bomb fallout, Chernobyl fallout, and atmospheric material from the Sellafield site, *J. Environ. Radioact.* 15: 103–108.

HILTON, J., LIVENS, F.R., SPEZZANO, P. and LEONARD, D.R.P. (1993) Retention of radioactive caesium by different soils in the catchment of a small lake. Science of the Total Environment 129, 253-266.

HOLLÄNDER, W., GARGER, E. ((eds.) 1996) Contamination of surfaces by resuspended material. ECP-1, Final report, Report EUR 16527, European Commission, Luxembourg.

HOWARD, B.J., BERESFORD, N.A. and HOVE, K. (1991) Transfer of radiocesium to ruminants in unimproved natural and semi-natural ecosystems and appropriate countermeasures. *Health Physics,* 61, 715-725

HOWARD B. J. (2000) The concept of radioecological sensitivity *Radiation Protection Dosimetry*, 92, 29-34.

HOWARD, B.J., STRAND P, ASSIMAKOPOULOUS P., BRÉCHIGNAC F., GASCO C., MÉTIVIER H., MOBERG L., SMITH J.T., TAMPONNET C., TRUEBA C., VOIGT G., WRIGHT S.M. (2002) Estimation of radioecological sensitivity. In: Actes du congrès ECORAD-2001, France Radioprotection - Colloques, v. 37, C1-1167/ C1-1173.

INTERNATIONAL ADVISORY COMMITTEE (1991) The International Chernobyl Project, Technical Report, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1994) Handbook of parameter values for the prediction of radionuclide transfer in temperate environments, Technical Report Series No. 364, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY et al. (1996) International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources, Safety Series No.115, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (2002) Modelling the migration and accumulation of radionuclides in forest ecosystems. IAEA-BIOMASS-1, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (2003a) Assessing radiation doses to the public from radionuclides in timber and wood products. IAEA-TECDOC-1376, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (2003b). Marine Environment Assessment of the Black Sea. Final report. Technical co-operation project RER/2/003, IAEA, Vienna, 358 p.

INTERNATIONAL ATOMIC ENERGY AGENCY (2005) Radiation Conditions of the Dnieper River Basin: Assessment by the IAEA project team and recommendations for Strategic Action Plan. IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY AND FOOD AND AGRICULTURAL ORGANISATION (1994) Guidelines for Agricultural Countermeasures Following an Accidental Release of Radionuclides, Technical Reports Series No. 363. IAEA, Vienna, 115p.

INTERNATIONAL NUCLEAR SAFETY ADVISORY GROUP (1986) Summary Report on the Post-Accident Review Meeting on the Chernobyl Accident, Safety Series No. 75-INSAG-1, IAEA, Vienna.

IZRAEL, YU.A., S.M. VAKULOVSKY, V.A. VETROV et al. (1990). Chernobyl: Radioactive Contamination of the Environment. Gidrometeoizdat, Leningrad.

IZRAEL YU. (ED.) (1998) Atlas of Radioactive Contamination of European Russia, Belarus and Ukraine, Federal Service for Geodesy and Cartography of Russia, Moscow, 143 p.

IZRAEL, YU.A. (2002) Radioactive Fallout after Nuclear Explosions and Accidents. Elsewier, Amsterdam, 282 p.

JACOB, P, MECKBACH, R (2000) Measurements after the Chernobyl accident regarding the exposure of an urban population. In: Restoration of environments affected by residues from radiological accidents: Approaches to decision making. IAEA-TECDOC-1131. IAEA, Vienna, pp 34-41.

JOHANSSON, K. (1994) Radiocaesium in Game Animals in the Nordic Countries. In: Nordic Radioecology. The Transfer of Radionuclides through Nordic Ecosystems to Man. Ed. by Dahlgaard, H., Elsewier, Amsterdam, pp. 287-301.

JOHANSSON, K. (2003) Personal communication to the Chernobyl Forum.

JONSSON, B., FORSETH, T. and UGEDAL, O. (1999) Chernobyl radioactivity persists in fish. *Nature* 400, 417.

KANIVETS, V.V., VOITSEKHOVITCH, O.V., SIMOV, V.G. and GOLUBEVA, Z.A. (1999) The post-Chernobyl budget of ^{137}Cs and ^{90}Sr in the Black Sea. Journal of Environmental Radioactivity 43, 121-135.

KASHPAROV V.A., OUGHTON D.H., ZVARICH S.I., PROTSAK V.P., LEVCHUK S.E. (1999) Kinetics of fuel particle weathering and ^{90}Sr mobility in the Chernobyl 30-km exclusion zone. Health Physics 76/3: 251-259.

KASHPAROV V.A., LUNDIN S.M., ZVARICH S.I., YOSCHENKO V.I., LEVTCHUK S.E., KHOMUTININ YU.V., MALOSHTAN I.N., PROTSAK V.P. (2003) Territory contamination with the radionuclides representing the fuel component of Chernobyl fallout. *The Science of The Total Environment*, 317: 105-119.

KASHPAROV V.A., AHAMDACH N., ZVARICH S.I., YOSCHENKO V.I., MALOSHTAN I.N., DEWIERE L. (2004) Kinetics of dissolution of Chernobyl fuel particles in soil in natural conditions. Journal of Environmental Radioactivity .72/3: 335-353.

KIRCHNER, G. (1994) Transport of iodine and cesium via the grass-cow-milk pathway after the Chernobyl accident. Health Physics, 66: 653-665.

KNAPINSKA-SKIBA, D., BOJANOWSKI, R., RADECKI, Z. and MILLWARD, G.E. (2001) Activity concentrations and fluxes of radiocaesium in the Southern Baltic Sea. Estuarine, Coastal and Shelf Science 53, 779-786.

KONOPLEV, A.V., COMANS, R.N.J., HILTON, J., MADRUGA, M.J. BULGAKOV, A.A, VOITSEKHOVITCH, O.V., SANSONE, U., SMITH, J.T. and KUDELSKY, A.V. (1996) Physico-chemical and hydraulic mechanisms of radionuclide mobilization in aquatic systems. In: Karaoglou, A., Desmet, G, Kelly, G.N. and Menzel, H.G. (eds), The radiological consequences of the Chernobyl accident: EUR 16544 EN, European Commission, Luxembourg, pp. 121-135.

KONOPLEV, A.V., A.A. BULGAKOV, V.G. ZHIRNOV et al. (1998) Study of the behavior of Cs-137 and Sr-90 in Svyatoe and Kozhanovskoe lakes in the Bryansk region. Meteorol. Gidrol. Nr. 11, 78-87.

KONOPLEV A.V., BULGAKOV A.A., HOFMAN. F.O. et al. (1999) Validation of models of radionuclide wash-off from contaminated watersheds using Chernobyl data. J. of Enviromental Research, 42: 131-141.

KONOPLEV A.V., AVILA R., BULGAKOV A.A., JOHANSON K.-J., KONOPLEVA I.V., POPOV V.E. (2000) Quantitative assessment of radiocaesium bioavailability in forest soils. *Radiochim. Acta,* 88, pp. 789-792.

KRYSHEV, I.I. (1995) Radioactive contamination of aquatic ecosystems following the Chernobyl accident. Journal of Environmental Radioactivity 27, 207-219.

KRYSHEV, I.I. and RYABOV, I.N. (1990) About the efficiency of trophic levels in the accumulation of Cs-137 in fish of the Chernobyl NPP cooling pond. In: Ryabov, I.N. and Ryabtsev, I.A. (eds), Biological and radioecological aspects of the consequences of the Chernobyl Accident, USSR Academy of Sciences, Moscow, pp. 116-121.

KRYSHEV, I.I. and SAZYKINA, T.G. (1994) Accumulation factors and biogeochemical aspects of migration of radionuclides in aquatic ecosystems in the areas impacted by the Chernobyl accident. Radiochimica Acta 66/67, 381-384.

KUDELSKY, A.V., SMITH, J.T., OVSIANNIKOVA, S.V. and HILTON, J. (1996) The mobility of Chernobyl-derived ^{137}Cs in a peatbog system within the catchment of the Pripyat River, Belarus. Science of the Total Environment, 188, 101-113.

KUDELSKY, A.V., SMITH, J.T., ZHUKOVA, O.M., MATVEYENKO, I.I. and PINCHUK, T.M. (1998) Contribution of river runoff to the natural remediation of contaminated territories (Belarus). Proceedings of the Academy of Sciences of Belarus, 42, 90-94 (in Russian).

KUZMENKO M.I., ROMANENKO V.D., DEREVETS V.V., VOLKOVA O.M. et al. (2001) Impact of radionuclide contamination on hydrobionts of the Chernobyl Exclusion zone, 318 p. (in Russian).

LOS I. and LIKHTAREV I. (1993) The peculiarities of urban environmental contamination and assessment of actions aimed at reduction of public exposure. Int. J. of Radiation Hygiene, v.1, No 1, pp. 51-59.

MAKHONKO, K.P., KOZLOVA, E.G., VOLOKITIN, A.A. (1996) Radioiodine accumulation on soil and reconstruction of doses from iodine exposure on the territory contaminated after the Chernobyl accident, *Radiat. & Risk* № 7, pp. 90–142.

MATSUNAGA, T., T. UENO, H. AMANO, Y. TKATCHENKO, A. KOVALYOV, M. WATANABE and Y. ONUMA (1998) Characteristics of Chernobyl-derived radionuclides in particulate form in surface waters in the exclusion zone around the Chernobyl Nuclear Power Plants. *J. Contaminant Hydrology*, 35,101-113.

MATSUNAGA, T., S. NAGAO, T. UENO, S. TAKEDA, H. AMANO, YU. TKACHENKO (2004) Association of dissolved radionuclides released by the Chernobyl accident with colloidal materials in surface water, *Applied Geochemistry*, 19, 1581-1599.

MINISTRY OF HEALTH (1999) Radiation Safety Standards (RSS-99), Sanitary Rules SR 2.6.1.758-99, Ministry of Health of the Russian Federation, Moscow.

MONTE, L. (1995) Evaluation of radionuclide transfer functions from drainage basins of freshwater systems. Journal of Environmental Radioactivity 26, 71-82.

MŰCK, K., PRŐHL, G., LIKHTAREV, I., KOVGAN, L., MECKBACH, R. and GOLIKOV, V. (2002) A consistent radionuclide vector after the Chernobyl accident. *Health Phys.* 82, pp. 141-156.

NASVIT O.I. (2002). Radioecological Situation in the Cooling Pond of Chornobyl NPP. In. Recent Research Activity about Chernobyl NPP Accident in Belarus, Ukraine and Russia. Kioto University, KURAI-KR-79, pp. 74-85.

NIELSEN, S.P., BENGTSON, P., BOJANOWSKY, R., HAGEL, P., HERRMANN, J., ILUS, E., JAKOBSON, E., MOTIEJUNAS, S., PANTELEEV, Y., SKUJINA, A. and SUPLINSKA, M. (1999) The radiological exposure of man from radioactivity in the Baltic Sea. The Science of the Total Environment 237/238, 133-141.

NYLÉN, T. (1996) Uptake, turnover and transport of radiocaesium in boreal forest ecosystems, Thesis, Swedish University of Agricultural Sciences, Uppsala Sweden, ISBN 91-576-5149-3.

OULD-DADA, Z. and N. M. BAGHINI (2001) Resuspension of small particles from tree surfaces. Atmospheric Environment, 35, 3799–3809.

PANFILOV, A. (1999) Countermeasures for radioactively contaminated forests in the Russian Federation. In: Contaminated forests, Ed. by LINKOV, I. and SCHELL, W.R., NATO Science Series Vol. 58, pp. 271 – 279.

POLLANEN R., VALKAMA I., TOIVONEN H. (1997) Transport of radioactive particles from the Chernobyl accident. *Atmospheric Environment*. 31, pp.3575-3590.

PRISTER B.S., BARJAKHTAR V.G., PERELYATNIKOVA L.V., VYNOGRADSKAJA V.D., RUDENKO V.A., V.A. GRYTSCHUK N.R. and T.N. IVANOVA (2003) Experimental substantiation in parametrization of the model describing ^{137}Cs and ^{90}Sr behavior in a soil-plant system. Environment Science and Pollution Research – Special issue 1, pp. 126–136.

PRÖHL, G. and HOFFMAN, F.O. (1996) Radionuclide interception and loss processes in vegetation. In: Modelling of Radionuclide Interception and Loss Processes in Vegetation and of Transfer in Semi-Natural Ecosystems. Second Report of the VAMP Terrestrial Working Group, IAEA TECDOC-857, IAEA, Vienna, pp. 9-48.

RAVILA, A. and HOLM, E. (1994) Radioactive elements in the forest industry. The Science of the Total Environment, 32: 339-356.

RENAUD PH., K. BEAUGELIN, H. MAUBERT ET PH. LEDENVIC (1999) Les retombees en France de l'accident de Tchernobyl, consequences radioecologiques et dosimetriques. IPSN Collection. EDP Sciences.

ROED, J., ANDERSSON, K. (2002). Personal communication to the UN Chernobyl Forum.

RUHM, W., Kammerer, L., Hiersche, L. and Wirth, E. (1997) The $^{137}Cs/^{134}Cs$ ratio in fungi as an indicator of the major mycelium location in forest soil, Journal of Environmental Radioactivity, 35: 129-148.

RYABOV, I., N. BELOVA, L. PELGUNOVA et al. (1996) Radiological phenomena of the Kojanovskoe Lake. In: The Radiological Consequences of the Chernobyl Accident. (A. Karaoglou, G. Desmet, G.N. Kelly et al., eds.). EUR 16544, pp. 213-216.

SALBU B., KREKLING T., LIND D.H., OUGHTON D.H., ET AL. (2001) High energy X-ray microscopy for characterization of fuel particles. Nuclear Instruments and Methods in Physics Research, A467-468: 1249-1252.

SANDALLS, F.J., SEGAL, M.G., VICTOROVA, N.V. (1993) Hot particles from Chernobyl: A review, *J. Environ. Radioact.* 18, pp. 5–22.

SANSONE, U. and VOITSEKHOVITCH, O.V. (1996) Modelling and study of the mechanisms of the transfer of radionuclides from the terrestrial ecosystem to and in water bodies around Chernobyl. Report EUR 16529 EN. European Commission, Luxembourg.

SANTSCHI, P.H., BOLLHALDER, S., ZINGY, S., LUCK, A. and FARRENKOTHER, K. (1990) The self cleaning capacity of surface waters after radionuclide fallout. Evidence from European lakes after Chernobyl 1986-88. Environmental Science and Techno*logy* 24, 519-527.

SANZHAROVA N.I., FESENKO S.V., ALEXAKHIN R.M., ANISIMOV V.S., KUZNETSOV V. K., CHERNYAEVA L.G. (1994) Changes in the forms of ^{137}Cs and its availability for plants as dependent on properties of fallout after the Chernobyl nuclear power plant accident. *The Science of the Total Environment*, 154, 9-22.

SANZHAROVA N.I., FESENKO S.V., KOTIK V.A., SPIRIDONOV S.I. (1996) Behaviour of radionuclides in meadows and efficiency of countermeasures. *Radiation Protection Dosimetry. 64,* 43-48.

SAURAS YERA, T., VALLEJO, V. R., VALCKE, E., COLLE, C., FÖRSTEL, H., MILAN, R., JOUGLET, H. (1999) ^{137}Cs and ^{90}Sr root uptake predictions under close-to-real controlled conditions. Journal of Environmental Radioactivity, 45: 191-217.

SAXÉN, R. and ILUS, E. (2001) Discharge of ^{137}Cs and ^{90}Sr by Finnish rivers to the Baltic Sea in 1986–1996. Journal of Environmental Radioactivity 54, 275-291.

SHAW, G. (2004) Personal communication to the Chernobyl Forum.

SHAW, G., R. AVILA, S. FESENKO, A. DVORNIK and T. ZHUCHENKO (2002) Modelling the Behaviour of Radiocaesium in Forest Ecosystems. In: Modelling Radioactivity in the Environment, ed. by E. M. Scott, Elsevier Science Publishers.

SHAW, G., VENTER, A., R. AVILA, R. BERGMAN, A. BULGAKOV, P CALMON, S. FESENKO, M. FRISSEL, F. GOOR, A. KONOPLEV, I. LINKOV, S. MAMIKHIN, L. MOBERG, A. ORLOV, A. RANTAVAARA, S. SPIRIDONOV, Y. THIRY (2004) Radionuclide Migration in Forest Ecosystems – Results of a Model Validation Study. Journal of Environmental Radioactivity, accepted for publication in the special issue on IAEA BIOMASS Programme.

SHCHEGLOV, A. I., O. B. TSVETNOVA and A. L. KLYASHTORIN (2001) Biogeochemical migration of technogenic radionuclides in forest ecosystems. Nauka, Moscow.

SHESTOPALOV V.M et al. (2002). Chernobyl disaster and groundwater. (ed. by V. Shestopalov) Balkema Publisher, 289 p.

SHESTOPALOV V.M., KASHPAROV V.A., IVANOV Y.A. (2003) Radionuclide Migration into the Geological Environment and Biota Accident. Environment Science and Pollution Research – Special issue 1, pp. 39 – 47.

SHEVCHOUK, V.E. GOURACHEVSKY (EDS.) (2001) 15 Years After Chernobyl Catastrophe: Consequences in the Republic of Belarus and Their Overcoming. National Report, Committee on the Problems of the Consequences of the Accident at the Chernobyl NPP, Minsk, 118 p.

SCHUCK, A.; PÄIVINEN, R.; HÄME, TUOMAS; VAN BRUSSELEN, J.; KENNEDY, P.; FOLVING, S. (2003) Compilation of a European forest map from Portugal to the Ural mountains based on earth observation data and forest statistics. Forest policy and economics, 5/2: 187 – 202.SHUTOV V., BAZUKIN, A., KADUKA, M. et al. (2004) Personal communication to the Chernobyl Forum.

SMITH, J.T., FESENKO, S.V., HOWARD, B.J., HORRILL, A.D., SANZHAROVA, N.I., ALEXAKHIN, R.M., ELDER, D.G. and NAYLOR, C. (1999) Temporal change in fallout ^{137}Cs in terrestrial and aquatic systems: a whole ecosystem approach. Environmental Science and Technology 33, 49-54.

SMITH, J.T., KUDELSKY, A.V., RYABOV, I.N. and HADDERINGH, R.H. (2000a) Radiocaesium concentration factors of Chernobyl-contaminated fish: a study of the influence of potassium, and “blind” testing of a previously developed model. Journal of Environmental Radioactivity 48, 359-369.

SMITH, J.T., COMANS, R.N.J., BERESFORD, N.A., WRIGHT, S.M., HOWARD, B.J. and CAMPLIN, W.C. (2000b) Chernobyl’s legacy in food and water. Nature 405, 141.

SMITH, J.T., VOITSEKHOVITCH, O.V., HÅKANSON, L. and HILTON, J. (2001) A critical review of measures to reduce radioactive doses from drinking water and consumption of freshwater foodstuffs. J. Environmental Radioactivity, 56, 11-32.

SMITH, J.T., WRIGHT, S.M., CROSS, M.A., MONTE, L., KUDELSKY, A.V., SAXÉN, R., VAKULOVSKY, S.M., TIMMS, D.N. (2004) Global analysis of the riverine transport of ^{90}Sr and ^{137}Cs. Environmental Science and Technology 38, 850-857.

STRAUME, T., MARCHETTI, A.A., ANSPAUGH, L.R., KHROUCH, V.T., GAVRILIN, YU.I., SHINKAREV, S.M., DROZDOVITCH, V.V., ULANOVSKY, A.V., KORNEEV, S.V., BREKESHEV, M.K., LEONOV, E.S., VOIGT, G., PANCHENKO, S.V., MINENKO, V.F. (1996) The feasibility of using ^{129}I to reconstruct ^{131}I deposition from the Chernobyl reactor accident, *Health Phys.* 71, pp.733–740.

TAMPONNET, C., C. PLASSARD, N. PAREKH and S. SANCHEZ (2001) Impact of microorganisms on the fate of radionuclides in rhizospheric soils. In: Radioactive Pollutants-Impact on the environment, (F. Bréchignac and B. Howard, eds). EDP Sciences, Les Ulis France; pp175-185.

TIKHOMIROV, F.A. and SHCHEGLOV, A.I. (1994) Main investigation results on the forest radioecology in the Kyshtym and Chernobyl accident zones. The Science of the Total Environment, 157: 45-57.

UKRAINIAN HYDRO-METEOROLOGICAL INSTITUTE (2004) Database of radiation measurements of aquatic samples.

UKRAINIAN INSTITUTE OF AGRICULTURAL RADIOLOGY (1998) Recommendations on Agriculture Management on Contaminated Territories, UIAR, Kyiv, 102 p. (In Ukrainian).

UNITED NATIONS, Sources, Effects and Risks of Ionizing Radiation (1988 Report to the General Assembly, with Annexes), Scientific Committee on the Effects of Atomic Radiation, UN, New York (1988) 309–374.

UNITED NATIONS, Sources and Effects of Ionizing Radiation (2000 Report to the General Assembly, with Scientific Annexes), Scientific Committee on the Effects of Atomic Radiation, UN, New York (2000) Volume II 451–566.

UNITED NATIONS DEVELOPMENT PROGRAMME (2001) Assessment and forecast of the water quality at the areas affected by Chernobyl accident (Bryansk Region) UNDP Project RUS/95/004.

VAKULOVSKY, S.M., VOITSEKHOVITCH, O.V., KATRICH, I. YU, MEDINETS, V.I., NIKITIN, A.I., CHUMICHEV, V.B. (1990) Radioactive contamination of water systems in the area affected by releases from the Chernobyl nuclear power plant accident. In: Environmental contamination following a major nuclear accident. IAEA, Vienna, V. 1, pp. 231-246.

VAKULOVSKY, S.M., NIKITIN, A.I., CHUMICHEV, V.B., KATRICH, I.YU., VOITSEKHOVITCH, O.A., MEDINETS, V.I., PISAREV, V.V., BOVKUM, L.A. and KHERSONSKY, E.S. (1994) Cs-137 and Sr-90 contamination of water bodies in the areas affected by releases from the Chernobyl Nuclear Power Plant accident: an overview. Journal of Environmental Radioactivity 23, 103-122.

VANDENHOVE ,H. (ED.) (2002) PHYTOR, Evaluation of Willow Plantation for the Phytorehabilitation of Contaminated Arable Lands and Flood Plane Areas. Final Report, SCK-CEN ERB IC15-CT98 0213. 75.

VOITSEKHOVITCH, O.V., BORZILOV, V.A. and KONOPLEV, A.V. (1991) Hydrological aspects of radionuclide migration in water bodies following the Chernobyl accident. In: Comparative Assessment of the Environmental Impact of Radionuclides Released during Three Major Nuclear Accident: Kyshtym, Windscale, Chernobyl, Report EUR 13574, European Commission, Luxembourg, pp. 528-548.

VOITSEKHOVITCH, O.V., SANSONE, U., ZHELESNYAK, M. and BUGAI, D. (1996) Water quality management of contaminated areas and its effects on doses from aquatic pathways. In: Karaoglou, A., Desmet, G, Kelly, G.N. and Menzel, H.G. (eds) The radiological consequences of the Chernobyl accident. Report EUR 16544 EN, European Commission, Luxembourg, pp. 401-410.

VOITSEKHOVITCH O.V., KANIVETS, V.V. (1997) Cs-137 in the bottom sediments of the Dnieper reservoirs. Maps.

VOITSEKHOVITCH, O.V., NASVIT, O., LOS, I. and BERKOVSKI, V. (1997) Present thoughts on aquatic countermeasures applied to regions of the Dnieper river catchment contaminated by the 1986 Chernobyl accident. In: G. Desmet, R. Blust, R.N.J. Comans, J.A. Fernandez, J. Hilton. & A. de Bettencourt (eds.) Freshwater and estuarine radioecology, Amsterdam: Elsevier, pp. 75-86.

ZHELEZNYAK M., SHEPELEVA, V., SIZONENKO, V. and MEZHUEVA, I. (1997) Simulation of countermeasures to diminish radionuclide fluxes from the Chernobyl zone via aquatic pathways. Radiation Protection Dosimetry, 73, 181-186.

ZIBOLD, G. (2004) Personal communication to the UN Chernobyl Forum.

ZIBOLD, G., KAMINSKI, S., KLEMT, E. and SMITH, J.T. (2002). Time-dependency of the ^{137}Cs activity concentration in freshwater lakes, measurement and prediction. Radioprotection – Colloques 37, 75-80.

3. ENVIRONMENTAL COUNTERMEASURES AND REMEDIATION

Since the very first days after the Chernobyl accident had happened, as soon as severe environmental radiation conditions were revealed and human-exposure levels broadly assessed, the need for application of urgent protection actions became evident. The range of countermeasures applied with the aim of protecting the public from Chernobyl-caused radiation was notably wide, i.e., from urgent evacuation in 1986 of inhabitants from the near area of highest radioactive contamination in order to save their lives and health, to long-term monitoring of radionuclides in foodstuffs in many European countries aiming to prevent consumption of food contaminated above established action levels. The whole spectrum of the applied countermeasures and their effectiveness has been considered in a number of international reports (UNSCEAR 1988; IAEA 1990; IAC 1991; IAEA 1996a; EC 1996; UNSCEAR 2000; IAEA 2001).

The subject of the present report is mainly the countermeasures that have been applied to the environment in order to reduce radiation impact on the man. At the time of the Chernobyl accident, the philosophy of radiation protection of non-human species was not developed sufficiently to be practically applied for justification of appropriate countermeasures. This methodology is currently still under development (ICRP 2003).

The report does not consider specifically the past emergency and mitigation actions at the destroyed reactor and at the whole Chernobyl NPP aiming to reduce radioactive release to the environment; these issues have been presented elsewhere (IAEA 1990). Neither it considers protection of workers involved in the emergency and mitigation actions nor performance and radiological efficiency of evacuation of 116 thousand residents of the most contaminated areas of the USSR in 1986 and subsequent relocation of 220 thousand residents more in 1989-1992 (UNSCEAR 2000) from contaminated areas to non-contaminated ones.

The ecosystems to which environmental countermeasures have been applied since 1986 and by the present time are urban, agricultural, forest and aquatic ones. Most of these countermeasures were driven by relevant international and national radiological criteria and derived reference levels.

3.1. Radiological criteria

Countermeasures called protection actions at the emergency stage and remediation actions at the post-emergency stage are actions taken to reduce the level of exposure as much as is reasonably achievable. A fundamental aspect of radiation-protection philosophy is to optimise the dose averted against the costs of applying the countermeasure. However, the costs and benefits of countermeasures are not always quantifiable in purely monetary terms. The advantages of countermeasures often include reassurance and a decrease in anxiety in the affected population. But, countermeasures may also have negative consequences, either directly to ecosystems (e.g., disruption of nutrient cycles) or to sectors of the population either economically or due to disruption of normal life.

3.1.1. International radiological criteria and standards

At the moment of the Chernobyl accident in 1986 the relevant international radiation-protection standards for protection of the public and the workers were presented in the then basic ICRP Publication 26 (ICRP 1977) and in the specific recommendations for protection of the public in the event of a major radiation accident, ICRP Publication 40 (ICRP 1984). The corresponding IAEA basic safety standards based on ICRP recommendations (ICRP 1977) were issued in 1982 (IAEA 1982). The basic principles of modern radiation protection, i.e., justification, optimization and limitation, and clear distinction between practice and intervention situations were introduced in these documents. At the level of radiological knowledge and radiation-protection philosophy existing at that time period, the annual limit of occupational exposure was established equal to 50 mSv and that for public exposure was 5 mSv. The latter value was perceived as a safe level of human exposure.

Special limits for public radiation protection in case of nuclear or radiological emergencies were not specifically established in these documents and instead it was recommended:

— By almost all means to reduce human accidental exposure in doses that may result in deterministic health effects (acute radiation syndrome, radiation damage to particular organs or tissues); and

— To intervene, i.e., to apply and subsequently withdraw countermeasures aiming at reduction of stochastic health effects (cancer, genetic anomalies), based on an optimisation assessment taking into account both the collective dose reduction and economic and social intervention costs.

The most relevant ICRP guidance (ICRP 1984) recommended some generic two-level criteria for intervention in the early accidental phase, i.e., for sheltering - 5 to 50 mSv of whole body dose or 50 to 500 mSv to particular organs, for administration of stable iodine aiming at thyroid protection against intake of radioiodines - 50 to 500 mSv to the thyroid, for evacuation - 50 to 500 mSv of whole body dose or 500 to 5000 mSv to particular organs. For intermediate accidental phase the generic criteria 5 to 50 mSv of whole body dose or 50 to 500 mSv to particular organs were recommended for control of foodstuff contamination with radionuclides, and 50 to 500 mSv of whole body dose for relocation.

Afterwards, in connection with public concerns of the radiological consequences of the Chernobyl accident, new additional international regulations have been developed. Thus, in 1989 the Codex Alimentarius Commission approved guidance levels for radionuclides in food moving in international trade for the first year after a major nuclear accident – Table 3.1 (CODEX 1989)[2].

New basic international radiation-protection standards for protection of the public and workers were developed by ICRP in 1990 after research data had proved that radiation-risk coefficients for stochastic human health effects were substantially higher than suggested before. Therefore, the annual limits of exposure were substantially (by a factor of 2.5 to 5) reduced and established equal to 20 mSv for workers and 1 mSv for members of the general public (ICRP 1991). The latter value is currently perceived as a safe level of human exposure.

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[2] The Codex Guideline Levels for Radionuclides in Foods Following Accidental Nuclear Contamination for Use in International Trade (CAC/GL 5-1989) are currently being revised.

TABLE 3.1. GUIDELINE LEVELS FOR RADIONUCLIDES IN FOODS FOLLOWING ACCIDENTAL NUCLEAR CONTAMINATION FOR USE IN INTERNATIONAL TRADE

<table>
<tr><th>Radionuclides</th><th>Foods for general consumption ($Bq\ g^{-1}$)</th><th>Milk and infant foods ($Bq\ g^{-1}$)</th></tr>
<tr><td>Cs-134, Cs-137</td><td rowspan="2">1</td><td>1</td></tr>
<tr><td>I-131</td><td rowspan="2">0.1</td></tr>
<tr><td>Sr-90</td><td>0.1</td></tr>
<tr><td>Pu-239, Am-241</td><td>0.01</td><td>0.001</td></tr>
</table>

Still, special limits for public protection in case of nuclear or radiological emergency were not established in these documents, and instead appropriate specific recommendations were later elaborated on intervention for protection of the public in a radiological emergency (ICRP 1993). In this guidance, the optimisation concept was confirmed as the basic one applicable in case of emergency and further elaborated with regard to dose averted as the intervention result – Fig. 3.1. ICRP discarded the previous 2-level intervention criteria and recommended instead some intervention levels (in terms of averted effective dose), i.e., 50 mSv for sheltering, 500 mSv (thyroid dose) for administration of stable iodine, 500 mSv for evacuation, 1000 mSv (lifetime dose) for relocation and 10 mSv per year for control of foodstuffs.

The more recent ICRP Publication 82 (ICRP 1999) considered public-radiation protection in conditions of prolonged radiation exposure, such as in areas contaminated due to the Chernobyl accident. In this document, the ICRP generally recommends keeping the optimisation principle, but also suggests generic radiological criteria for making decisions on countermeasure application. In particular, it is proposed to use the value of the existing annual dose, including external and internal doses from natural and man-made radionuclides, of 10 mSv as the generic dose below which intervention is hardly expedient. This does not exclude intervention at lower annual existing dose, if site-specific optimisation analysis proves countermeasure expediency. *Inter alia*, the ICRP recommended a generic intervention-exemption level for radionuclides in commodities dominating human exposure equal to 1 mSv per year. This criterion could be applied for justification of the reference levels of radionuclides in food.

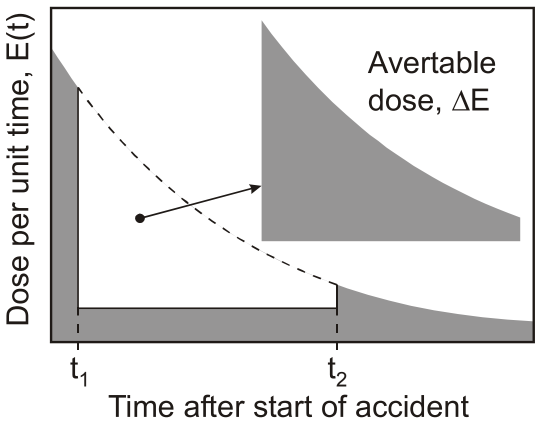


FIG. 3.1. Avertable dose and effective dose accumulated per unit time as a function of time when the protective measure is introduced at time t_1 and lifted again at time t_2.

3.1.2. National radiological criteria and standards

Limitations of human exposure caused by the Chernobyl accident, including standards for radionuclides in food, drinking water, timber, etc., were introduced soon after the accident firstly by the USSR, but also by many European countries, i.e., Nordic countries, EU countries and Eastern European countries (UNSCEAR 1988).

In accordance with the Standards of Radiation Safety (USSR Ministry of Health 1977) in force in 1986, the USSR Ministry of Health introduced a temporary limit of average equivalent whole body dose of 100 mSv for the first year after the Chernobyl accident (from 26 April 1986 till 26 April 1987), then 30 mSv for the second year and 25 mSv in each of 1988 and 1989 (IAC 1991). In all, till 1 January 1990 the dose of general public not exceeding 173 mSv was allowed from the radioactive fallout of the Chernobyl accident.

In order to limit internal exposure of the population, the temporary permissible levels (TPLs) of radionuclide content in food products and drinking water were elaborated in the USSR. Table 3.2 presents the TPLs for main food products (IAC 1991; Balonov 1993). The first TPL set approved by the USSR Ministry of Health on 6 May 1986 concerned the ^{131}I-activity concentrations as a dominating factor of human internal exposure in the early period of the accident, and was aimed at limitation of the thyroid dose of children to 300 mGy. The next TPL set adopted on 30 May 1986 concerned content of all beta emitters in food products caused by surface contamination, but was substantiated with primary attention to ecologically mobile and long-lived caesium radionuclides. The later TPL sets put in force since 1988 (TPL-88) and 1991 (TPL-91) concerned the sum of ^{134}Cs and ^{137}Cs activities. The TPL-91 for caesium radionuclides was supplemented by TPLs for ^{90}Sr.

Annual consumption by rural inhabitants of the usual food ration, all components of which contained caesium radionuclides at the level of TPL-86, would cause internal dose of less than 50 mSv, TPL-88 – less than 8 mSv, and TPL-91 – less than 5 mSv.

TABLE 3.2. TEMPORARY PERMISSIBLE LEVELS (TPL, Bq kg-1) OF RADIONUCLIDE CONTENT IN FOOD PRODUCTS AND DRINKING WATER ESTABLISHED IN THE USSR IN 1986–1991, AFTER THE CHERNOBYL ACCIDENT (IAEA 1991; BALONOV 1993)

TPL	4104-88	129-252	TPL-88	TPL-91	
Date of adoption	06.05.1986	30.05.1986	15.12.1987	22.01.1991	
Nuclide	^{131}I	β-emitters	^{134}Cs + ^{137}Cs	^{134}Cs + ^{137}Cs	^{90}Sr
Drinking water	3700	370	18.5	18.5	3.7
Milk	370–3700	370–3700	370	370	37
Dairy products	18500–74000	3700–18500	370–1850	370–1850	37–185
Meat and meat products	–	3700	1850-3000	740	–
Fish	37000	3700	1850	740	–
Eggs	–	37000	1850	740	–
Vegetables, fruits, potato, root-crops	–	3700	740	600	37
Bread, flour, cereals	-	370	370	370	37

TABLE 3.3. CURRENT ACTION LEVELS (Bq kg^{-1}) FOR CAESIUM RADIONUCLIDES IN FOOD PRODUCTS ESTABLISHED AFTER THE CHERNOBYL ACCIDENT

<table>
<tr><th>Country, International body</th><th>CAC</th><th>EU</th><th>Belarus</th><th>Russia</th><th>Ukraine</th></tr>
<tr><td>Year of adoption</td><td>1989</td><td>1986</td><td>1999</td><td>2001</td><td>1997</td></tr>
<tr><td>Milk</td><td rowspan="8">1000</td><td rowspan="2">370</td><td>100</td><td>100</td><td>100</td></tr>
<tr><td>Infant food</td><td>37</td><td>40–60</td><td>40</td></tr>
<tr><td>Dairy products</td><td rowspan="6">600</td><td>50–200</td><td>100–500</td><td>100</td></tr>
<tr><td>Meat and meat products</td><td>180–500</td><td>160</td><td>200</td></tr>
<tr><td>Fish</td><td>150</td><td>130</td><td>150</td></tr>
<tr><td>Eggs</td><td>–</td><td>80</td><td>6 Bq/egg</td></tr>
<tr><td>Vegetables, fruits, potato, root-crops</td><td>40–100</td><td>40–120</td><td>40–70</td></tr>
<tr><td>Bread, flour, cereals</td><td>40</td><td>40–60</td><td>20</td></tr>
</table>

Action levels for ^{131}I in foods established in some European countries in May 1986 varied in the range of 500 to 5000 Bq kg^{-1} among countries. Later on, the EU authorities established two values for caesium radionuclides in imported foods, one for milk and infant food and another for all other food products, see Table 3.3 (EC 1986; IAC 1991). Similar values were introduced in Nordic countries with an exception for wild foods (reindeer meat, game, freshwater fish, forest berries, fungi and nuts) that are important products for some local populations and especially for indigenous people. Thus, in the first month Sweden imposed action levels of 5 kBq kg^{-1} for ^{131}I and 10 kBq kg^{-1} for ^{137}Cs in imported food; for domestic foods the respective values were 2 and 1 kBq kg^{-1}. In the middle of May, action levels of 300 Bq kg^{-1} ^{137}Cs for all food and 2 kBq kg^{-1} for ^{131}I in milk and dairy products were introduced. For wild foods produced or consumed in the Nordic countries, the action levels varied between 1500 and 6000 Bq kg^{-1} in different countries and time periods.

Along with the standards for food products, the standards for agricultural raw materials, wood (see Section 3.3), herbs, and for beta contamination of different surfaces were introduced in the USSR (IAC 1991).

The general policy of the USSR and later on of the authorities in the separate republics was to reduce both the radiological criteria and TPLs along with natural improvement of radiological conditions due to radionuclide decay and penetration/fixation in soil. Gradual TPL reduction has been used as an instrument to force producers to apply technologies that decreased radionuclide content in products in order to limit associated human exposure. The TPLs were substantiated by expert weighting between intentions to reduce internal dose in populations and not to terminate profitable agricultural production and forestry in the controlled areas. Different reference levels for numerous groups of food products were established with the aim not to restrict consumption of any foods, unless the dose criterion might be exceeded.

At the end of 1991, the USSR had split into separate countries, among them Belarus, Russia and Ukraine, which had been strongly affected by the Chernobyl accident, and afterwards each country implemented its own policy of radiation protection of the public. Because of acceptance by ICRP in 1990 of the annual effective dose limit for the public in practice situations equal to 1 mSv, this level was considered by the authorities of the three countries as safe also in post-emergency conditions. Therefore, it is still used in national legislations as an intervention level of annual dose caused by Chernobyl fallout for introduction of countermeasures, including long-term remediation measures.

Current national TPLs for food products, drinking water and wood in the three countries are comparable to each other, see Table 3.3, and all of them are substantially lower than both EU maximum permissible levels for import (EC 1986) and Codex Alimentarius Commission's guidance levels for radionuclides in food moving in international trade (CAC 1989).

The state authorities have struggled to meet established TPLs for products and the dose criteria via implementation of environmental countermeasures described below and by inspection of foods throughout each country.

3.2. Urban decontamination

Decontamination of settlements was one of the main countermeasures applied against external exposure of the public and clean-up workers during the initial stage of response to the Chernobyl accident. The immediate purpose of settlement decontamination was the removal of radiation sources distributed in urban environments inhabited by humans.

Analysis of the external dose-formation process in different population groups living in contaminated areas revealed that a significant fraction of dose is received by people from sources located in soil, on coated surfaces like asphalt and concrete and to a small extent on building walls and roofs. This is why most dose-effective decontamination technologies involved removal of the upper soil layer.

The decontamination efficiency can be characterised by means of the following parameters: Dose Rate Reduction Factor (DRRF), which presents relative reduction of dose rate above a surface following decontamination, and Dose Reduction Factor (DRF) that presents reduction of the effective external dose to an individual from gamma-emitting radionuclides deposited in the environment.

3.2.1. Decontamination research

In order to ensure high decontamination effectiveness and to keep the associated costs low, several research projects have been implemented aiming to determine the values of the DRRF and DRF factors for particular decontamination technologies applied to different surfaces and objects in the anthropogenic environment (Balonov et al. 1992; Hubert et al. 1996; Roed et al. 1998). Reports from these experimental and theoretical works contain validated models of urban decontamination, and sets of model parameters and practical recommendations for clean up in different time periods after urban radioactive contamination. A preliminary remediation assessment based on well developed cost-benefit techniques is recommended in order to justify decontamination and optimize its implementation.

According to these and other studies, contributions of different urban surfaces to human external doses and associated opportunities for dose reduction are determined by settlement and house design, construction materials applied, the populations' habits, mode of radionuclide deposition (dry or wet), radionuclide and physicochemical composition of the fallout, and on time, see Section 2.2.

Following dry deposition, street cleaning, removal of trees and shrubs and ploughing gardens are efficient and inexpensive means of achieving very significant reductions in dose and would rate highly in a list of short-term priorities. Roofs are important contributors to dose but the cost of cleaning roofs is high and this would not rank highly in a list of priorities. Walls contribute little to dose, are expensive and difficult to decontaminate and would therefore carry a very low rating.

In the case of wet deposition, the garden and lawn both in the short and the long term will be given first priority, because a considerable reduction in dose (~60%) can be achieved at relatively low cost. Street cleaning would also be useful.

While planning decontamination for the long term, it is important to take into account the contribution of external dose to the total (external + internal) dose. In areas dominated by clay soils, transfer of cesium radionuclides along the food chain and associated internal doses are low. In these areas the relative decrease of total dose is close to the DRF value. In contrast, in sandy and peaty soil areas where long-term internal exposure dominates, the relative decrease of the total dose due to village decontamination is expected to be less significant.

3.2.2. Chernobyl experience

Large-scale decontamination was performed in 1986-1989 in cities and villages of the USSR most contaminated after the Chernobyl accident. This activity was performed usually by military personnel and included washing of building with water or special solutions, cleaning of residential areas, removal of contaminated soil, cleaning and washing of roads, and decontamination of open water supplies. Special attention was paid to kindergartens, schools, hospitals, and other buildings frequently visited by large numbers of persons. In total, about one thousand settlements were treated; this included cleans tens of thousands of residence and social buildings and more than a thousand agricultural farms (Balonov et al. 1992; Vovk et al. 1993; Antsipov et al. 2000).

In the early period following the accident, inhalation of resuspended radioactive particles of soil and nuclear fuel could significantly contribute to internal dose. To suppress dust formation dispersion of organic solutions over contaminated plots was used in order to create an invisible polymer film after drying. This method was implemented on the Chernobyl NPP and in the 30-km zone during Spring and Summer 1986. Streets in cities were watered to prevent dust formation and to remove radionuclides to the sewerage system. The effectiveness of early decontamination efforts in 1986 still remains to be quantified. However, according to Los and Likhtarev (1993) daily washing of streets in Kyiv decreased collective external dose to its 3 million inhabitants by 3000 man-Sv, and decontamination of schools and school areas saved 600 man-Sv more.

Depending on decontamination technologies the dose rate over different measured plots was decreased by a factor of 1.5 to 15. But the high cost of these activities hindered their complete application to the contaminated areas. Due to these limitations, the actual effectiveness of the decrease in annual external dose was 10 to 20% for the average population and ranged from about 30% for children visiting kindergarten and schools to less than 10% for outdoor workers (herders, foresters, etc.). These data were confirmed by individual external dose measurements conducted before and after large-scale decontamination campaigns in 1989 in the Bryansk Oblast, Russia (Balonov et al. 1992).

Regular monitoring of decontaminated plots in settlements during five years showed that after 1986 there was no significant recontamination and the exposure rate was decreasing over the long term according to the regularities described in Section 4.1 of this report. The averted collective external dose to 90,000 inhabitants of the 93 more contaminated settlements of the Bryansk Oblast was estimated to be about 1000 man-Sv (Balonov et al. 1992).

Since 1990, large-scale decontamination in the countries of the former Soviet Union was canceled, but particular contaminated plots and buildings with measured high contamination levels were specifically cleaned. Some decontamination activity still continues in Belarus aimed at mostly social buildings and areas: hospitals, schools, recreation areas etc. However, in some contaminated Belarusian villages clean up of dwellings and farms have also been performed (Antsipov et al. 2000).

Another area of continuing decontamination activity is clean up of industrial equipment and premises contaminated as a result of ventilation systems being operated during the release/deposition period in 1986 and immediately afterwards. Some 20 to 30 industrial buildings and ventilation systems have been decontaminated annually in Belarus (Antsipov et al. 2000).

3.2.3. Recommended decontamination technologies

In accordance with present radiation-protection methodology, a decision on intervention (decontamination) and selection of optimal decontamination technologies should be made with consideration for costs of all actions and social factors. Calculated cost relates to various decontamination technologies for which an assessment of the averted dose has been made. Benefit (averted collective effective dose) and detriment (expenses, collective dose of decontamination workers) are to be compared for each decontamination technology by means of cost-benefit analysis (ICRP 1977) or multi-attributive analysis (ICRP 1989), which may include qualitative social factors.

The priorities that different procedures would be given in a decontamination strategy should be environment-specific. Nevertheless, based on accumulated experience and research, the following generic set of the major simple decontamination procedures could be recommended for the long term:

(1) Removal of the upper 5-10 cm layer (depending on the activity depth distribution) of soil in courtyards in front of residential buildings, around public buildings, schools and kindergartens, and from roadsides inside a settlement. The removed most contaminated layer of soil should be placed into holes specially dug on the territory of a private homestead or on the territory of a settlement. And the clean soil from the holes should be used to cover decontaminated areas. Such a technology excludes the formation of special burials of radioactive waste.

(2) Private fruit gardens should be treated by deep ploughing or removal of the upper 5-10 cm layer of soil. At the present time vegetable gardens have been ploughed up many times, and in this case the activity distribution in soil will be uniform in a layer 20-30 cm deep.

(3) Covering the decontaminated parts of courtyards, etc. with a layer of clean sand, or, where possible, with a layer of gravel to attenuate residual radiation (see Item 1).

(4) Cleaning roofs or their replacement.

These procedures can be applied both for decontaminating single private homestead lands and houses, and also for decontaminating settlements as a whole. It is evident that in the latter case the influence of the decontamination upon further reduction in external radiation dose will be greater. Achievable decontamination factors for various urban surfaces are presented in Table 3.4. Detailed data on the efficiency, technology, necessary equipment, cost and time expenses, quantity of radioactive waste, and other parameters of separate decontamination procedures are contained in the report (Roed et al. 1995).

TABLE 3.4. ACHIEVABLE DECONTAMINATION FACTORS (DRRF, DIMENSIONLESS) FOR VARIOUS URBAN SURFACES (ROED ET AL. 1995)

Surface	Technique	DRRF, dimensionless
Windows	Washing	10
Walls	Sandblasting	10–100
Roofs	Hosing and/or sandblasting	1–100
Gardens	Digging	6
Gardens	Removal of surface	4–10
Trees and Shrubs	Cut back or remove	~10
Streets	Sweeping and vacuum cleaning	1–50
Streets (asphalt)	Lining	>100

Radioactive wastes generated from urban decontamination should be disposed of in accordance with established sanitary requirements. In case of large-scale decontamination, a temporary storage should be arranged on special isolated grounds from which future activity release into the environment is negligible. The site should be marked by the international symbol of radiation hazard.

3.3.Agricultural countermeasures

The implementation of agricultural countermeasures after the Chernobyl accident has been extensive, both in the most severely affected countries of the former Soviet Union and in western Europe. The main aim of agricultural countermeasures was the production of food products with radionuclide-activity concentrations below action levels[3]. The application of countermeasures in intensive agricultural production systems was largely confined to Belarus, Russia and Ukraine, although some food bans were initially applied in Western Europe. Many countermeasures were used extensively in the first few years after the accident, and their application continues today. In addition, in these three countries, countermeasures have been applied to extensive, private food production using unimproved meadows by rural families, where high ^{137}Cs activity concentrations have persisted for many years (IAC 1991; IAEA 1996a; IAEA 2001).

The high and persistent transfer of ^{137}Cs has also occurred in many contaminated, extensive areas of Western Europe. In these countries, countermeasures have largely been focused on animal-food products from extensive systems, including unimproved pastures used for free ranging grazing animals.

3.3.1. Early phase

From 2-5 May 1986, about 50,000 cattle, 13,000 pigs, 3300 sheep and 700 horses were evacuated from the 30-km zone together with the people (Nadtochiy et al. 2003). In the 30-km zone, more than 20,000 remaining agricultural and domestic animals, including cats and dogs, were killed and buried. Due to a lack of forage for the evacuated animals and difficulties in managing large number of animals in the territories to which they were moved, many were subsequently slaughtered (Alexakhin et al. 1991, Prister et al. 1993). In the acute period after the accident, it was not possible to differentiate the different levels of contamination in animals and in the period of May-July 1986, the total number of slaughtered animals reached 95,500 cattle and 23,000 pigs.

[3] Referred to in the countries of the former Soviet Union as Temporary Permissible Levels (TPL).

Many corpses were buried and some corpses were stored in refrigerators, but this had great hygienic, practical and economical difficulties. Condemnation of meat was an immediately available and effective countermeasure to reduce ingested dose from animal products and was widely used both in the USSR and elsewhere. However, this was very expensive and resulted in large quantities of contaminated waste.

In the first weeks after the accident the main aim of countermeasure application in the USSR was to lower ^{131}I-activity concentrations in milk, or to prevent contaminated milk entering into the foodchain. Recommendations on how to achieve this were by (USSRSAC 1986):

— exclusion from animals' diet of contaminated pasture grasses by changing from pasture to indoor feeding of uncontaminated feed;

— radiation monitoring and subsequent rejection of milk at processing plants in which ^{131}I-activity concentrations were above action levels (3700 Bq L^{-1} at that time); and

— processing rejected milk (mainly converting milk to storable products such as condensed or dried milk, cheese or butter).

In the first few days after the accident, the countermeasures were largely directed toward collective milk and few private farmers were involved. Information on countermeasures for milk was confined to managers and local authorities and was not distributed to the private farming system of the rural population. This resulted in limited application of the countermeasures with some delay, especially in rural settlements for privately produced milk, resulting in a low effectiveness in some areas.

Within a few weeks of the accident, feeding of animals with "clean" fodder began, because this had the potential to reduce ^{137}Cs in cattle to acceptable levels within 1-2 months. However, this countermeasure was not in widespread use at this stage, partly due to a lack of uncontaminated feed early in the growing season.

As early as the beginning of June 1986, maps were constructed of the density of radioactive deposition in the contaminated regions. This allowed estimates to be made of the extent of contamination of pasture and identification of where contaminated milk would occur.

During the growing period of 1986, when there was still substantial surface contamination of plants, the major countermeasures in agriculture were of a restrictive nature. In the first few months, severely contaminated land was taken out of use, and recommendations were developed on suitable countermeasures which would allow continued production on less heavily contaminated land. In the more heavily affected regions a ban was imposed on keeping dairy cattle. To reduce contamination levels in crops, an effective method was to delay harvesting of forage and food crops. Radiation control of products was introduced at each stage of food production, storage and processing (IAC 1991, Alexakhin 1993).

Based on a radiological survey performed from May to July 1986, approximately 130,000, 17,300 and 57,000 ha of agricultural land were initially excluded from economic use in Belarus, Russia and Ukraine (Baryakhtar 1997).

From June 1986, other countermeasures aimed at reducing ^{137}Cs uptake into farm products were implemented as follows:

— banning cattle slaughter in regions where ^{137}Cs-contamination levels exceeded 555 kBq m^{-2}, animals had to be fed clean food for 1.5 months before slaughter;

— minimizing external exposure and formation of contaminated dust by omitting some procedures normally used in crop production;
— limiting the use of contaminated manure for fertilization;
— preparation of silage from maize instead of hay;
— restriction on the consumption of milk produced in the private sector;
— obligatory radiological monitoring of agricultural products; and
— obligatory milk processing.

Decontamination by removal of the top soil layer was not found to be appropriate for agricultural lands because of its high cost, destruction of soil fertility and severe ecological problems related to burial of the contaminated soil.

As early as August-September 1986, each collective farm received maps of contamination levels of their agricultural land and guidance on potential contamination of products, including instructions on farming of private plots (IAC 1991, Alexakhin 1993).

In Western Europe, advice was initially given on avoiding consumption of drinking water from local supplies in some countries.

Sweden received some of the higher levels of deposition outside of the countries of the former Soviet Union. Initially, Sweden imposed action levels on ^{131}I and ^{137}Cs activities in imported and domestic foods, see Section 3.1. A range of other responses were applied: (1) cattle were not put onto pasture, if the ground deposition exceeded 10 kBq m^{-2} of ^{131}I and 3 kBq m^{-2} of radiocaesium, (2) advice was given not to consume fresh leafy vegetables and to wash other fresh vegetables, (3) restrictions were placed on the use of sewage sludge as fertilizer for soil (4) deep ploughing was recommended and (5) a higher cutting level for harvesting grass was advised.

In Norway, crops in fields were monitored after harvesting and those with radiocaesium above 600 Bq kg^{-1} fresh weight were discarded and ploughed in. Also hay and silage harvested in June was monitored, and that with activity concentrations exceeding guidelines was not used as forage.

In Germany, some milk in Bavaria was diverted into food-processing plants to be converted into milk powder. It was intended to use the milk powder as feed for pigs, but this was not done due to the high radiocaesium content.

In the UK, advice was issued to regulate the consumption of red grouse, and restrictions were imposed on the movement and slaughter of upland sheep from a number of the more contaminated areas of the UK.

In Austria there was advice not to feed fresh grass to cows for a short period in May 1986 (Mück 2002).

3.3.2. Later phase

Radiological surveys of agricultural products showed that by the end of 1986 four oblasts of Russia (Bryansk, Tula, Kaluga, and Orel), five oblasts of Ukraine (Kyiv, Zhytomir, Rovno, Volyn and Chernigov) and three oblasts of Belarus (Gomel, Mogilev and Brest) had food products which exceeded action levels for radiocaesium. In the more contaminated raions of Gomel, Mogilev , Bryansk,, Kyiv and Zhitomir Oblasts in the first year after the accident the proportion of grain and milk exceeding the action levels was about 80 % (IAC 1991; IAEA 2001; Nadtochiy et al. 2003).

Additionally, in the early 1990s in Ukraine 101,285 ha of agricultural lands (about 30% of this area had a ^{137}Cs-contamination level above 555 kBq m^{-2}) were also withdrawn from agricultural use. Private cattle were moved with the people from these settlements. Provision of "clean" foodstuffs produced in the collective sector or imported from "clean" regions was organized for residents of these settlements.

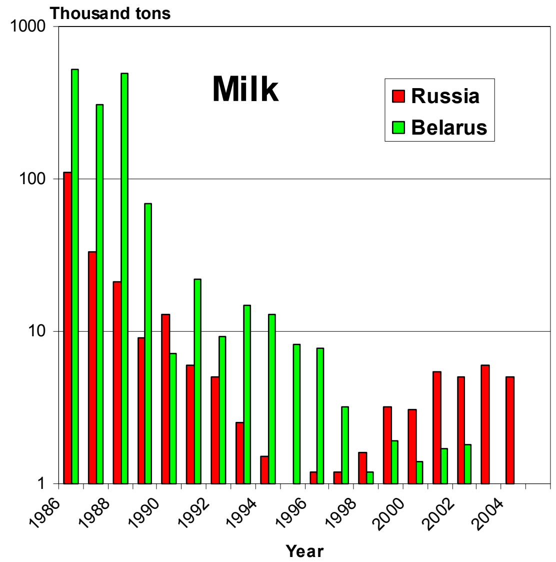
In Russia, in 1987-1988, further evacuations of agricultural animals were carried out, but on a more elective method than in Ukraine. All sheep in the areas contaminated at over 555 kBq m^{-2} were removed, because of the high transfer of radiocaesium to these ruminants. For cattle, in the regions above 555 kBq m^{-2}, 6880 animals were removed but many families retained their animals.

In Belarus in 1989, 52 settlements were relocated after decontamination and countermeasure use was found to be inadequate to lower doses to an acceptable level. Additionally, in 1991 under two new laws some people were allowed to resettle away from contaminated areas and other settlements were moved. In total, 470 settlements were moved. For all these resettlements, the agricultural animals accompanied their owners to the new locations when possible.

Application of countermeasures in contaminated areas had two major radiation-protection aims. The first was to guarantee foodstuff production corresponding to action levels and providing an annual effective dose to local inhabitants lower than 1 mSv. The second was to minimise the total flux of radionuclides in agricultural production. Generally, the earlier agricultural countermeasures were applied, the more cost-effective they were (Prister et al. 2000).

From 1987, high radiocaesium-activity concentrations in agricultural products were only observed in animal products; application of countermeasures aimed at lowering ^{137}Cs activity concentrations in milk and meat was the key focus of the remediation strategy for intensive agriculture. Potatoes and root vegetables were being produced in which the radiocaesium levels were acceptably low. In the second year, radiocaesium-activity concentration in grain was much lower than in the first year, and, therefore, countermeasure application ensured that most grain was below the action levels. By 1991, less than 0.1 % of grain had radiocaesium contents above 370 Bq kg^{-1} in all three countries.

The most difficult issue remaining was the production of milk in compliance with the standards. However, large scale application of a range of countermeasures (described below) made it possible to achieve a sharp decrease in the amount of animal products with radiocaesium-activity concentrations above action levels in all three countries. The changes with time in milk exceeding action levels can be seen in Fig. 3.2, but it must be realized that the values of action levels have been reduced with time in each of the three countries, so the data are not directly comparable. Changes in the action levels in each country are shown in Fig. 3.3.



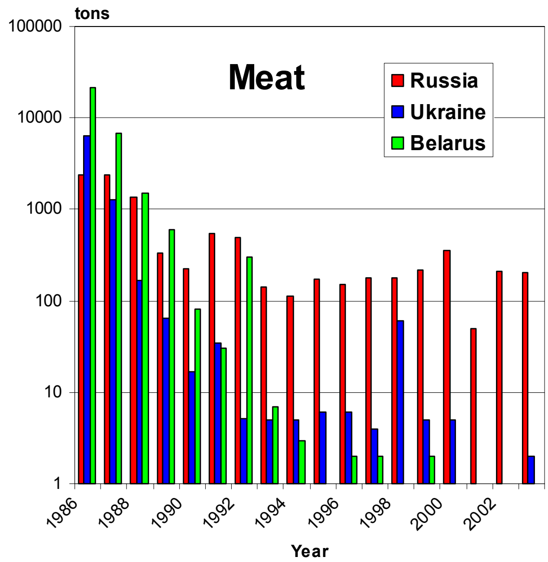
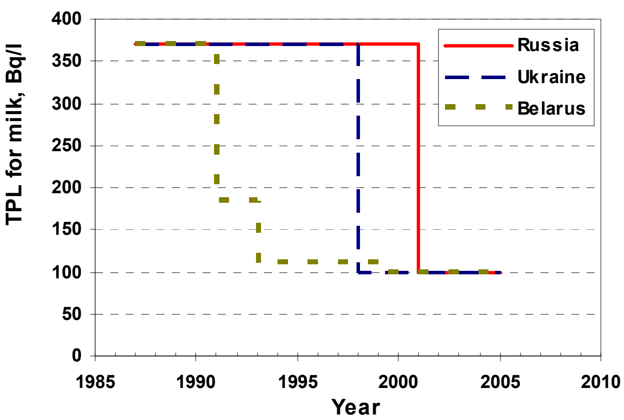


FIG. 3.2. Amounts of milk and meat exceeding action levels in Russia (all milk and meat - collective and private), Ukraine and Belarus (only milk and meat entering processing plants) after the Chernobyl accident (Nadtochiy et al. 2003).



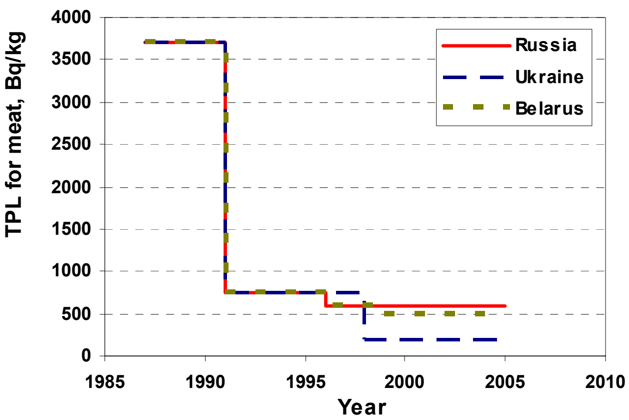


FIG. 3.3. Changes with time in action levels (TPL) in the USSR and later in the independent countries (Shevchuk et al. 2001).

Differences in the time trend in Fig. 3.2 among the countries largely relate to changes in the action level, but also to the scale of countermeasure application. This is particularly clear for Russian milk, where radiocaesium-activity concentrations rose after 1997 due to a reduction in countermeasure use. The recent reduction in meat above action levels in Ukraine and Belarus is because animals are monitored before slaughter to ensure the meat is below the required level. In Russia, where animals are also monitored before slaughter, the concentration data are higher, because they refer to both private and collective meat. The small tonnage of meat in each country now above the action levels is largely due to slaughter of animals which have been injured.

The maximum effect from countermeasure application was achieved in 1986-1992. Thereafter, because of financial constraints in the mid 1990s, the use of agricultural countermeasures was drastically reduced, and their application rates were inadequate not only for a countermeasure but also for conventional food production. However, by optimizing available resources ^{137}Cs-countermeasure effectiveness remained at a level sufficient to maintain an acceptable ^{137}Cs content in most animal products (Fig. 3.2).

3.3.3. Countermeasures in intensive agricultural production

The main countermeasures used in the USSR and later in the independent three countries are briefly described below. The priority was focused on chemical amendments to improve soil fertility and to reduce the uptake of radiocaesium by crops and plants used for fodder. The extent to which each measure was used varied among the three countries. The recommendations on which countermeasures to use were repeatedly revised and updated (Alexakhin 1991; Prister 1998; Bogdevitch 2003).

3.3.3.1. Soil treatment

Soil treatment reduces uptake of radiocaesium (and radiostrontium). The procedure can involve ploughing, reseeding and/or the application of nitrogen, phosphorus, potassium (NPK) fertilisers and lime. Ploughing diluted the radioactive contamination, which was originally in the upper soil layers where most plant roots absorb their nutrients. Both deep and shallow ploughing were used extensively and skim and burial ploughing were also used. The use of fertilisers increases plant production, thereby diluting the radioactivity in the plant. In addition, the use of fertilisers reduces plant-root uptake by decreasing the Cs:K ratio in the soil solution (Alexakhin 1993).

When soil treatment includes all the above measures it is commonly called radical improvement, and this has been found to be the most efficient, practical countermeasure for meadows contaminated by Chernobyl fallout. In the first few years after the accident, the focus was on radical improvement including greatly increased fertilisation rates. Commonly, high value legume and cereal grasses were grown on the treated land. The nature of action and efficiency of radical improvement of hay-land and pastures strongly depends on the types of meadow and soil properties. Traditional surface improvement, involving soil discing, fertilization and surface liming was less effective. Acid soil was limed. Some marshy plots were drained, deep ploughed, improved and used as grassland. In the nineties, there was a greater focus on site-specific characteristics to ensure that the soil treatment used was the most appropriate and effective for the prevailing conditions. With time, repeated fertilisation of already treated soils was necessary, but the appropriate application rates were carefully assessed. However, actual rates of application were sometimes constrained by availability of funds (Alexakhin 1993; Vidal et al. 2001).

Areas which received additional fertilisers in each of the three more affected countries are indicated in Fig. 3.4; areas receiving radical improvement are indicated in Fig. 3.5. The average amount of additional K fertilizers added was about 60 kg ha^{-1} of K_2O annually between 1986 and 1994. In the mid-nineties, the productivity of arable land fell, because a worsening economic condition prevented the implementation of countermeasures at the previous rates; this resulted in an increasing proportion of contaminated products. In some areas of Russia, this halted the previous decrease in the amounts of milk and meat exceeding radiation-safety standards (see Fig. 3.2). For example, in the more contaminated raions such as Novozybkov Raion (Bryansk Oblast) because of insufficient use of K fertilizers, ^{137}Cs-activity concentrations in agricultural products in 1995-1996 increased by more than 50 % compared to the period of optimal countermeasures application (1991-1992).

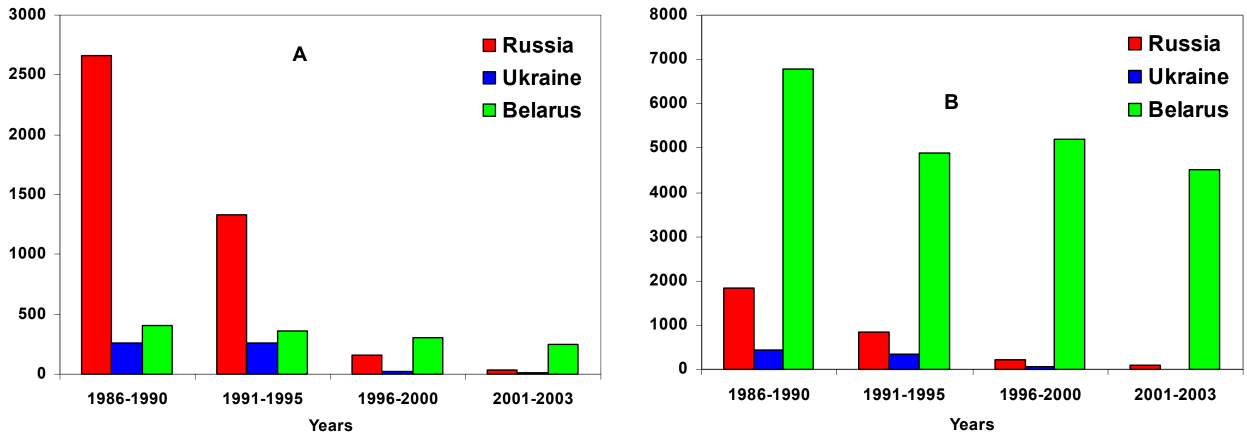


FIG. 3.4. Changes in the extent of agricultural areas treated with liming (A) and mineral fertilizers (B) in the USSR and later the three independent countries affected by the Chernobyl accident, thousand ha. (Shevchuk et al. 2001).

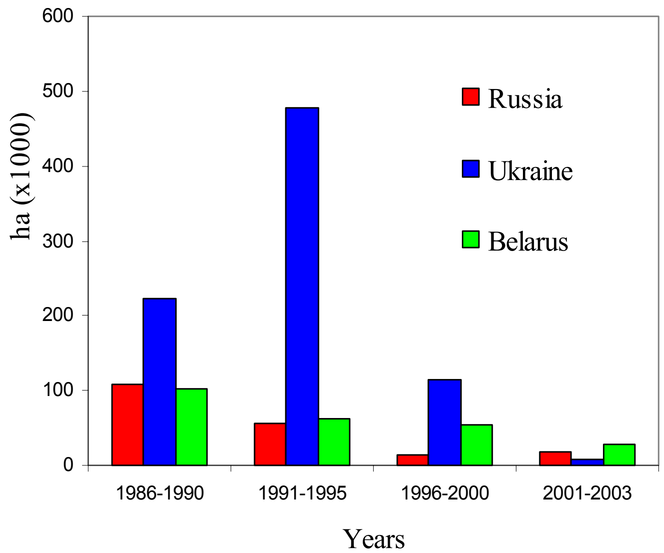


FIG. 3.5 Areas of radical improvement in the USSR and later the three independent countries affected by the Chernobyl accident, (Shevchuk et al. 2001).

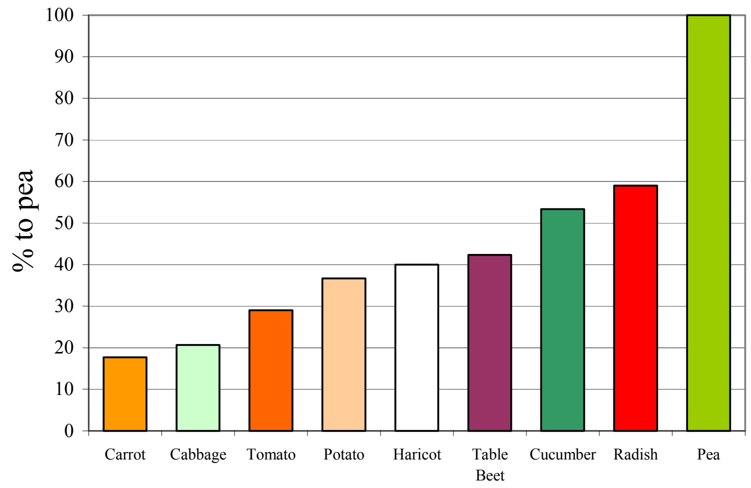


FIG. 3.6. Comparison of ^{137}Cs uptake in different crops, normalised to that of peas. (Bogdevich et al. 2003).

The effectiveness of soil treatment is influenced by soil type, nutrient status and pH, and also the plant species selected for reseeding. In addition, clearly the application rates of NPK fertilisers and lime affected the reduction achieved. Several studies have shown that the reduction factors achieved for soil-plant transfer of radiocaesium following radical improvement, liming and fertilisation were in the range of 2-4 for poor, sandy soils and 3-6 for more organic soils. An added benefit was the reduction in external dose rate by a factor of 2-3 due to the dilution of the surface contamination layer after ploughing.

Even though problems of ^{90}Sr are less acute than those of ^{137}Cs, some countermeasures have been developed and a reduction of 2-4 in soil-plant transfer of radiostrontium following discing, ploughing and reseeding has been achieved.

Despite these countermeasures, in the more highly contaminated raions of the Bryansk Oblast radiocaesium contamination of 20 % of pasture and hay on farms in the south-western zone in 1997-2000 still exceeded the action levels. Concentrations of ^{137}Cs in hay varied between 650 and 66,000 Bq kg^{-1} dry weight.

3.3.3.2. Change in fodder crops grown on contaminated land

Some plant species take up less radiocaesium than others as can be seen from experimental data collated in Belarus from 1997-2002 (Fig. 3.6). The extent of the difference is considerable and fodder crops, such as lupine, peas, buckwheat and clover, which accumulate high amounts of radiocaesium, were completely or partly excluded from cultivation.

In Belarus, rape seed is grown on contaminated areas with the aim of producing two products: edible oil and protein cake as an animal fodder. Varieties of rape seed are grown which are known to have a 2-3 fold lower ^{137}Cs and ^{90}Sr uptake rate than many varieties. When the rape seed is grown, additional fertilisers (liming 6 t ha^{-1} and fertilization with $N_{90}P_{90}K_{180}$) are used to reduce radiocaesium and radiostrontium uptake into the plant by a factor of about two. This reduces contamination of the seed which is used for the protein cake. During processing of the rape seed, both radiocaesium and radiostrontium are effectively removed, and negligible amounts remain. The production of rapeseed oil in this way has proved to be an effective, economically viable way to use contaminated land and is profitable for both the farmer and the processing industry. During the last decade the area under rape-seed cultivation has increased four fold to 22,000 ha. (Bogdevich et al. 2002).

3.3.3.3. Clean feeding

The provision of uncontaminated feed or pasture to previously contaminated animals for an appropriate period before slaughter (so called "clean feeding") effectively reduces radionuclide contamination in meat and milk at a rate depending on the animal's biological half-life for each radionuclide. Radiocaesium-activity concentration in milk responds rapidly to changes in the diet, as the biological half life is a few days. For meat the response time is longer due to the longer biological half time in muscle (Prister et al. 1993).

Clean feeding reduces uptake of the contaminating radionuclides; it has been one of the most important and frequently used countermeasures for meat from agricultural animals in both the countries of the former Soviet Union and in Western Europe after the Chernobyl accident. Official estimates of the number of cattle treated were between 5000 and 20,000 annually in the Russian Federation and 20,000 in Ukraine (supported by the government up to 1996) (IAC 2001). Clean feeding is routinely used in all three countries for meat production and is combined with live monitoring of animals so that if animal's muscles are above action levels they can be returned to the farm for further clean feeding.

3.3.3.4. Administration of Cs-binders

Hexacyanoferrate compounds (commonly referred to as "Prussian Blue") are highly effective radiocaesium binders, which may be added to the diet of dairy cows, sheep and goats, as well as to meat-producing animals, to reduce radiocaesium transfer to milk and meat by reducing absorption in the gut. They have a low toxicity and are therefore safe to use. Many different formulations of hexacyanoferrates have been developed in different countries, partially to identify the most effective compound and partly to produce a cheaper, locally available product. Hexacyanoferrate compounds can achieve reduction factors in animal products of up to 10 (IAEA 1997).

Prussian blue has been added to the diet of animals as a powder, incorporated into pelleted feed during manufacturing, or mixed with sawdust. In Russia, a locally manufactured hexacyanoferrate called ferrocyn (a mixture of 5% $KFe[Fe(CN)_6]$ and 95% $Fe_4[Fe(CN)_6]$) was developed. It has been administered as 98% pure powder, salt licks (10% ferrocyn) and in sawdust with 10% adsorbed ferrocyn (called bifege) (Ratnikov et al. 1998).

The number of cattle treated annually with Prussian blue in each of the three countries is shown in Fig. 3.7. In addition, slow release boli containing hexacyanoferrate have been developed which are introduced into the animals rumen and gradually release the Cs-binder over a few months. The boli, originally developed in Norway, consist of a compressed mixture of 15% hexacyanoferrate, 10% beeswax and 75% barite (Hove and Hansen 1993).

Prussian blue has been used to reduce ^{137}Cs contamination of animal products since the beginning of the 1990s. Prussian blue application has been especially useful and effective in settlements where there is a lack of meadows suitable for radical improvement. In initial trials, Prussian blue reduced ^{137}Cs transfer from fodder to milk and meat by a factor of 1.5-6.0 (Hove et al. 1995). In Belarus, a special concentrate with Prussian blue is produced and distributed at a rate of 0.5 kg of concentrate per cow daily and an average value for reduction factor of 3 for milk has been achieved. Boli are given to dairy cows in intensive systems in both Belarus and Russia.

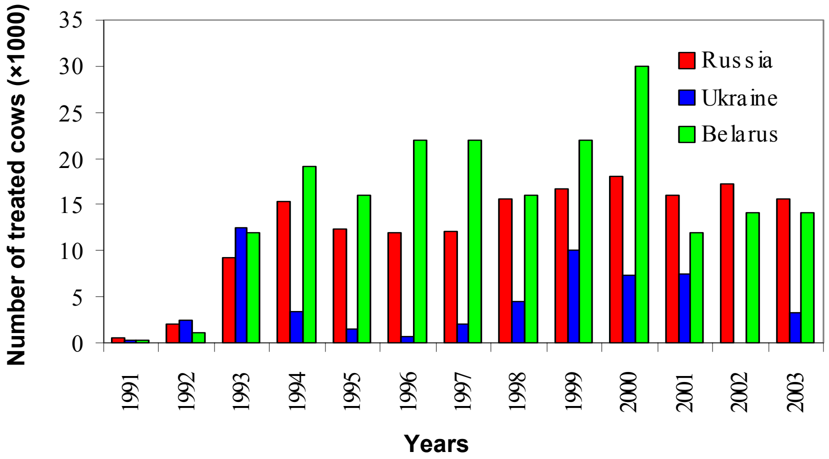


FIG. 3.7. Changes with time in the use of Prussian blue in the three countries of the former Soviet Union.

Prussian blue has not been used as extensively in Ukraine as in Russia and Belarus; and its use was confined to the early nineties. This is because in Ukraine no local source of Prussian blue is available and the cost of purchasing it from Western Europe was considered too high. Therefore, instead they have used locally available clay-mineral binders, on a small scale, which are somewhat less effective than Prussian blue but cheaper.

3.3.4. Summary of countermeasure effectiveness in intensive production

The effectiveness of different agricultural countermeasures in actual use on farms has been summarised in Table 3.5. The reduction factors (ratio of activity concentration in the product before and after countermeasures application) achieved by each measure are given.

TABLE 3.5. SUMMARY OF THE REDUCTION FACTORS OF DIFFERENT COUNTERMEASURES USED IN THE THREE COUNTRIES OF THE FORMER SOVIET UNION (ALEXAKHIN 1993; SHEVCHUK ET AL. 2001; DEVILLE-CAVELIN ET AL. 2001; BOGDEVITCH ET AL. 2002)

Countermeasure	^{137}Cs	^{90}Sr
Normal ploughing (first year)	2.5–4.0	
Skim and burial ploughing	8–16	
Liming	1.5–3.0	1.5–2.6
Application of mineral fertilisers	1.5–3.0	0.8–2.0
Application of organic fertilisers	1.5–2.0	1.2–1.5
Radical improvement:		
– First application	1.5–9.0*	1.5–3.5
– Further applications	2.0–3.0	1.5–2.0
Surface improvement:		
– First application	2.0–3.0*	2.0–2.5
– Further applications	1.5–2.0	1.5–2.0
Change in fodder crops	3–9	
Clean feeding	2–5 (time dependent)	2–5
Administration of Cs binders	2–5	–
Processing milk to butter	4–6	5–10
Processing rapeseed to oil	250	600

* For wet peat up to 15 with drainage.

3.3.5. Countermeasures in extensive production

Extensive production in the three countries of the former Soviet Union is largely confined to the grazing of privately owned cows on poor, unimproved meadows. Because of the poor productivity of these areas, radiocaesium uptake is relatively high compared to land used by collective farms. Radical improvement of meadows used by private cows has been applied in all three countries since the early 1990s. Clean feeding is not generally used by private farmers, although on occasions collective farms may supply private farmers with uncontaminated feed or pastures. Prussian Blue is used by private farmers in both Russia and Belarus. In Russia, all three Prussian Blue delivery systems are used, according to availability and preference (Jacob et al. 2001).

In extensive systems, such as upland grazed areas in Western Europe, the most commonly used countermeasures for free-ranging animals have been clean feeding, administration of caesium binders, monitoring of live animals, management restrictions, and change in slaughter times. Many of these countermeasures are still in use in 2004. The application of long-term countermeasures has been most extensive in Norway and Sweden, but has also been applied in the UK and Ireland.

AFCF is a highly effective hexacyanoferrate compound achieving up to a 5-fold reduction in lamb and reindeer meat and up to a 3-fold reduction in cow's milk and 5-fold reduction in goat's milk. The use of AFCF has been temporarily authorised in the EU and some other countries. AFCF as a caesium binder is effective in extensive production systems, in contrast to many other countermeasures where the applicability is limited. Boli are particularly favourable for infrequently handled free-grazing animals, as the boli can be administered when animals are gathered for routine handling operations. For use in extensive systems, the boli can be given a protective surface coating of wax to delay the onset of AFCF release, so that effectiveness is increased at the time when animals are collected for slaughter (Hansen et al. 1996). Brynildsen et al (1996) estimated that the use of boli as a countermeasure for sheep was 2.5 times as cost effective as feeding with uncontaminated feed. Salt licks containing AFCF have also been used, but are less effective (Hove 1993).

Management regimes have been modified for some animals in contaminated areas. For instance, slaughter times are modified to ensure that the ^{137}Cs-activity concentrations are relatively low. In the UK, the movement and slaughter of upland sheep has been restricted in some upland areas after the accident, and the animals are monitored to ensure that their ^{137}Cs-activity concentration is below the action levels before they are slaughtered.

Live monitoring of animal-derived products (monitoring of live animals and/or of milk and tissues after slaughter) has been used to ensure that countermeasures have been effective. The use of monitoring is also important in maintaining public confidence in the products from affected areas.

An example of the long-term consequences of the accident can be seen in Fig. 3.8 which shows the number of reindeer in Sweden, which had radiocaesium-activity concentrations above action levels, and the number of slaughtered animals. The high number of slaughtered animals in the first year was in part due to the low action level of 300 Bq kg^{-1} fresh weight which was subsequently increased to 1500 Bq kg^{-1} from 1987. The decline has been achieved partly by extensive use of countermeasures including clean feeding and change of slaughter time.

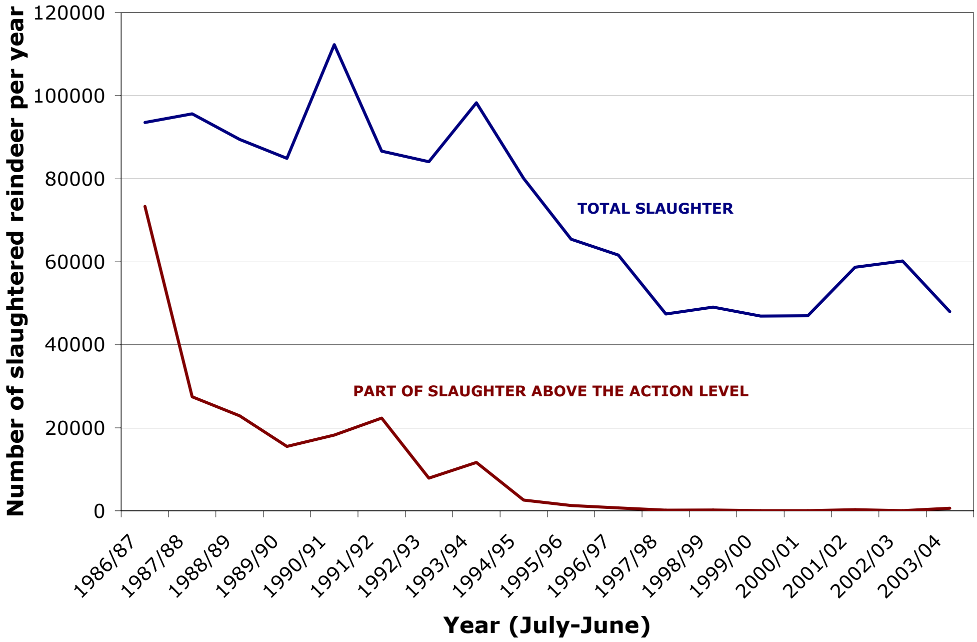


FIG. 3.8 Change with time in the number of reindeer in Sweden which had radiocaesium activity concentrations above action levels and the number of slaughtered animals (The Swedish Board of Agriculture, Jönköping, Sweden).

3.3.6. Current status of agricultural countermeasures

Currently, in all three countries of the former Soviet Union clean feeding remains an important countermeasure to ensure that meat from intensive farms can be marketed.

In Belarus, fertilisation with P-K is used on collective farms, and milk from these farms above action levels is processed into butter. Radical improvement is used on private farms together with Prussian blue for milk. Rape seed production is currently limited by processing capacity, although this may be increased in the future.

In Ukraine, the only remaining countermeasure used on intensive systems is clean feeding of meat-producing animals prior to slaughter. Any milk above action levels is used within the settlements, partially to feed pigs. All other countermeasures are directed at private farmers using extensive areas. These currently comprise radical improvement of meadows and the use of clay mineral Cs-binders for private milk.

In Russia, fertilizers (largely K) are supplied to intensive farms. For private farms, Prussian blue is provided for private milk and on request for private meat intended for market.

In all contaminated settlements, it is possible to get local produce monitored, although the capacity and availability of the service varies.

In Western Europe, countermeasures for animals in extensive systems are still used in Norway and Sweden, and the movement and slaughter of upland sheep is still restricted in certain upland areas of Great Britain.

3.3.7. A wider perspective of remediation including socio-economic issues

Experience after the Chernobyl accident has shown that restoration strategies need to consider a wide range of different issues to ensure the long-term sustainability of large and varied contaminated areas (Oughton et al. 2004). The selection of robust and practicable restoration strategies should take into account not only radiological criteria but also (1) practicability, including effectiveness, technical feasibility and acceptability of the countermeasure; (2) cost benefit; (3) ethical and environmental considerations; (4) requirements for effective public communication; (5) spatial variation in many of these factors, and (6) the contrasting needs of people in urban, rural and industrial environments (www.strategy-ec.org.uk/). When not only radiological factors but also social and economic ones are taken into account, better acceptability of countermeasures by the public can be achieved.

A number of EC and UN projects have applied some of the above considerations in trying to provide appropriate information and interaction with people in contaminated territories and to involve them in making decisions about responding to enhanced doses and what can be done to live sustainably in contaminated areas. In particular, this introduces the possibility of self-help and the opportunity for people to decide for themselves whether they wish to modify their behaviour to reduce their doses. The EC project ETHOS (Heriard-Dubreuil et al. 1999; Lochard 2004) identified dissemination of a practical radiological culture within all segments of the population as a prerequisite, especially within professionals in charge of public health. The EU Tacis project, ENVREG 9602, in Belarus and Ukraine sought to minimize environmental and secondary medical effects resulting from the Chernobyl disaster by improving the public perception and awareness of these effects. Deliverables included hyperlinked training materials on CD-ROM and services to the local populations, notably training-the-trainers.

Most recently, the EC project CORE was initiated to address long-term rehabilitation and sustainable development of Bragin, Chechersk, Slavgorod and Stolin Raions in Belarus. Core community projects include health care, radiological safety, information and education. In addition, critical socio-economic constraints are being addressed, specifically using a micro-crediting system for small businesses and farmers, cost-effective production of “clean” products, creation of Rural Entrepreneurs Centre, and the promotion of community economic initiatives.

Chernobyl debate is increasingly about socio-economic issues and communicating technical information in an understandable way. ETHOS, ENVREG and CORE all have a strong community focus and target Chernobyl-affected communities and other local stakeholders. In principle, community feedback, income generation, health and well-being should indicate which approaches proved/are proving successful and by how much. In general, the holistic philosophy of these projects considering both environmental and social problems is in line with the recent UN initiative known as “Strategy for Recovery” (UNDP-UNICEF, 2002).

3.3.8. Current status and future of abandoned land

The extent of recovery of abandoned land is discussed for each of the three countries of the former Soviet Union below. Currently in 2004, 16,100, 11,000 and 6,095 ha of previously abandoned land in Belarus, Russia and Ukraine, respectively, have been returned to economic use (Nadtochiy et al. 2003). In general, there is currently little effort being devoted to any further rehabilitation of abandoned areas.

3.3.8.1. The exclusion and resettlement zones of Belarus

The 30-km exclusion zone covers a total of 215,000 ha in Belarus. The people who used to reside here were evacuated in 1986. Since May 1986, lands in the exclusion zone have been taken out of agricultural and other production. The Polessje State Radiation and Ecological Reservation (PSRER) set up by Government decree in 1988 constitutes mostly the 30-km zone, but also includes some other areas with high transuranium contamination. Access to the PSRER is strictly forbidden and very few, mostly old, people are currently present without permission in the area. Pursuant to the Law 'On legal regime of territories contaminated as a result of the Chernobyl NPP catastrophe,' most of the land in the exclusion zone cannot be brought back into economic production within a millennia, because of contamination with long-lived transuranium radionuclides. In the exclusion zone, only activities related to: ensuring radiation safety; to fight forest fires; preventing the transfer of radioactive substances and environmental protection, as well as scientific research and experimental work, are permitted.

While the exclusion zone is the most contaminated, compact area adjacent to the Chernobyl NPP (Bragin, Khoiniki and Narovlya Raions of the Gomel Oblast), a further resettlement zone was identified in the early nineties from which more people were evacuated; this zone covers a total area of 450,000 ha in 15 raions of 2 oblasts.

A total area of agricultural land of 265,000 ha received deposition of ^{137}Cs over 1480 kBq m^{-2}, and/or ^{90}Sr – over 111 kBq m^{-2}, and/or plutonium isotopes – over 3.7 kBq m^{-2}. All this land was excluded from agricultural use.

The remaining abandoned agricultural land in the resettlement zone could be used for agriculture in future. The present state of ecosystems and economic infrastructure of the resettlement zone is characterized by a general deterioration in the former agricultural lands, drainage systems and roads. Due to lack of drainage, there has also been a gradual elevation in the water table. Normal ecological succession has lead to an increase in the number of perennial weeds and shrubs. Unlike the exclusion zone, in the resettlement zone strictly limited access for certain maintenance activities are permitted. These are activities to maintain roads, electricity transmission lines, etc.

In Belarus, it is considered to be important to bring lands back to agricultural use, if possible. At the request of collective and state farms, if supported by local authorities, a survey of former agricultural lands was conducted to see whether it is possible to rehabilitate the land for agricultural use. This was based on radiological criteria only.

During the post-accident period, a total of 14,600 ha of previously withdrawn land was returned back into circulation by 2001 (Shevchuk et al. 2001); recently this has increased to about 16,000 ha. This land is closely adjacent to populated settlements. In these rehabilitated sites, the soil fertility has been restored and a variety of countermeasures used to minimize radiocaesium and radiostrontium uptake based on official guidelines (Bogdevitch 2003).

Most of the agricultural and other land of the resettlement zone was transferred to the authority of the Ministry of Forestry. This is because much of the resettlement zone is considered suitable for forest production.

According to an assessment of Bogdevitch et al. (1998), a total of about 35,000 ha of more fertile agricultural land may be suitable for further rehabilitation. However, economic support for recovery and use of countermeasures has declined significantly over recent years. Use of countermeasures is now confined to radical improvement of meadows, Prussian blue for cows, liming and fertilization.

The situation with contamination of agricultural lands is periodically updated. Methodologies of rehabilitation of abandoned land are being developed and improved in particular with respect to economic evaluation. The main obstacles for agricultural use of abandoned land are the destroyed infrastructure, the high-expected production cost and low market demand for the agriculture goods. A large-scale rehabilitation of excluded land will only be possible if there is a general improvement in the economic situation of the country.

3.3.8.2. Rehabilitation of contaminated lands of Ukraine

The first priority of rehabilitation is directed to land where people are living. Afterwards, consideration has been given to the potential rehabilitation of abandoned areas. The approach has been that such areas be rehabilitated if this procedure is expedient with respect to economic and social criteria. The main condition for human occupancy of such areas without any restrictions is that the additional annual dose should not exceed 1 mSv.

The efficiency of countermeasures is determined by the following criteria:

— **radiological** – reduction of radionuclide content in the product and associated individual and collective dose;

— **economic** – increased product market value; and

— **social and psychological** – public opinion on a given countermeasure.

Currently, in 2004 by radiological criteria alone a significant part of the abandoned agricultural lands (more than 70 %) could be returned to economic use. When economic and social criteria are assessed, the amount of land which could be rehabilitated declines and is shown in Table 3.6. The table shows a scheme for rehabilitation based on technical criteria over a seven-year period The first phase from 1998-2000 was implemented but that for the second phase was not due to changing economic and social conditions.

In the 30-km zone, the limiting radionuclide is now ^{90}Sr rather than ^{137}Cs. Under radiological criteria, the southwest part of the zone can be used without restrictions. However, in reality, legal restrictions, the lack of a suitable infrastructure, and consideration of economic and social-psychological factors prevent its rehabilitation.

TABLE 3.6. REHABILITATION OF ZONES OF OBLIGATORY RESETTLEMENT (OUTSIDE 30-KM ZONE)

Area	Abandoned lands (ha)	Can be rehabilitated judged by radiological, economic and social criteria (ha)
	Kyiv Oblast	
1998–2000 (done)		3475
2001–2005		4720
Total	29342	8205
	Zhytomyr Oblast	
1998–2000 (done)		2620
2001–2005		4960
Total	71943	7580

The same restrictions apply for the other abandoned areas where legal restrictions are also in place which, together with deteriorating economic conditions, currently prevent the application of countermeasures on the remaining identified abandoned areas. The pressure to bring the abandoned land back into production is also reduced by the current oversupply of agriculturally productive land in Ukraine, and the presence in southern Ukraine of land which is much more productive.

Some people have returned to abandoned areas to live and others live outside it but use the land for agricultural activity such as hay production. Application of countermeasures is not supported in the abandoned areas, but there is sanitary and regulatory control of these activities.

3.3.8.3. Abandoned zones in Russia

Areas in Russia with high levels of radioactive soil contamination were abandoned in stages from 1986-1989 and in total 17,000 ha of agricultural lands were excluded from economic usage. The abandoned areas belonged to 17 rural settlements with about 3000 inhabitants at the time of the accident and 12 collective farms.

In 1987-1989, considerable efforts were made to retain the highly contaminated areas in economic use and so most of the abandoned area were subject to intensive countermeasure application. However, these efforts were only partially successful, and the land was gradually more fully abandoned and then the intensity of countermeasure application declined in the 1990s. Overall, about 11,000 ha have been returned to agricultural use up to 1995 after new radiation surveys and appropriate justification. These decisions were made individually for each contaminated field. Special attention was paid to highly contaminated fields surrounded by fields with relatively low levels of contamination, because there was a natural inclination to use these fields. The assessments were based on Russian radiation safety standards including standards (TPL-93) governing quality of agricultural products (SCRSI 1993).

Between 1995 and 2004, there has been no further rehabilitation of the abandoned areas. Officially, they are abandoned but unofficially some local people are living in these areas and using them for agricultural production, but without the benefit of countermeasures.

Recently, a technical project of gradual rehabilitation of the remaining abandoned areas, in which the mean ^{137}Cs-soil deposition varies from 1540-3500 kBq m^{-2} has been proposed by Russian Institute of Agricultural Radiology and Agroecology. The criteria for agricultural production include ensuring that ^{137}Cs-activity concentrations would be less than the TPL, as well as a requirement that application of countermeasures for each contaminated field would be optimised.

During the first stage up to 2015, production of grain and potatoes was suggested using agricultural workers who lived elsewhere but would come into the contaminated area as necessary. The soil-based countermeasures (liming, K fertilization) should allow the production of plant products with sufficiently low levels of ^{137}Cs on most of the abandoned area. From 2015, the implementation of animal breeding with a gradual extension was planned and from 2025 the re-establishment of populated settlements could commence. Thus, by 2045 all abandoned lands could be used once more, although the application of different countermeasures would be needed up to 2055 to ensure that annual doses to the local inhabitants were less than 1 mSv.

3.4. Forest countermeasures

Countermeasures to be applied to forested areas contaminated with radionuclides are only likely to be implemented if they are something that a forester or landowner can accept on a practical basis – i.e., actions likely to fit in with normal forest-management practices. For countermeasures to be successful they must also be accepted by the general public. As forest countermeasures are labour consuming and expensive, they will not be implemented quickly, but will have to be planned carefully. They are likely to be long term activities and their beneficial effects will take time to be realized.

3.4.1. Studies on forest countermeasures

Prior to the Chernobyl accident countermeasures to offset doses due to large-scale contamination of forests had not been given significant attention. Several international projects in the 1990s gave rise to a number of publications in which suggestions and recommendations were given for possible countermeasures to be applied in forests (Guillitte et al. 1993, 1994; Amiro et al. 1999; Rafferty and Synnott 1998). However, in the three countries of the former Soviet Union actions had already been taken to restrict activities in the more contaminated zones which included significant areas of forestry (Fesenko and Brown 2000). These actions were, in general, rather simple and involved restrictions on basic activities such as accessing forests and gathering wild foods and firewood. A major question remains as to whether any more complex or technologically based countermeasures can be applied in practice, and whether many of the ideas developed by researchers will remain as theoretical possibilities rather than methods which can be applied in real forests at a realistic scale. The following section describes some of the more feasible countermeasures which have been devised for forests contaminated with radiocaesium. This is illustrated in Section 3.4.3 by studies in which countermeasures were actually put into practice.

3.4.2. Countermeasures for forests contaminated with radiocaesium

There are several categories of countermeasures which are, in principle, applicable to forest ecosystems (Tikhomirov and Shcheglov 1994; Panfilov 1999). A selection of these is shown in Table 3.7. These can be broadly categorised into a) management and b) technological countermeasures.

3.4.2.1. Management-based countermeasures

Under the broad heading of management-based countermeasures the principal remedial methods applied after Chernobyl involved restrictions of various activities normally carried out in forests. Restriction of access to contaminated forests and restriction of the use of forest products were the main countermeasures applied in the USSR and later on in the three independent countries (Fesenko and Brown 2000). These restrictions can be categorised as follows:

- Restricted access, including restrictions on public and forest-worker access. This has been assisted by the provision of information from local monitoring programmes and education on issues such as food preparation (Beresford et al. 1999).
- Restricted harvesting of food products by the public. The most commonly obtained food products include game, berries and mushrooms. The relative importance of these varies from country to country. In the three countries of the fromer Soviet Union mushrooms are particularly important and can often be severely contaminated (see Section 2.3).

TABLE 3.7. SELECTED COUNTERMEASURES WHICH HAVE BEEN CONSIDERED FOR APPLICATION IN CONTAMINATED FORESTS (SHAW ET AL. 2001)

Countermeasure	Category	Caveats	Benefit(s)	Cost(s)
Normal Operation:	Management	–	No loss of productivity or amenity	No dose reduction, negative social costs
Minimum Management: Forest fire protection, disease protection, and necessary hunting	Management	–	Creation of nature reserve, and reduced worker dose	(Worker dose), Loss of productivity, negative social costs, costs for hunting
Delay cutting of mature trees	Management / Agrotechnical	Marginal feasibility	Reduce contamination of wood due to: i)radioactive decay ii)fixation of Cs in soil iii)loss from soil and wood	Delay in revenue
Early clear cutting & replanting or self-regeneration	Management / Agrotechnical	Must consider tree age at time of contamination. Possibly in combination with soil mixing	Reduce tree contamination: i)lower soil-tree transfer ii)delayed harvest time iii)alternative tree crop	Higher dose to workers during replanting Operational costs
Soil improvement: harrowing after thinning or clear cutting	Agrotechnical	Cost-effectiveness is dependent on area to be treated. Possibly in combination with fertiliser application	Improves tree growth,therefore growth dilution. Dilutes radionuclide activity concentrations. in soil surface layer Decrease in mushroom, berries, understorey-game	Operational costs Worker doses Environmental or ecological costs (e.g. nitrate & other nutrients lost).
Application of PK fertiliser and/or liming	Agrotechnical	PK: May only be effective for Cs Especially effective for younger stands lime: particularly useful for Sr-90	Reduction of uptake to trees, herbs, etc. Maybe better growth and dilution effect Higher fixation	Cost of fertiliser Worker dose Negative ecological effects
Limit public access	Management	NB. people normally residing in forests not considered	Reduction in dose possible increase in public confidence!	Loss of amenity/social value Loss of food Negative social impacts
Salt licks	Agrotechnical		reduction in caesium uptake by grazing animals	Continuing cost of providing licks
Ban hunting	Management		reduction in dose due to ingestion of game	need to find alternative supply of meat
Ban mushroom collection	Management		reduction in internal dose	Need to find alternative mushroom supply

— Restricted collection of firewood by the public. This not only exposes people to *in situ* gamma radiation while collecting firewood, but can also lead to further exposures in the home and garden when the wood is burned and the ash is disposed of, sometimes being used as a fertiliser.

— Alteration of hunting practices. The consumption of fungi by animals such as roe deer leads to strong seasonal trends in their body burden of radiocaesium (see Section 2.3). Thus, excessive exposures can be avoided by eating the meat only at seasons in which fungi are not available as a food source for the animals.

— Fire prevention is a fundamentally important part of forest management under any circumstances, but it is also important after large scale deposition to avoid secondary contamination of the environment. This could result from burning of trees and especially forest litter which is one of the major repositories of radiocaesium in the forest system (see section 2.3). One of the ways in which forest fires can be avoided is by minimising human presence in the forest, so this countermeasure is strongly linked to restricting access as described above.

3.4.2.2. Technologically based countermeasures

This category of countermeasures includes the use of machinery and/or chemical treatments to alter the distribution or transfer of radiocaesium in the forest. Many mechanical operations are carried out as part of normal forestry practice and examples of these have been described by Hubbard et al. (2002) with reference to their use as countermeasures. Similarly, applications of fertilisers and pesticides may be made at different times in the forest-cropping cycle as part of normal management practice. However, the cost effectiveness of many technological countermeasures is questionable, especially when applied on a large scale (Shaw et al. 2001). Thus, it is to be expected that such countermeasures will be restricted to small-scale cases only, if they are feasible at all. Such cases might include small areas of urban woodland, such as parkland, which is likely to be visited by many more people than extensive and remote forest areas.

Technological countermeasures might include mechanical removal of leaf litter or scraping of soil layers, clear cutting and ploughing, and the application of calcium and potassium containing fertilisers. It is evident, however, that any of these methods can damage the ecological functioning of the forest when applied outside of the normal schedule of forestry operations. This, and the high economic costs of such operations, means that the practical use of such techniques as countermeasures remains largely speculative, and such measures have not been applied after the Chernobyl accident other than in small-scale experiments. Indeed, the results of cost-benefit calculations indicate that the management options likely to result in the least overall detriment are those which limit access and consumption of forest foods. Options, which involve technological intervention, application of chemicals, or altering the harvesting patterns in forests are unlikely to be used in practice.

3.4.3. Examples of forest countermeasures

Case studies in which forest countermeasures, particularly technologically based countermeasures, have actually been applied in practice are rare. This illustrates the difficulty of implementing practical remedial measures in forests in contrast to agriculture in which the application of fertilisers, in particular, has been used with some success (see Section 3.2). In practice, restrictive countermeasures have been applied in the USSR and later on in the three independent countries, as well as in a limited number of other countries, such as Sweden.

In the Bryansk Oblast of Russia, individual restrictions on forestry and for the population living near forests were recommended according to the level of ^{137}Cs deposition. For forests receiving deposition greater than 1480 kBq m^{-2} access was only allowed for forest conservation, fire fighting and control of pests and diseases. All forestry activity was stopped and public access, including collection of forest plants, was prohibited. For forests receiving deposition between 555 – 1480 kBq m^{-2} collection of forest products was also prohibited but limited forestry activities continued. At deposition levels from 185 – 555 kBq m^{-2} harvesting of trees was continued on the basis of radiological surveys which were used to identify individual areas in which external doses to forestry workers and contamination of wood were acceptable. However, the collection of berries and mushrooms by the public was only permitted in forests with less than 74 kBq m^{-2}.

One of the major effects of the restrictions which were enforced on a large scale up to 1990 was a negative impact on rural populations. At the beginning of 1990 the population began gathering mushrooms and berries again over the whole Bryansk Oblast. However, in areas where the original ^{137}Cs deposition was between 555 and 1480 kBq m^{-2} restrictions on gathering forest food products are still in force. This example illustrates a major difficulty in implementing countermeasures involving restrictions on public activities which inevitably lead to a disturbance of normal societal behaviour patterns. Furthermore, wood production is still under the official control of local forest authorities (Fesenko and Brown 2000); currently applicable permissible levels for contamination of wood and forest products in the Russian Federation are shown in Table 3.8. Similar restrictions and permissible levels have been implemented in different oblasts of Belarus, notably the Gomel and Mogilev Oblasts.

The use of caesium binders in domestic animals, particularly Prussian Blue, has been one of the more effective techniques used to reduce doses from contaminated forests in the three countries of the former Soviet Union. The principles underlying this method are described in Section 3.3, and they are equally applicable to the problem of marginal grazing of domestic animals in forests. Typically, reductions in ^{137}Cs-activity concentrations of a factor of 5 in milk and a factor of 3 in meat can be achieved at optimum dosage (Fesenko and Brown 2000).

TABLE 3.8. TEMPORARY PERMISSIBLE LEVELS FOR ^{137}Cs IN WOOD AND FOREST FOOD PRODUCTS IN THE RUSSIAN FEDERATION (MINISTRY OF HEALTH 1997)

Forest product	Temporary Permissible Level (Bq kg^{-1})
Round wood including bark	11,100
Unsawn timber with bark removed	3,100
Sawn wood (planks)	3,100
Construction wood	370
Wood used for pulp and paper production	3,100
Wood products for household and industrial processing	2,200
Wood products for packing and food storage	1,850
Firewood	1,400
Mushrooms and berries (fresh weight)	1,480
Mushrooms and berries (dry weight)	7,400
Medical plants and medical raw stuff	7,400
Seeds of trees and bushes	7,400

One example of intervention in normal forest related practices in countries outside the former Soviet Union is the case of roe deer hunting in Sweden. In 1988 the average muscle content of roe deer shot in the autumn was 12,000 Bq kg^{-1} in the Gävle area. The intervention level for such foodstuffs in Sweden is 1,500 Bq kg^{-1}. Such high levels of contamination of roe deer meat were due to the preferential consumption of fungi by the deer in the autumn. As a result of experiments, the Swedish authorities recommended a change of hunting season for roe deer to the spring and this change was applied voluntarily by the hunting community in the early 1990s. As a result, the radiocaesium content in roe deer meat in Gävle was reduced by approximately six times. The recommendation to shift the hunting season to the spring has remained in place until the present day (Johanson 1994).

In addition, the management of reindeer by the Sami people in northern Sweden has been altered in a variety of ways to help reduce the radiocaesium content of animals before slaughter. This includes provision of clean fodder for sufficient time to reduce the body burden below the intervention level. A similar result can be achieved by altering the time of slaughter, sometimes in combination with feeding of clean fodder (Åhman 1994).

3.5. Aquatic countermeasures

Following a radioactive fallout, there are a number of different intervention measures to reduce radioactive doses to the public via the surface-water pathway. These actions may be grouped into two main categories: those aimed at reducing doses from radioactivity in drinking water and from consumption of aquatic foodstuffs.

In the context of an atmospheric fallout of radionuclides to both terrestrial and aquatic systems, it has been shown (Berkovski et al. 1996; Voitsekhovitch et al. 1996; Stone et al. 1997) that doses from terrestrial foodstuffs are in general much more significant than doses from drinking water and aquatic foodstuffs. However, for the Dnieper system, the river water transported radionuclides to areas which were not significantly contaminated by atmospheric fallout. This created a significant stress in the population and demand on decision makers to reduce radionuclide fluxes from the zone via the aquatic system. Many remediation measures were put in place, but because actions were not taken on the basis of dose reduction, most of these measures were ineffective. Moreover, radiation exposures to workers implementing these countermeasures were high.

Measures to reduce doses via drinking water may, however, be required, particularly in the short-term (timescale, weeks) after a fallout, when activity concentrations in surface waters are relatively high. Because of the importance of short lived radionuclides, early intervention measures, particularly changing supplies, can significantly reduce doses to the population. Measures to reduce doses via freshwater foodstuffs may be required over longer timescales as a result of bioaccumulation of radionuclides through the aquatic food chain.

Reviews of aquatic countermeasures (e.g., Voitsekhovitch et al. 1992; Waters et al. 1996; Smith et al. 2001, Voitsekhovitch 2001) have considered both direct (restrictions) and indirect intervention measures to reduce doses at the following stages of the aquatic dose pathway.

- Restrictions on water use or changing to alternative supplies;
- Restrictions on fish consumption;
- Water flow control measures (for example, dykes and drainage systems);
- Uptake by fish and aquatic foodstuffs from contaminated water; and

— Preparation of fish prior to consumption.

To our knowledge, no countermeasures were required, or applied, in marine systems after Chernobyl.

3.5.1. Measures to reduce doses at the water supply and treatment stage

Restrictions were placed on the use of water from the Dnieper River for the first year after the accident. Abstraction of drinking water for Kyiv was switched to the Desna River with use of a pipeline built during the first weeks after the accident. A summary of the measures taken by the Ukrainian authorities to switch to alternative supplies from less contaminated rivers and from groundwater can be found in (Voitsekhovitch et al. 1992, 2001).

Radionuclides may be removed from drinking-water supplies during the water-treatment process. Suspended particles are removed during water treatment, and filtration can remove dissolved radionuclides. In the Dnieper Waterworks Station activated charcoal and zeolite were added to water-filtration systems. It was found that activated charcoal was effective in removing ^{131}I and ^{106}Ru, and zeolite was effective in removing ^{137} Cs, ^{134}Cs and ^{90}Sr. These sorbents were effective for the first three months, after which they became saturated and their efficiency declined (Tsarik 1993; Voitsekhovitch et.al. 1997). The average removal of these radionuclides from water (dissolved phase) was up to a factor of two.

After the accident, the upper gates on the Kyiv Reservoir dam were opened to release surface water. It was believed at the time that the surface water was relatively low in radionuclide content, because suspended particles had sunk to deeper waters. Therefore, the release of water would allow room in the reservoir to contain runoff water from inflowing rivers which was believed to be highly contaminated. In fact, because of direct atmospheric deposition to the reservoir surface, surface waters in the reservoir were much more contaminated than deep waters. As noted by Voitsekhovitch et al. (1997), "a better approach to lowering the water level within the Kiev Reservoir would have been to open the bottom dam gates and close the surface gates. This would have reduced the levels of radioactivity in downstream drinking water in the first weeks after the accident." Although this countermeasure was not efficiently implemented after the Chernobyl accident, regulation of flow, given the correct information of contamination, could effectively reduce activity concentrations in drinking water, as it takes some time (days or more) for lakes and reservoirs to become fully mixed.

In a large river-reservoir system such as the Dnieper, control of water flows in the system can significantly reduce transfers of radioactivity downstream (Zheleznyak et al. 1997). This study showed that in the Dnieper, the time it takes for water to travel from the Kyiv Reservoir to the Black Sea can be varied between 3 and 10 months. Over the time that the water takes to travel downstream, radioactive pollution reduces by decay of short-lived radionuclides and transfers (particularly of radiocaesium) to reservoir bed sediments.

3.5.2. Measures to reduce direct and secondary contamination of surface waters

Standard anti-soil erosion measures can be used to reduce runoff of radionuclides attached to eroded soil particles. Note, however, that typically less than 50% of radiocaesium and less than 10% of radiostrontium and radioiodine were in the particulate phase, and this limits the potential effectiveness of this countermeasure. It is also noted that the dissolved, rather than particulate, form of these radionuclides is important in determining activity concentrations in drinking water and freshwater biota.

Dredging of canal-bed traps to intercept suspended particles in contaminated rivers was carried out on the Pripyat River (Voitsekhovitch 2001). These canal-bed traps were found to be highly inefficient for two reasons: (1) flow rates were too high to trap small suspended particles carrying much of the radionuclide contamination; and (2) a significant proportion of the radionuclide activity (and most of the "available" activity) was in dissolved forms and thus would not have been intercepted by the sediment traps.

One hundred and thirty zeolite-containing dykes were constructed on smaller rivers and streams around Chernobyl in order to intercept dissolved radionuclides. These were found to be very ineffective: only 5-10% of the ^{90}Sr and ^{137}Cs in the small rivers and streams were adsorbed by these zeolite barriers (Voitsekhovitch et al. 1997). In addition, the rivers and streams on which they were placed were later found to contribute only a few percent to the total radionuclide load in the Pripyat-Dnieper system.

After the Chernobyl accident, spring flooding of the highly contaminated Pripyat flood plain resulted in increases in ^{90}Sr-activity concentrations in the Pripyat River from annual average activity concentrations of around 1 Bq L^{-1} to a maximum of around 8 Bq L^{-1} for a flood event covering an approximately 2-week period (Vakulovsky et al. 1994). In 1993, a dyke was constructed around the highly contaminated flood plain on the left bank of the Pripyat. This prevented flooding of this area and proved effective in reducing ^{90}Sr wash off to the river during flood events (Voitsekhovitch et al. 1997). A second dyke was constructed on the right bank of the Pripyat in 1999. Annual average ^{90}Sr-activity concentration in Kyiv Reservoir water, however, was below 1 Bq L^{-1} for all years from 1987 onwards. The radiological significance of the ^{90}Sr-activity concentrations in Kyiv Reservoir water, even during the short flood events, is therefore very low, though it has been argued that the averted *collective* dose to the large number of users of the river-reservoir system is significant.

It is potentially possible to increase the sedimentation of radionuclides from lakes and reservoirs by introduction of strongly sorbing material, such as a zeolite or an (uncontaminated) mineral soil. This method has not to our knowledge been tested. Using a model for the removal of radiocaesium from lakes by settling of suspended particles Smith et al. (2001) identified two problems with this method of reducing radiocaesium in lakes: (1) large, deep lakes would require extremely large amounts of sorbent; and (2) secondary contamination of the lake by remobilisation of activity from the catchment and/or bottom sediments would require repeat applications in most systems.

3.5.3. Measures to reduce uptake by fish and aquatic foodstuffs

Bans on consumption of freshwater fish have been applied in the restricted zones of Chernobyl (Ryabov 1992). In some areas, selective bans on the more contaminated predatory fish have been applied. It is believed that such bans are often ignored by fishermen. Bans on the sale of freshwater fish were applied in some areas of Norway (Brittain et al. 1991). Use of farmed fish could be used as an alternative source of freshwater fish in areas affected by fishing bans. Farmed fish, fed with uncontaminated food, do not significantly accumulate radionuclides (Camplin et al. 1989).

Addition of lime to reduce radioactivity in fish was tested in 18 Swedish lakes (Håkanson and Andersson 1992). The results of the experiments showed that liming had no significant effect on uptake of ^{137}Cs in fish in comparison with control lakes. Although uptake of ^{90}Sr was not studied in these experiments, it is expected that increased calcium concentration in lakes may have an effect on ^{90}Sr concentration in fish. Experience of lake liming, in conjunction with artificial feeding of fish in Ukraine, has been summarised by Voitsekhovitch (2001).

It is known that the concentration factor of radiocaesium in fish is inversely related to the potassium content of the surrounding water. After the Chernobyl accident, potassium was added to 13 lakes in Sweden, either as potash or as an additive in mixed lime (Håkanson and Andersson 1992). The results of the potash treatment were somewhat inconclusive with a small reduction in activity concentrations in perch fry observed during the two-year experiment. It was found that in lakes with short water-retention times it was difficult to maintain high levels of K^+ in the lake.

In an experiment on Lake Svyatoe (a 'closed lake') in Belarus, Kudelsky and coworkers (Kudelsky et al. 2002; Smith et al. 2003) added potassium-chloride fertilizer onto the frozen lake surface. Results showed a significant (factor of 3) overall reduction in ^{137}Cs concentration in fish during the first years after the experiment. However, as expected, the ^{137}Cs in the water increased by a factor of 2-3 after the countermeasure application. It is likely that potassium treatment is only feasible in lakes with very long water-residence times, which allow increased potassium concentrations to be maintained. Also, the increased ^{137}Cs in water is unlikely to be acceptable in lakes where water is abstracted for drinking.

Manipulation of the aquatic food web by intensive fishing was carried out in four lakes in Sweden (Hakanson and Andersson 1992), and as a complementary measure in an additional three lakes. This resulted in a reduction of the fish population by about 5-10 kg per hectare. The species reduced were mainly pike, perch and roach. No effect of intensive fishing on ^{137}Cs concentrations in fish was observed. Fertilisation was carried out in two Swedish lakes using "Osmocoat" (5% P and 15% N). The concentrations of total phosphorus generally showed no change in the long-term mean value: it appears that the fertilisation remedy was not conducted sufficiently effectively. No effect was observed on ^{137}Cs-activity concentrations in fish.

Different methods of food preparation may affect the quantity of radionuclides in consumed food (Rantavaara 1989). Ryabov (1992) suggested bans on consumption of smoked and dried fish, because these processes increase concentrations of radionuclides (per unit of weight consumed). Other preparation processes may reduce radioactivity in fish by, approximately, a factor of two. An effective measure to reduce consumption of radiostrontium is to remove the bony parts of fish prior to cooking, because strontium is mainly concentrated in the bones and skin. Various other food-preparation methods are discussed in IAEA/FAO guidance (IAEA/FAO 1994)

3.5.4. Groundwater

To our knowledge, there have been no measures to protect groundwater supplies after an atmospheric deposition of radioactivity. Groundwater residence times are long enough that shorter lived radionuclides such as ^{131}I will have decayed long before they affect drinking water. Only very small amounts of radiostrontium and radiocaesium percolate from surface soils to groundwater after atmospheric deposition. A study by Bugai and coworkers (Waters et al. 1996) showed that after the Chernobyl accident, exposure to ^{90}Sr and ^{137}Cs via the groundwater pathway was insignificant in comparison with other pathways (food, external exposure, etc.).

Measures were taken to protect groundwater from seepage of radionuclides from the "Shelter" and from waste sites in the Exclusion Zone. These focused mainly on the construction of engineering and geochemical barriers around the local hot spots to reduce groundwater fluxes to the river network. Actions to stop atmospheric precipitation inside the "Shelter" and drainage of rainwater collected in the bottom rooms of the "Shelter" have to be considered also as a preventive measure to reduce groundwater contamination around the Chernobyl NPP industrial site.

3.5.5. Irrigation water

As discussed previously, irrigation did not significantly add to radionuclide contamination of crops which had previously been affected by atmospheric deposition of radionuclides. Thus, in practice, no countermeasures were directly applied to irrigation waters. However experience described in (Voitsekhovitch 2001) showed that the change from sprinkling to drainage irrigation of agricultural plants (for instance for vegetables) can reduce the transfer of radionuclides from water to crops by several times. This, in combination with improved fertilisation of irrigated lands can effectively reduce radioactivity in crops irrigated by water from reservoirs affected by radioactive pollution.

3.6. Conclusions and recommendations

The Chernobyl accident led to an extensive set of actions from the USSR authorities by introducing a range of short- and long-term environmental countermeasures to reduce the negative consequences. The countermeasures involved a great amount of human, economic and scientific recourses. Unfortunately, there was not always openness and transparency for the public and information was withheld. This can in part explain some of the problems experienced later on in communication with the public and the mistrust of the competent authorities. This sort of an information crisis also led, in many other countries outside Russia, Belarus and Ukraine, to a distrust in authority which in many countries has lead to investigation in how to deal with such major accidents in an open and transparent way and how the affected people can be part of decision-making processes.

Unique experience of countermeasure application after the Chernobyl accident has already been widely used both at national and international levels in order to improve preparedness for future nuclear and radiological emergencies, e.g., (CAC 1989; IAEA 1994, 1996b, 1997; ICRP 1993); however, more should be done in order to account comprehensively both for positive and negative lessons from the Chernobyl experience.

3.6.1. Conclusions

(1) The Chernobyl accident led to an extensive set of actions from the USSR and later independent country authorities by introducing a range of short- and long-term environmental countermeasures aimed to reduce the accident's negative consequences. The countermeasures involved large amounts of human, economic and scientific recourses.

(2) When social and economic factors along with the radiological ones are taken into account during planning and application of countermeasures, better acceptability of them by the public has been achieved.

(3) The unprecedented scale and long-term consequences of the Chernobyl accident required development of some additional radiation-safety standards, which have been developed following changes of radiation conditions, and promoted harmonisation of standards worldwide.

(4) Countermeasures applied in the early phase of the Chernobyl accident were only partially effective in reducing radioiodine intake via milk, because of the lack of timely information about the accident and advice on appropriate actions, particularly for private farmers.

(5) The most effective countermeasures in the early phase were exclusion of contaminated pasture grasses from animal diets and rejection of milk (with further processing) based on radiation-monitoring data. Feeding animals with "clean" fodder was effectively performed in some affected countries. The slaughtering of cattle was unjustified from a radiological point of view and had great hygienic, practical and economic problems.

(6) The greatest long-term problem has been radiocaesium contamination of milk and meat. In the USSR and later in the three independent countries, this has been addressed by the treatment of land used for fodder crops, clean feeding and application of caesium binders to animals that enabled most farming practices to continue in affected areas and resulted in a large dose reduction. While in the long term environmental radiation conditions change slowly, the efficiency of environmental countermeasures is also assumed to remain stable.

(7) Decontamination of settlements was widely applied in radiocontaminated regions of the USSR during the first years after the Chernobyl accident as a means of reducing external exposure of the public; this was cost-effective with regard to external dose reduction when its planning and implementation was preceded by a remediation assessment based on cost-benefit considerations and external dosimetry data.

(8) The decontamination of urban environments has produced a considerable amount of low-level radioactive waste that creates a problem of disposal. Secondary contamination of cleaned-up plots has not been observed.

(9) The following forest-related restrictions widely applied in the USSR and later in the three independent countries and partially in Scandinavia have reduced human exposure due to residence in radioactively contaminated forests and use of forest products:

— Restrictions on public and forest-worker access as a countermeasure against external exposure;

— Restricted harvesting by the public of food products, such as game, berries and mushrooms, contributed to a reduction of internal doses. In the affected countries mushrooms are readily consumed and, therefore, this restriction has been particularly important;

— Restricted collection of firewood by the public to prevent exposures in the home and garden when the wood is burned and the ash is disposed of or used as a fertiliser;

— Alteration of hunting practices aimed to avoid consumption of meat with high seasonal levels of radiocaesium;

— Fire prevention, especially in areas with large scale radionuclide deposition, in order to avoid secondary contamination of the environment.

(10) However, the experience in the three countries of the former Soviet Union has shown that such restrictions can also result in significant negative social consequences, and advice from the authorities to the general public may be ignored as a result. This situation can be offset by the provision of suitable educational programmes targeted at the local scale to emphasise the relevance of suggested changes in use of individual areas in the forest.

(11) It is unlikely that any technologically based forest countermeasures, i.e., the use of machinery and/or chemical treatments to alter the distribution or transfer of radiocaesium in the forest, will be practicable on a large scale.

(12) Numerous countermeasures put in place in the months and years after the accident to protect water systems from transfers of radioactivity from contaminated soils were, in general, ineffective and expensive, and led to relatively high exposures to workers implementing the countermeasures.

(13) The most effective countermeasure for aquatic pathways was the early restriction of drinking-water abstraction and changing to alternative supplies. Restrictions on consumption of freshwater fish have also proved effective in Scandinavia and Germany, though in Belarus, Russia and Ukraine such restrictions may not always have been adhered to.

(14) It is unlikely that any future countermeasures to protect surface waters will be justifiable in terms of economic cost per unit of dose reduction. It is expected that restrictions on consumption of fish will remain, in a few cases (for so-called. closed lakes), for several more decades.

3.6.2. Recommendations

3.6.2.1. Countries affected by the Chernobyl accident

(1) All kinds of long-term remediation measures and regular countermeasures in the areas contaminated with radionuclides should be applied, if they are radiologically justified and optimised. In optimising countermeasures, social and economic factors should be taken into account, along with formal cost-benefit analysis, aimed at achieving acceptability of countermeasures by the public.

(2) Authorities and the general public should be particularly informed on radiation-risk factors and technological possibilities to reduce them in the long term via remediation and countermeasures. Local authorities and the public should be involved in discussion and decision-making.

(3) In the long term after the Chernobyl accident, remediation measures and regular countermeasures remain efficient and justified mainly in the agricultural areas with poor (sandy and peaty) soils and resulting high radionuclide transfer from soil to plants.

(4) Particular attention must be given to the production of private farms of several hundred settlements and about 50 intensive farms in Belarus, Russia and Ukraine, where radionuclide concentrations in milk still exceed national action levels.

(5) Among long-term remediation measures, radical improvement of pastures and grasslands, as well as draining of wet peaty areas, is of high efficiency. The more efficient regular agricultural countermeasures are pre-slaughter clean feeding of animals accompanied with *in-vivo* monitoring, application of Prussian Blue to cattle and enhanced application of mineral fertilisers in plant growing.

(6) Restricting harvesting of wild food products such as game, berries, mushrooms and fish from ‘closed lakes’ by the public still may be needed in areas where their activity concentrations exceed national action levels.

(7) Advice on diet aimed to reduce consumption of highly contaminated wild food products and on simple cooking procedures to remove radioactive caesium still is an important countermeasure in reducing internal exposure.

(8) It is necessary to identify sustainable ways to make use of the more affected areas that reflect the radiation hazard, but also revive the economic potential for the benefit of the community.

3.6.2.2. Worldwide

(9) Unique experience of countermeasure application after the Chernobyl accident should be carefully documented and used for preparation of international and national guidance for authorities and experts responsible for radiation protection of the public and the environment.

(10) Practically all long-term agricultural countermeasures implemented in the large scale on contaminated lands of the three more affected countries can be recommended for use in case of future accidents. However, the effectiveness of soil-based countermeasures varies at each site. Therefore, analysis of soil properties and agricultural practice before application is of great importance.

(11) Recommendations for decontamination of the urban environment in case of large-scale radioactive contamination should be distributed to management of nuclear facilities that have the potential of substantial accidental radioactive release (NPPs and reprocessing plants) and to authorities of adjacent regions.

3.6.2.3. Research

(12) Generally, physical and chemical processes involved in environmental countermeasures and remediation technologies, both of mechanical nature (radionuclide removal, mixing with soil, etc.), chemical nature (soil liming, fertilisation, etc.) or their combinations, are understood well enough in order to be modeled and applied in similar circumstances worldwide. Much less understood are the biological processes that could be used in environmental remediation, e.g., re-profiling of agricultural production, bioremediation, etc. These processes require more research.

(13) An important issue that requires more sociological research is perception by the public of the introduction, performance and withdrawal of countermeasures in case of emergency, as well as development of social measures aiming at involvement of the public in these processes at all the stages beginning with the decision making.

(14) There still is substantial diversity in international and national radiological criteria and safety standards applicable to remediation of affected areas in case of environmental contamination with radionuclides. Experience of radiological protection of the public after the Chernobyl accident has clearly shown the need for further international harmonisation of appropriate radiological criteria and safety standards.

REFERENCES TO SECTION 3

ÅHMAN, B. (1994) Radiocaesium in reindeer (Rangifer Tarandus Tarandus) after fallout from the Chernobyl accident. Thesis, Swedish University of Agricultural Sciences, ISBN 91-576-4837-9.

ALEXAKHIN, R.M. (ED.) (1991) Recommendations for 1991-1995 on Agriculture Management in Areas Subjected to Contamination as a Result of the Accident at the Chernobyl NPP, State Commission of the USSR Council of Ministers on Food and Procurement, Moscow, 57 p. (In Russian).

ALEXAKHIN, R. A., KORNEEV, N. A. (Eds.) (1991) Agricultural Radioecology. Ecology, Moscow, 397 p. (in Russian).

ALEXAKHIN, R.M. (1993) Countermeasures in agricultural production as an effective means of mitigating the radiological consequences of the Chernobyl accident. The Science of the Total Environment, 137, 9-20.

AMIRO, B.D., GREBEN'KOV, A. and VANDENHOVE, H. (1999) Countermeasures and risks associated with contaminated forests. In: Contaminated forests, Ed. by LINKOV, I. and SCHELL, W.R., NATO Science Series, Vol. 58, pp 395 – 401.

ANTSIPOV G., TABACHNY L., BALONOV M. and ROED J. (2000) Evaluation of the effectiveness of decontamination activities in the CIS countries for objects contaminated as a result of the Chernobyl accident. In: Proceedings of the EU/CIS Workshop on Restoration of Contaminated Territories Resulting from the Chernobyl Accident, Report EUR 18193 EN, Brussels, pp 10-15.

BALONOV, M. (1993) Overview of Dose to the Soviet Population from the Chernobyl Accident and Protective Actions Applied, In: The Chernobyl Papers, Vol. I, Doses to the Soviet Population and Early Health Effects Studies. Ed. by MERWIN, S. and BALONOV, M., Research Enterprises, Richland, pp 23–45.

BALONOV M.I., GOLIKOV V.Y., ERKIN V.G., PARCHOMENKO V.I., PONOMAREV A. (1992) Theory and practice of a large-scale programme for the decontamination of the settlements affected by the Chernobyl accident. In: Proc. of the Int. Seminar on Intervention Levels and Countermeasures for Nuclear Accidents. Report EUR 14469, pp. 397-415.

BARYAKHTAR V.G., 1997. Chernobyl Catastrophe. Export Publishing House, Kyiv, 572 p.

BERESFORD, N. A., HOWARD, B. J., BARNETT, C. L. et al. (1999) Reducing the consumption of ^{137}Cs via forest fungi – provision of self-help advice. In: Contaminated forests, Ed. by LINKOV, I. and SCHELL, W.R., NATO Science Series, Vol. 58, pp 359 – 368.

BERKOVSKI, V., VOITSEKHOVITCH, O.A., NASVIT, O., ZHELEZNYAK, M. and SANSONE, U. (1996) Exposures from aquatic pathways. In: The radiological consequences of the Chernobyl accident. Ed. by A. Karaoglou, G. Desmet, G.N. Kelly and H.G. Menzel. Report EUR 16544 EN, European Commission, Luxembourg, pp. 283-294.

BOGDEVITCH, I.M. (2003) Guidelines on Agricultural and Industrial Production Under Radioactive Contamination in The Republic of Belarus, Minsk, 72 p. (In Russian).

BOGDEVITCH, I.M., SHMIGELSKAYA, I.D. and TARASIUK S.V. Rational using of the contaminated soils after Chernobyl accident in Belarus. Natural resources (1998) pp. 15-29 (In Russian).

BOGDEVITCH, I., SANZHAROVA, N., PRISTER, B., TARASIUK S. (2002) Countermeasures on natural and agricultural areas after Chernobyl accident. In: Role of GIS in Lifting the Cloud off Chernobyl, J. Kolejka (ed.), Kluwer Academic Publishers, pp 147-158.

BOGDEVITCH, I. M., PUTYATIN , YU. V., SHMIGELSKAYA, I. D., SERAYA, T. M., CENCEVICKI, F. A. (2003) Food crop production on personal plots of land contaminated with radionuclides. Recommendations of the Institute for Soil Science and Agrochemistry of the National Academy of Sciences of Belarus, Minsk, 27 p. (in Russian).

BRITTAIN, J.E., STORRUSTE, A. and LARSEN, E. (1991) Radiocaesium in Brown Trout (*Salmo trutta*) from a subalpine lake ecosystem after the Chernobyl reactor accident. *Journal of Environmental Radioactivity,* 14, 181-191.

BRYNILDSEN, L. I., SELNAES, T. D., STRAND, P. & HOVE, K. (1996). Countermeasures for radiocesium in animal products in Norway after the Chernobyl accident - techniques, effectiveness, and costs. *Health Physics 70*, 665-672.

CAMPLIN, W.C., LEONARD, D.R.P., TIPPLE, J.R. AND DUCKETT, L. (1989) *Radioactivity in freshwater systems in Cumbria (UK) following the Chernobyl accident*. MAFF Fisheries Research Data Report No. 18. 90 p.

CODEX ALIMENTARIUS COMMISSION (1989) Guideline Levels for Radionuclides in Foods Following Accidental Nuclear Contamination for Use in International Trade (CAC/GL 5-1989).

CORE: http://www.core-chernobyl.org/eng/

DEVILLE-CAVELIN, G., ALEXAKHIN, R.M., BOGDEVITCH, I.M., PRISTER, B.S., BIESOLD, H., PEREPELYATNIKOVA, L.V., SANZHAROVA, N.I., TARASIUK, S.V. (2001) Countermeasures in Agriculture: assessment of efficiency. Proceeding of the International Conference “Fifteen Years after the Chernobyl Accident. Lessons Learned”, Kiev, pp. 118-128.

ENVREG 9602: http://mns.gov.ua/chornobyl/envreg-9602icd/?m=17

EUROPEAN COMMISSION (1986). Council Regulation (EEC) No 1707/86 OJ No L 146 of 31.05.1986, pp. 88-90.

FESENKO, S. and BROWN, J. (2000) Review of countermeasures options for semi-natural environments: forest and natural meadows. NRPB-M1123, National Radiological Protection Board, Chilton, Didcot, Oxon, UK.

GUIDANCE FOR AGRICULTURAL WORKERS AND POPULATION INHABITING THE AREA OF THE RADIOACTIVE TRAIL OF THE CHERNOBYL NPP. (1986) RIARAE, USSR State Agroindustrial Committee, Moscow, 15 p. (In Russian).

GUILLITTE, O. and WILLDRODT, C. (1993) An assessment of experimental and potential countermeasures to reduce radionuclide transfers in forest ecosystems. Science of the Total Environment, 137, 273–288.

GUILLITTE, O., TIKHOMIROV, F. A., SHAW, G. and VETROV, V. (1994) Principles and practices of countermeasures to be carried out following radioactive contamination of forest areas. Science of the Total Environment, 157, 399 - 406.

HÅKANSON, L. and ANDERSSON, T. (1992) Remedial measures against radioactive caesium in Swedish lake fish after Chernobyl. *Aquatic Sciences, 54,* 141-164.

HANSEN, H.S., HOVE, K. and BARVIK, K. (1996). The effect of sustained release boli with ammoniumiron (III)-hexacyanoferrate(II) on radiocesium accumulation in sheep grazing contaminated pasture. *Health Physics*, *71,* 705-712.

HERIARD DUBREUIL G., LOCHARD J., GIRARD P., GUYONNET J.F., LE CARDINAL G., LEPICARD S. et al. (1999) Chernobyl Post-Accident Management: The ETHOS Project. Health Physics, 77, pp. 361-372.

HOVE, K. (1993). Chemical methods for reduction of the transfer of radionuclides to farm animals in semi-natural environments. *Science of the Total Environment*, 137, 235-248.

HOVE, K. and HANSEN, H.S. (1993). Reduction of radiocaesium transfer to animal products using sustained release boli with ammoniumiron(III)-hexacyanoferrate(II). *Acta vetinaria scandinavia, 34,* 287-297.

HOVE, K., STRAND, P., SALBU, B., OUGHTON, D., ASTASHEVA, N., SOBOLEV, A., VASILIEV, A., RATNIKOV, A., ALEKSAKHIN, R., JIGAREVA, T., AVERIN, V., FIRSAKOVA, S., CRICK, M. and RICHARDS, J.I. (1995). Use of caesium binders to reduce radiocaesium contamination of milk and meat in Belarus, Russia and Ukraine. In: Environmental Impact of Radioactive Releases. IAEA-SM-339/153, Vienna, pp 539-547.

HUBBARD, L., RANTAVAARA, A., ANDERSSON, K., ROED, J. (2002) Tools for forming strategies for remediation of forests and park areas in northern Europe after radioactive contamination: background and techniques. Nordic Nuclear Safety Research, NKS-52. ISBN 87-7893-105-3.

HUBERT, P., ANISIMOVA, L., ANTSIPOV, G., RAMZAEV, V., SOBOTOVICH, V. (EDS.) (1996) Strategies for Decontamination, Final report on Experimental Collaboration Project No.4, EUR 16530 en, European Commission, Luxembourg.

INTERNATIONAL ADVISORY GROUP (1991), International Chernobyl Project, Technical Report, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1982) Basic Safety Standards for Radiation Protection, Safety Series No.9, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1990) Recovery Operations in the Event of a Nuclear Accident or Radiological Emergency. IAEA-SM-316, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1996a), One Decade after Chernobyl: Summing up the Consequences of the Accident, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1996b) International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources, Safety Series No.115, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1997) The use of Prussian Blue to reduce radicaesium contamination of milk and meat produced on territories affected by Chernobyl accident. Report of United Nations Project E 11, TECDOC-926, IAEA, Vienna, 80 p.

INTERNATIONAL ATOMIC ENERGY AGENCY (2001) Present and future environmental impact of the Chernobyl accident, TECDOC-1240, IAEA, Vienna, 128 p.

INTERNATIONAL ATOMIC ENERGY AGENCY AND FOOD AND AGRICULTURAL ORGANISATION (1994) Guidelines for Agricultural Countermeasures Following an Accidental Release of Radionuclides, Technical Reports Series No. 363. IAEA, Vienna, 115 p.

INTERNATIONAL COMISSION ON RADIOLOGICAL PROTECTION (1977) Recommendations of the International Commission on Radiological Protection, ICRP Publication 26, Pergamon Press, Oxford and New York.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (1984) Protection of the Public in the Event of Major Radiation Accidents: Principles for Planning, ICRP Publication 40, Pergamon Press, Oxford and New York.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (1989) Optimization and Decision-Making in Radiological Protection, Publication 55, Annals of the ICRP, Vol. 20, No. 1.

INTERNATIONAL COMISSION ON RADIOLOGICAL PROTECTION (1991) 1990 Recommendations of the International Commission on Radiological Protection, ICRP Publication 60, Pergamon Press, Oxford and New York.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (1993) Principles for Intervention for Protection of the Public in a Radiological Emergency, ICRP Publication 63, Pergamon Press, Oxford and New York.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (1999) Protection of the Public in Situations of Prolonged Radiation Exposure, ICRP Publication 82, Annals of the ICRP, Vol. 29, Nos. 1-2.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (2003) A framework for assessing the impact of Ionizing radiation on non-human species. ICRP Publication 91, Annals of the ICRP, Vol. 33, No. 3.

JACOB P., FESENKO S., FIRSAKOVA S. K., LIKHTAREV I. A., SCHOTOLA C., ALEXAKHIN R. M., ZHUCHENKO Y. M., KOVGAN L., SANZHAROVA N. I. AND AGEYETS V. (2001) Remediation strategies for rural territories contaminated by the Chernobyl accident. J. Environmental Radioactivity, v. 56, pp. 51-76.

JOHANSSON, K. (1994). *Radiocaesium in Game Animals in the Nordic Countries.* In: Nordic Radioecology. The Transfer of Radionuclides through Nordic Ecosystems to Man. Ed. by Dahlgaard, H., Elsewier, Amsterdam, 287-301.

KUDELSKY A.V., SMITH J.T., PETROVICH A.A. (2002). An experiment to test the addition of potassium to a non-draining lake as a countermeasure to ^{137}Cs accumulation in fish. *Radioprotection – Colloques* vol. 37, 621-626.

LOCHARD, J. (2004) Living in Contaminated Territories: A Lesson in Stakeholder Involvement. In: 'Current Trends in Radiation Protection', EDP Sciences, pp. 211-220.

LOS I. AND LIKHTAREV I. (1993) The peculiarities of urban environmental contamination and assessment of actions aimed at reduction of public exposure. Int. J. of Radiation Hygiene, v.1, No 1, pp. 51-59.

MINISTRY OF HEALTH (1997) Permissible levels of ^{137}Cs and ^{90}Sr contents in products of forestry. Hygienic standards, GH 2.6.1.670-97. Ministry of Health of the Russian Federation, Moscow (in Russian).

MÜCK, K. (2002) Environmental restoration by natural effects – Advantages and limits. In: Radiation Legacy of the 20th Century: Environmental Restoration, IAEA TECDOC- 1280, IAEA, Vienna, pp. 115-126.

NADTOCHIY P.P., MALINOVSKIY A.S., MOGAR A.O., LAZAREV N.M., KASHPAROV V.A., MELNIK A.I. (2003). Experience of liquidation of the Chernobyl accident consequences, Svit, Kyiv, 371 p. (In Ukrainian).

OUGHTON, D., FORSBERG, E-M, BAY, I, KAISER, M. and HOWARD, B. (2004) An ethical dimension to sustainable restoration and long-term management of contaminated areas. J. Environmental Radioactivity, 74, pp 171-183.

PANFILOV, A. (1999) Countermeasures for radioactively contaminated forests in the Russian Federation. In: Contaminated forests, Ed. by LINKOV, I. and SCHELL, W.R., NATO Science Series, Vol. 58 (1999) 271 – 279.

PRISTER, B. (ED.) (1998) Recommendations on Agriculture Management on Contaminated Territories, Ukrainian Institute of Agricultural Radiology, Kyiv, 102 p. (In Ukrainian).

PRISTER, B., PEREPELYATNIKOV, G.P., PEREPELYATNIKOVA, L.V. (1993). Countermeasures used in the Ukraine to produce forage and animal food products with radionuclide levels below intervention limits after the Chernobyl accident. *Science of the Total Environment, 137,* 183-198.

PRISTER B., ALEKSAKHIN R., FIRSAKOVA S., HOWARD B. (2000) Short and Long Term Environmental Assesment. In: Proceedings of the EU/CIS Workshop on Restoration of Contaminated Territories Resulting from the Chernobyl Accident, Report EUR 18193 EN, Brussels, pp 103-114.

RAFFERTY, B. and SYNNOTT, H. (1998) FORECO – countermeasures applied in forest ecosystems and their secondary effects. Serie Documenti 6/1998, ANPA. Rome, Agenzia Nazionale per la Protezzione dell'Ambiente (32 p).

RANTAVAARA, A.H. (1989) *Transfer of radionuclides during Processing and Preparation of Foods: Finnish Studies since 1986.* In: Proc. Int. Seminar on Radioactivity Transfer during Food Processing and Culinary Preparation. Report CEC, Luxembourg, pp 69-94.

RATNIKOV, AN., VASILIEV, A.V., KRASNOVA, E.G., PASTERNAK, A.D., HOWARD, B.J., HOVE, K. and STRAND, P. (1998). The use of hexacyanoferrates in different forms to reduce radiocaesium contamination of animal products in Russia. *The Science of theTtotal Environment, 223,* 167-176.

ROED J., ANDERSSON K. and PRIP H. (1995) Practical Means for Decontamination 9 years After a Nuclear Accident. Risoe-R-828(EN), Roskilde.

ROED J., ANDERSSON K., BARKOVSKY A., FOGH C., MISHINE A., OLSEN S., PONOMAREV A., PRIP H., RAMZAEV V., VOROBIEV B. (1998) Mechanical Decontamination Tests in Areas Affected by the Chernobyl Accident. Riso-R-1029(EN), Roskilde.

RYABOV, I.N. (1992) Analysis of countermeasures to prevent intake of radionuclides via consumption of fish from the region affected by the Chernobyl accident. In: Proc. Int. Seminar on Intervention Levels and Countermeasures for Nuclear Accidents. EUR-14469, Luxembourg: European Commission, pp. 379-390.

SHAW, G., ROBINSON, C., HOLM, E., FRISSEL, M. J. and CRICK, M. (2001) A cost-benefit analysis of long-term management options for forests following contamination with ^{137}Cs. J. Environmental Radioactivity, 56, 185 – 208.

SHEVCHUK, V.E., GOURACHEVSKIY, V.L., (EDS.) (2001) 15 Years After Chernobyl Catastrophe: Consequences in the Republic of Belarus and Their Overcoming, National

Report, Committee on the Problems of the Consequences of the Accident at the Chernobyl NPP, Minsk, 118 p.

SMITH, J.T., VOITSEKHOVITCH, O.V., HÅKANSON, L. and HILTON, J. (2001) A critical review of measures to reduce radioactive doses from drinking water and consumption of freshwater foodstuffs. J. Environmental Radioactivity, 56, 11-32.

SMITH J.T., KUDELSKY A.V., RYABOV I.N., HADDERINGH R.H., BULGAKOV A.A. (2003) Application of potassium chloride to a Chernobyl – contaminated lake: modelling the dynamics of radiocaesium in an aquatic ecosystem and decontamination of fish. *Science of the Total Environment,* 305, 217-227.

STATE COMMITTEE OF RUSSIA FOR SANITARY INSPECTION (1993) TPL-1993 Temporary Permissible Levels of Caesium-134 and -137 and Strontium-90 in Food Products, Hygienic Standard GS 2.6.005-93. State Committee of Russia for Sanitary Inspection, Moscow.

STONE, D., SMITH, J.T., JACKSON, D. and IBBOTSON, A.T. (1997) *Scoping study on the relative importance of freshwater dose pathways following a major nuclear accident*. Whitehaven: Westlakes Research Institute.

TIKHOMIROV, F.A. AND SHCHEGLOV, A.I. (1994). Main investigation results on the forest radioecology in the Kyshtym and Chernobyl accident zones. Science of the Total Environment, 157, 45-57.

TSARIK, N. (1993) Supplying water and treating sewage in Kiev after the Chernobyl accident. *Journal of the American Water Works Association*, *85,* 42-45.

UNITED NATIONS (1988) Sources, Effects and Risks of Ionizing Radiation (1988 Report to the General Assembly, with Annexes), Scientific Committee on the Effects of Atomic Radiation, UN, New York, pp 309–374.

UNITED NATIONS (2000) Sources and Effects of Ionizing Radiation (2000 Report to the General Assembly, with Scientific Annexes), Scientific Committee on the Effects of Atomic Radiation, UN, New York.

UNITED NATIONS DEVELOPMENT PROGRAMME AND UNITED NATIONS CHILDREN'S FUND (2002) The Human Consequences of the Chernobyl Nuclear Accident. Strategy for Recovery.

USSR MINISTRY OF HEALTH (1977) Standards of Radiation Safety SRS-76, Atomizdat, Moscow (in Russian).

VAKULOVSKY, S.M., NIKITIN, A.I., CHUMICHEV, V.B., KATRICH, I.YU., VOITSEKHOVITCH, O.A., MEDINETS, V.I., PISAREV, V.V., BOVKUM, L.A. and KHERSONSKY, E.S. (1994) Cs-137 and Sr-90 contamination of water bodies in the areas affected by releases from the Chernobyl Nuclear Power Plant accident: an overview. *J. Environmental Radioactivity* 23, 103-122.

VIDAL M., CAMPS M., GREBENSHIKOVA N., SANZHAROVA N., IVANOV Y., VANDECASTEELE C., SHAND C., RIGOL A., FIRSAKOVA S., FESENKO S., LEVCHUK S., CHESHIRE M., SAURAS T., RAURET G. (2001) Soil- and plant-based countermeasures to reduce ^{137}Cs and ^{90}Sr uptake by grasses in natural meadows: the REDUP project. Journal of Environmental Radioactivity. 56/1-2, 139-156.

VOITSEKHOVITCH, O.V. (2001) *Management of surface water quality in the areas affected by the Chernobyl accident*, Ukrainian Hydrometeorological Institute, Kiev, 135 p. (in Russian).

VOITSEKHOVITCH, O., KANIVETS, V. et al. (1992) Hydrological Processes and their Influence on Radionuclide Behaviour and Transport by Surface Water Pathways as Applied to Water Protection after Chernobyl Accident .// In: “Hydrological Impact of NPP” Proc. of the UNESCO International Symposium. UNESCO, Paris, pp 85-105.

VOITSEKHOVITCH, O.V., SANSONE, U., ZHELESNYAK, M. and BUGAI, D. (1996) Water quality management of contaminated areas and its effects on doses from aquatic pathways. In: The radiological consequences of the Chernobyl accident. Ed. by A. Karaoglou, G. Desmet, G.N. Kelly and H.G. Menzel. Report EUR 16544 EN, European Commission, Luxembourg, pp. 401-410.

VOITSEKHOVITCH, O.V., NASVIT, O., LOS’Y, Y. and BERKOVSKI, V. (1997) Present thoughts on aquatic countermeasures applied to regions of the Dnieper river catchment contaminated by the 1986 Chernobyl accident. In: G. Desmet, R. Blust, R.N.J. Comans, J.A. Fernandez, J. Hilton. & A. de Bettencourt, *Freshwater and estuarine radioecology*, (pp. 75-86). Amsterdam: Elsevier.

VOVK I., BLAGOEV V., LYASHENKO A. and KOVALEV I. (1993) Technical Approaches to Decontamination of Terrestrial Environments in the CIS (former USSR). Science of the Total Environment, 137, pp. 49-63.

WATERS R., GIBSON D., BUGAY, B., DGEPO, S., SKALSKY, A., VOITSEKHOVITCH, O. (1996). A review of Post-Accident Measures Affecting Transport and Isolation of Radionuclides Released from the Chernobyl Accident. In: Proc. of International Symposium on Environmental Contamination in the Central and Eastern Europe, Budapest-94. Budapest.

ZHELEZNYAK M., SHEPELEVA, V., SIZONENKO, V. and MEZHUEVA, I. (1997) Simulation of countermeasures to diminish radionuclide fluxes from the Chernobyl zone via aquatic pathways. *Radiation Protection Dosimetry, 73,* 181-186.

4. HUMAN EXPOSURE LEVELS

4.1. Introduction

4.1.1. Populations and areas of concern

Following the Chernobyl accident, both the workers and the general public were affected by radiation that resulted or still can result in adverse health effects. UNSCEAR in its Report-2000 selected the following three categories of exposed populations: (1) the workers involved in the accident, either during the emergency period or during the clean-up phase; (2) the inhabitants of contaminated areas that were evacuated in 1986; and (3) the inhabitants of contaminated areas who were not evacuated (UNSCEAR 2000).

In this document we consider primarily members of the general public exposed from radionuclides deposited in the environment, and not workers who were involved in the emergency response to the accident or in the cleanup following the accident and exposed predominantly on-site, i.e. at the Chernobyl NPP and in the Exclusion Zone. For information on Chernobyl worker populations, the reader is referred to the comprehensive material provided by the UNSCEAR (1988, 2000) and by the Chernobyl Forum in its report considering human health effects (WHO 2005).

Information on doses received by members of general public, both evacuated from the accident area and those who permanently live in radiocontaminated areas, is required for the following health-related purposes:

— Substantiation of countermeasures and remediation programmes;
— Forecast of expected adverse health effects and justification of corresponding health protection measures;
— Information for the public and for the authorities; and
— Epidemiological and other medical studies of radiation-caused adverse health effects.

In the present report, the methodologies and data specifically required for the estimation of mean doses to population groups living in particular settelements and selected by the factors influencing either external or internal dose or both are presented. These factors usually are age, gender, occupation, food habits, etc. Dose distributions among group members and collective doses are considered as well. In contrast, individual doses of the members of the public used mainly in analytical epidemiological studies are presented in the Chernobyl Forum Report on health consequences of the Chernobyl accident (WHO 2005). On these subjects, substantial progress has been achieved since publication of the comprehensive UNSCEAR report in 2000.

As mentioned in Section 2.1, atlases (Izrael 1998; De Cort et al. 1998) have been prepared that show the deposition of ^{137}Cs and other radionuclides throughout the former Soviet Union and the countries of Europe. The results indicate that the more affected countries are Belarus, Ukraine, and Russia. In addition, the countries of Sweden, Finland, Austria, Norway, Bulgaria, Switzerland, Greece, Slovenia, Italy, and Moldova had areas that can be considered to have been "contaminated," that is, at the level of more than 37 kBq m^{-2} (>1 Ci km^{-2}) of ^{137}Cs, see Table 2.2. Much of the information on doses from the Chernobyl accident will be focused on the three primary countries.

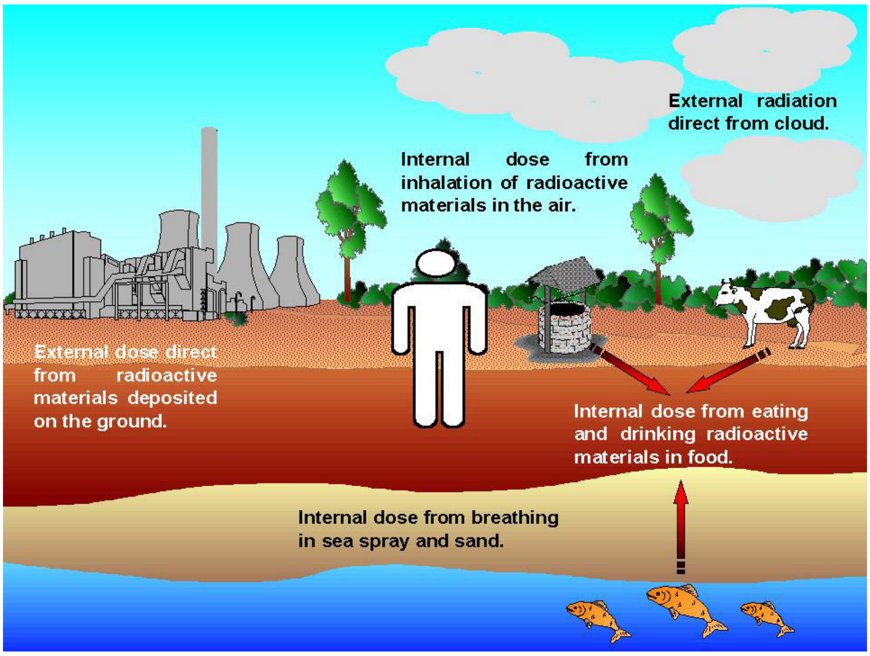


FIG. 4.1. Schematic diagram of pathways of exposure to man from environmental releases of radioactive materials.

4.1.2. Exposure pathways

Following the Chernobyl accident there were several pathways whereby humans were exposed to radioactive materials, Fig. 4.1. The main pathways are listed below in the approximate time sequence within which the doses would have been received.

(1) External dose from cloud passage;

(2) Internal dose from inhalation during cloud passage and of resuspended materials;

(3) External dose from radionuclides deposited upon soil and other surfaces; and

(4) Internal dose from the consumption of contaminated food and water.

Under most exposure conditions for members of the general public the two more important pathways are items three and four above: dose from radiations from the decay of radionuclides deposited upon the soil and other surfaces and from the ingestion of contaminated food and water. If persons would have been evacuated quickly after passage of the initial cloud, then the more important pathways would have been the first two items because the latter two pathways would have been prevented. Except for such unusual conditions, however, the first two pathways are not very significant and can generally be ignored.

4.1.3. Concepts of dose

Methods of calculating radiation dose have been refined over the years, and specific concepts have evolved (ICRP 1991; UNSCEAR 2000). The fundamental measure of radiation dose to an organ or tissue is *absorbed dose*, which is the amount of energy absorbed by that organ or tissue divided by its weight. The international unit of absorbed dose is the gray (Gy), which is equal to one joule per kilogram. Because this is a rather large amount of dose, it is common to use units of mGy (one thousandth of a Gy) or μGy (one millionth of a Gy).

Because many organs and tissues were exposed as a result of the Chernobyl accident, it has been very common to use an additional concept, that of *effective dose,* which is the sum of the products of absorbed dose to each organ multiplied by a radiation-weighting factor and a tissue-weighting factor. The former varies by radiation type and is related to the density of ionisations created; the latter is an approximation of the relative probability that an absorbed dose to a particular organ might lead to the production of a cancer. The sum of all tissue-weighting factors is equal to 1.0.

The concepts mentioned above are applied to individuals. If many individuals have been exposed to an event, such as happened following the Chernobyl accident, an additional concept of *collective dose* has been applied. The collective dose is the sum of the doses to all individuals within a particular group, which might be the residents of a particular country or the persons involved in some type of activity, such as cleaning up the consequences of the accident. This concept is most often applied to effective doses, and the common unit of collective effective dose is the man-Sv (or person-Sv).

Finally, the UNSCEAR (2000) has employed the concept of *dose commitment* to examine the long-term consequences of some practice or accident. For example, at the very moment that the Chernobyl accident occurred, it can be considered that a dose commitment occurred at the moment of the release of the radioactive materials. This is true even though it will take many years for the doses to be received by the persons alive at that time and persons not yet born or conceived.

4.1.4. Background radiation levels

Living organisms are continually exposed to ionizing radiation from natural sources, which include cosmic rays and terrestrial radionuclides (such as ^{40}K, ^{238}U, ^{232}Th and their progeny including ^{222}Rn (radon)). Table 4.1 shows the average annual dose and typical dose range worldwide from natural sources.

In addition to natural sources, radiation exposure occurs as a result of human activities. Table 4.2 shows the annual individual effective doses in 2000 on a worldwide basis. Diagnostic medical exposure is the largest non-natural source of radiation. The residual global effects of the Chernobyl accident are now very small but, of course, are higher in European countries and especially in the areas of Belarus, Russia and Ukraine.

TABLE 4.1. RADIATION DOSES FROM NATURAL SOURCES (UNSCEAR 2000)

Source	Worldwide average annual effective dose (mSv)	Typical range (mSv)
External exposure		
Cosmic rays	0.4	0.3–1.0
Terrestrial gamma rays	0.5	0.3–0.6
Internal exposure		
Inhalation (mainly radon)	1.2	0.2–10
Ingestion	0.3	0.2–0.8
Total	2.4	1-10

TABLE 4.2. EFFECTIVE DOSE IN YEAR 2000 FROM NATURAL AND HUMAN SOURCES (UNSCEAR 2000)

Source	Worldwide average annual per caput effective dose (mSv)	Range or trend in exposure
Natural background	2.4	Typically range from 1-10 mSv.
Diagnostic medical examinations	0.4	Ranges from 0.04-1 mSv at lowest and highest levels of health care.
Atmospheric nuclear testing	0.005	Has decreased from a maximum of 0.15 mSv in 1963. Higher in northern hemisphere.
Chornobyl accident	0.002	Has decreased from a maximum of 0.04 mSv in 1986 (in northern hemisphere). Higher at locations nearer the accident site.
Nuclear power production	0.0002	Has increased with expansion of nuclear programme but decreased with improved practice.

4.1.5. Decrease of dose rate with time

To calculate the dose for particular time periods, it is necessary to predict the decrease of dose rate with time. The most obvious mechanism acting to cause such a decrease is radioactive decay of the radionuclides. Additional dose-rate-reduction factors are usually called ecological half lives. For example, external gamma-exposure rates decrease with time due to the weathering of long-lived radionuclides, such as ^{137}Cs, into the soil and subsequent migration down the soil column; this results in increased absorption of the emitted radiations within the soil. Typically, a two component exponential function describes this process (Golikov et al. 2002; Likhtarev et al. 2002).

The rate at which ^{137}Cs is ingested with time also decreases at a rate faster than radioactive decay. This additional long term decrease is due mainly to the binding of ^{137}Cs to soil particles from which the caesium atoms are no longer biologically available. As with the external dose rate, the decrease of ^{137}Cs in milk or in humans living in areas contaminated by the Chernobyl accident also shows a two-component exponential decrease with time (Balonov et al. 1996; Likhtarev et al. 2000). (See Chapter 2 for more information.)

4.1.6. Critical groups

As with all situations that involve the exposure of large segments of the population to natural or man-made radioactivity, there is always a significant spread in the amount of radiation dose received by various members of the population living within the same geographic area. Those individuals with the higher doses are frequently called the *critical group,* and these persons may have doses twice or even higher than the average dose to all members of the population considered. Usually such persons can be identified in advance, and, in some cases, special protective measures might be considered.

For external dose members of the critical group are those who spend a considerable amount of time outdoors, either for occupational or recreational reasons; in addition, people living and/or working in buildings with minimal shielding might be members of the critical group.

For exposure to radioiodine isotopes, the critical group consists of infants drinking goat's milk. Infants have a thyroid gland weighing only two grams that concentrates roughly 30% of the radioiodine consumed; goats are more efficient than cows at secreting radioiodine into milk. For exposure to radiocaesiums critical groups have been identified as those who consume large quantities of local animal products, such as milk and meat, and "wild" products, such as game meat, mushrooms, wild berries, and lake fish.

4.2. External exposure

4.2.1. Formulation of the model of external exposure

In any situation of human external exposure caused by releases of radioactive substances into the environment, the following three types of data are necessary for assessment of organ or effective dose:

— parameters that describe the external gamma-radiation field;

— parameters describing human behaviour in this field; and

— conversion factors from dose in air to organ or effective dose.

The basic model for human external exposure in case of radioactive contamination of the environment is the model for exposure above an open plot of undisturbed soil; the absorbed dose in air $D(t)$ at the height of 1 m above the soil surface is used as the basic parameter to describe the radiation field. The value of this basic parameter is influenced not only by the surface activity of deposited radionuclides, but also by such natural factors as the initial penetration of radionuclides in soil, their radioactive decay, vertical migration of long-lived radionuclides, and the presence of snow cover.

The values of the radiation field are different, when population exposure occurs in an altered or disturbed environment. In models this factor is taken into account by using location factors, LF_i, defined as the ratio of the dose rate in air at point i inside a settlement to a similar value above a plot of undisturbed soil (Meckbach and Jacob 1988). Human behaviour in the radiation field is described by occupancy factors, OF_{ik}, which represent the fraction of time spent by individuals of the k-th population group at the i-th point of the settlement of interest. The third type of data necessary for assessment of effective external dose is conversion factors, CF_k, that relate actually measured values (the absorbed dose in air) with the criterion of radiation impact—the effective dose to the k-th population group.

On this basis, a deterministic model for assessment of the effective external dose rate E_k for representatives of the k-th population group is represented in Fig. 4.2.

4.2.2. Input data for the estimation of effective external dose

Numeric values of the parameters listed above have been determined from long-term dosimetric investigations in more highly contaminated regions after the Chernobyl accident.

4.2.2.1. Dynamics of external gamma-dose rate over open undisturbed soil

Immediately after the accident external gamma-exposure rates were relatively high, and contributions from many short-lived radionuclides were important. Thus, in the contaminated areas outside the Chernobyl NPP boundaries the initial dose rate over lawns and meadows ranged between 3-10 µGy h^{-1} in areas contaminated at about 37 kBq m^{-2} (1 Ci km^{-2}) of ^{137}Cs and up to 10,000 µGy h^{-1} within the 30-km exclusion area with higher depositions. Exposure rates decreased rapidly, due to radioactive decay of short-lived radionuclides, as shown in Fig. 4.3.

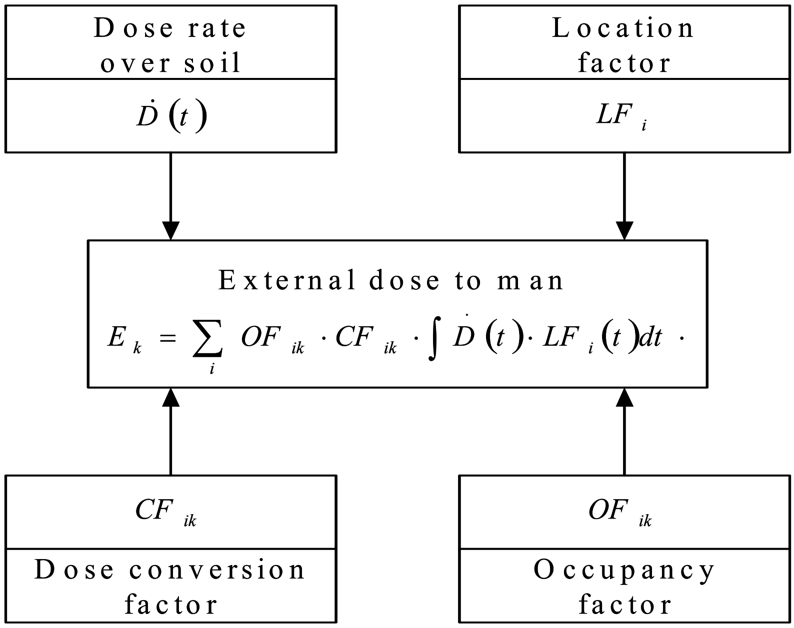


FIG. 4.2. Model of external exposure of the k-th population group (i is a location index). (Balonov et al. 1996).

Because of different isotopic compositions of radionuclide fallout in different geographical areas (Izrael et al. 1990, Mück et al. 2002, Likhtarev et al. 2002), the contribution of short-lived radionuclides to the overall dose rate was highly variable. In the 30-km zone, ^{132}Te+^{132}I, ^{131}I, and ^{140}Ba+^{140}La dominated during the 1st month and then ^{95}Zr+^{95}Nb for another half a year before ^{137}Cs and ^{134}Cs became dominant (Fig. 4.4). In contrast, in the far zone just the radioiodine isotopes dominated during the 1st month; afterwards ^{137}Cs and ^{134}Cs dominated with a moderate contribution from ^{103}Ru and ^{106}Ru (Fig. 4.5). Since 1987, more than 90% of dose rate in air has been attributed to gamma radiation of long-lived ^{137}Cs and ^{134}Cs. Thus, the radionuclide composition of the deposited activity was a major factor in determining external exposure of the population in the early period of time after the accident. Model estimations of gamma-dose rate in free air (90% confidence interval) based on the radionuclide composition of the deposited activity agree well with the measured values during the first month after deposition (see Fig. 4.6).

The influence of radionuclide migration into soil on the gamma-dose rate has been determined using gamma spectrometric analyses of over 400 soil samples taken during 1986–1999 in the contaminated areas of Germany (Bavaria), Sweden, Russia, and Ukraine (Jacob and Likhtarev 1996; Golikov et al. 2002; Likhtarev et al. 2002). The analysis also included data on the ^{137}Cs distribution in soil at sites in the north-east region of the USA, whose contamination was attributed to nuclear tests at the Nevada Test Site (Miller et al. 1990), and in Bavaria (Germany) which was due to global fallout. The last two data sets were obtained 20 to 30 years after deposition and this allows for long-term predictions to be applied to the Chernobyl depositions. The measurement sites were considered to be representative for reference sites, i.e., open undisturbed fields.

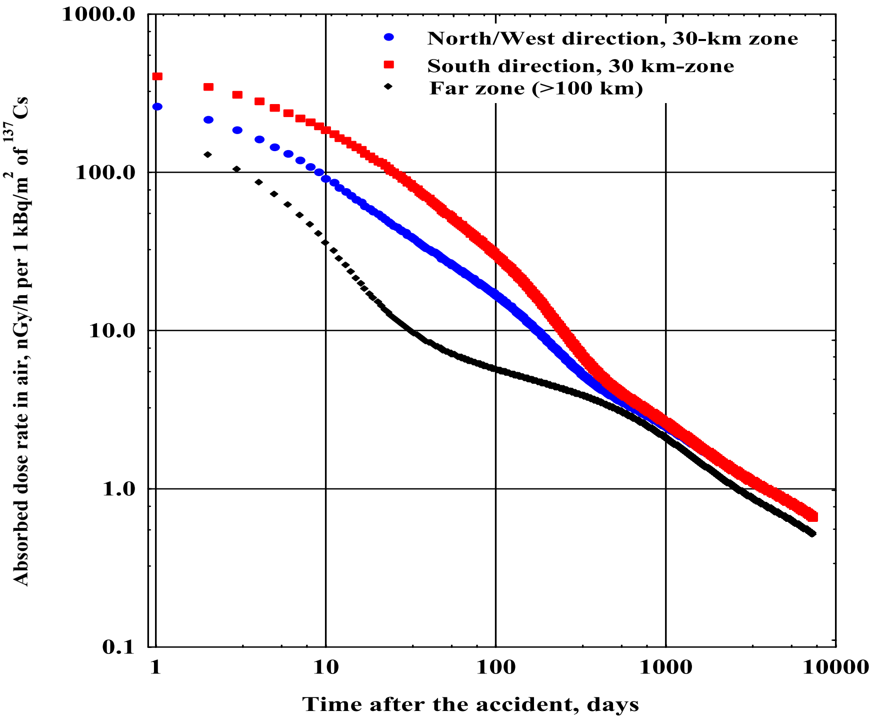


FIG. 4.3. Dynamics of standardised dose rate in air over undisturbed soil after the Chernobyl accident in different geographical areas (Golikov 2004).

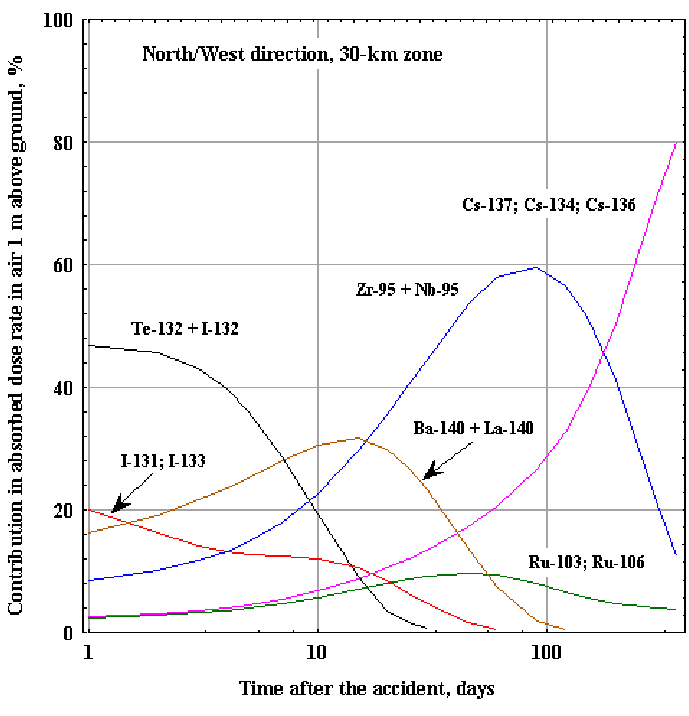


FIG. 4.4. Relative contribution of gamma radiation from individual radionuclides to the external gamma-dose rate in air during the first year after the Chernobyl accident (north/west direction, 30-km zone) (Golikov 2004).

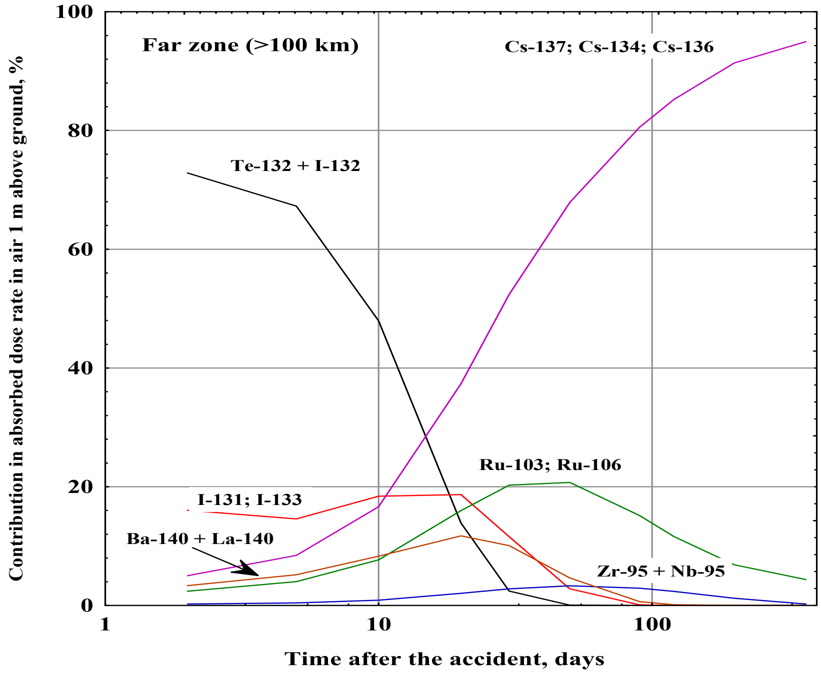


FIG. 4.5. Relative contribution of gamma radiation from individual radionuclides to the external gamma-dose rate in air during first year after the Chernobyl accident (far zone with distance from the Chernobyl Power Plant of more than 100 km). (Golikov 2004).

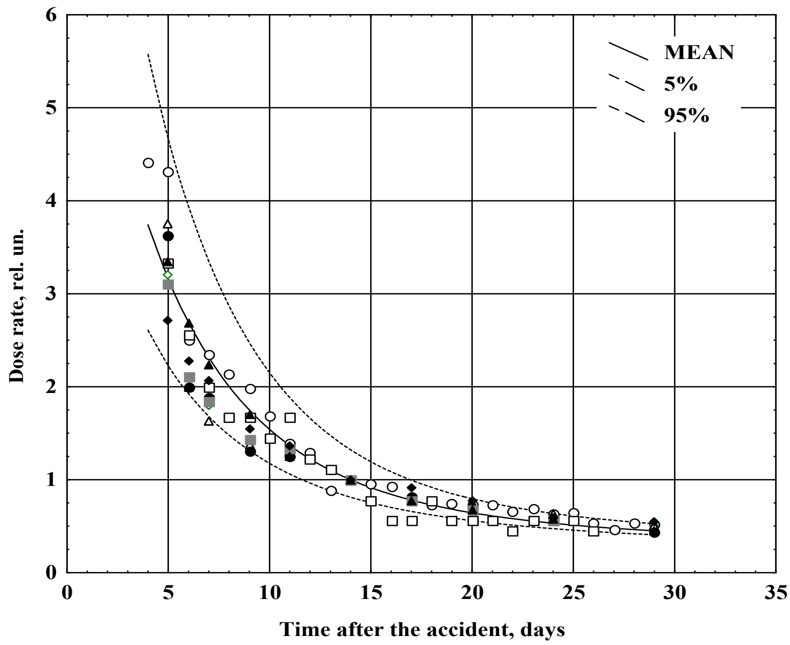


FIG. 4.6. Dose rate in air during the first days after the accident in several rural settlements in Bryansk and Tula Oblasts (normalized to the dose rate on 10 May 1986). Points indicate dose-rate measurements and curves represent calculated values according to the isotopic composition (Golikov et al. 2002).

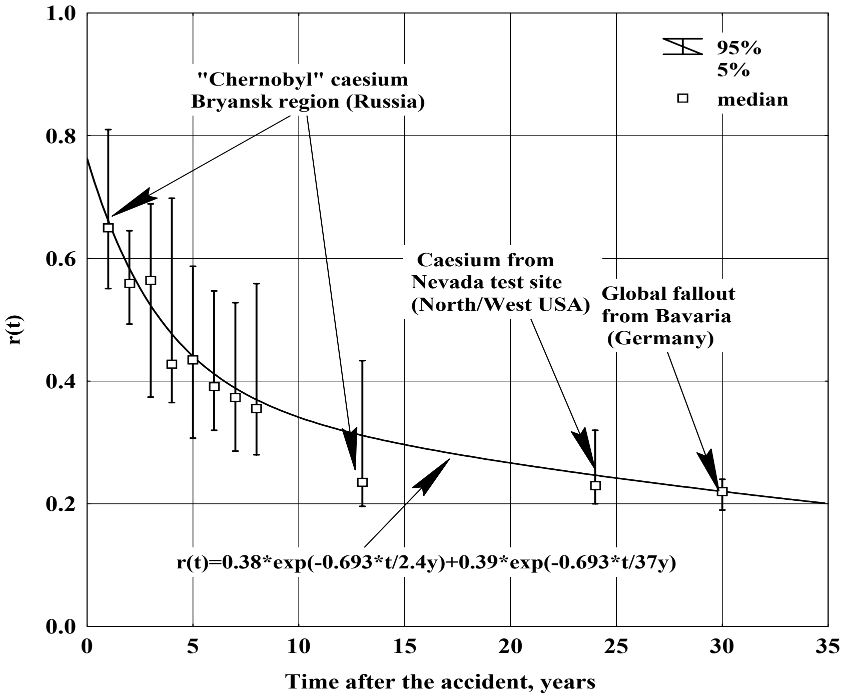


FIG. 4.7. Reduction of the ^{137}Cs-gamma-dose rate in air due to caesium migration in undisturbed soil relative to the dose rate caused by a plane source on the air -soil interface. From Golikov et al. (2002).

For a few years after the accident the dose rate over open plots of undisturbed soil decreased by a factor of 100 or more compared with the initial level (see Fig. 4.3). At that time, the dose rate was mainly determined by gamma radiation of caesium radionuclides, i.e., ^{137}Cs (half-life 30 y) and ^{134}Cs (half-life 2.1 y) and later on, one decade and more after the accident, mainly by the longer-lived ^{137}Cs. Long-term studies of external gamma-exposure rates during the past 17 years have shown that the external gamma-exposure rate is decreasing faster than that due to radioactive decay alone. Golikov et al. (2002) and Likhtarev et al. (2002) have calculated a reference function for ^{137}Cs-gamma-radiation-dose rate that has 40–50% of the exposure rate decreasing with an ecological half life of 1.5–2.5 years and the remaining 50–60% decreasing with an ecological half life of 40–50 years, as indicated in Fig. 4.7. The latter value is rather uncertain. It corresponds to less uncertain effective half life that takes into account both radioactive decay of ^{137}Cs and its gradual deepening in soil of 17-19 years.

4.2.2.2. Dynamics of external gamma-dose rate in anthropogenic areas

In settlements in urban and rural areas, the characteristics of the radiation field differ considerably from those over an open plot of undisturbed land, which is used as the reference site and starting point for calculation of external dose of people from deposited activity. These differences were attributable to varying source distributions as a result of deposition, run-off, weathering, and shielding. All such effects can be summarized within the term “location factors.”

Location factors for typical Western European buildings have been assessed by calculations (Meckbach et al. 1988; Meckbach and Jacob 1988; Hedeman Jensen 1985). Gamma spectrometric measurements performed in Germany and Sweden (Jacob et al. 1987; Karlberg 1987; Jacob and Meckbach 1990, 2000) have allowed the determination of location factors in urban environments and to trace their variation with time for several years after the Chernobyl accident. The characteristic feature, and advantage, of these investigations is that they began immediately after the accident, whereas systematic investigations of location factors in the contaminated areas of Russia, Ukraine and Belarus began 2–3 years after the accident. Results of one such later investigation in Novozybkov (Bryansk Oblast, Russia) are presented in Fig. 2.13 (Section 2).

4.2.2.3. *Behaviour of people in the radiation field*

The influence of the behaviour of different social population groups on the level of exposure can be taken into account, if the frequency with which people of the k-th population group remain at the location of the i-th type is known. The times spent in various types of location (indoor, outdoors on streets or in yards, etc.) by members of different population groups has been assessed on the basis of responses to a questionnaire. Data collected included: age, sex, occupation, information about dwelling, etc. An example of the results is shown in Table 4.3, where values of occupancy factors for the summer period are presented for different groups of the rural populations of Russia, Ukraine and Belarus (Jacob and Likhtarev 1996).

4.2.2.4. *Effective dose per unit gamma dose in air*

Mean values of conversion factors, CF_k, which convert the gamma-dose rate in air to the effective dose rate in a member of population (age) group k were obtained for the three groups of population by use of phantom experiments (Jacob and Likhtarev 1996) and Monte-Carlo calculations (Jacob et al. 1990). The values were 0.75 Sv Gy^{-1} for adults, 0.80 Sv Gy^{-1} for school children (7–17 years) and 0.90 Sv Gy^{-1} for pre-school children (0–7 years). For the calculation of effective doses, conversion factors, CF_k, were used that are independent of location and time after the accident.

4.2.3. Results

4.2.3.1. *Dynamics of external effective dose*

Shortly after the deposition of fallout the gamma-radiation field was dominated by emissions from short-lived radionuclides, as discussed above (see Figs. 4.4 and 4.5). And the mixtures at different locations varied widely. Thus, the radionuclide composition of the deposited activity was a major factor in determining the external exposure of the population during the early period of time after the accident.

TABLE 4.3. VALUES OF OCCUPANCY FACTORS FOR THE SUMMER PERIOD FOR DIFFERENT GROUPS OF THE RURAL POPULATIONS OF RUSSIA, UKRAINE AND BELARUS. THE FIRST NUMBER CORRESPONDS TO DATA FOR RUSSIA, THE SECOND FOR BELARUS, AND THE THIRD FOR UKRAINE. FROM JACOB AND LIKHTAREV (1996)

Location	Indoor workers	Outdoor workers	Pensioners	School children	Preschool children
Inside houses	0.65/0.77/0.56	0.50/0.40/0.46	0.56/0.44/0.54	0.57/0.44/0.75	0.64/-/0.81
Outside houses (living area)	0.32/0.19/0.40	0.27/0.25/0.29	0.40/0.42/0.41	0.39/0.45/0.21	0.36/-/0.19
Outside settlement	0.03/0.04/0.04	0.23/0.35/0.25	0.04/0.14/0.05	0.04/0.11/0.04	0/-/0

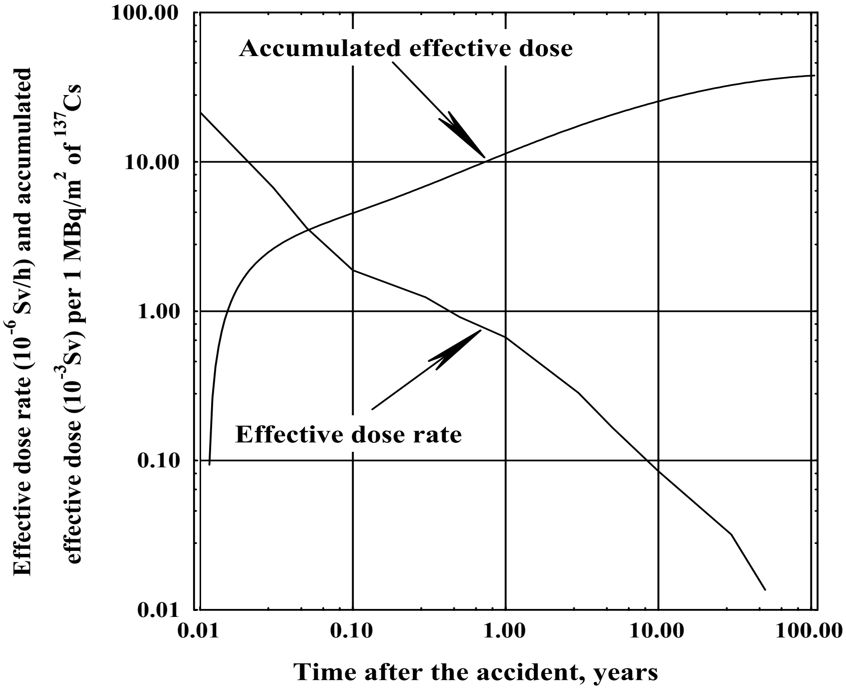


FIG. 4.8. Model prediction of time dependence of external effective gamma-dose rate and accumulated external effective dose for the urban population of Bryansk Oblast in Russia (Golikov et al. 2002).

Another relevant parameter in the mid-term period is the dependence of location factors on time, due to the relatively fast migration processes of radionuclides during this period. The dose rate over different urban surfaces caused by gamma radiation of ^{137}Cs was decreasing during the first years after deposition with an exponential half-time of 1 to 2 years – (see Fig. 2.13 (Golikov et al. 2002). In 5–7 years after deposition the change in dose rate with time had stabilized—this was due to the decay of the short-lived radionuclides and the fixation of caesium radionuclides within the soil column.

According to measurements and evaluations within the first year after the accident the external dose rate had decreased by a factor of approximately thirty, mainly due to radioactive decay of short-lived radionuclides (see Fig. 4.8). During the following decade the external dose rate decreased because of the radioactive decay of ^{134}Cs+^{137}Cs and the migration of radiocaesium into the soil. Afterwards the external dose rate was mainly due to ^{137}Cs. After such a long time radiocaesium is fixed within the soil matrix, and this results in a slow migration into the soil and correspondingly in a slow decrease of the external dose rate. On the basis of such measurements it is predicted that, of the total external dose to be accumulated during 70 years following the accident, about 30% was accumulated during the first year and about 70% during the first 15 years (Fig. 4.8) (Golikov et al. 2002).

4.2.3.2. Measurement of individual external dose by thermoluminescence dosimeters

As a rule, before the Chernobyl accident individual external doses were measured only for occupational exposures. After the Chernobyl accident, individual external doses of members the population were also measured. For this purpose thermoluminescent (TL) dosimeters were distributed to the inhabitants of the more contaminated areas of Russia, Ukraine and Belarus (Erkin and Lebedev 1993; Skryabin et al. 1995; Likhtarev et al. 1996; Chumak et al. 1999; Golikov et al. 1999). Inhabitants wore TL dosimeters for about one month in the spring/summer periods. Examples of such results are presented in Figs. 4.9 and 4.10 for rural and urban areas, respectively. According to these results it can be concluded that the urban population has been exposed to a lower dose by a factor of 1.5–2 compared to the rural population living in areas with similar levels of radioactive contamination. This arises because of the better shielding features of urban buildings and different occupational habits.

The critical group in relation to external irradiation is composed of individuals in an occupation or with habits that result in spending a significant amount of time outside in areas of undisturbed soil, in forests or meadows, and who also live in houses with the least protective properties. Presently, the average external dose for any population group does not exceed the average dose in a settlement by more than a factor of two. Typical critical groups are foresters (factor: 1.7), herdsmen (factor: 1.6) and field-crop workers (factor: 1.3) living in one-storey wooden houses (Jacob and Likhtarev 1996; Balonov et al. 1996).

Analysis of results of measurements in inhabitants of settlements showed that the distribution of individual doses can be described by a log-normal function (Golikov et al. 2002). Fig. 4.11 presents a comparison of model calculations with individual TL measurements performed in 1993 in four villages of the Bryansk Oblast (565 measurements). The distributions of the ratio of individual external doses to the mean value of measured doses in each of the villages is almost identical. Thus, the resulting log-normal distribution with a geometric standard deviation of about 1.5 (attributed mainly to the stochastic variability of individual doses) may be assumed to be typical for rural settlements in the zone of the Chernobyl accident.

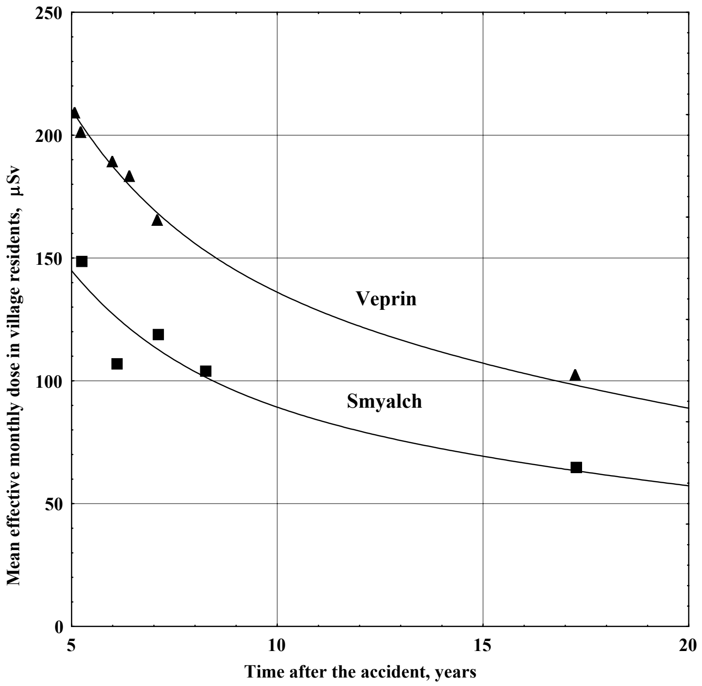


FIG. 4.9. Results of TL measurements of mean monthly doses among inhabitants living in wooden houses of the villages Veprin and Smyalch (Bryansk Oblast, Russia) in different time periods following deposition (Golikov et al. 1999).

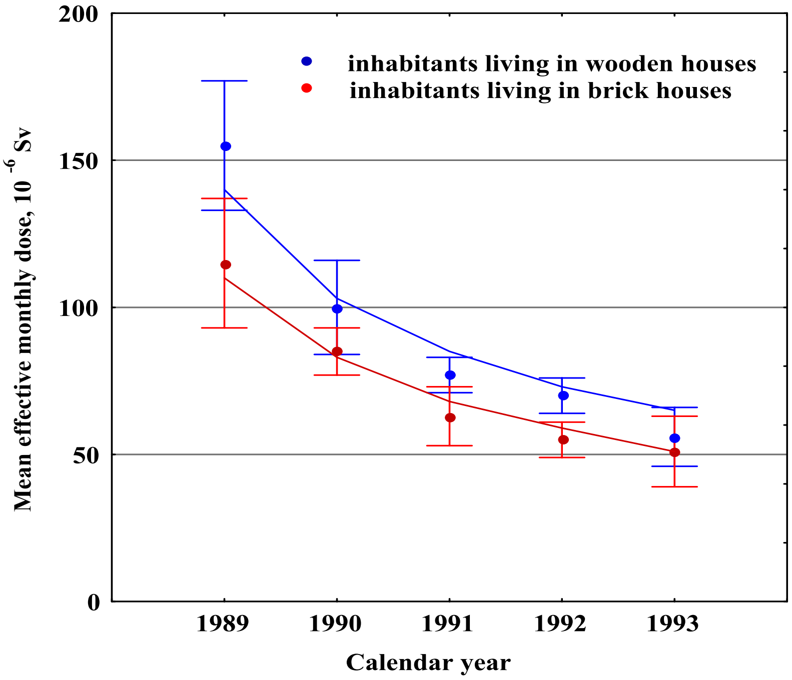


FIG. 4.10. Effective dose rates for indoor workers in Novozybkov (Russia). The points with error bars represent average values and 95% confidence intervals (+/- two standard errors) of TL measurements (Golikov et al. 1999).

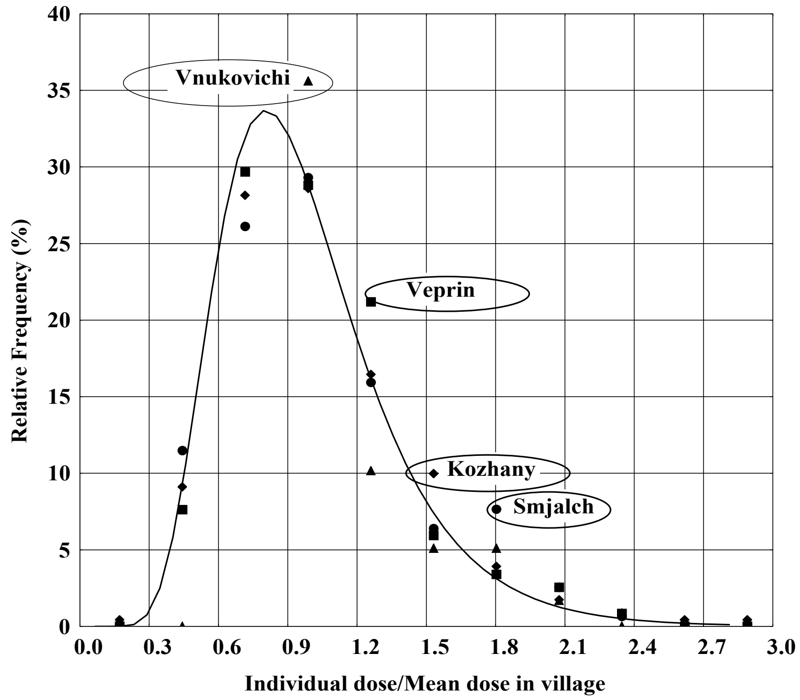


FIG. 4.11. Frequency distributions of monthly effective external doses of individual persons as measured in summer 1993 with TL dosimeters in four villages of the Bryansk Oblast (points) and calculated by the stochastic model (curve). Doses are normalized to the arithmetic mean of the individual doses determined for each of the villages. From Golikov et al. (2002).

4.2.3.3. Levels of external exposure

To illustrate actual levels of external exposure and differences in level of exposure among various population groups, Table 4.4 presents calculated values of effective external doses during different time intervals for rural and urban populations in Russia and Ukraine, and Table 4.5 presents the ratio of average effective doses in separate population groups to the mean dose in a settlement. Calculations of dose for different time intervals were performed on the basis of the model described above for the assessment of external dose in a population.

Presently, the average annual external dose to residents of a rural settlement with substantial ^{137}Cs-soil deposition equal at the present time to ~700 kBq·m^{-2} (~20 Ci·km^{-2}) is 0.9 mSv. For the critical group, the dose value exceeds the annual dose limit of 1 mSv set for the population for normal conditions of operation of ionizing radiation sources. The external dose due to Chernobyl deposition accumulated by the present time is 70% to 75% of the total lifetime dose (70 years) for persons born in 1986 and living all the time in contaminated areas.

TABLE 4.4. AVERAGE EFFECTIVE EXTERNAL DOSE OF THE ADULT POPULATION IN THE INTERMEDIATE (100 KM <DISTANCE <1000 KM) ZONE OF CHERNOBYL CONTAMINATION

Country	Population	E/σ_{137}, μSv per kBq·m^{-2} of ^{137}Cs*				
		1986	1987–1995	1996–2005	2006–2056	1986–2056
Russia (Golikov et al. 1999; 2002)	Rural	14	25	10	19	68
	Urban	9	14	5	9	37
Ukraine (Likhtarev et al. 2002)	Rural	24	36	13	14	88
	Urban	17	25	9	10	61

* σ_{137} is given as for 1986.

TABLE 4.5. RATIO OF AVERAGE EFFECTIVE EXTERNAL DOSES IN SEPARATE POPULATION GROUPS TO THE MEAN DOSE IN A SETTLEMENT (BALONOV ET AL. 1996)

Type of dwelling	Indoor workers	Outdoor workers	Herders, foresters	School children
Wooden	0.8	1.2	1.7	0.8
1-2 storey, brick	0.7	1.0	1.5	0.9
Multi-storey	0.6	0.8	1.3	0.7

4.3. Internal dose

4.3.1. Model for internal dose

The general form of models used to calculate internal doses is shown in Fig. 4.12 (Balonov et al. 1996). The main pathways of radionuclide intake into the body of a person of age k are inhalation with average inhalation rate, IR_k, of air with time-dependent concentration, AC_r, of radionuclide r and ingestion of the set of f food products and water with consumption rates, CR_{fk}, with time-dependent specific activity, SA_{fr}.

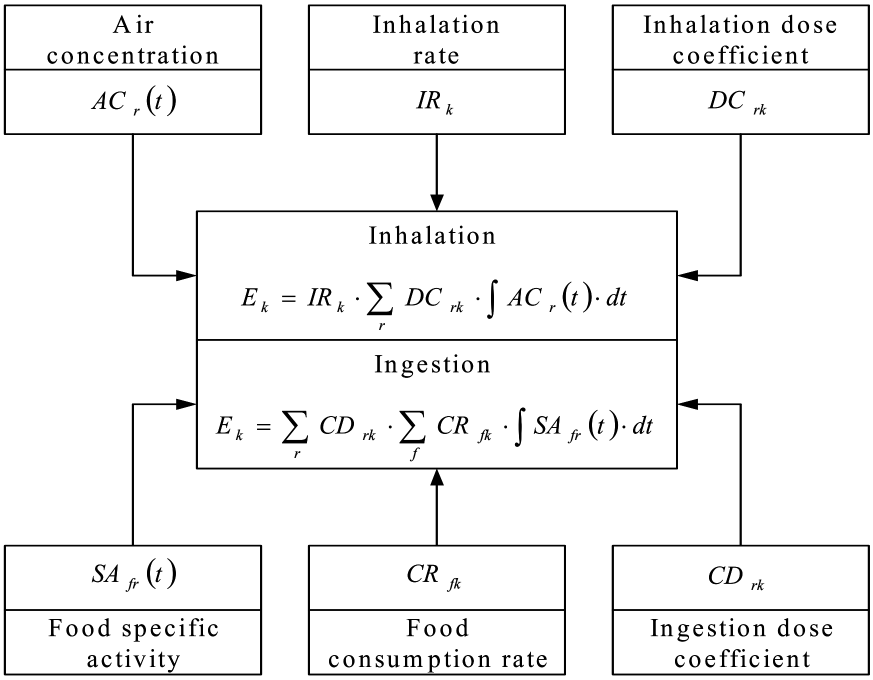


FIG. 4.12. Model for calculation of internal exposure for persons exposed to Chernobyl fallout (after Balonov et al. 1996).

Data on air concentrations and food-activity concentrations have been discussed in previous sections and will be summarized briefly below. Data on food-consumption rates are taken from the literature (UNSCEAR 1988; Likhtarev et al. 2000) or from special surveys of the affected populations (Bruk et al. 1999; Travnikova et al. 2004). Other data needed for dosimetric calculations are taken from publications of the International Commission on Radiological Protection for age-specific inhalation rates (ICRP 1994), and for age-specific dose coefficients (ICRP 2001). The latter values are for both inhalation and ingestion and give the dose per unit radionuclide inhaled or ingested. These values are calculated in terms of *committed dose;* that is, the dose that will be received over the next 50 years for adults or until age 70 for younger persons. For most radionuclides, but not for ^{90}Sr or ^{239}Pu, the biological residence time within the body is short, and the committed dose is only slightly larger than the dose accrued over the course of one year. Strontium and plutonium nuclides, and a few others, are metabolized slowly, and the full committed dose is not actually received for many years.

Another method of calculating internal dose is to use direct measurements of the radionuclide of interest in the human body. This was done for ^{131}I in the human thyroid in the three most affected countries (Likhtarev et al. 1993; Zvonova and Balonov 1993; Gavrilin et al. 1999) and for ^{137}Cs (e.g., Balonov et al. 1995; Likhtarev et al. 2000). Especially for the thyroid, direct measurements are not sufficient information to calculate doses, and such information must be supplemented by suitable intake models to determine the past and future concentration of the radionuclide in the body and its organs.

Predictions of future intakes of long-lived radionuclides in the body must be made to be able to predict future doses. Information on the long-term transfer of the important radionuclide, ^{137}Cs, from the environment to the human body can be made on the basis of experience with this radionuclide in global and local fallout (UNSCEAR 2000). Also, enough time has now passed since the Chernobyl accident so that measurements specific to Chernobyl can be used to predict the future course of concentration of ^{137}Cs in foods and the human body. For example, Likhtarev et al. (2000) on the basis of 126,000 samples of milk collected during 1987–1997 observed a two-component exponential loss curve with 90% of ^{137}Cs activity disappearing according to a half life of 2.9±0.3 year and 10% with 15±7.6 years. The second value is very uncertain due to the short time of observation compared to the radiological half life of ^{137}Cs of 30 years. These data are in general agreement with those observed in Russia (Bruk et al. 1998; Fesenko et al. 2004).

4.3.2. Monitoring data as input for assessment of internal dose

A unique feature of Chernobyl-related monitoring of human internal exposure was extensive application of whole-body measurements of radionuclide content in the human body and its organs (mainly, thyroid); these measurements had been performed along with regular measurements of radionuclides in food, drinking-water samples, and other components of the environment. This combination of various kinds of monitoring data allowed substantial improvement in the precision of reconstruction of internal dose.

To assess internal dose from inhalation, the air-concentration measurements described in a previous Section have been used. The most important aspect was assessment of dose for the first days after the accident, when concentration of radionuclides in air had been relatively high. Later on, the assessment of doses via inhalation has been needed predominantly for the resuspension of radionuclides with low mobility in the food chain, such as plutonium.

Assessment of radionuclide intake with food and drinking water was primarily based on numerous measurements of ^{131}I, $^{134,137}Cs$ and ^{90}Sr, which have been performed all over Europe and especially in the three most affected countries (Belarus, Russia and Ukraine). Gamma spectroscopy for ^{131}I and $^{134,137}Cs$ and radiochemical analyses for ^{90}Sr have been the main types of measurements. In some laboratories beta spectroscopy has been successfully applied to determine different radionuclides in samples; when radionuclide composition was well known, total-beta-activity measurements were also implemented. In most of the measurements, ^{137}Cs in raw animal products (milk, meat, etc.) has been determined; the number of these measurements performed since 1986 and available for dose estimations comprises a few million. Generic data of radionuclide measurements in food are presented in Sections pertaining to the terrestrial environment.

Activity concentration of soluble radionuclides (mainly ^{131}I, $^{134,137}Cs$, ^{90}Sr) in drinking water had been extensively determined in 1986 both in surface and underground sources (see Section 2.5). Later on, these activity concentrations were relatively low, and their contributions to internal dose were usually negligible compared with that associated with the intake of food.

In May–June 1986, both special and non-specific radiological equipment was widely applied to measure ^{131}I activities in thyroids of residents of areas with substantial radionuclide deposition. In total, more than 300,000 ^{131}I measurements in the thyroid were performed in the three more affected countries, and a substantial number of measurements were also performed in other European countries. Special attention was paid to measurements of children and adolescents. After careful calibration had been ensured, data on large-scale measurements have been used as the main basis for the reconstruction of thyroid dose.

Most of numerous whole-body measurements performed since 1986 in different European countries have been aimed at the determination of $^{134,137}Cs$. The number of measurements exceeded one million, most of them performed in the three more affected countries. The measurement data have been widely used both for model validation concerning radionuclide intake and evaluation of the effectiveness of countermeasures. In the more contaminated regions of Belarus, Russia and Ukraine, whole-body measurement data have been used to determine more precise estimates of human doses both for radiation-protection purposes and as part of epidemiological studies.

Strontium-90 and plutonium radionuclides, which do not emit gamma radiation easily detectable by whole-body counters, have been measured in excreta samples, or, since the 1990s, in samples taken at autopsy. Several hundred samples of human-bone tissue were analysed by radiochemical methods for $^{90}Sr/^{90}Y$ content. Activities of plutonium radionuclides have been successfully determined in several tens of samples of human lungs, liver and bones (Ivanova et al. 1995; Kutkov et al. 1995).

Reduced monitoring programmes for radiation-protection purposes and specifically for justification of remediation efforts are continuing in the affected areas.

4.3.3. Avoidance of dose by human behaviour

In addition to the countermeasures that were employed to reduce levels of contamination in urban environments and in agricultural foodstuffs, changes in human habits after the accident were also effective in reducing dose to residents of the contaminated areas. The most obvious and highly effective method immediately after the accident would have been to stop the consumption of milk to reduce the intake of ^{131}I. The effectiveness of this is not well documented, and it is only in some of the more affected regions that the residents of the three countries were advised of this option in a timely manner.

The longer term option of reducing the consumption of food products known to be more highly contaminated by $^{134,137}Cs$ appears to have been more successful, at least during 1987–1993 (Balonov and Travnikova 1990; Likhtarev et al. 2000). Such foods were typically locally produced milk and beef or of the “wild” variety, including game meat, mushrooms, and berries. Later, due to deteriorating economic conditions and the loss of fear, such self-imposed restrictions became less widespread.

4.3.4. Results for doses to individuals

4.3.4.1. Thyroid doses due to radioiodines

One of the major impacts of the accident was exposure of the human thyroid. Doses were accumulated rather quickly due to the rapid transfer of iodine through the food chain and the short half life of ^{131}I of eight days; other radioiodines of interest in terms of thyroid dose also have short half lives. The importance of thyroid doses was recognized by national authorities throughout the world, and early efforts focused on this issue. Estimates of country-average individual thyroid doses to infants and adults have been provided by UNSCEAR in its early report (UNSCEAR 1988). Even more attention has been paid to thyroid-dose reconstruction since the early 1990s, when an increase of thyroid-cancer morbidity in children and adolescents residing in areas of Belarus, Ukraine and Russia contaminated with Chernobyl fallout was discovered (Thomas et al. 1999; UNSCEAR 2000; WHO 2005).

In association with radio-epidemiological studies the main patterns of thyroid-dose formation were clarified and published in the 1990s (Zvonova and Balonov 1993; Likhtarev et al. 1993; Gavrilin et al. 1999) and summarized in (UNSCEAR 2000). Nevertheless, important new work in that area have appeared recently (Zvonova et al. 2000; Kruk et al. 2004; Likhtarev et al. 2005). The general approach to internal dose reconstruction has been elaborated in (ICRU 2002).

Methodologies of thyroid-dose reconstruction for the Chernobyl-affected populations developed in parallel in Belarus, Russian and Ukraine with participation of the US and EU experts; these methodologies have a number of commonalities and some substantial distinctions that complicate their desired integration. Firstly, in all three countries there are available many tens of thousands of ^{131}I-thyroid measurements, although of different quality, that are used as the basic data for thyroid-dose reconstruction. In Russia, they additionally used data of ^{131}I in milk measurements. Due to immediate use of human and environmental ^{131}I measurements the reconstructed doses are rather realistic as opposed to conservative.

Another commonality is the use of several age groups living in one settlement or in a group of close settlements as a unit for mean thyroid-dose reconstruction. When there is a substantial number of human and environmental ^{131}I measurements available in a settlement, they are immediately used for dose reconstruction. The subsidiary quantity used for dose reconstruction in the settlements where historical ^{131}I measurements are not available are ^{137}Cs-soil deposition values as indicators of area radioactive contamination.

However, methodologies of thyroid-dose reconstruction for settlements without environmental or human-^{131}I measurements are substantially different in the three countries. In Ukraine, where most of the radioiodine was deposited in dry weather conditions, Likhtarev et al. developed a model with linear dependence of thyroid dose on ^{137}Cs-soil deposition (Likhtarev et al. 2005). In Belarus, where both dry and wet deposition of radioiodines occurred, a semi-empirical model based on non-linear dependence of thyroid dose on ^{137}Cs-soil deposition was developed by Gavrilin et al. (1999) and widely applied. In another recently published paper devoted to the same problem (Kruk et al. 2004), a comprehensive radioecological model of radioiodine environmental transfer was developed and successfully applied for thyroid-dose reconstruction. In Russia, where wet deposition of radioiodines had dominated, a linear semi-empirical model of dependence of ^{131}I-activity concentration in milk and of thyroid dose on ^{137}Cs-soil deposition more than 37 kBq m^{-2} was developed (Zvonova et al. 2000) and applied (Balonov and Zvonova 2002). Despite differences in the applied methodological approaches, general agreement among numerical results, except for areas of low contamination, is satisfactory (WHO 2005).

The thyroid doses resulting from the Chernobyl accident comprise four contributions: (1) the internal dose from intakes of ^{131}I; (2) the internal dose from intakes of short-lived radioiodines (^{132}I, ^{133}I, and ^{135}I) and of short-lived radiotelluriums (^{131}Te and ^{132}Te); (3) the external dose from the deposition of radionuclides on the ground and other materials; and (4) the internal dose from intakes of long-lived radionuclides such as ^{134}Cs and ^{137}Cs.

For most residents of the Chernobyl-affected areas, the internal thyroid dose resulting from intakes of ^{131}I is by far the most important and has received almost all of the attention. The dose from ^{131}I was mainly due to the consumption of fresh cow's milk and, to less extent, of green vegetables; children on average received a dose that was much greater than that received by adults, because of their small thyroid mass and a consumption rate of fresh cow's milk that was similar to that of adults.

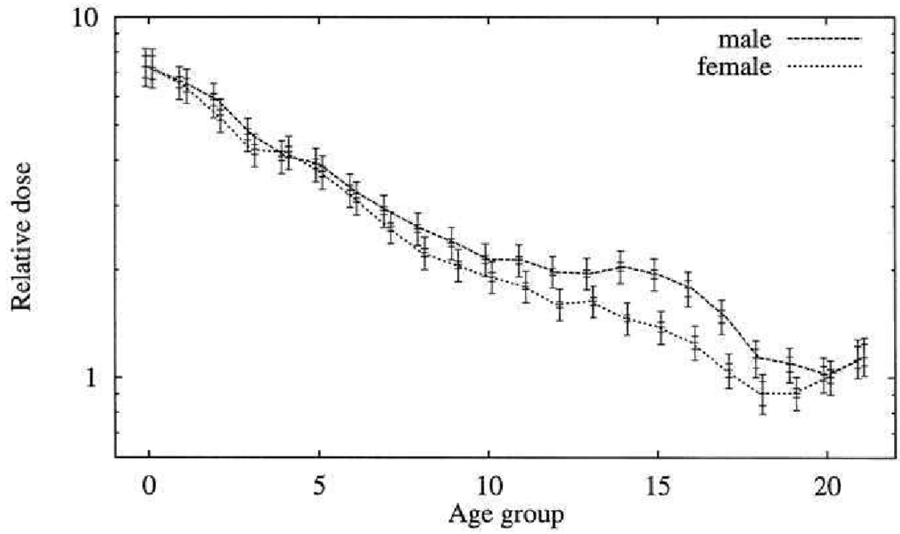


FIG.4.13. Age-sex dependence of the mean thyroid dose of inhabitants of a settlement standardised to the mean dose in adults from the same settlement (Heidenreich et al. 2001).

An example of age and gender dependence of the mean-thyroid dose of inhabitants of a settlement based on 60,000 measurements of ^{131}I in thyroid performed in Ukraine in May 1986 is presented in Fig. 4.13 (Heidenreich et al. 2001). Mean-thyroid dose in infants is by a factor of about 7 larger than in young adults (19-30 y) residing in the same rural or urban settlements; this ratio monotonically decreases with age as an exponential function with some deviation to the top in adolescents. Differences in age dependence between males and females seem to be insignificant. Similar patterns were revealed both from Belarusian and Russian series of measurements of ^{131}I in thyroid (Zvonova and Balonov 1993; Gavrilin et al. 1999).

As the rural population of contaminated areas depends more on local agricultural production than does the urban population, thyroid doses caused predominantly by consumption of contaminated milk and diary products are by a factor of about two higher in rural than in urban populations (UNSCEAR 2000).

Although the largest contribution to thyroid dose resulted from intakes of ^{131}I, it is also important to take into consideration the internal dose from short-lived radioiodines, i.e., ^{132}I, ^{133}I, and ^{135}I. Among members of the public, the highest relative contribution to the thyroid doses from short-lived radionuclides was expected among the residents of Pripyat. This cohort exposed to radioiodines via inhalation only was evacuated about 1.5 days after the accident. Analysis of the direct thyroid and lung spectrometric measurements performed on 65 Pripyat evacuees has specified that the contribution of short-lived radionuclides to the thyroid dose of Pripyat evacuees is about 20% for persons who did not employ stable iodine to block their thyroids and more than 50% for persons who took KI pills soon after the accident (Balonov et al. 2003). The total thyroid dose among the Pripyat evacuees, however, was relatively small compared with populations consuming contaminated food.

For populations permanently residing in contaminated areas, the contribution of short-lived radionuclides to thyroid dose was minor, as most of the thyroid exposure resulted from the week-long consumption of contaminated milk and other foodstuffs. During transport of radioiodines along food chains short-lived radioiodines decayed and the contribution of short-lived radioiodines is estimated to have been of the order of 1% of the ^{131}I-thyroid dose (Balonov et al. 2003; Gavrilin et al. 2004).

TABLE 4.6. DISTRIBUTION OF INDIVIDUAL THYROID DOSES FOR AGE GROUPS OF CHILDREN AND ADOLESCENTS FROM KYIV, ZHYTOMYR AND CHERNIGOV OBLASTS OF THE UKRAINE WITH ^{131}I IN THYROID MEASUREMENTS (LIKHTAREV ET AL. 2005)

Category and age group	Number of measurements	Percent of children with thyroid dose (Gy) in interval				
		≤ 0.2	>0.2–1	>1–5	>5–10	>10
Settlements not evacuated						
Rural areas						
1–4 years	9119	40	43	15	1.7	0.9
5–9 years	13 460	62	31	6.5	0.44	0.07
10–18 years	26 904	73	23	3.7	0.16	<0.01
Urban areas						
1–4 years	5147	58	33	7.5	1.0	0.7
5–9 years	11 421	82	15	2.6	0.23	0.04
10–18 years	24 442	91	7.7	1.4	0.12	<0.01
Evacuated settlements						
1–4 years	1475	30	45	22	2.7	1.0
5–9 years	2432	55	36	8.4	0.6	0.08
10–18 years	4732	73	23	3.6	0.13	0.02

The variety of individual-thyroid doses is illustrated by Table 4.6 relevant to children and adolescents residing in Northern regions of Ukraine, i.e., Kyiv, Zhytomir and Chernigov regions, mostly affected by radiation after the Chernobyl accident (Likhtarev et al. 2005). Dose distributions presented in Table 4.4 are based on about 100,000 human-thyroid measurements. The range of thyroid dose in all groups is wide, between less than 0.2 Gy and more than 10 Gy. The latter dose group includes about 1% of younger children, less than 0.1% of children of 5-9 y, and less than 0.01% of adolescents. Doses in adults are lower by a factor of about 1.5 than those of adolescents, see Fig 4.13. In all age groups presented in Table 4.4 and especially in the younger ones, doses were high enough to cause both short-term functional thyroid changes and remote thyroid-cancer effects in some individuals (Thomas et al. 1999; UNSCEAR 2000; WHO 2005).

Similar data for Belarus and Russia are available (Gavrilin et al. 1999; Balonov and Zvonova 2002). Substantially more detail on the calculation of thyroid doses to individuals is provided in the Section “Dosimetry”of the Chernobyl Forum report on health effects (WHO 2005).

Generally it can be stated that adequate methodologies for thyroid-dose reconstruction of people who resided in Spring 1986 in the contaminated areas of Belarus, Russia and Ukraine have been developed and published. These estimates of both individual and collective doses are widely used by research scientists and national health authorities both in forecasts of thyroid morbidity and in radio-epidemiological studies.

4.3.4.2. Long-term internal doses from terrestrial pathways

Inhabitants of areas contaminated with radionuclides in 1986 are still experiencing internal exposure due to consumption of local foodstuffs containing ^{137}Cs and, to a lesser extent, ^{90}Sr. According to model estimates and direct human measurements (Ivanova et al. 1995), inhalation of plutonium radionuclides and ^{241}Am does not significantly contribute to human doses.

TABLE 4.7. RECONSTRUCTION AND PROGNOSIS OF THE AVERAGE EFFECTIVE INTERNAL DOSE OF THE ADULT RURAL POPULATION IN THE INTERMEDIATE (100 KM <DISTANCE <1000 KM) ZONE OF CHERNOBYL CONTAMINATION

Country, reference	Soil type	E/σ_{137}, μSv per kBq·m^{-2} of Cs-137*				
		1986	1987–1995	1996–2005	2006–2056	1986–2056
Russia (Balonov et al. 1996)	Soddy-podzolic sandy	90	60	12	16	180
	Black	10	5	1	1	17
Ukraine (Jacob and Likhtarev 1996; Likhtarev et al. 2000)	Peat-bog	19	167	32	31	249
	Sandy	19	28	5	5	57
	Clay	19	17	3	3	42
	Black	19	6	1	1	27

* σ_{137} is given as for 1986.

Generic dose-conversion parameters have been developed to reconstruct broadly the past, assess the current, and forecast the future average effective internal dose. Examples for the adult rural population of a settlement located in the intermediate (100 km < Distance < 1000 km) zone of contamination based on experimental data and models developed in Russia and Ukraine are given in Table 4.7 (Balonov et al. 1996; Jacob and Likhtarev 1996; Likhtarev et al. 2000). Values for each indicated time period are given separately for various soil types as the ratios of the mean internal dose (E) to the mean ^{137}Cs-soil deposition in a settlement as of 1986 (σ_{137}) (μSv per kBq·m^{-2}).

In a series of experimental whole body measurements and associated annual internal dose calculations it was stated that long term doses caused by ingestion of food containing caesium radionuclides of children are usually lower by a factor of about 1.1 to 1.5 than these in adults and adolescents, see e.g. (IAC 1991; Shutov et al. 1993).

The mean internal doses to residents of rural settlements strongly depend on soil properties. For assessment purposes, soils are classified into three major soil types: (1) black or chernozem soil, (2) podzol soil (including both podzol sandy and podzol loamy soils), and (3) peat-bog or peat soil. In accordance with the environmental behaviour of ^{137}Cs, internal exposure exceeds external dose in areas with peaty soils. Contributions due to internal and external exposure are comparable in areas with light sandy soils, and the contribution of internal exposure to the total (external and internal) dose does not exceed 10% in areas with dominantly black soils. According to numerous studies, the contribution of ^{90}Sr to the internal dose regardless of natural conditions is usually less than 5%.

The parameters obtained from independent sets of Russian and Ukrainian data significantly differ for some soil types and time periods (see Table 4.7). Some of these discrepancies can be explained by different meteorological conditions (mainly dry deposition in Ukraine and wet deposition in Russia) that occurred in different parts of the Chernobyl affected areas and different food-consumption habits.

Multiplication of the parameters presented in Table 4.7 by the mean ^{137}Cs-soil deposition (as of 1986) gives an estimate of the internal effective dose caused by radiation from ^{137}Cs and ^{134}Cs (for Russia, also from ^{90}Sr and ^{89}Sr) but not for radioiodines. Dose estimates are given for the assumption that countermeasures against internal exposure were not applied. In broad terms, the more important factors controlling internal dose to the rural population are the dominant soil type and the amount of ^{137}Cs deposition.

In towns and cities, internal dose is partially determined by radioactive contamination of foodstuffs produced in surrounding districts. However, importation of foodstuffs from non-contaminated areas has significantly reduced the intake of radionuclides, and internal doses received by urban populations are typically a factor of two to three less than in rural settlements with an equal level of radioactive contamination.

The deviation in dose to critical groups compared to settlement-average values varies by a factor of about three for internal exposure. The group mostly subjected to internal exposure from ^{137}Cs are adults consuming both locally produced agricultural animal food (e.g., milk, dairy products, etc.) and natural food (e.g., mushrooms, lake fish, berries, etc.) in amounts exceeding average-consumption rates.

At present, inhabitants of areas of low contamination (less than 0.04 MBq m^{-2} of ^{137}Cs) are receiving from ingestion of local food up to 0.004 mSv per year in black soil areas, up to 0.04 mSv per year in sandy soil areas and about 0.1 mSv per year in villages located in peaty soil areas. In the period of 2002–2056, they will receive an additional internal dose of less than 0.1 mSv in black soil areas, up to 0.7 mSv in sandy soil areas and about 1 to 2 mSv in villages located in peaty soil areas.

To avoid presentation of dosimetric data on a site-by-site basis, mean effective doses to adult residents of rural and urban localities have been determined as a function of soil-^{137}Cs deposition and predominant soil type; such data are given in Tables 4.8 and 4.9. The ^{137}Cs-soil deposition is sub-divided into two ranges: 0.04 to 0.6 MBq m^{-2} (1 to 15 Ci km^{-2}) and above 0.6 MBq m^{-2}, i.e., which were 0.6 to 4 MBq m^{-2} (15 to 100 Ci km^{-2}) in 1986. The level 0.04 MBq m^{-2} is considered as a conventional border between "non-contaminated" and "contaminated" areas. In areas contaminated with ^{137}Cs above 0.6 MBq m^{-2}, application of active countermeasures, i.e., agricultural restrictions, decontamination measures, recommendations to restrict consumption of locally gathered natural food (forest mushrooms and berries, lake fish, etc.), has been mandatory.

The dosimetric models presented in Table 4.8 predict that, by 2001, the residents had already received at least 75% of their lifetime internal dose due to ^{137}Cs, ^{134}Cs, ^{90}Sr and ^{89}Sr. During the coming years (2001–2056), they will receive the remaining 25%, i.e., less than 1 mSv for black soil, up to 7 mSv for podzol soils and up to 30 mSv for peat soil. In the more contaminated podsol soil areas, an effective dose up to 50 mSv can still be expected.

As can be seen from Table 4.9, the more elevated internal doses in some of the settlements are above the national action level of 1 mSv per year. For some population groups in contaminated areas, wild food (forest mushrooms, game, forest berries, fish) can make an important contribution to dose (Balonov et al. 1996, Jacob and Likhtarev 1996, Travnikova et al. 2004). Bruk et al. (1999) studied ^{137}Cs intake of the rural population in Bryansk Oblast, Russia, and found that natural foods contributed about 20% of total uptake in 1987, but up to 80% in 1994–1999. The relative contribution of wild food to internal dose has risen gradually because of substantial reduction of radionuclide content in agricultural food derived from vegetables and animals, while contamination of wild food has decreased much more slowly. In the latter period, the higher contributions to ^{137}Cs intake (and, by inference, internal dose) were forest mushrooms followed by forest berries, game and lake fish.

TABLE 4.8. PAST (1986–2000) AND FUTURE (2001–2056) MEAN CHERNOBYL-RELATED EFFECTIVE INTERNAL DOSES (mSv) OF ADULT RESIDENTS OF AREAS WITH Cs-137 SOIL DEPOSITION ABOVE 0.04 MBq m^{-2} (1 Ci km^{-2}) IN 1986 (IAEA 2005)

Population	^{137}Cs in soil, MBq m^{-2}	Soil type/Time period					
		Black		Podzol		Peat	
		1986–2000	2001–2056	1986–2000	2001–2056	1986–2000	2001–2056
Rural	0.04–0.6	1–10	0.1–1	3–30	0.5–7	8–100	2–30
	0.6–4	–		30–100	7–50	–	
Urban	0.04–0.6	1–8	0.1–0.6	2–20	0.3–5	6–80	1–20

TABLE 4.9. ANNUAL (2001) MEAN CHERNOBYL-RELATED EFFECTIVE INTERNAL DOSES (mSv) OF ADULT RESIDENTS OF AREAS WITH ^{137}Cs-SOIL DEPOSITION ABOVE 0.04 MBq m^{-2} (1 Ci km^{-2}) IN 1986 (IAEA 2005)

Population	^{137}Cs in soil, MBq m^{-2}	Soil type		
		Black	Podzol	Peat
Rural	0.04–0.6	0.004–0.06	0.03–0.4	0.1–2
	0.6–4	–	0.4–2	–
Urban	0.04–0.6	0.003–0.04	0.02–0.2	0.1–1

Travnikova et al. (2004) found similar trends in a study in 1996 of residents of Kozhany Village (Bryansk Oblast) located on the coast of a highly contaminated lake where natural foods contributed as average from 50 to 80% of ^{137}Cs intake. Men were more likely to eat natural foods than women, and there was a positive correlation between consumption of mushrooms and fish that indicates a liking of many inhabitants for “gifts of nature.” The average annual internal dose due to ^{137}Cs was estimated to be 1.2 mSv for men and 0.7 mSv for women in 1996.

4.3.4.3. Long-term doses from aquatic pathways

Human exposure via the aquatic pathway occurs as a result of consumption of drinking water, fish, and agricultural products grown using irrigation water from contaminated water bodies. Use of water bodies as a source of drinking water for livestock and flooding of agricultural land can also lead to human exposure via terrestrial pathways.

In the middle and lower areas of the Dnieper River catchment, which were not significantly subjected to direct radionuclide contamination in 1986, a significant proportion (10–20%) of the Chernobyl exposures were attributed to aquatic pathways. Although these doses were estimated to be very low, there was an inadequate understanding of the real risks of using water from contaminated aquatic systems. This created an (unexpected) stress in the population concerning the safety of the water system. In areas close to Chernobyl, radiation exposures via the aquatic pathway are much higher, but are again minor in comparison with terrestrial pathways.

Three pathways of exposure due to aquatic systems need to be considered:

— **Consumption of drinking water from rivers, lakes, reservoirs and wells in the contaminated areas:** The more significant exposures via consumption of drinking water resulted from use of water from the Dnieper Basin, and, in particular, the reservoirs of the Dnieper River system. The Dnieper cascade is a source of drinking water for more than 8 million people. The main consumers of drinking water from the

Dnieper River are in the Dnipropetrovsk and Donetsk Oblasts. In Kyiv, water from the Dnieper and Desna Rivers is used by about 750,000 people. The remaining part of the population use water mainly from groundwater sources.

— **Consumption of fish:** The Dnieper River reservoirs are used intensively for commercial fishing. The annual catch is more than 25,000 tonnes. There was no significant decrease in fishing from most of these reservoirs during the first decade after the accident. During the first 2–3 years, however, restrictions were placed on consumption of fish from the Kyiv Reservoir. In some smaller lakes, both in the former Soviet Union and in parts of Western Europe, fishing was prohibited during the first months and even years after the accident.

— **Consumption of agricultural products grown on lands irrigated by water from the Dnieper reservoirs:** In the Dnieper Basin there are more than 1.8 million ha of irrigated agricultural land. Almost 72% of this territory is irrigated with water from the Kakhovka Reservoir in the Dnieper River-Reservoir system. Accumulation of radionuclides in plants on irrigated fields can take place because of root uptake of radionuclides introduced with irrigation water and due to direct incorporation of radionuclides through leaves following sprinkler irrigation. However, recent studies show that, in the case of irrigated lands of southern Ukraine, radioactivity in irrigation water did not add significant radioactivity to crops in comparison with that which had been initially deposited in atmospheric fallout and subsequently taken up from the soil.

The contribution of aquatic pathways to the dietary intake of ^{137}Cs and ^{90}Sr is usually quite small, even in areas that were seriously affected by Chernobyl fallout. For the relatively large rural population, however, where people consume fish from local rivers and lakes, exposures could be significant. In addition, collective doses to the large urban and rural populations using water from the Pripyat-Dnieper River-Reservoir system were relatively high. Because of the high fallout within the catchment of the Pripyat and Dnieper Rivers, this system has been intensively monitored and doses via aquatic pathways have been estimated.

Contaminated rivers could potentially have led to significant doses in the first months after the accident through consumption of drinking water, mainly through contamination by short-lived radionuclides. The most significant individual dose was from ^{131}I and was estimated to be up to 0.5–1.0 mSv for the citizens of Kyiv during the first few weeks after the Chernobyl accident.

After the end of the first month following the accident, the main contributors to doses via aquatic pathways became ^{137}Cs and ^{90}Sr. Estimated doses due to these radionuclides in the Dnieper River-Reservoir system were made on the basis of monitoring data and predictions of flood frequencies. A worst case scenario of a series of high floods during the first decade after the accident (1986–1995) was assumed. Estimates were that individual doses via aquatic pathways would not have exceeded 1–5 μSv y^{-1}. Thus, long term doses via the drinking-water pathway were small in comparison with doses (mainly from short-lived radionuclides) in the early phase.

The contribution of different exposure pathways to dose is shown in Fig. 4.14 for the village of Svetilovichy in the Gomel Oblast of Belarus. In this case, consumption of freshwater fish forms an important part of the diet, and hence doses via this pathway can be significant for some individuals.

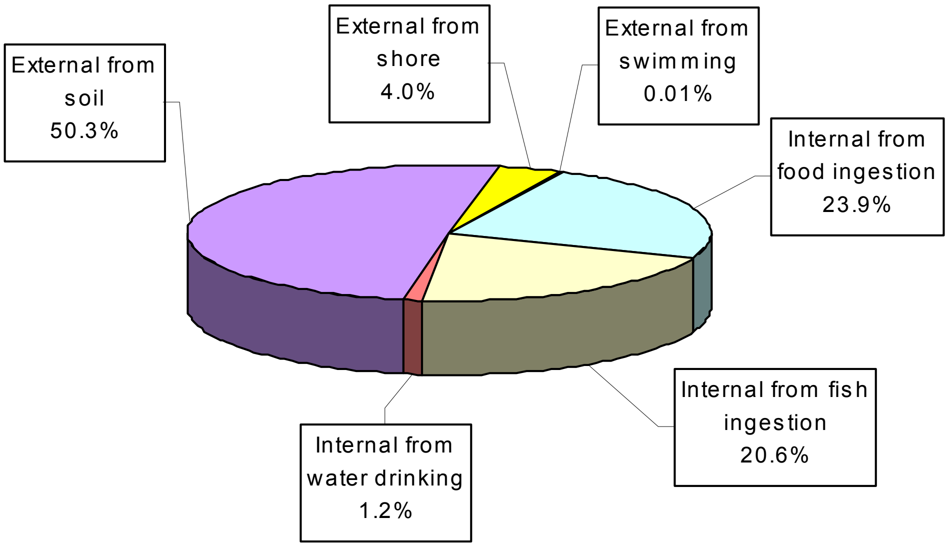


FIG. 4.14. Contributions of different pathways to effective dose to the critical group of the population for the settlement Svetilovichi, Gomel Oblast, Belarus. (Drozdovich et al. 2002; IAEA 2005).

4.4. Total (external and internal) exposure

The generalised data of both external and internal (not including dose to the thyroid) exposures of the general public presented above in Tables 4.4 and 4.9, respectively, have been summarised in Table 4.10 in order to estimate broadly the mean-individual total (external + internal) effective doses accumulated by residents of radioactively contaminated areas during 1986-2000 and to forecast doses for 2001-2056. Table 4.11 gives estimates of the annual total dose in 2001. In both tables data are given for levels of ^{137}Cs-soil deposition existing in 1986 in currently inhabited areas of Belarus, Russia and Ukraine, separately for rural and urban population and for different soil types without account for current countermeasures. Both accumulated and current annual total doses are presented for adults, as in the long term children generally receive lower external and internal doses from ^{137}Cs environmental contamination (in contrast to thyroid internal doses from radioiodine intake), because of their occupation (see Tables 4.3 and 4.5), food habits and metabolic features.

As can be seen from Table 4.10 both accumulated and predicted mean doses in settlement residents vary in the range of two orders of magnitude depending on radioactive contamination of the area, dominating soil type and settlement type. Thus, in 1986-2000 the dose range comprised from 2 mSv in towns located in black soil areas up to 300 mSv in villages located in areas with podzol sandy soils. According to the forecast, doses expected in 2001-2056 are substantially lower than already received ones, i.e., in the range of 1 to 100 mSv. In total, if countermeasures were not applied, the population of some more contaminated villages in Belarus and Russia would receive life-time effective doses up to 400 mSv, not including dose to the thyroid. However, intensive application of countermeasures such as settlement decontamination and agricultural countermeasures has reduced dose levels by a factor of about two. For comparison, a worldwide average lifetime dose from natural background radiation is about 170 mSv with a typical range of 70 to 700 mSv in various regions.

TABLE 4.10. PAST (1986-2000) AND FUTURE (2001-2056) MEAN CHERNOBYL-RELATED TOTAL EFFECTIVE DOSES (mSv) OF ADULT RESIDENTS OF AREAS WITH ^{137}Cs-SOIL DEPOSITION ABOVE 0.04 MBq m^{-2} (1 Ci km^{-2}) IN 1986 (IAEA 2005)

Population	^{137}Cs in soil, MBq m^{-2}	Soil type					
		Black		Podzol		Peat	
		1986–2000	2001–2056	1986–2000	2001–2056	1986–2000	2001–2056
Rural	0.04–0.6	3–40	1–14	5–60	1–20	10–150	3–40
	0.6–4	–		60–300	20–100	–	
Urban	0.04–0.6	2–30	1–9	4–40	1–13	8–100	2–20

TABLE 4.11. ANNUAL (2001) MEAN CHERNOBYL-RELATED TOTAL EFFECTIVE DOSES (mSv) OF ADULT RESIDENTS OF AREAS WITH ^{137}Cs-SOIL DEPOSITION ABOVE 0.04 MBq m^{-2} (1 Ci km^{-2}) IN 1986 (IAEA 2005)

Population	^{137}Cs in soil, MBq m^{-2}	Soil type		
		Black	Podzol	Peat
Rural	0.04–0.6	0.05–0.8	0.1–1	0.02–2
	0.6–4	–	1–5	–
Urban	0.04–0.6	0.03–0.4	0.05–0.6	0.1–1

With account for local demographic data (IAC 1991), ^{137}Cs soil deposition maps (see Section 2.1) and current level of countermeasure application (see Section 3), the vast majority of about five million population residing in contaminated areas of Belarus, Russia and Ukraine (see Table 2.2) currently, i.e., in early 2000s, receive annual effective dose less than 1 mSv (national action levels in three countries). For comparison, a worldwide average annual dose from natural background radiation is about 2.4 mSv with the typical range of 1 to 10 mSv in various regions (UNSCEAR 2000).

The number of residents of the contaminated areas in the three more affected countries that currently receive more than 1 mSv annually can be broadly estimated as about 100,000 persons. As future reduction of both external dose rate and radionuclide (mainly ^{137}Cs) activity concentrations in food is rather slow (see Sections 2.3 to 2.5 and 4.2, respectively), reduction of human exposure levels is expected to be slow as well, i.e., about 3 to 5% per year for current countermeasures.

4.5. Collective doses

4.5.1. Thyroid

A summary of the collective doses to the thyroid for the three more contaminated countries based on thyroid-dose reconstruction techniques described in Sub-Section 4.3.4.1 is shown in Table 4.12. The total is 1.6 million man-Gy with nearly half received by the group of persons exposed in Ukraine. The present estimation of the collective thyroid dose does not differ from that made in (UNSCEAR 2000).

TABLE 4.12. COLLECTIVE THYROID DOSES TO THE THREE COUNTRIES MOST CONTAMINATED BY THE CHERNOBYL ACCIDENT. FROM UNSCEAR (2000)

Country	Collective thyroid dose, man-Gy
Russian Federation	300,000
Belarus	550,000
Ukraine	740,000
Total	1,600,000

TABLE 4.13. ESTIMATED COLLECTIVE EFFECTIVE DOSES IN 1986–2005 TO THE POPULATIONS OF CONTAMINATED AREAS OF BELARUS, RUSSIA AND UKRAINE (^{137}Cs SOIL DEPOSITION IN 1986 MORE THAN 37 kBq m^{-2}; EXCLUDING THYROID DOSE). MODIFIED FROM UNSCEAR (2000), ANNEX J, TABLE 34, ACCORDING TO DOSIMETRIC MODELS PRESENTED ABOVE

Country	Population, million persons	Collective dose (thousand man-Sv)		
		External	Internal	Total
Belarus	1.9	11.9	6.8	18.7
Russia	2.0	10.5	6.0	16.5
Ukraine	1.3	7.6	9.2	16.8
Total	5.2	30	22	52

TABLE 4.14. COLLECTIVE DOSE COMMITMENT (CDC_{70}) CAUSED BY ^{90}Sr AND ^{137}Cs FLOWING FROM THE PRIPYAT RIVER (BERKOVSKY ET AL. 1996A,B)

Region	Population, (in millions of people)	^{90}Sr CDC_{70} (man-Sv)	^{137}Cs CDC_{70} (man-Sv)	Ratio ^{90}Sr CDC_{70} (^{137}Cs CDC_{70})$^{-1}$
Chernigov	1.4	4	2	2
Kyiv	4.5	290	190	1.5
Cherkassy	1.5	115	50	2.3
Kirovograd	1.2	140	40	3.5
Poltava	1.7	130	60	2.2
Dnepropetrovsk	3.8	610	75	8
Zaporojie	2	320	35	9
Nikolaev	1.3	150	20	8
Kharkov	3.2	60	4	15
Lugansk	2.9	15	1	15
Donetsk	5.3	330	20	17
Kherson	1.2	100	20	5
Crimea	2.5	175	5	35
Total	32.5	2500	500	5

4.5.2. Total (external + internal) dose from terrestrial pathways

Estimates of collective dose accumulated in 1986-2005 via the terrestrial pathways of external dose and ingestion of contaminated foods are given in Table 4.13 for the three countries of greatest interest. The total collective dose was estimated to be 43 thousand man-Sv in 1986–1995, including 24 thousand man-Sv from external exposure and 19 thousand man-Sv from internal exposure according to UNSCEAR (2000), Annex J, Table 34. According to models of exposure dynamics presented above (Golikov 2002; IAEA 2005), the estimated collective effective external doses in 1986–2005 are by a factor of about 1.2 and collective effective internal doses are by a factor of 1.1 to 1.5 (depending on soil properties and applied countermeasures) more than these obtained in 1986–1995. In total, collective dose increased by 9 thousand man-Sv or by 21% during the second decade compared with the first decade after the accident and reached 52 thousand man-Sv. This is in good agreement with the predictions made by the UNSCEAR (1988) in 1988.

The recent estimation of collective dose based both on human and environmental measurements implicitly accounts for substantial, but not specified, amounts of collective doses saved by the institution of countermeasures, that included evacuation, relocation, prohibition on the use of foodstuffs, and longer term remediation of contaminated areas.

4.5.3. Internal dose from aquatic pathways

The most important aquatic system (the Dnieper Basin) occupies a large area with a population of about 32 million people who use the water for drinking, fishing and irrigation. Estimates were made of the collective dose to people from these three pathways for a period of 70 years after the accident, i.e., from 1986 to 2056 (Zheleznyak et al. 1994, Berkovski et al 1996a). A long-term hydrological scenario was analysed using a computer model (Zheleznyak et al. 1992). Historical data were used to account for the natural variability in river flow. Dose-assessment studies were carried out to estimate the collective dose from the three pathways (Berkovski et al 1996a,b). The results of such calculations are given in Table 4.14.

Dose estimates for the Dnieper system show that if there had been no action to reduce radionuclide fluxes to the river, the collective dose commitment for the population of Ukraine (mainly due to radiocaesium and radiostrontium) could have reached 3000 man Sv. Protective measures (see Countermeasure Section 3), which were carried out during 1992–1993 on the left-bank flood plain of the Pripyat River decreased exposure by approximately 700 man Sv. Other protective measures on the right-bank in the Chernobyl Exclusion Zone (during 1999–2001) will further reduce collective doses by 200–300 man Sv (Voitsekhovich et al. 1996).

4.6. Conclusions and rcommendations

4.6.1. Conclusions

(1) The collective effective dose (not including dose to the thyroid) received by about five million residents living in the areas of Belarus, Russia and Ukraine contaminated from the Chernobyl accident (^{137}Cs deposition on soil of >37 kBq m^{-2}) was approximately 40,000 man-Sv during the period of 1986–1995. The groups of exposed persons within each country received an approximately equal collective dose. The additional amount of collective effective dose projected to be received during 1996– 2006 is about 9000 man Sv.

(2) The collective dose to the thyroid was nearly 2 million man-Gy with nearly half received by persons exposed in Ukraine.

(3) The main pathways leading to human exposure were external exposure from radionuclides deposited on the ground and by the ingestion of contaminated terrestrial food products. Inhalation and ingestion of drinking water, fish, and products contaminated with irrigation water were generally minor pathways.

(4) The range in thyroid doses in different settlements and in all age-gender groups is wide, between less than 0.1 Gy and more than 10 Gy. In some groups and especially in younger children, doses were high enough to cause both short-term functional thyroid changes and remote thyroid-cancer effects in some individuals.

(5) The internal thyroid dose from intake of ^{131}I was mainly due to the consumption of fresh cow's milk and, to less extent, of green vegetables; children on average received a dose that was much greater than that received by adults, because of their small thyroid mass and a consumption rate of fresh cow's milk that was similar to that of adults.

(6) For populations permanently residing in contaminated areas and exposed predominantly via ingestion, the contribution of short-lived radioiodines, i.e., ^{132}I, ^{133}I, and ^{135}I to thyroid dose was minor, i.e., about 1% of the ^{131}I thyroid dose. This is because during transport of radioiodines along food chains, the short-lived radioiodines decayed. The highest relative contribution (20 to 50%) to the thyroid doses of the public from short-lived radionuclides was received by the residents of Pripyat via the inhalation pathway only.

(7) According to both measurement and modelling data, the urban population has been exposed to a lower external dose by a factor of 1.5–2 compared to the rural population living in areas with similar levels of radioactive contamination. This arises because of the better shielding features of urban buildings and different occupational habits. As members of the urban population depend less on local agricultural products and wild food than persons residing in rural areas, both effective and thyroid internal doses caused predominantly by ingestion are by a factor of two to three lower in the urban population.

(8) The initial high rates of exposure declined rapidly due to the decay of short-lived radionuclides and to the movement of radiocaesiums into the soil profile. The latter caused a decrease in the rate of external dose due to increased shielding. In addition, as caesium moves into the soil column it binds to soil particles, and this reduced the availability of caesium to plants.

(9) The great majority of dose from the accident has already been accumulated.

(10) Persons who received effective doses (not including dose to the thyroid) larger than average by a factor of two to three were those who lived in rural areas in single story homes and who ate large amounts of "wild" foods, such as game meats, mushrooms, and berries.

(11) The long term internal doses to residents of rural settlements strongly depend on soil properties. Contributions due to internal and external exposure are comparable in areas with light sandy soils, and the contribution of internal exposure to the total (external and internal) dose does not exceed 10% in areas with dominantly black soils. The contribution of ^{90}Sr to the internal dose regardless of natural conditions is usually less than 5%.

(12) Long term internal doses of children caused by ingestion of food containing caesium radionuclides are usually lower by a factor of about 1.1 to 1.5 than those in adults and adolescents.

(13) Both accumulated and predicted mean doses in settlement residents vary in the range of two orders of magnitude depending on radioactive contamination of the area, dominating soil type and settlement type. In 1986-2000 the accumulated dose range comprised from 2 mSv in towns located in black soil areas up to 300 mSv in villages located in areas with podzol sandy soils. The doses expected in 2001–2056 are substantially lower than those already received, i.e., in the range of 1 to 100 mSv.

(14) If countermeasures were not applied, the population of some more contaminated villages would receive life-time (70 years) effective doses up to 400 mSv. Intensive application of countermeasures, such as settlement decontamination and agricultural countermeasures, has substantially reduced dose. For comparison, a worldwide average life time dose from natural background radiation is about 170 mSv with the typical range of 70 to 700 mSv in various regions.

(15) The vast majority of about five million persons residing in contaminated areas of Belarus, Russia and Ukraine currently, i.e., in early 2000s, receive annual effective dose less than 1 mSv (national action levels in the three countries). For comparison, a worldwide average annual dose from natural background radiation is about 2.4 mSv with the typical range of 1 to 10 mSv in various regions (UNSCEAR 2000).

(16) The number of residents of the contaminated areas in the three more affected countries that currently receive more than 1 mSv annually can be broadly estimated as about 100,000 persons. As future reduction of both external dose rate and radionuclide (mainly ^{137}Cs) activity concentrations in food is rather slow, reduction of human exposure levels is expected to be slow as well, i.e., of about 3 to 5% per year for current countermeasures.

(17) Based upon available information it does not appear that doses associated with “hot particles” have been significant.

(18) The assessment of the experts working on the Chernobyl Forum agrees with that of the UNSCEAR (2000) experts in terms of the dose received by the populations of the three more affected countries: Belarus, Ukraine, and the Russian Federation.

4.6.2. Recommendations

(1) Large-scale monitoring of foodstuffs, whole-body counting of individuals, and provision of TL detectors to members of the general population are no longer necessary. Critical groups in areas of high contamination and/or high transfer of radiocaesium to foods are known. Representative members of these critical groups should be monitored by dosimeters for external dose and by whole body counting for internal dose.

(2) Sentinel individuals in more highly contaminated areas not scheduled for further remediation might be identified with the goal of continued periodic whole body counts and monitoring for external dose. The goal would be to follow the expected continued decrease in external and internal dose and to determine whether such decreases are due to radioactive decay alone or to further ecological elimination.

REFERENCES TO SECTION 4

BALONOV, M., JACOB, P., LIKHTAREV, I., MINENKO, V., (1996) “Pathways, levels and trends of population exposure after the Chernobyl accident,” In: The radiological consequences of the Chernobyl accident. Ed. by A. Karaoglou, G. Desmet, G.N. Kelly and H.G. Menzel. Rept. EUR 16544 EN, European Commission, Luxembourg, pp. 235–249.

BALONOV, M.I., BRUK, G., GOLIKOV, V., ERKIN, V.G., ZVONOVA, I.A., PARKHOMENKO, V.I., SHUTOV V.N. (1995) Long term exposure of the population of the Russian Federation as a consequence of the accident at the Chernobyl power plant. In: Environmental Impact of Radioactive Releases. IAEA, Vienna, pp. 397–411.

BALONOV, M.I., TRAVNIKOVA I.G. (1990) The role of agricultural and natural ecosystems in the internal dose formation in the inhabitants of a controlled area. In: Transfer of radionuclides in natural and semi-natural environments. Ed. by G. Desmet, P. Nassimbeni and M. Belli, Elsevier Applied Science, London and New York, pp. 419–430.

BALONOV M., KAIDANOVSKY G., ZVONOVA I., KOVTUN A., BOUVILLE A., LUCKYANOV N., AND VOILLEQUÉ, P. (2003) Contributions of short-lived radioiodines to thyroid doses received by evacuees from the Chernobyl area estimated using early *in-vivo* activity measurements. Radiat. Prot. Dosim. Vol. 105, No. 1-4, 593-600.

BALONOV, M.I., ZVONOVA, I.A. (EDS.) (2002) Mean Thyroid Doses for Inhabitants of Different Age Who Lived in 1986 in Settlements of the Bryansk, Tula, Orel and Kaluga Regions Contaminated by Radionuclides as a Result of the Chernobyl Accident, Radiation and Risk, Special Issue, Obninsk-Moscow, 94 p. (in Russian).

BERKOVSKI, V., VOITSEKHOVITCH, O., NASVIT, O., ZHELEZNYAK, M., SANSONE, U. (1996a), Exposures from aquatic pathways. In: The radiological consequences of Chernobyl accident. Ed. by A. Karaoglou, G. Desmet, G.N. Kelly and H.G. Menzel. Rept. EUR 16544 EN, European Commission. Luxembourg, p.p. 283-294 (1996).

BERKOVSKI, V., RATIA, G., NASVIT O. (1996b), Internal Doses to Ukrainian Population using Dnieper River Water. Health Phys. 71(1): 37-44.

BRUK G. Ya., SHUTOV V.N., BALONOV M.I., BASALEYAVA L.N. and KISLOV M.V. (1998) Dynamics of ^{137}Cs content in agricultural food products produced in regions of Russia contaminated after the Chernobyl accident. Radiation Protection Dosimetry, 76, 169-178.

BRUK, G.YA, SHUTOV, V.N., TRAVNIKOVA, I.G., BALONOV, M.I., KADUKA, M.V., BASALAEVA, L.N. (1999) The role of the forest products in the formation of internal exposure doses to the population of Russia after the Chernobyl accident. In: I. Linkov and W.R. Shell (eds.), Contaminated Forests, Nato Science Series, Kluwer Academic Publishers, Netherlands, 1999, v. 58, pp. 343–352.

CHUMAK, V.V., LIKHTAREV, I.A., PAVLENKO, J.V. (1999) Monitoring of individual doses of populations residing in the territories contaminated after the Chernobyl accident, Radiat. Prot. Dosim. 85, 137–139.

DE CORT, M., DUBOIS, G., FRIDMAN, SH.D., GERMENCHUK, M.G., IZRAEL, YU.A. JANSSENS, A., JONES, A.R., KELLY, G.N., KVASNIKOVA, E.V., MATVEENKO, I.I., NAZAROV, I.M., POKUMEIKO, YU.M., SITAK, V.A., STUKIN, E.D., TABACHNY, L.YA., TSATUROV, YU.S., AVDYUSHIN, S.I., Atlas of caesium deposition on Europe after the Chernobyl accident, Rept. EUR 16733, European Commission, Luxembourg (1998).

DROZDOVITCH, V.V., MINENKO, V.F., KUKHTA, T.S., ULANOVSKY, A.V., BOUVILLE, A. (2002), Estimates of internal doses from radiocesium received by the population of Belarus following the Chernobyl accident. Submitted to Health Phys.

ERKIN VG, LEBEDEV OV (1993) Thermoluminescent dosimeter measurements of external doses to the population of the Bryansk region after the Chernobyl accident. In: Balonov MI, Merwin SE (eds) The Chernobyl Papers. Vol. 1. Research Enterprises; Publishing Segment, Richland, pp 289–311.

FESENKO S.V. PAKHOMOV A. YU, PASTERNAK A.D., GORYAENOV V.A., FESENKO G.A., PANOV A.V. (2004) Regularities of changing in ^{137}Cs activity concentrations in the long term after the accident at the Chernobyl NPP. Radiation biology. Radioecology, 44, 1, pp. 35-49.

GAVRILIN, YU.I., KHROUCH, V.T., SHINKAREV, S.M., KRYSENKO, N.A., SKRYABIN, A.M., BOUVILLE, A., ANSPAUGH, L.R. (1999) Chernobyl accident: Reconstruction of thyroid dose for inhabitants of the Republic of Belarus, *Health Phys.,* 76, 105–119.

GAVRILIN Y, KHROUCH V, SHINKAREV S, DROZDOVITCH V, MINENKO V, SHEMIAKHINA E, ULANOVSKY A, BOUVILLE A, ANSPAUGH L, VOILLEQUÉ P, LUCKYANOV N. (2004) Individual thyroid dose estimation for a case-control study of Chernobyl-related thyroid cancer among children of Belarus. Part 1: ^{131}I, short-lived radioiodines (^{132}I, ^{133}I, ^{135}I), and short-lived radiotelluriums (^{131m}Te and ^{132}Te). Health Physics 86: 565-585.

GOLIKOV V.YU. (2004) Personal communication to the Chernobyl Forum.

GOLIKOV, V., BALONOV, M., ERKIN, V., JACOB, P. (1999), Model validation for external doses due to environmental contaminations by the Chernobyl accident. Health Physics, v. 77, Nr. 6, pp. 654–661.

GOLIKOV V.YU., BALONOV M. I., JACOB P. (2002), External exposure of the population living in areas of Russia contaminated due to the Chernobyl accident, *Radiat. Environ. Biophys.,* 41, 185–193.

HEDEMANN-JENSEN, P. (1985) Shielding factors for gamma radiation from activity deposited on structures and ground surfaces. Nucl. Techn. 68: 29-39.

HEIDENREICH W, KAIRO I.A., CHEPURNY N.I., JACOB P, SPAK V., GULKO G.M. AND PARETZKE, H. (2001) Age- and sex- specific relative thyroid radiation exposure to ^{131}I in Ukraine after the Chernobyl accident. Health Physics 80(3): 242-250.

INTERNATIONAL ADVISORY COMMITTEE (1991) The International Chernobyl Project, Technical Report, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (2005) Environmental Radioactivity in the Dnieper River Basin: Assessment of the IAEA Project Team and Recommendations for the Strategic Action Plan. IAEA, Vienna, in print.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (1991) 1990 Recommendations of the International Commission on Radiological Protection, ICRP Publication 60, *Annals of the ICRP* 21 (1–3), Pergamon Press Oxford.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (1994) Human Respiratory Tract Model for Radiological Protection, ICRP Publication 66, Pergamon, Oxford.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (2001) The ICRP Database of Dose Coefficients: Workers and Members of the Public, ICRP CD-ROM System, Version 2.01, Pergamon, Oxford.

INTERNATIONAL COMMISSION ON RADIOLOGICAL UNITS AND MEASUREMENTS (2002). Retrospective Assessment of Exposures to Ionising radiation. ICRU Report 68, ed. by P. Jacob, Journal of the ICRU, 2, Nr. 2, pp. 1-136.

IVANOVA, N.P., SHVYDKO, N.S., ERSHOV, E.B. and BALONOV, M.I. (1995), Population doses in Russia from plutonium fallout following the Chernobyl accident. Radiation Protection Dosimetry, 58, pp. 255–260.

IZRAEL, YU.A.; VAKULOVSKII, S.M.; VETROV, V.A.; PETROV, V.N.; ROVINSKII, F.YA.; STUKIN, YE.D. (1990) Chernobyl: Radioactive contamination of the environment. Gidrometeoizdat: Leningrad, (in Russian).

IZRAEL YU. (ED.) (1998) Atlas of Radioactive Contamination of European Russia, Belarus and Ukraine, Federal Service for Geodesy and Cartography of Russia, Moscow, 143 p.

JACOB P., MECKBACH R. and MILLER H.M. (1987) Reduction of External Exposures from Deposited Chernobyl Radioactivity Due to Run-Off, Weathering, Street-Cleaning and Migration in the Soil. Radiation Protection Dosimetry, 21 (1-3) 51-57.

JACOB, P.; MECKBACH R. (1990a) External exposure from deposited radionuclides. Proceedings of the seminar on methods and codes for assessing the off-site consequences of nuclear accidents, May 7-11, 1990. Athens.

JACOB P., ROSENBAUM H., PETOUSSI N. and ZANKL M. (1990b) Calculation of Organ Doses from Environmental Gamma Rays Using Human Phantoms and Monte Carlo Methods. Part II: Radionuclides Distributed in the Air or Deposited on the Ground. GSF-Bericht 12/90.

JACOB, P., LIKHTAREV, I. (eds.) (1996) Pathway analysis and dose distributions. Joint study project No. 5. Final report, Report., EUR 16541, European Communities, Luxembourg.

JACOB, P.; MECKBACH R. (2000) Measurements after the Chernobyl accident in relation to an exposure of an urban population. TECDOC-1131, IAEA, Vienna, pp. 34-41.

KARLBERG, O. (1987) Weathering and migration of Chernobyl fallout in Sweden. Radiation Protection Dosimetry, 21: 75-78.

KRUK J.E., PROHL G., KENIGSBERG JA. (2004) A radioecological model for thyroid dose reconstruction of the Belarus population following the Chernobyl accident. *Radiat. Environ. Biophys*. 43:101-110.

KUTKOV V., SKRAYBIN A., POGODIN R., AREFIEVA Z. and MURAVYEV YU. (1995) Inhalation of the aerosol of nuclear fuel particles from the Chernobyl nuclear power plant by adult persons from the Gomel region of Belarus. In: Environmental Impact of Radioactive Releases. IAEA, Vienna, pp. 107–115.

LIKHTAREV, I.A., SHANDALA, N.K., GULKO, G.M., KAIRO, I.A., CHEPURNY, N.I. (1993) Ukrainian thyroid doses after the Chernobyl accident, Health Physics, 64, 594–599.

LIKHTAREV, I.A., KOVGAN, L., NOVAK, D., VAVILOV, S., JACOB, P., PARETZKE, H.G. (1996a) Effective doses due to external irradiation from the Chernobyl accident for different population groups of Ukraine, Health Physics, 70, 87–98.

LIKHTAREV, I.A., KOVGAN, L.N., VAVILOV, S.E., PEREVOZNIKOV, O.N., LITVINETS, L.N., ANSPAUGH, L.R., JACOB, P., PRÖHL, G. (2000) Internal exposure from the ingestion of foods contaminated by ^{137}Cs after the Chernobyl accident—Report 2.

Ingestion doses of the rural population of Ukraine up to 12 y after the accident. Health Physics, 79, 341–357.

LIKHTAREV, I.A., KOVGAN, L.N., JACOB, P., ANSPAUGH, L.R. (2002) Chernobyl accident: Retrospective and prospective estimates of external dose of the population of Ukraine, Health Physics, 82, 290–303.

LIKHTAREV, I.A., KOVGAN, L., VAVILOV, S.E., CHEPURNY, N.I., BOUVILLE A., LUCKYANOV N., JACOB, P., VOILLEQUÉ, P. AND VOIGT, G. (2005) Post-Chernobyl thyroid cancers in Ukraine. Report 1: Estimation of thyroid doses. Radiation Research, 163: 125-136.

MECKBACH, R., JACOB, P. AND PARETZKE, H. (1988) Gamma exposures due to radionuclides deposited in urban environments. Part I: Kerma rates from contaminated urban surfaces. Radiation Protection Dosimetry, 25: 167-169.

MECKBACH, R., JACOB, P. (1988) Gamma exposures due to radionuclides deposited in urban environments. Part II: Location factors for different deposition patterns. Radiation Protection Dosimetry, 25: 181-190.

MILLER KM, KUIPER IL, HELFER IK (1990) Cs-137 fallout depth distributions in forest versus field sites: implication for external dose rates. J. Environ. Radioactivity 12: 23–47.

MÜCK, K., PRÖHL, G., LIKHTAREV, I., KOVGAN, L., MECKBACH, R. and GOLIKOV, V. (2002) A consistent radionuclide vector after the Chernobyl accident. *Health Phys.* 82, 141-156.

SHUTOV V.N., BRUK G.YA., BALONOV M.I., PARHOMENKO V.I. AND PAVLOV I.JU. (1993) Cesium and strontium radionuclide migration in the agricultural ecosystem and estimation doses to the population. In: The Chernobyl Papers, Vol.1. Doses to the Soviet Population and Early Health Effects Studies. Ed. by S.E. Merwin and M.I. Balonov, Richland, Research Enterprises, pp. 167-218.

SKRYABIN A.M.; MASYAKIN V.B.; OSYPENKO A.N.; VLASOVA N.C.; SAVKIN M.N.;GRINEV M.; LEBEDEV A.; ANDROSOVA A.; KONSTANTINOV Y.O.; ERKIN V.G.; KORELINA N.F.; MOSKALEV O.S.; ROBINSON C.A.; PROSSER S.L.; JONES K.A.; MORREY M. (1995) Distribution of doses received in rural areas affected by the Chernobyl accident. NRPB-R277; Chilton; Didcot; Oxon.

TRAVNIKOVA, I.G., BAZJUKIN, A.N., BRUK, G.JA, SHUTOV, V.N., BALONOV, M.I., SKUTERUD, L., MEHLI, H. and STRAND, P. (2004) Lake fish as the main contributor of internal dose to lakeshore residents in the Chernobyl contaminated area. Journal of Environmental Radioactivity, 77, Nr.1, pp. 63–75.

THOMAS, G., KARAOGLOU, A., WILLIAMS, E.D. (EDS.) (1999) Radiation And Thyroid Cancer, World Scientific, Singapore, 524 p.

UNITED NATIONS, Sources, Effects and Risks of Ionizing Radiation (1988 Report to the General Assembly, with Annexes), Scientific Committee on the Effects of Atomic Radiation, UN, New York (1988) 309–374.

UNITED NATIONS, Sources and Effects of Ionizing Radiation (2000 Report to the General Assembly, with Scientific Annexes), Scientific Committee on the Effects of Atomic Radiation, UN, New York (2000).

VOITSEKHOVITCH, O.V., SANSONE, U., ZHELEZNYAK, M. and BUGAI, D. (1996) Water quality management of contaminated areas and its effects on doses from aquatic pathways. In: The radiological consequences of the Chernobyl accident. Ed. by A. Karaoglou, G. Desmet, G.N. Kelly and H.G. Menzel. Rept. EUR 16544 EN, European Commission.

WORLD HEALTH ORGANIZATION (2005) Health Effects of the Chernobyl Accident and Special Health Care Programmes, Report of the Chernobyl Forum Expert Group “Health” (EGH), WHO, Geneva.

ZHELEZNYAK, M.K et al. (1992), Mathematical modeling of radionuclides dispersion in the Pripyat-Dnieper aquatic system after the Chernobyl accident, Science of the Total Environment, 112, 89-114.

ZHELEZNYAK, M., KUZMENKO, YU., TKALICH, P., DZUBA, N., GOFMAN, D., GOLOVANOV, I., MARINETS, A., MEZHUEVA, I. (1994), Modelling of radionuclides transport in the set of river reservoirs. –In: *Computational Methods in Water Resources,* X, A. Peters et al. (eds.) , vol. 2, Kluwer Academic, pp. 1189 - 1196.

ZVONOVA, I.A., BALONOV, M.I. (1993) Radioiodine dosimetry and prediction of consequences of thyroid exposure of the Russian population following the Chernobyl accident, In: The Chernobyl Papers (MERWIN, S.E., BALONOV, M.I., Eds.) Research Enterprises, Richland pp. 71–125.

ZVONOVA I., BALONOV M., BRATILOVA A., VLASOV A., PITKEVICH V., VLASOV O., SHISHKANOV N. (2000) Methodology of Thyroid Dose Reconstruction for Population of Russia after the Chernobyl Accident. In: Harmonization of Radiation, Human Life and the Ecosystem, Proc. of 10th International Congress of the IRPA, International Conference Center Hiroshima, Hiroshima, P-11-265.

5. RADIATION-INDUCED EFFECTS ON PLANTS AND ANIMALS

5.1. Prior knowledge of radiation effects on biota

The biological effects of radiation on plants and animals have long been of interest to scientists; in fact, much of the information on effects on humans has evolved from studies on plants and animals. Additional effects research followed the development of nuclear energy and concerns about the possible impacts of increased, but authorised, discharges of waste radionuclides into the terrestrial and aquatic environments. The magnitude of these authorised releases has always been controlled on the basis of limitation of human exposure, but it has been recognised that animals and plants have also been exposed—frequently to a higher degree than humans. By the mid 1970s sufficient information had been accrued on the effects of ionising radiation on plants and animals that several authoritative reviews had been compiled to summarise the findings (IAEA 1976, 1988, 1992; UNSCEAR 1996).

Some broad generalizations about effects from radiation exposure can be gleaned from the research that has been conducted over the last 100 years. Foremost are the relatively large differences in doses required to cause lethality among various taxonomic groups (Fig. 5.1). Considerable variation in response occurs within a taxon due to enhanced radiosensitivity of some individuals or life stages. Wide ranges in doses are also observed within a group or taxon when progressing from minor to severe effects.

Fig. 5.2 summarizes information on the doses delivered over a short time period required to produce damage of different degrees in various plant communities, soil invertebrates and rodents. Within the plant kingdom, trees are generally more sensitive than shrubs, which in turn are more sensitive than herbaceous species. Primitive forms such as lichens, mosses, and liverworts are more resistant than vascular plants. Radiation-resistant plants frequently have molecular and cellular characteristics that enhance their ability to tolerate radiation stress, and differences in plant-community response can be explained, in part, by the early work of Sparrow (1961). He showed that characteristics such as large chromosomes, normal (rather than diffuse) centromeres, small chromosome number, uni-nucleated cells, diploid or haploid cells, sexual reproduction, long intermitotic and slow rates of meiosis are associated with high radiosensitivity in plants, but that sensitivity can be modified in time due to seasonal processes (e.g., dormancy, or the onset of growth in spring; Table 5.1).

Scientific reviews, e.g., (IAEA 1992), indicated that mammals are the most sensitive organisms and that reproduction is a more sensitive endpoint than mortality. For acute exposures of mammals, mortality generally occurs at doses >3 Gy while reproduction is affected at doses <0.3 Gy. Chronic exposures alter the responses, with mortality occurring at >0.1 Gy d^{-1} and reproduction effected at <0.01 Gy d^{-1}. Among aquatic organisms, fish are the most sensitive, with gametogenesis and embryo development being the more sensitive stages. Effects on animal populations can be reduced by their mobility (in terms of moving from areas of high exposure to areas of low exposure). Comparatively stationary soil invertebrates do not have such abilities and can receive substantial doses relative to the rest of the animal kingdom, particularly because the soil is a sink for most radioactive contamination.

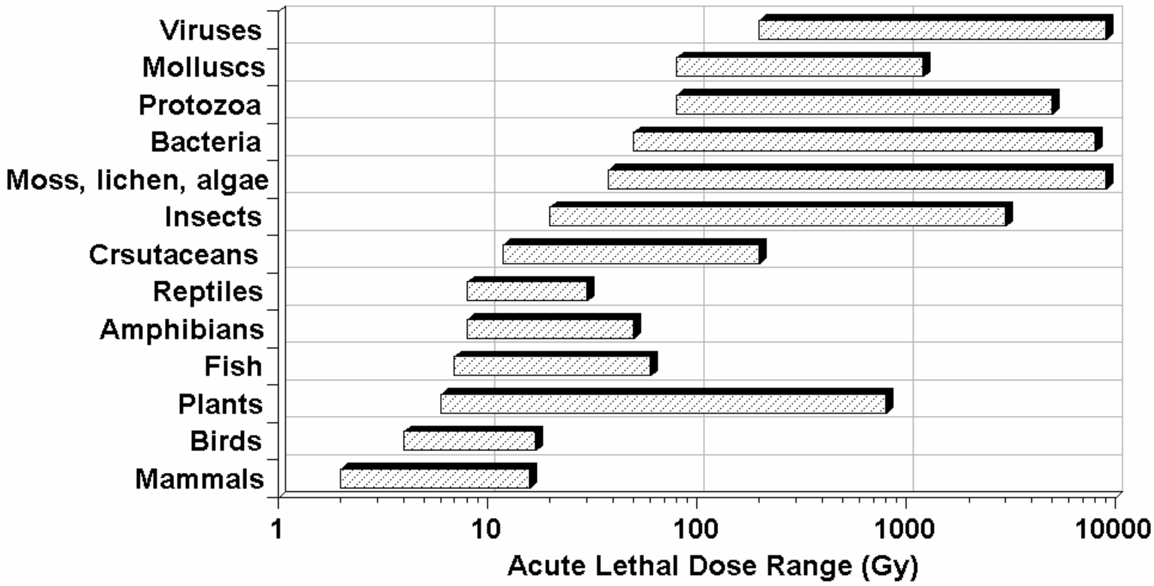


FIG. 5.1. Acute dose ranges that result in 100% mortality in various taxonomic groups. Humans are among the most sensitive mammals, and therefore among the most sensitive organisms (Whicker and Schultz 1982).

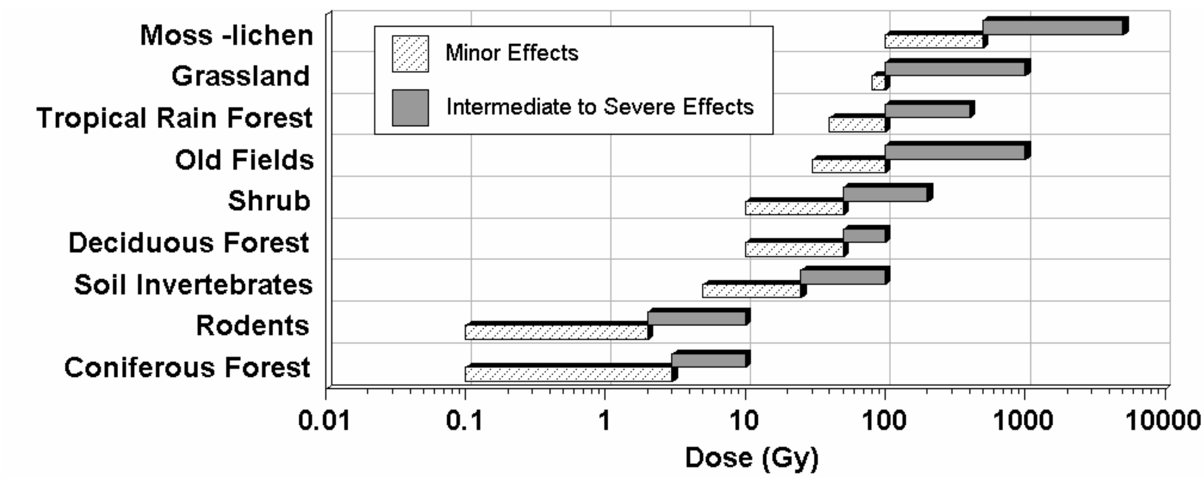


FIG. 5.2. Range of short term radiation doses (delivered over 5 to 60 d) that produced effects in various plant communities, rodents and soil invertebrates. Minor effects include chromosomal damage, changes in productivity, reproduction and physiology. Intermediate effects include changes in species composition and diversity through selective mortality. Severe effects (massive mortality) begin at the upper range of intermediate effects (Whicker and Fraley 1974, and reviewed in Whicker 1997).

TABLE 5.1. PRINCIPAL NUCLEAR CHARACTERISTICS AND FACTORS INFLUENCING THE SENSITIVITY OF PLANTS TO RADIATION (SPARROW 1961, AND ADAPTED FROM WHICKER AND SCHULTZ 1982)

Factors increasing sensitivity	Factors decreasing sensitivity
Large nucleus (high DNA content)	Small nucleus (low DNA content)
Much heterochromatin	Little heterochromatin
Large chromosomes	Small chromosomes
Acrocentric chromosomes	Metacentric chromosomes
Normal centromere	Polycentric or diffuse centromere
Uninucleate cells	Multinucleate cells
Low chromosome number	High chromosome number
Diploid or haploid	High polypolid
Sexual reproduction	Asexual reproduction
Long intermitotic time	Short intermitotic time
Long dormant period	Short or no dormant period
Slow meiosis	Fast meiosis

The response of a plant or animal to radiation depends on the dose received, as well as its radiosensitivity. The former is largely determined by its habitat preference in relation to the evolving distribution of radioactive contaminants as a function of time, as well as the organism's propensity to accumulate radionuclides into its organs and tissues. Because of their particular use of the habitat, plants and animals within a contaminated area may receive radiation doses that can be substantially greater than those of humans occupying the same area (e.g., humans gain some shielding from housing and may obtain food and water from less contaminated sources; IAEA 1992).

Although all exposures to ionising radiations have the potential to damage biological tissue, protraction of a given total absorbed dose in time can, depending on the dose rate, result in a reduction in response due to the intervention of cellular and tissue-repair processes. This has led to the conventional, but somewhat artificial, distinction between the so-called acute and chronic radiation-exposure regimes. In general, an acute radiation exposure is one that usually occurs at a high dose rate and in a short period of time relative to that within which obvious effects occur. Chronic exposures are taken to be continuous in time, often over a significant portion of an organism's lifespan, or throughout some particular lifestage (e.g., embryonic development), and usually at a sufficiently low dose rate that the cumulative dose does not produce acute effects.

The earlier reviews noted above were consistent in concluding that it is unlikely that there will be any significant detrimental effects:

— to terrestrial and aquatic plant populations, and aquatic animal populations, at chronic dose rates less than 10 mGy d^{-1}; or,

— to terrestrial animal populations at dose rates less than 1 mGy d^{-1}.

It should be emphasised, however, that these dose rates were not intended for use as limits in any system to provide for the protection of the environment; they were simply the dose rates below which the available evidence, admittedly limited in the range of organisms and the biological responses investigated, indicated little likelihood of any significant response. The above dose rates are with reference to population-level effects, not to impacts to individual organisms.

The more recent reviews of the effects of irradiation on individual organisms carried out in the frame of two European Community projects, FASSET (Framework for the Assessment of Environmental Impact) and EPIC (Environmental Protection from Ionising Contaminants in the Arctic) have produced broadly consistent conclusions (Woodhead and Zinger 2003; Sazykina et al. 2003; Real et al. 2004). Although minor effects may be seen at lower dose rates in sensitive cell systems or individuals of sensitive species (e.g., haematological cell counts in mammals, immune response in fish, growth in pines and chromosome aberrations in many organisms), the threshold dose rate for significant effects in most studies is about 0.1 mGy h^{-1} (2.4 mGy d^{-1}). Detrimental responses then increase progressively with increasing dose rate and usually become clear at >1 mGy h^{-1} (24 mGy d^{-1}) given over a large fraction of the lifespan. The significance of the minor morbidity and cytogenetic effects to the individual, or to populations more generally, seen at dose rates less than 2.4 mGy d^{-1}, has yet to be determined (Real et al. 2004).

The recently compiled EPIC database covers a very wide range of radiation-dose rates (from below 10^{-5} Gy d^{-1} up to more than 1 Gy d^{-1}) to wild flora and fauna observed in northern parts of Russia, including wildlife in the Chernobyl contaminated areas (Sazykina et al. 2003). The general conclusion from the EPIC database is that the threshold for deterministic radiation effects in wildlife lies somewhere in the range of 0.5–1 mGy d^{-1} for chronic low-LET radiation.

These broad conclusions concerning the impact of radiation on plants and animals provide an appropriate context within which to consider the available information on the effects that have been observed from the increased radiation exposures following the accident at Chernobyl.

5.2. The temporal dynamics of radiation exposure from the Chernobyl accident

It is critical to frame any discussion of Chernobyl environmental effects within the specific time period of interest. Effects observed now, nearly 20 years after the accident, are drastically different from those that occurred during the first 20 days. Three distinct phases of radiation exposure have been identified in the area local to the accident (UNSCEAR 1996). In the first 20 days, radiation exposures were essentially acute because of the large quantities of short-lived radionuclides present in the passing cloud of contamination (^{99}Mo, ^{132}Te/I, ^{133}Xe, ^{131}I and ^{140}Ba/La). Most of these short-lived, highly radioactive nuclides deposited onto plant and ground surfaces resulting in the accumulation of large doses that measurably impacted biota. High exposures to the thyroids of vertebrate animals also occurred during the first days/weeks following the accident from the inhalation and ingestion of radioactive iodine isotopes or their radioactive precursors.

The measured exposure rates on the accident day in the immediate vicinity of the damaged reactor are given in Fig. 5.3. These exposure rates were mainly due to γ-irradiation from deposited radionuclides, and range up to about 20 Gy d^{-1}. However, for surface tissues and small biological targets (e.g., mature needles and growing buds of pine trees), there was a considerable additional dose rate from the β-radiation of the deposited radionuclides. Taking into account the high dose rates during the relatively short exposure period from the short-lived radioisotopes, this first phase of 20 to 30 days can be generally characterised as an acute exposure regime that had pronounced effects on biota.

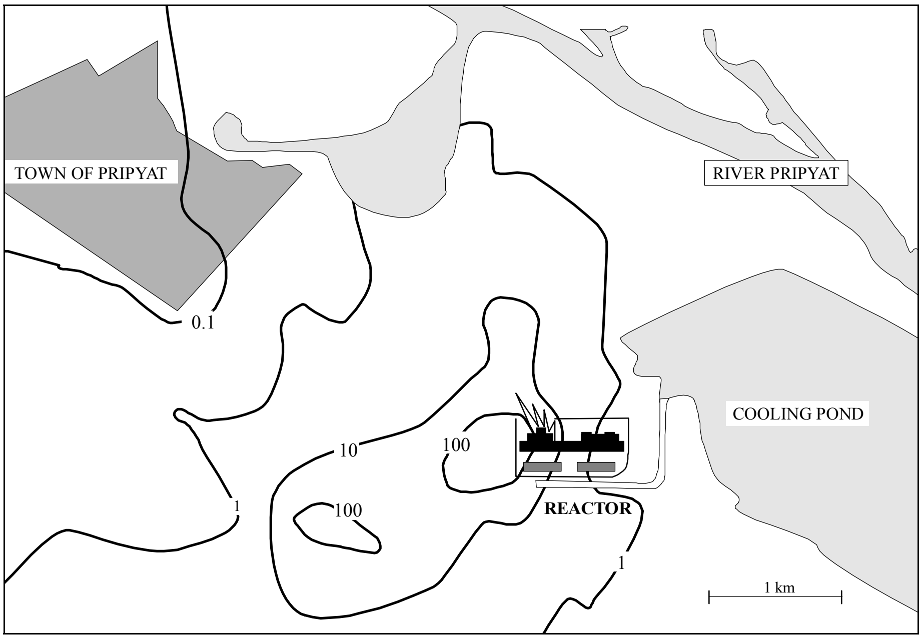


FIG. 5.3. Measured exposure rates in air on 26 April 1986 in the local area of the Chernobyl reactor. Units of isolines are R h^{-1} (UNSCEAR 2000; 1 R h^{-1} is approx 0.2 Gy d^{-1}).

The second phase of radiation exposure extended through the summer and autumn of 1986, during which time the short-lived radionuclides decayed and longer-lived radionuclides were transported to different components of the environment by physical, chemical and biological processes. Dominant transportation processes included rain-induced transfer of radionuclides from plant surfaces onto soil, and bioaccumulation through plant tissues. Although the dose rates at the soil surface declined to much less than 10% of the initial values due to radioactive decay of the short-lived isotopes (see Fig. 4.3), damaging total doses were still accumulated. The modifying effect of radionuclide wash-off by rain to radiation damage of conifers is shown in Fig. 5.4.

In general, approximately 80% of the total radiation dose accumulated by plants and animals was received within 3 months of the accident, and over 95% of this was due to β-radiation (UNSCEAR 1996). This finding agrees with earlier studies on the importance of β-radiation, relative to the γ-component, to the total dose from radioactive fallout. For example, when a 10-hour old mixture of fresh fission products was experimentally deposited onto cereal plants at differing stages of growth, at a density of 7 GBq m^{-2}, the ratio of resulting β- and γ-dose rates, measured with thermoluminescent dosimeters, varied from 1 to 130 (Prister et al. 1982).

Measurements made with thermoluminescent dosimeters on the soil surface at sites within the 30-km exclusion zone indicated that the ratio of β-dose to γ-dose was about 26:1, (i.e., 96% of the total dose was from β-radiation). For a γ-dose rate of 0.01 mGy h^{-1} at the soil surface, 15 days after the accident, the total cumulative dose in the first month from β- and γ-radiation was estimated to be 0.5 ± 0.2 Gy, and 0.6 and 0.7 Gy at the end of the second and third months, respectively (Krivolutsky et al. 1999).

FIG. 5.4. Photograph of a small conifer showing the upper portion damaged by the initial deposition of radiation onto the crown of the plant, and the lower part of the plant damaged from the irradiation of surface deposited material subsequently washed from the plant's crown, leaving the middle section of the plant unaffected (photo: T. Hinton).

In the third (and continuing) phase of radiation exposure, dose rates have been chronic, less than 1% of the initial values, and derived mainly from ^{137}Cs contamination. With time, the decay of the short-lived radionuclides and the migration of much of the remaining ^{137}Cs into the soil have meant that the contributions to the total radiation exposure from the β- and γ-radiations have tended to become more comparable. The balance does depend, however, on the degree of bioaccumulation of ^{137}Cs in organisms and the behaviour of the organism in relation to the main source of external exposure from the soil. Aside from the spatial heterogeneity in dose rate arising from the initial deposition, large variations in the radiation exposure of different organisms occurred at different times due to their habitat niche (e.g., birds in the canopy versus rodents on the ground). Immigration of animals into the 30-km zone and recruitment of plants and animals from those that are present means that new animals are constantly being introduced into the radioactively contaminated conditions that exist around Chernobyl today. Current conditions are presented in Section 5.8.

5.3. Radiation-induced effects on plants

Doses received by plants from the Chernobyl fallout were influenced by the physical properties of the various radionuclides (i.e., their half-lives, radiation emissions, etc.), the physiological stage of the plant species at the time of the accident, and the different species-dependent propensities to take up radionuclides into critical plant tissues. The occurrence of the accident in late April heightened the damaging effects of the fallout because it coincided with the period of accelerated growth and reproduction in plants. The deposition of β-emitting contamination onto critical plant tissues resulted in their receiving a significantly larger dose than animals living in the same environment (Prister et al. 1982; Prister et al. 1991). Large, apparent, inconsistencies in dose-response observations occurred when the beta-irradiation component was not appropriately accounted for (Grodzinsky et al. 1991).

Within the 30-km zone of Chernobyl, deposition of total beta activity and associated doses to plants were sufficient (0.7–3.9 GBq m^{-2}) to cause short-term sterility and reduction in productivity of some species (Prister et al. 1991). By August 1986, crops that had been sown prior to the accident began to emerge. Growth and development problems were observed in plants growing in fields with contamination densities of 0.1 to 2.6 GBq m^{-2}, and with estimated dose rates initially received by plants reaching 300 mGy d^{-1}. Spot necroses on leaves, withered tips of leaves, inhibition of photosynthesis, transpiration and metabolite synthesis were detected, as well as an increased incidence of chromosome aberrations in meristem cells (Shevchenko et al. 1996). The frequency of various anomalies in winter wheat exceeded 40% in 1986–1987, with some abnormalities apparent for several years afterwards (Grodzinsky and Gudkov 2001).

Coniferous trees were already known to be among the more radiosensitive plants, and pine forests 1.5 to 2 km west of the Chernobyl Nuclear Reactor received sufficient dose (>80 Gy) to cause mortality (Tikhomirov and Shcheglov 1994), at dose rates that exceeded 20 Gy d^{-1} (UNSCEAR 2000). The first signs of radiation injury in pine trees in close proximity to the reactor were yellowing and needle death, which appeared within 2-3 weeks. During Summer 1986, the area of radiation damage expanded in the northwest direction up to 5 km; serious damage was observed at a distance of 7 km. The colour of the dead pine stands resulted in the forest being referred to as the “red forest.”

Tikhomirov and Shcheglov (1994) and Arkhipov et al. (1994) found that mortality rate, reproduction anomalies, stand viability, and re-establishment of pine-tree canopies, were dependent on absorbed dose. Acute irradiation of *Pinus silvestris* at doses of 0.5 Gy caused detectable cytogenetic damage; at >1 Gy growth rates were reduced and morphological damage occurred; and at >2 Gy the reproductive abilities of trees were altered. Doses of less than 0.1 Gy did not cause any visible damage to the trees. Table 5.2 shows the variation in activity concentration and dose among pine trees within the 30-km zone. The radiosensitivity of spruce trees was observed to be greater than that of pines. At absorbed doses as low as 0.7 to 1 Gy, spruce trees had malformed needles, buds, and shoot growth (Kozubov et al. 1990).

Of the absorbed dose to critical parts of trees, 90% was due to β-irradiation from the deposited radionuclides and 10% to γ-irradiation. As early as 1987, recovery processes were evident in the surviving tree canopies and young forests were re-established in the same place as the perished trees by replanting via reclamation efforts (Arkhipov et al. 1994). In the decimated pine stands a sudden invasion of pests occurred that later spread to adjoining areas. The deceased pine stands have now been replaced by grassland with a slow invasion of self-seeding, deciduous trees. Four distinct zones of radiation-induced damage to conifers were discernable (Table 5.3).

5.4. Radiation effects on soil invertebrates

Although between 60 and 90% of the initial fallout was captured by the forest canopy and other plants (Tikhomirov and Shchelgov 1994), within weeks to a few months, the processes of wash-off by rain and leaf-fall moved the majority of the contamination to the litter and soil layers (see Section 2.4 for more details), where soil and litter invertebrates were exposed to high radiation levels for protracted time periods. The potential for impact on soil invertebrates was particularly large, because the timing of the accident coincided with their most radiosensitive life stages: reproduction, and molting following their winter dormancy.

TABLE 5.2. RADIOACTIVE CONTAMINATION (kBq kg^{-1}) OF CONIFEROUS TREES AS A FUNCTION OF DISTANCE FROM THE CHERNOBYL REACTOR (AZIMUTH 205 TO 260 DEGREES), WITH CORRESPONDING ESTIMATES OF THE AIR DOSE RATE (mGy h^{-1}) IN OCTOBER 1987, AND THE ACCUMULATED EXTERNAL DOSE (Gy) (ADAPTED FROM KRYSHEV ET AL. 1992)

Distance from ChNPP (km)	Air Dose Rate (mGy h^{-1})*	External Dose (Gy)*	Activity Concentration of Needles (kBq kg^{-1})					
			^{144}Ce	^{106}Ru	^{95}Zr	^{95}Nb	^{134}Cs	^{137}Cs
2.0	2.2	126	13400	4100	800	1500	1500	4100
4.0	0.10	5	150	60	8	15	17	72
16.0	$3.5\ 10^{-4}$	0.014	1.5	0.6	0.1	0.17	0.18	0.55

* Dose rate and dose of gamma radiation at 1 m height from the soil surface.

TABLE 5.3. ZONES AND CORRESPONDING DAMAGE TO CONIFEROUS FOREST IN THE AREA AROUND THE CHERNOBYL NUCLEAR POWER PLANT (FROM KOZUBOV ET AL. 1990)

Zone and Classification	External Gamma Dose* (Gy)	Air Dose Rate* (mGy h^{-1})	Internal Dose to Needles (Gy)
Conifer Death (4 km^2); Complete death of pines; partial damage to deciduous trees	over 80 - 100	over 4	over 100
Sublethal (38 km^2); Death of most growth points, partial death of coniferous trees, morphological changes to deciduous trees	10 – 20	2 – 4	50 – 100
Medium Damage; (120 km^2); Suppressed reproductive ability, dried needles, morphological changes	4 – 5	0.4 – 2	20 –50
Minor Damage; Disturbances in growth, reproduction and morphology of coniferous trees	0.5 – 1.2	< 0.2	< 10

*Dose rate and dose of gamma radiation at 1 m height from the soil surface.

Within two months after the accident, the numbers of invertebrates in the litter layer of forests 3 to 7 km from the nuclear reactor were reduced by a factor of 30 (Krivolutsky et al. 1999), and reproduction was strongly impacted (larvae and nymphs were absent). Doses of approximately 30 Gy (estimated from TLDs placed in the soil) had catastrophic effects on the invertebrate community, causing mortality of eggs and early life stages, as well as reproductive failure in adults. Within a year, reproduction of invertebrates in the forest litter resumed, due in part to the migration of invertebrates from less contaminated sites. After 2.5 years, the ratio of young to adult invertebrates in the litter layer, as well as the total mass of invertebrates per unit area, were no different from control sites; however, species diversity remained markedly lower (Krivolutsky et al. 1999).

The diversity of invertebrate species within the soil facilitates an analysis of community-level effects (i.e., changes in species composition and abundance). For example, only five species of invertebrates were found in 10 soil cores taken from pine stands in July 1986, 3 km from the Chernobyl nuclear power plant, compared to 23 species at a control site 70 km away. The mean density of litter fauna was reduced from 104 individuals per 225 cm^2 core at the control location to 2.2 at the 3-km site. Six species were found in all 10 cores taken from the control site, whereas no one species was found in all 10 cores from the 3-km location (Krivolutsky and Pokarzhevsky 1992). The number of invertebrate species found in the heavily contaminated sites was only half that of controls in 1993, and complete species diversity did not recover until 1995, almost 10 years after the accident (Krivolutsky et al., 1999).

Compared with invertebrates within the forest-litter layer, those residing in arable soil were not as impacted. A four-fold reduction in earthworm number was found in arable soils, but no catastrophic mortality in any group of soil invertebrates was observed. There was no reduction in soil invertebrates below a 5-cm depth in the soil. Radionuclides had not yet migrated into deeper soil layers, and the overlying soil shielded the invertebrates from beta irradiation, the main contributor (94%) to total dose. The dose to invertebrates in forest litter was 3-10 fold higher than to those residing in surface soil (Krivolutsky et al. 1999).

Although, researchers were unclear if sterility of invertebrates occurred in the heavily contaminated sites at Chernobyl (Krivolutsky et al. 1999), the 30-Gy cumulative dose reported for Chernobyl field studies is within the range of experimental doses used to control pest insects by external irradiation. A recent review indicated that most insect, mite, and tick families require a sterilization dose less than 200 Gy (Bakri et al. 2005), although the sterilization dose for some insects and related arthropods is much lower and ranges widely among and within orders. As was found for plants (Sparrow et al. 1961) radiosensitivity among insects is related to the average interphase nuclear volume (Bakri et al. 2005).

5.5. Radiation effects on farm animals

Ruminants, both domesticated (cattle, goats, sheep) and wild (elk, deer), generally receive high doses in radioactively contaminated environments, because they consume large amounts of vegetation, and many radionuclides accumulate in their bodies. For example, each day a single cow consumes about 30% of the grass from an area of 150 m^2. Ingestion of radionuclides leads to exposure of gut, thyroid and other body organs. Injuries to cattle are a major fallout consequence on rural populations because of livestock loss, but also because of the associated social and psychological implications (Prister 1999; Ilyazov et al. 2002).

In the period shortly after the accident, domestic livestock within the 30-km zone were exposed to high levels of radioactive iodine (^{131}I and ^{133}I with half-lives of 8 d and 21 h, respectively); this results in significant internal and external doses from β- and γ-radiation (Table 5.4). A thyroid dose of 76 Gy by the two isotopes of iodine is sufficient to cause serious damage to the gland (Belov and Kirshin 1987). Soils of Ukraine and Belarus are naturally low in stable iodine, cobalt and manganese. In conditions of endemic deficiency of stable iodine, the transfer of radioactive iodine from blood to the thyroid gland may be 2 to 3 times greater than normal (Prister et al. 1991). These conditions accentuated the consequences of the accident.

Depressed thyroid function in cattle was related to the dose received (69% reduction in function with a thyroid dose of 50 Gy, and an 82% reduction in animals that received a dose of 280 Gy). The concentration of thyroid hormones in the blood of animals was lower than the physiological norm during the whole lactation period. Radiation damage of the thyroid gland was confirmed by histological studies (i.e., hyperplasia of connective tissue and sometimes adipose tissue, vascular hyperaemia, and necrosis of epithelium). Animals with practically no thyroid tissue were observed in Ukraine. Disruptions of the hormonal status in calves born to cows with irradiated thyroid glands were especially pronounced (Astasheva et al. 1991). Similar effects were observed in cattle evacuated from the Belarussian portion of the 30-km zone (Ilyazov et al. 2002).

TABLE 5.4. DOSES TO CATTLE THAT STAYED IN THE 30-KM ZONE OF CHERNOBYL FROM 26 APRIL TO 3 MAY 1986 (KRYSHEV ET AL. 1992)

Distance from Reactor (km)	Surface Activity (10^8 Bq m^{-2})	Absorbed Dose (Gy)		
		Thyroid	GI Tract	Whole body Internal
3	8.4	300	2.5	1.4
10	6.1	230	1.8	1.0
14	3.5	260	1.0	0.6
12	2.4	180	0.7	0.4
35	1.2	90	0.4	0.2

Although most livestock was evacuated from the area after the accident, several hundred cattle were maintained in the more contaminated areas for a 2–4 month period. By Autumn 1986, some of these animals had died; others showed impaired immune responses, lowered body temperatures and cardiovascular disorders. Hypothyroidism lasted until 1989, and may have been responsible for reproductive failures in animals that received a thyroid dose >180 Gy (Ilyazov et al. 2002). Offspring of highly exposed cows had reduced weight, reduced daily weight gains, and signs of dwarfism. Reproduction returned to normal in the spring of 1989. Hemaetological parameters were normal for animals kept in areas with ^{137}Cs contamination of 0.2 to 1.4 MBq m^{-2} (5 to 40 Ci km^{-2}) (Astasheva et al. 1991).

Chronic radiation damage was observed in over 2000 sheep and 300 horses (3-8 years old), removed from the highly contaminated Khoiniki Raion of Belarus 1.5 years after the accident (Ilyazov et al. 2002). Doses were not estimated. In sheep, a depression of general condition, emaciation, heavy breathing, decrease of temperature and other abnormalities were found. Leukopaenia, erythropaenia, thrombocytopaenia and eosinophilia, increase in blood sugar concentrations 1.5 to 2 times greater than normal, and a significant decrease of thyroid-hormone concentrations compared with normal levels were observed. The offspring weight and fleece clip yield of irradiated sheep were half as much as, or less than, those of healthy individuals. In horses, the damage resulted in depression of general condition, oedema, leukopenia, thrombocytopenia, eosinophilia and myelocytosis. Seventy percent of the animals had thyroid-hormone concentrations in blood serum lower than the detection level of the assay methods (Ilyazov et al. 2002).

Numerous news reports of radiation-induced teratogenesis (birth defects) in cattle and pigs occurred in regions where total doses did not exceed 0.05 Gy y^{-1}. Scientific evidence indicated that increased birth defects were not distinguishable from background frequencies at such low doses (Prister 1999). Additionally, data for 1989 show that livestock birth defects in the contaminated area of the Zhytomir Oblast were no higher than in the uncontaminated areas of the same oblast. Photos of a six-footed calf were widely disseminated and the abnormality was attributed to the accident. The calf, however, was born in June 1986, and thus the process of differentiation and organ formation within the womb was completed prior to the accident. Therefore, this much publicized observation of teratogenesis was caused by factors other than radiation from the Chernobyl accident.

5.6. Radiation effects on other terrestrial animals

Four months after the accident, surveys and autopsies of wildlife and of abandoned domestic animals that remained within the 10-km exclusion zone of Chernobyl were conducted (Krivolutksy et al. 1999). Fifty species of birds were identified, including some rare ones; all appeared normal in appearance and behaviour. No dead birds were found. Swallows and house sparrows were found to be producing progeny that also appeared normal. Forty-five species of mammals from six orders were observed and no unusual appearances or behaviours were noted.

Some wildlife and domestic animals were shot and autopsied in August and September 1986. Dogs and chickens showed signs of chronic radiation syndrome (reduced body mass; reduced fat reserves; increase in mass of lymph nodes, liver and spleen; haematomas present in liver and spleen; and thickening of the lining of the lower intestine). No eggs were found in the nests of chickens, nor eggs in their ovaries.

During Fall 1986, the number of small rodents on highly contaminated research plots decreased by a factor of 2 to 10. Estimates of absorbed dose during the first five months after the accident ranged from 12 to 110 Gy for gamma and 580 to 4500 Gy for beta irradiation. Numbers of animals were recovering by Spring 1987, mainly due to immigration from less affected areas. In 1986 and 1987, the percentage of pre-implantation deaths in rodents from the highly contaminated areas increased 2- to 3-fold compared with controls. Resorption of embryos also increased markedly in rodents from the impacted areas; however, the number of progeny per female did not differ from controls (Taskaev and Testov 1999).

5.7. Radiation effects on aquatic organisms

Cooling water for the Chernobyl nuclear power plant was obtained from a 21.7 km^2 man-made reservoir located southeast of the plant site. The cooling reservoir became heavily contaminated following the accident (see Section 2.5 for details) with over $6.5 \pm 2.7 \times 10^{15}$ Bq of a mixture of radionuclides in the water and sediments (Kryshev 1995). Aquatic organisms were exposed to external irradiation from radionuclides in water, contaminated bottom sediments, and irradiation from contaminated aquatic plants. Internal irradiation occurred as organisms took up radioactively contaminated food and water or inadvertently consumed contaminated sediments. The resultant doses to aquatic biota over the first 60 days following the accident are depicted in Figure 5.5.

The maximum dose rates for aquatic organisms (excluding fish) were reported in the first two weeks after the accident, when short-lived isotopes (primarily ^{131}I) contributed 60 to 80% of the dose. During the second week, the contribution of short-lived radionuclides to doses of aquatic organisms decreased by a factor of two. Maximum dose rates to fish were delayed (Fig. 5.5) due to the time required for their food webs to become contaminated with longer-lived radionuclides (largely $^{134,137}Cs$, $^{144}Ce/Pr$, $^{106}Ru/Rh$ and $^{90}Sr/Y$). Differences in dose rates among fish species occurred due to their trophic positions. Non-predatory fish (carp, gold fish, bleak) reached estimated peak dose rates of 3 mGy d^{-1} from internal contamination in 1986, followed by significant reductions in 1987. Dose rates in predatory fish (perch), however, increased in 1987 and did not start to decline until 1988 (Kryshev et al. 1992). Accumulated doses were greatest for the first generation of fish born in 1986 and 1987. Bottom-dwelling fish (gold fish, silver bream, bream, carp) that received significant irradiation from the bottom sediments attained accumulated total doses of approximately 10 Gy.

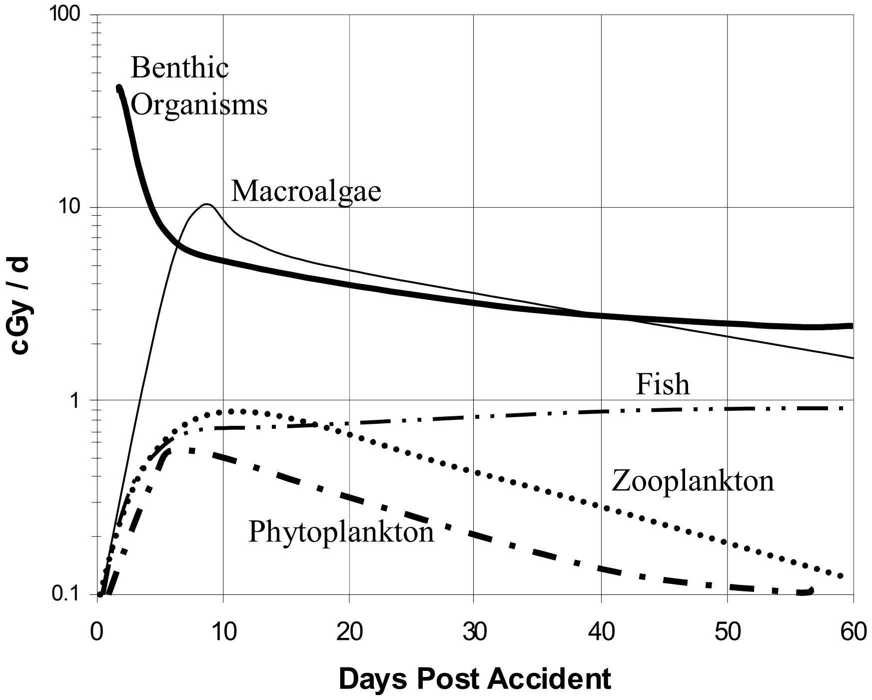


FIG. 5.5. The dynamics of absorbed dose rate (cGy d^{-1}) of organisms within the Chernobyl reactor cooling pond during the first 60 days following the accident. Data are model results based on concentrations of radionuclides in the water column and lake sediments (adapted from Kryshev et al. 1992).

TABLE 5.5. CHRONIC EFFECTS OF IONIZING RADIATION ON REPRODUCTION IN FISH, DERIVED FROM THE FASSET DATABASE (COPPLESTONE ET AL. 2003)

Dose Rate (μGy h^{-1})	Dose Rate (mGy d^{-1})	Reproductive Effects
0 - 99	0 – 2.4	– Background dose group, normal cell types, normal damage and normal mortality observed
100 – 199	2.4 – 4.8	– No data available
200 – 499	4.8 – 12	– Reduced spermatogonia and sperm in tissues
500 – 999	12 – 24	– Delayed spawning, reduction in testes mass
1,000 – 1,999	24 – 48	– Mean lifetime fecundity decreased, early onset of infertility
2,000 – 4,999	48 – 120	– Reduced number of viable offspring – Increased number of embryos with abnormalities – Increased number of smolts in which sex was undifferentiated – Increased brood size reported – Increased mortality of embryos
5,000 – 9,999	120 – 240	– Reduction in number of young fish surviving to 1 month of age – Increased vertebral abnormalities
> 10,000	> 240	– Interbrood time tends to decrease with increasing dose rate – Significant reduction in neonatal survival – Sterility in adult fish – Destruction of germ cells within 50 days in medaka fish – High mortality of fry, germ cells not evident – Significant decrease in number of male salmon returning to spawn - After 4 y, female salmon had significantly reduced fecundity

In 1990 the reproductive capacity of young silver carp was analyzed (Ryabov 1992). The fish were in live boxes within the cooling pond at the time of the accident. By 1988, the fish reached sexual maturity. Over the entire post-accident period they received a dose of 7-8 Gy. Biochemical analyses of muscles, liver and gonads indicated no difference from controls. The amount of fertilized spawn was 94%, 11% of the developing spawn were abnormal. Female fertility was 40% higher than the controls, but 8% of the irradiated sires were sterile. The level of fluctuating asymmetry in offspring did not differ from the controls, although the level of cytogenetic damage (22.7%) significantly exceeded controls (5-7%). In contrast, Pechkurenkov (1991) reported that the number of cells with chromosome aberrations in 1986-1987 in carp, bream flat and silver carp was within the norm. It is worth noting that the cooling pond was subject not only to radioactive contamination, but also to chemical pollution.

Recent reviews of chronic effects of ionizing radiation on reproduction in fish, with the Chernobyl data included (Table 5.5), have been summarized.

5.8. Genetic effects in animals and plants

Quality data are relatively sparse concerning the incidence of Chernobyl-related induced mutations in plants and animals. An increased mutation level was apparent in 1987 in the form of various morphological abnormalities observed in plants of Canada flea-bane, common yarrow, and mouse millet. Examples of abnormalities include unusual branching of stems; doubling the number of racemes; abnormal colour and size of leaves and flowers; and development of “witch’s brooms” in pine trees. Similar effects in the 5-km zone near the reactor also appeared in deciduous trees (leaf gigantism, changes in leaf shapes; Fig 5.6). Morphological changes were observed at an initial gamma-exposure rate of 20-30 mR h^{-1}. At 75-150 mR h^{-1} enhancement of vegetative reproduction (heather) and gigantism of some plant species were observed (Arkhipov et al. 1994; Kozubov and Taskaev 1994; Tikhomirov and Shcheglov 1994; Tikhomirov et al. 1993).

Cytogenetic analysis of cells from the root meristem of winter rye and wheat germ of the 1986 harvest demonstrated a dose dependency in the number of aberrant cells. A significant excess over the control level of aberrations was observed at an absorbed dose of 3.1 Gy, inhibition of mitotic activity occurred at 1.3 Gy, and germination was reduced at 12 Gy (Geraskin et al. 2003). The analysis of three successive generations of winter rye and wheat on the most contaminated plots revealed that the rate of aberrant cells in the intercalary meristem in the second and third generations were higher than in the first.

From 1986 to 1992, mutation dynamics were studied in populations of *Arabidopsis thaliana Heynh.* (L.) within the 30-km zone (Abramov et al. 1992). On all study plots in the first 2-3 years after the accident *Arabidopsis* populations exhibited an increased mutation burden. In later years, the level of lethal mutations declined; nevertheless the mutation rate in 1992 was still 4-8 times higher than the spontaneous level. The dose dependence of the mutation rate was best approximated by a power function with a power index of less than 1.

Zainullin et al. (1992) observed elevated levels of sex-linked recessive lethal mutations in natural *Drosophila melanogaster* populations living under conditions of increased background radiation due to the Chernobyl accident. The mutation levels were increased in 1986-1987 in flies inhabiting the more contaminated areas with initial exposure rates of 200 mR h^{-1} and more. In the subsequent two years mutation frequencies gradually returned to normal.

FIG. 5.6. Typical morphological abnormality seen on conifer trees. Such enhancement of vegetative growth and gigantism of some plant parts were not uncommon (photo: T. Hinton, 1991).

Studies of adverse genetic effects in wild mice were reported by Shevchenko et al. (1992) and Pomerantseva et al. (1997). These involved mice caught during 1986-1991 within a 30-km radius of the Chernobyl reactor with different levels of gamma radiation and in 1992-1993 on a site in the Bryansk Oblast, Russia. The estimated total doses of gamma and beta radiation varied widely and reached 3-4 Gy per month in 1986-1987. One endpoint was dominant lethality, measured by embryo mortality in the offspring of wild male mice mated to unexposed female laboratory mice. The dominant lethality rate was elevated for a period of a few weeks following capture in mice sampled at the most contaminated site. At dose rates of about 2 mGy h^{-1}, two of 122 captured males produced no offspring and were assumed to be sterile. The remainder showed a period of temporary infertility and reduced testes mass, which, however, recovered with time after capture.

The frequencies of reciprocal translocations in mouse spermatocytes were consistent with previous studies. For all collected mice, a dose-rate-dependent incidence of increased reciprocal translocations (scored in spermatocytes at meiotic metaphase I) was observed. The frequency of mice harboring recessive lethal mutations decreased with time post-accident (Pomerantseva et al. 1997). Radiation-related gene mutation is unlikely to have any adverse effect on populations, at the dose rates that prevail now.

Advances in the sophistication and associated technologies of detecting molecular and chromosomal damage have occurred since the early genetic studies prior to the Chernobyl accident. Such advances have allowed researchers on the genetic consequences of the Chernobyl accident to examine endpoints not previously considered. Most prominent, and controversial, is the mutation frequencies in repeat DNA sequences termed ‘minisatellite loci’ or ‘expanded simple tandem repeats’ (ESTR). These are repeat DNA sequences that are distributed throughout the germline and that have a high background (spontaneous) mutation rate. Presently, ESTRs are considered to have no function, although this is a matter of much interest and discussion (ICRP 2003; CERRIE 2004). Minisatellite mutations have only rarely been associated with recognisable genetic disease (Bridges 2001).

Although laboratory examination of mutations in mouse ESTR loci show clear evidence of a mutational dose-response (Fan et al. 1995; Dubrova et al. 1998), to our knowledge no convincing data on elevated levels of minisatellite mutations in plants or animals residing in the Chernobyl-affected areas have been published so far in peer-reviewed scientific literature. In general, quantitative interpretation of the ESTR data is difficult because of conflicting findings, their weak association with genetic disease, dosimetric uncertainties and methodological problems (CERRIE 2004). This is an area of science that requires additional research.

5.9. Secondary impacts and current conditions

Prior to the accident, much of the area around Chernobyl was covered in 30- to 40-year old pine stands that, from a successional standpoint, represented mature, stable ecosystems. The high dose rates from ionizing radiation during the first few weeks following the accident altered the balanced community by killing sensitive individuals, altering reproduction rates, destroying some resources (e.g., pine stands), making other resources more available (e.g., soil water), and opening niches for immigration of new individuals. All these components, and many more, were interwoven in a complex web of action and reaction that altered populations and communities of organisms.

Irradiation is an environmental stress, in many ways similar to other environmental stresses, such as pollution by metals or the destruction caused from forest fires. If such stressors are sufficient, the community organization is changed and generally reverts to an earlier successional state. However, when the stress is subsequently reduced and sufficient time passes, recovery occurs, and the ecosystem again regains stability advancing toward a more mature state. The change in species diversity observed within the soil invertebrate communities, presented above, is perhaps the most obvious published example of community-level change and subsequent recovery following the Chernobyl accident. The death of pine stands close to the Chernobyl reactor and the subsequent establishment of grasslands and deciduous trees are striking visual examples.

Age and sex distributions, diversity, abundance, and gross physiological conditions of small mammal populations in the 30-km zone appear to be similar to background locations in other parts of Ukraine (Baker et al. 1996; Baker and Chesser 2000; Jackson et al. 2004). Reports on the current genetic conditions of rodents within the zone are contradicting. For example, Shevchenko et al. (1992) found significant disorders in spermatogenesis, while Baker et al. (1996) found no reproductive inhibition or germ-line mutations.

Layered on top of the impacts from the irradiation was the abrupt and drastic change that occurred when humans were removed from the 30-km zone. The town adjacent to the Chernobyl reactor, Pripyat, was abandoned when over 50,000 people were evacuated. Agricultural activity, forestry, hunting and fishing within the 30-km zone were stopped, because of the radioactive contamination of the products. Only activities designed to mitigate the consequences of the accident were carried out, as well as those supporting the living conditions of the clean-up workers, including substantial road construction.

FIG. 5.7. Wild boar (above) and wolf (below) inhabiting the Chernobyl Exclusion Zone are not afraid of people because of long term hunt prohibition. Photos – courtesy of Sergey Gaschak, 2004.

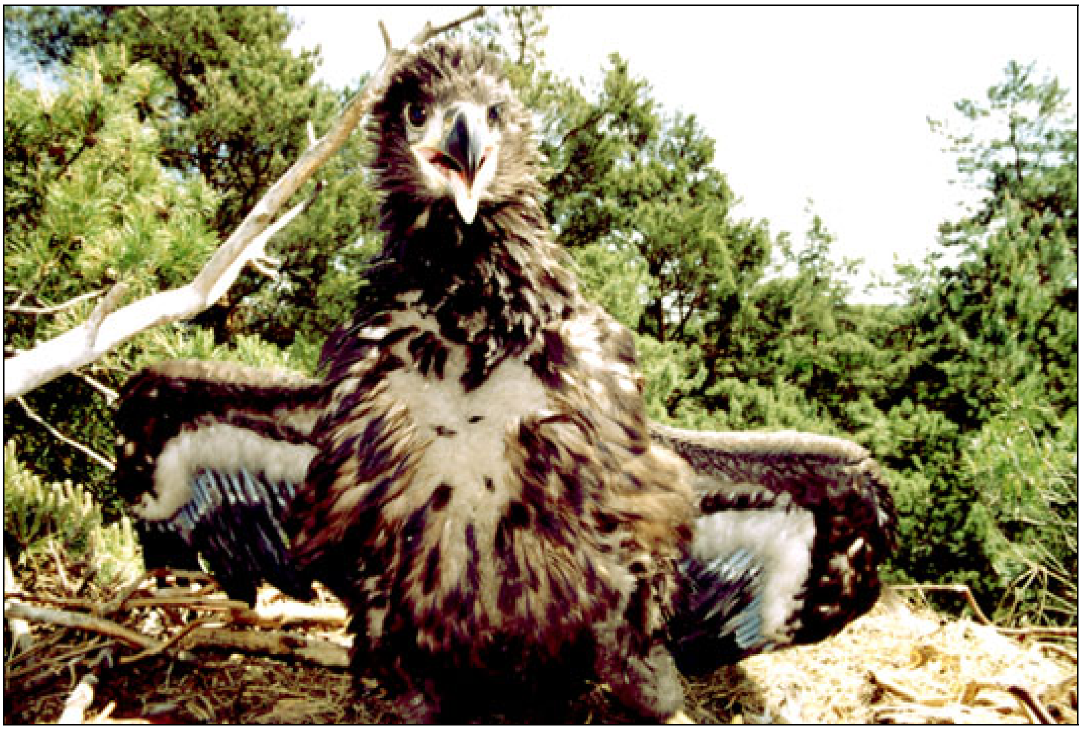


FIG. 5.8. A white-tailed eagle chick observed recently in the Chernobyl Exclusion Zone. Before 1986, these rare predatory birds had rarely been found in this area. Photo – courtesy of Sergey Gaschak, 2004.

For some years after the accident, the agricultural fields still yielded domesticated produce, and many animal species, especially rodents and wild boars, consumed the abandoned cereal crops, potatoes and grasses as an additional source of forage. This advantage, along with the special reserve regulations established in the exclusion zone (e.g., a ban on hunting), tended to compensate for the adverse biological effects of radiation, and promoted an increase in the populations of wild animals. Significant population increases of game mammals (wild boars, roe deer, red deer, elk, wolfs, foxes, hares, beaver, etc., - Fig. 5.7) and bird species (black grouse, ducks, etc.) were observed soon after the Chernobyl accident (Gaichenko et al. 1990; Sushenya et al. 1995).

More than 400 species of vertebrate animals, including 67 ichthyoids, 11 amphibians, 7 reptilians, 251 birds and 73 mammals inhabit the territory of the evacuated town of Pripyat and its vicinity; more than fifty of them belong to a list of those protected according with to national Ukrainian and European Red Books. The Chernobyl Exclusion Zone has become a breeding area of white-tailed eagle, spotted eagle, eagle owl, crane and black stork – Fig. 5.8 (Gaschak et al. 2002).

In the Pripyat River floodplain, a developed system of artificial drainage channels now supports about a hundred families of beavers. Recognising the value of the abandoned land around Chernobyl, 28 endangered Przhevalsky wild horses were introduced in 1998. After six years their number doubled (Gaschak et al. 2002). In both the Ukrainian and Belarusian parts of the exclusion zone, State radioecological reserves have been created with a regime of nature protection.

As has been shown many times before, when humans are removed--nature flourishes. This phenomenon exists in U.S. National Parks such as Yellowstone and the Grand Tetons, and at large U.S. Department of Energy sites where the general public has been excluded for over 50 years. Human presence in any environment is a disturbance to the natural biota. Normal activities of farming, hunting, logging, and road building, to name but a few, fragment, pollute, and generally stress the processes and mechanisms of natural environments. The removal of humans alleviates one of the more persistent and ever-growing stresses experienced by natural ecosystems.

On the other hand, the absence of forest management, and the associated increase in forest fires have substantial impacts to natural communities. After the human population was evacuated, both wood cutting and the construction of mineralized fire-prevention strips ceased. The number of dead trees increased and created conditions that enhanced the development of forest diseases and pests (borers, bark beetles, etc.). The amount of dead-wood and brushwood has gradually increased in the unmanaged forests. The degradation of the forests resulted in enormous forest fires during the dry summer period in 1992 when the area of the burnt forests amounted to 170 km^2 (i.e., about one-sixth the woodlands; Frantsevich et al. 1995).

Without a permanent residency of humans for 20 years, the ecosystems around the Chernobyl site are now flourishing. The 30-km zone has become a wildlife sanctuary (Baker and Chesser 2000), and it looks like the nature park it has become.

5.10. Conclusions and recommendations

5.10.1. Conclusions

(1) Irradiation from radionuclides released from the Chernobyl accident caused numerous acute adverse effects in the biota located in areas of higher exposure, i.e., up to a distance of few tens of kilometres from the release point. Beyond the exclusion zone, no acute radiation-induced effects on biota have been reported.

(2) The environmental response to the Chernobyl accident was a complex interaction among radiation dose, dose rate and its temporal and spatial variations, as well as the radiosensitivities of the different taxons. Both individual and population effects caused by radiation-induced cell death have been observed in plants and animals as follows:

- Increased mortality of coniferous plants, soil invertebrates and mammals;
- Reproductive losses in plants and animals; and
- Chronic radiation syndrome of animals (mammals, birds, etc.).

No adverse radiation-induced effect has been reported in plants and animals exposed to a cumulative dose of less than 0.3 Gy during the first month after the accident.

(3) Following the natural reduction of exposure levels due to radionuclide decay and migration into soil, populations have been recovering from acute radiation effects. By the next growing season following the accident, population viability of plants and animals substantially recovered as a result of the combined effects of reproduction and immigration. A few years were needed for recovery from major radiation-induced adverse effects in plants and animals.

(4) The acute radiobiological effects observed in the Chernobyl-accident area are consistent with radiobiological data obtained in experimental studies or observed in natural conditions in other areas affected by ionising radiation. Thus, rapidly developing cell systems such as meristems of plants and insect larva were predominantly affected by radiation. At the organism level, the young plants and animals have been found to be more sensitive to acute effects of radiation.

(5) Genetic effects of radiation, in both somatic and germ cells, have been observed in plants and animals of the exclusion zone during the first few years after the Chernobyl accident. Both in the exclusion zone, and beyond, different cytogenetic anomalies attributable to radiation continue to be reported from experimental studies performed on plants and animals. Whether the observed cytogenetic anomalies in somatic cells have any detrimental biological significance is not known.

(6) The recovery of affected biota in the exclusion zone has been confounded by the overriding response to the removal of human activities, e.g., termination of agricultural and industrial activities accompanied with the environmental pollution in the more affected areas. As a result, populations of many plants and animals have eventually expanded, and the present environmental conditions have positive impact on the biota in the exclusion zone.

5.10.2. Recommendations for future research

(1) In order to develop further a system of environmental protection against radiation, the long-term impact of radiation on plant and animal populations should be further investigated in the exclusion zone of the Chernobyl accident; this is a unique area for radioecological and radiobiological research in an otherwise natural setting.

(2) In particular multigenerational studies of the recently identified radiobiological phenomena of genome instability and of the radiation effect on the genetic structure of plant and animal populations might bring fundamentally new scientific information.

(3) There is a need to develop standardised methods for dose reconstruction for biota, e.g., in the form of a unified dosimetric protocol.

5.10.3. Recommendations for countermeasures/remediation

(1) Protective actions for the farm animals in case of a nuclear or radiological emergency should be developed based on the Chernobyl experience and internationally harmonised.

(2) There is nothing that can be done to remedy the radiological conditions of the exclusion zone of the Chernobyl NPP that would not have adverse impacts to plants and animals.

REFERENCES TO SECTION 5

ABRAMOV, V.I., FEDORENKO, O.M. and SHEVCHENKO, V.A. (1992) Genetic consequences of radioactive contamination for populations of *Arabidopsis.* Sci. Total Environ. 112: 19-28.

ARKHIPOV, N.P., KUCHMA, N.D., ASKBRANT, S., PASTERNAK, P.S., MUSICA, V.V. (1994) Acute and long-term effects of irradiation on pine (*Pinus Silvestris*) stands post-Chernobyl. Sci. Total Environ. 157: 383-386.

ASTASHEVA, N.P., LAZAREV, N.M., KHRAMTSOVA, L.K., et al. (1991) Influence of radiation released during the Chernobyl NPP accident on clinical and physiological status of agricultural animals. In: Problems of agricultural radiology. UIAR, Kiev, pp.176-180 (in Russian).

BAKER, R. J., HAMILTON M. J., VAN DEN BLUSSCHE R. A., WIGGINS L. E., SUGG D. W, SMITH, M. H., LOMAKIN M. D., GASCHAK S. P, BUNDOVA E. G., RUDENSKAYA G. A, and CHESSER R. K. (1996) Small mammals from the most radioactive sites near the Chornobyl nuclear power plant. J. Mammology. 77: 155-170.

BAKER R. J. and CHESSER R. K. (2000) The Chernobyl nuclear disaster and subsequent creation of a wildlife preserve. Environ. Toxicol. Chem. 19: 1231-1232.

BAKRI A, HEATHER N, HENDRICHS J, AND FERRIS I. (2005) Fifty years of radiation biology in entomology: Lessons learned from IDIDAS. Annals of the Entomological Society of America, 98:1-12.

BELOV, A.D. and KIRSHIN, V.A. (1987) Veterinary radiobiology. M: Atomizdat, 287 p. (in Russian).

BRIDGES, B. A. (2001) Radiation and germline mutations at repeat sequences: are we in the middle of a paradigm shift? Radiat. Res. 156: 631-641.

CERRIE (2004) Report of the Committee Examining Radiation Risks of Internal Emitters. www.cerrie.org. ISBN 0-85951-545-1, London.

COPPLESTONE, D., ZINGER, I. AND JACKSON, D. (2003) The challenge of protecting non-human biota from exposure to ionising radiation. In: SRP 40th Anniversary Meeting: ALARP: Principles and Practices, St. Catherine's College, Oxford. Dubrova, Y. E., Jeffreys, A. J., and Malashenko, A. M. (1993) Mouse minisatellite mutations induced by ionising radiation. Nat. Genet 5: 92-94.

DUBROVA, Y.E, PLUMB, M., BROWN, J., FENNELLY, J., BOIS, P. AND GOODHEAD, D. (1998) State specificity, dose response, and doubling dose for mouse minisatellite germ-line mutation induced by acute radiation. Proc. Natl. Acad. Sci USA 95: 6251-6265.

FAN, Y. J., WNAG, Z., SADAMOTO, S., NINOMIYA, Y., KOTOMURA, N. KAMIYA, K., DHI, K., KOMIANAMI R. and NIWA O. (1995) Dose response of a radiation induction of a germline mutation at hyper variable mouse ministatellite locus. Int. J. Radiat. Biol. 68: 177-183.

FRANTSEVICH, L.I., DIDUKH, YA.P., et al., (1995) Secondary ecological changes associated with resettlement of population. In: The Chernobyl Disaster, ed. by Barjakhtar, V.G., Naukova Dumka, Kiev, pp.335-340 (in Ukrainian).

GAICHENKO, V. A., KRYZHANOVSKY, V. I., STOVBCHATY, V. N. et al. (1990) Ecological situation in the 30-km zone of Chernobyl NPP and its alteration over last 3 post-accidental years. In: Reports of the Second All-Union scientific and technical meeting on the results of elimination of consequences of the accident at the Chernobyl NPP. Chernobyl, 6 (3): 4-11 (in Russian).

GASCHAK, S. P., ZALISSKY, A. A., VISHNEVSKY, D. A. et al. (2002) Fauna of vertebrates in the Chornobyl zone of Ukraine. Preprint of the International Chornobyl center for nuclear safety, radioactive waste and radioecology, Slavutych, 80 p (in Ukrainian).

GERASKIN, S.A., DIKAREV, V.G., ZYABLITSKAYA, E.YA., OUDALOVA, A.A., SPIRIN, E.V. and ALEXAKHIN, R.M. (2003) Genetic consequences of radioactive contamination by the Chernobyl fallout to agricultural crops. J. Environ. Radioact. 66: 155-169.

GRODZINSKY, D.M., KOLOMIETS, O.D., KUTLAKHMEDOV, YU.A. (1991) Anthropogenic radionuclide anomaly and plants. Lybid, Kiev, 160 p. (in Russian).

GRODZINSKY, D.N. AND GUDKOV, I.N. (2001) Radibiological effects on plants in the contaminated territory. In: The Chornobyl Exclusion zone. Nat. Acad. of Sci. Ukraine, Kiev, pp.325-377.

INTERNATIONAL ATOMIC ENERGY AGENCY (1976) Effects of ionising radiation on aquatic organisms and ecosystems. Technical Reports Series No. 172, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1988) Assessing the impact of deep sea disposal of low level radioactive waste on living marine resources. Technical Reports Series No. 288, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (1992) Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards, Technical Reports Series No. 332. IAEA, Vienna.

INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (2003) Biological effects after prenatal Irradiation (Embryo and Fetus), ICRP Publication 90, Annals of the ICRP, 33 Nos.1-2.

ILYAZOV, R.G., SIROTKIN, A.N., et al. (2002) Ecological and radiobiological consequences of the Chemobyl catastrophe for stock-breeding and ways of its overcoming. Ed. by Ilyazov, R.G., Fan, Kazan, 330 p. (in Russian).

JACKSON D. COPPLESTONE D. and STONE D. M. (2004). Effects of chronic radiation exposure on small mammals in the Chernobyl Exclusion Zone. Nuclear Energy, 43 (5) 281-287.

KOZUBOV, G., TASKAEV, A. et al. (1990) radiation exposure of the coniferous forest in the area exposed to the Chernobyl contamination. Komy Scientific Centre of the Academy of Sciences, Syktyvkar (in Russian).

KOZUBOV G.M., TASKAEV A.I. (1994) Radiobiological and radioecological investigations of woody plants. St. Peterburg: Nauka, 256 p (in Russian).

KRIVOLUTSKY, D. AND POKARZHEVSKY, A. (1992) Effect of radioactive fallout on soil animal populations in the 30 km zone of the Chernobyl atomic power station. Sci. Total Environ., 112, pp. 69-77.

KRIVOLUTSKY, D., MARTUSHOV, V. and RYABTSEV, I. (1999) Influence of radioactive contamination on fauna in the area of the Chernobyl NPP during first years after the accident (1986-1988). In: Bioindicators of Radioactive Contamination, Nauka, Moscow, pp. 106-122 (in Russian).

KRYSHEV, I., ALEXAKHIN, R., MAKHONKO, K. ET AL, (1992) Radioecological Consequences of the Chernobyl Accident. Nuclear Society, Moscow.

KRYSHEV, I. (1995) Radioactive contamination of aquatic ecosystems following the Chernobyl accident. J. Environ. Radioact., 227, pp.207-219.

PECHKURENKOV, V.L. (1991) The influence of the Chernobyl disaster on fish populations in a cooling pond. Radiobiologiya 31: 704-708 (in Russian).

POMERANTSEVA, M.D., RAMAIYA, L.K. AND CHEKHOVICH, A.V. (1997) Genetic disorders in house mouse germ cells after the Chernobyl catastrophe. Mutat. Res. 381: 97-103.

PRISTER, B.S. (1999) Consequences of the accident at the Chemobyl NPP for agriculture of Ukraine. Center of privatisation and economic reform, v. 20, Kiev, 103 p (in Russian).

PRISTER, B.S., SHEVCHENKO, V.A., KALCHENKO, V.A. (1982) Genetic effects of radionuclides on agricultural crops. In: Progress of modem genetics. USSR Academy of Science, Moscow, pp.138-148.

PRISTER, B.S., LOSCHILOV, N.A., NEMETS, O.F. AND POYARKOV, V.A., (1991) Fundamentals of agricultural radiology, 2nd Edition, Urozhay, Kiev, 472 p. (in Russian).

REAL, A., SUNDELL-BERGMAN, S., KNOWLES, J. F., WOODHEAD, D. S. and ZINGER, I. (2004) Effects of ionising radiation exposure on plants, fish and mammals: relevant data for environmental radiation protection. J. Radiol. Prot., 24, A123–A137.

RYABOV, I.N. (1992) Effect of radioactive contamination on hydrobionts within the thirty-kilometer zone of the Chernobyl NPP. Radiobiologiya, 32: 662-667 (in Russian).

SAZYKINA, T.G., JAWORSKA, A. and BROWN, J.E. (Eds.) (2003). Dose-effect relationships for reference (or related) Arctic biota. Deliverable Report 5 for the EPIC project (Contract no. ICA2-CT-200-10032). Norwegian Radiation Protection Authority, Østerås, 119 p.

SHEVCHENKO, V.A., POMERANTSEVA, M.D., RAMAIYA, L.K., et al. (1992) Genetic disorders in mice exposed to radiation in the vicinity of the Chernobyl nuclear power station. Sci. Total Environ. 112: 45-56.

SHEVCHENKO, V.A., ABRAMOV, V.I., KALCHENKO, V.A. et al. (1996) Genetic consequences of radioactive contamination of the environment after the Chemobyl accident for populations of plants. Radiation biology. Radioecology, 36, pp. 531-545.

SPARROW, A. H. (1961) Correlation of nuclear volume and DNA content with higher plant tolerance to chronic radiation. SCIENCE 134: 282-284.

SUSCHENYA L.M. et al. (editors) (1995) Animal kingdom in the accident zone of Chernobyl NPP. Minsk: Navuka i technica, 263 p. (in Russian).

TASKAEV, A. AND TESTOV, B., (1999). Number and reproduction of mouse-like rodents in the Chernobyl accident area. In: Bioindicators of Radioactive Contamination, Nauka, Moscow, pp. 200-205 (in Russian).

TIKHOMIROV, F.A., SHCHEGLOV, A.I., SIDOROV, V.P. (1993) Forests and forestry: radiation protection measures with special reference to the Chernobyl accident zone. Sci. Total Environ. 137: 289-305.

TIKHOMIROV, F.A. and SHCHEGLOV, A.I. (1994). Main investigation results on the forest radioecology in the Kyshtym and Chernobyl accidents zones. Sci. Total Environ. 157: 45-47.

UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION (1996) Sources and Effects of Ionizing Radiation. Scientific Annex within 1996 UNSCEAR Report to the General Assembly, NY.

UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION (2000) Sources and Effects of Ionizing Radiation. Scientific Annex J within 2000 UNSCEAR Report to the General Assembly, NY.

WHICKER, F. W. and FRALEY, L. (1974) Effects of ionizing radiation on terrestrial plant communities. Adv. Radiat. Biol. 4:317-366.

WHICKER, F. W. and SCHULTZ, V. (1982) Radioecology: Nuclear Energy and the Environment. CRC Press, Boca Raton, Fl.

WHICKER, F. W. (1997) Impacts on plant and animal populations. In: Health Impacts of Large Releases of Radionuclides. CIBA Foundation Symposium 203. John Wiley & Sons, NY, pp 74-93.

WOODHEAD, D. and ZINGER, I. (2003) Radiation effects on plants and animals. FASSET Deliverable 4 to the Project "FASSET" Framework for the assessment of Environmental Impact, contract No FIGE-CT-2000-00102. Swedish Radiation Protection Authority.

ZAINULLIN, V.G., SHEVCHENKO, V.A., MYASNYANKINA, E.N., GENERALOVA, M.V. and RAKIN, A.O. (1992) The mutation frequency of *Drosophila melanogaster* populations living under conditions of increased background radiation due to the Chernobyl accident. Sci. Total Environ. 112: 37-44.

6. ENVIRONMENTAL ASPECTS OF DISMANTLEMENT OF THE SHELTER AND RADIOACTIVE WASTE MANAGEMENT

The destruction of the Unit 4 reactor at the Chernobyl Nuclear Power Plant (ChNPP) resulted in the generation of radioactive contamination and radioactive waste[4] in the Unit, the ChNPP site and surrounding area (further referred to as Exclusion Zone). The future development of the Exclusion Zone depends on the future strategy for conversion of Unit 4 into an ecologically safe system, i.e., the development of a New Safe Confinement (NSC), the dismantlement of the current Shelter, removal of fuel-containing material (FCM), and eventual decommissioning of the reactor site.

In particular, the envisaged long-term strategy for Unit 4 rests on implementation of the NSC concept, to cover the unstable Shelter, and also on the related radioactive waste-management activities at the ChNPP site and the Exclusion Zone. Currently Units 1, 2, and 3 (1000 MW RBMK reactors) are shutdown with a view to be decommissioned, and two additional reactors (Units 5 and 6) that had been near completion were abandoned in 1986 following the accident.

This Section addresses the current status and expected future environmental aspects associated with Unit 4 and the management of radioactive waste from the accident at the ChNPP site and the Exclusion Zone.

6.1. Current status and the future of Unit 4 and the Shelter

6.1.1. ChNPP Unit 4 after the accident

In the course of the 1986 accident, a small part of the nuclear fuel (3.5% according to past estimates (UNSCEAR 2000) or 1.5% according to recent estimates (Kashparov et al. 2003) and a substantial fraction of volatile radionuclides, see Section 2.1, was released from damaged Unit 4. The remainder of damaged nuclear fuel, more than 95% of the fuel mass at the moment of the accident equal to 190 t of uranium, i.e., about 180 t, left in the remains of the reactor (UNSCEAR 2000). Uncertainty of this estimate is discussed below in Subsection 6.1.5.

The first measures taken after the accident to control the fire and the radionuclide releases consisted of dumping neutron-absorbing compounds and fire-control material into the crater formed by the destruction of the reactor (UNSCEAR 2000) (see Fig. 6.1). The total amount of material dumped on the reactor was approximately 5,000 t, including about 40 t of boron compounds, 2400 t of lead, 1800 t of sand and clay, and 600 t of dolomite, as well as sodium phosphate and polymer liquids (UNSCEAR 2000).

The radiation conditions at the ChNPP site in mid-May 1986 could be described as high level air-dose rate and air-activity concentration caused by relatively uniform area contamination by finely dispersed nuclear fuel and aerosols of short-lived radionuclides, as well as the presence of dispersed nuclear fuel particles or fragments. These fragments consisted of discrete and non-uniform contamination from the reactor core, reactor constructional material, and graphite.

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[4] Radioactive waste at the Exclusion Zone refers to the waste excluding the waste associated with the decommissioning of ChNPP units 1, 2, 3.

FIG .6.1. Destroyed reactor after the accident in 1986.

After the accident the debris of the destroyed reactor building was collected along with fragments of the reactor core, etc., and the soil-surface layer. Thousands of cubic meters of radioactive waste generated by this work were disposed in the Pioneer Wall and Cascade Wall (Fig. 6.2). Construction of the walls around the damaged reactor reduced dose-rate levels by a factor of 10 to 20 (Borovoy et al. 1999). The completion of the Pioneer Wall and the Cascade Wall (Fig. 6.2) and a significant reduction in radiation levels allowed construction of the Shelter.

The Shelter, which was aimed at environmental containment of the damaged reactor, was erected in an extremely short period of time between May and November 1986 under conditions of severe radiation exposure to the personnel (Fig. 6.2). Steps taken to save time and cost during the construction, and the high dose rates inside the structure, led to lack of reliable and comprehensive data on the stability of the damaged older structures to support current needs; the need for remote control concreting; and the impossibility of using welding in some specific situations.

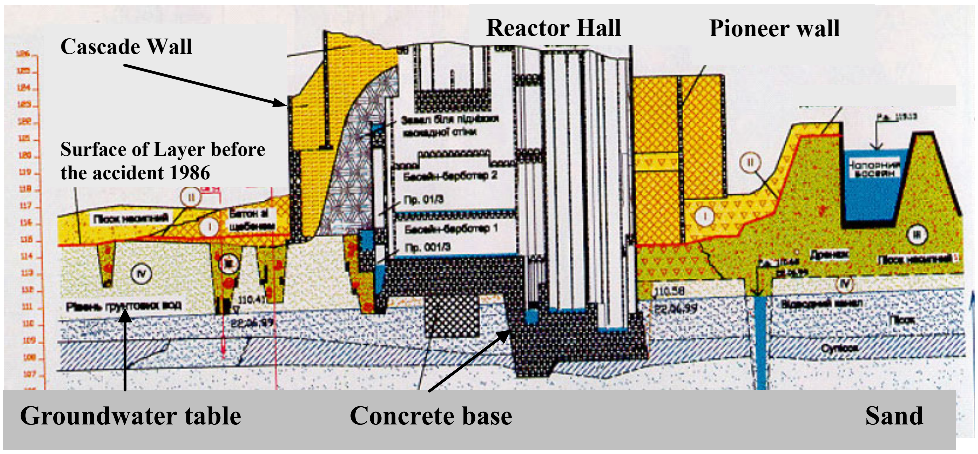

FIG. 6.2. Cross-section of the Chernobyl Unit 4 and the Shelter.

FIG. 6.3. Opening at the "Shelter" allowing infiltration of atmospheric water.

6.1.2. Current status of damaged Unit 4 and Shelter

A Shelter (Novokschenov 2002) was designed and constructed using steel beams and plates as structural elements. Its foundation rests at some points on the original structural elements of Unit 4, whose structural integrity following the accident is not well known, and at other points it rests on debris remaining from the accident. Thus, the ability of the Shelter structure to withstand natural events, such as earthquakes and tornados, is known only with large uncertainties. In addition to uncertainties in structural stability at the time of its construction, structural elements of the Shelter have degraded as a result of moisture-induced corrosion during the nearly twenty years since the accident.

The Shelter has approximately 1000 m^2 of openings in its surface. These openings allow approximately 2000 m^3 y^{-1} of precipitation to percolate through the radioactively contaminated debris and eventually pool in rooms in the lower levels of Unit 4 (see Fig. 6.3) (Borovoy 2001). Condensation within Unit 4 of approximately 1650 m^3 y^{-1} of water and periodic spraying of 180 m^3 y^{-1} of liquid-dust suppressant contributes to the quantities of water percolating through the Unit 4 debris and collecting in its basement. The collected water is contaminated with ^{137}Cs, ^{90}Sr, and transuranic elements resulting in average concentrations of 1.6×10^{10} Bq m^{-3} of ^{137}Cs, 2.0×10^9 Bq m^{-3} of ^{90}Sr, 1.5×10^5 Bq m^{-3} of Pu, and 6 mg L^{-1} of uranium. About 2100 m3 y^{-1} of the collected water evaporates, and about 1300 m^3 y^{-1} leaks through the foundation into the soil beneath Unit 4 (EIA 2003). The existing ChNPP radioactive waste-management system is not capable of treating liquid radioactive waste that contains transuranic elements.

The inside conditions of Unit 4 (Fig. 6.4) are hazardous and present significant risks to workers and the environment. General area radiation-dose rates range from 2 μSv h^{-1} to 0.1 Sv h^{-1} inside the Shelter (Borovoy 2001). Workers' individual occupational radiation exposures during current operations at Unit 4 are controlled so they do not exceed the dose limit of 20 mSv y^{-1} (NRSU 1998).

Unit 4 is ventilated during current activities through monitored exhaust above the reactor room. The unfiltered exhaust is normally below permitted limits for atmospheric discharge, and a filtration system exists for use should the exhaust approach permitted discharge limits. The ventilation system is zoned so that air flows from outside the Shelter through spaces of increasing levels of contamination.

FIG. 6.4. Reactor room of Unit 4 after the accident.

Unit 4 and the associated cascade walls have an accumulation of fuel-containing material (FCM) including large core fragments that could conceivably lead to criticality under flooded conditions. Such a criticality accident is considered unlikely; if criticality should occur, it migh lead to exposure of some workers inside Unit 4 to a whole body external dose of only a few mSv because, workers avoid the spaces with criticality risk. It has been estimated that in such a case there would be no significant consequences inside and outside of the Exclusion Zone, if a criticality were to occur within Unit 4 (Gmal et al. 1997; Borovoy 2001; NEA 2002).

A number of activities have been performed in recent years to stabilize and improve the conditions of the Shelter. These include repair of the Unit 3/4 ventilation-stack foundation and bracing, reinforcement of the B1 and B2 beams (Fig. 6.5), improvement of the physical protection and access-control system; design of an integrated automated control system (control of building structures conditions, seismic control, nuclear safety control, and stationary radiation control), modernization of the dust-suppression system, and additional structural stabilization. Computerized control systems were installed in the Shelter (NEA 2002) to monitor gamma radiation, neutron flux, temperature, heat flux, concentrations of hydrogen, carbon oxide, and air moisture, mechanical stability of structures, etc. This has been achieved with significant support of Ukraine and donor countries[5].

The magnitude and importance of possible future radioactive releases from the Shelter (in case of its collapse) significantly depend on the radiological and physical-chemical properties of the radioactive material, including dust that may arise from the area inside the Shelter. Now, nearly 20 years after the accident, dust has penetrated concrete walls, floors, and ceilings, and is in the air in the form of aerosols. Thus, in a number of Shelter premises, the fuel-containing dust has become the main source of radiation hazard. Research (Borovoy 2001; Bogatov and Borovoy 2000) shows the typical size of these particles (AMAD) is from 1 to 10 µm. Hence, most of the material is expected to be respirable, which increases its inhalation hazard, and in particular by potential winds that may be generated in case of Shelter roof collapse.

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[5] Contributors to the work of the Chernobyl Shelter Fund include Austria, Belgium, Canada, Denmark, European Commission, Finland, France, Germany, United Kingdom, Greece, Ireland, Italy, Kuwait, Luxembourg, Netherlands, Norway, Poland, Spain, Sweden, Switzerland, Ukraine, and USA. Additional donors to this Fund include Iceland, Israel, Korea, Portugal, Slovakia, and Slovenia.

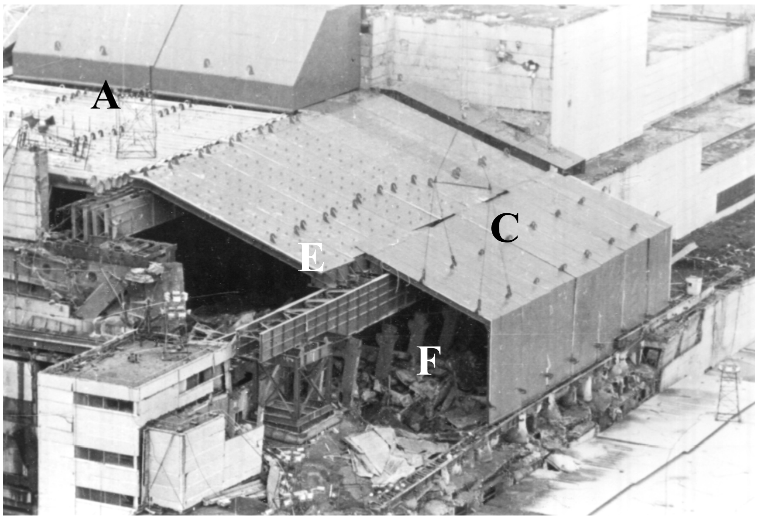


FIG. 6.5. Major structural components of the Chernobyl Shelter: (A) pipe roofing, (B) southern panels, (C) southern hockey sticks, (D) B1/B2 beams, (E) mammoth beam, and (F) octopus beam.

Should the Shelter collapse, it would also complicate continuing accident-recovery efforts, and the resulting radioactive dust cloud would have adverse environmental impacts. Further analysis of the environmental release is sensitive to the source term assumed in the dust cloud that would be generated as a result of the collapse. Different studies give different possible radioactive dust releases to the environment, ranging from about 500 to 2000 kg of particulate dust, which could contain from 8 kg to 50 kg of finely dispersed nuclear fuel. Regardless of the source-term assumption, almost all material that might be raised into the atmosphere by a Shelter collapse is expected to be deposited within the Exclusion Zone (Pretzsch 1997; STC 1998).

Another concern related to the FCM is its possible transport out of the Shelter into the groundwater through accumulated water. The potential for FCM to dissolve with the accumulated water was confirmed when bright yellow stains and faded pieces of FCM were found on the surface of solidified fuel-lava streams in Unit 4 (Borovoy et al. 1999); and subsequent analysis proved the presence of soluble uranium compounds. Until recently, this FCM was considered to be a glassy mass that was very insoluble. The possibility of leaching of radionuclides from the FCM and mobile radionuclides as ^{90}Sr to migrate and reach the Pripyat River are expected to be very low (NEA 2002). The expected significance of this phenomenon is not known and therefore monitoring of the evolving groundwater condition at and around the Shelter is important.

Additional studies of the water table showed that it has risen by up to 1.5 m in a few years to about 4 m from the ground level and it may still be rising. This effect was considered mainly as a result of the construction of a 3.5 km long and 35 m deep wall around Unit 4 that aimed to protect Kyiv reservoir from potential contamination through ground water (NEA 2002).

The main hazard of the Shelter is a possible collapse of its top structures and release of radioactive dust into the environment; therefore, a dust-suppression system was installed under the Shelter roof that periodically sprayed the dust-suppression solutions and fixatives. The system has operated since January 1990, and more than 1000 tons of dust suppressant have been sprayed during this period.

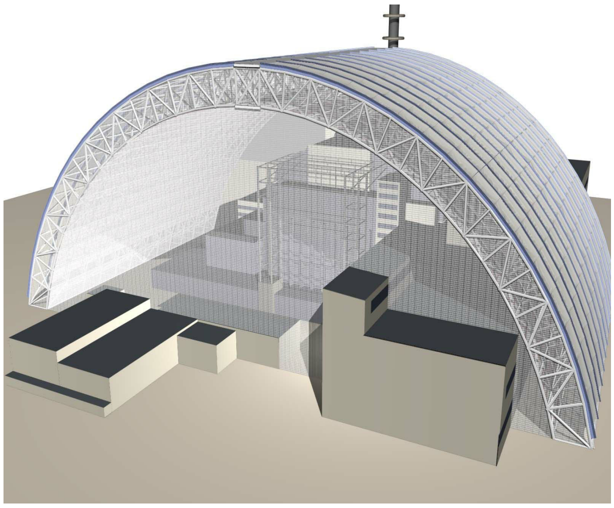


FIG. 6.6. Planned New Safe Confinement (NSC).

6.1.3. Long-term strategy for Shelter and New Safe Confinement (NSC)

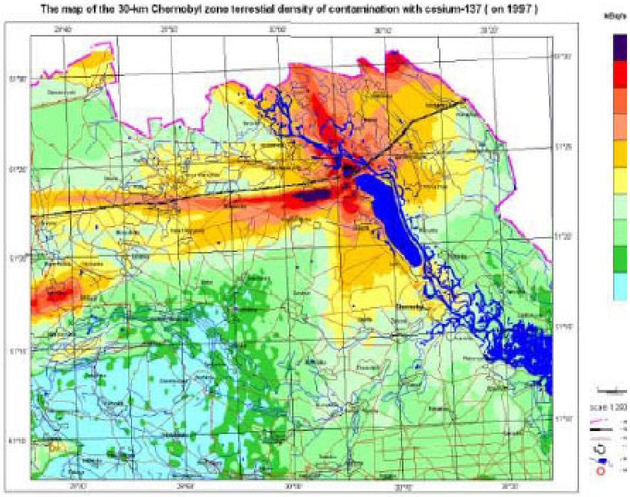
In order to avoid potential collapse of the Shelter in the future, some measures have been implemented and additional measures are planned to strengthen unstable parts of the Shelter with structures having stability from 15 to 40 years (SIP 2004). In addition a NSC is planned to be built as a cover over the existing Shelter as a longer-term solution (see Fig. 6.6). The Ukrainian Government supported the concept of a multifunctional facility with at least 100 years service life. This facility aims to reduce the probability of Shelter collapse; reduce the consequences of Shelter collapse; improve nuclear safety; improve worker and environmental safety, and convert Unit 4 into an environmentally safe site. The construction of the NSC is expected to allow for the dismantlement of the current Shelter, removal of FCM from Unit 4, and eventual decommissioning of the reactor.

The specific operational aspects related to the construction and operation of the NSC, including their maintenance in the long-term, have not been identified yet. It is important to note that the NSC design is based on the current plans for removal of the FCM that depend on the availability of the final geological disposal facility about 50 years from now (SIP 2004). This extended dormancy period could result in the dispersal of the critical human resources needed to remove and dispose FCM safely. Accordingly, reconsideration needs to be given to removing the FCM and structural material as soon as possible after the construction of the NSC.

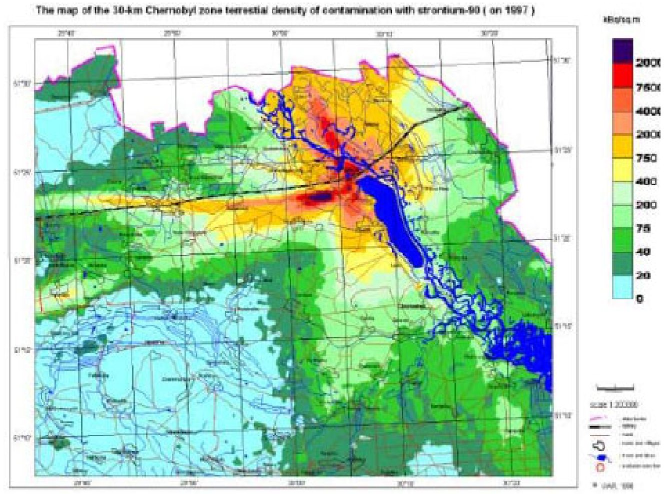
6.1.4. Environmental aspects

6.1.4.1. Current status of the Shelter

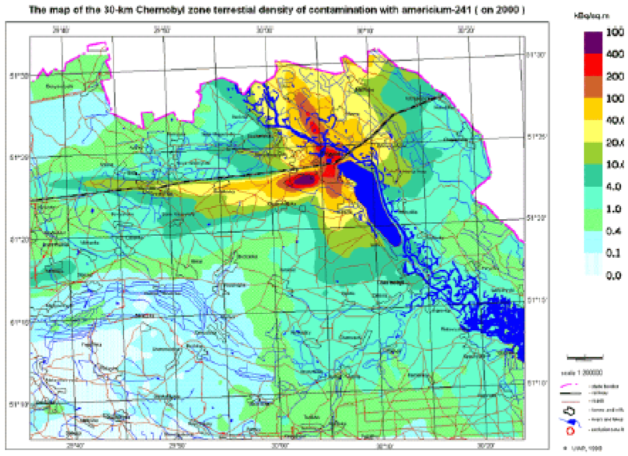
The current radiological impact of the Shelter and the proposed NSC needs to take into consideration the existing contamination surrounding Unit 4. At present the environmental contamination around the ChNPP site is determined by the initial radioactive contamination of the area from the accidental release of 1986; the routine releases of radionuclides through the ventilation system of the Shelter; and the engineering and other specific activities, which are carried out in the Exclusion Zone. The main dose-contributing radionuclides within the Exclusion Zone around the ChNPP site are ^{137}Cs, ^{90}Sr, ^{241}Am, and $^{239+240}$Pu (see also Chapter 2), the current distribution of these radionuclides is shown in Fig. 6.7 (Kashparov et al. 2003).



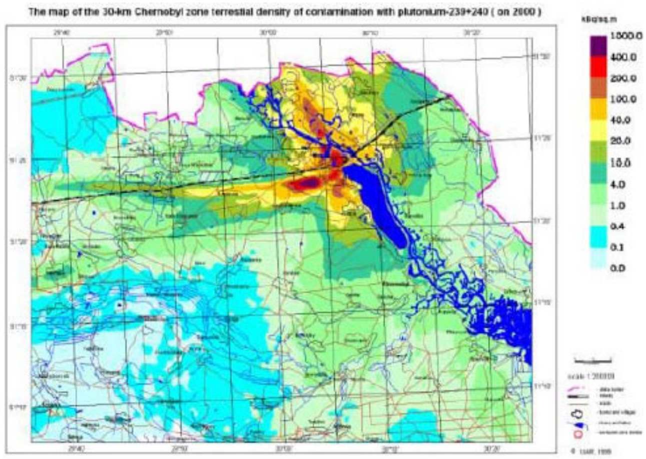
^{137}**Cs in soils of Chernobyl Exclusion Zone, kBq m**$^{-2}$



^{90}**Sr in soils of Chernobyl Exclusion Zone, kBq m**$^{-2}$



^{241}**Am in soils of Chernobyl Exclusion Zone, kBq m**$^{-2}$



$^{239+240}$**Pu in soils of Chernobyl Exclusion Zone, kBq m**$^{-2}$

FIG. 6.7. Surface contamination by radioactive fallout within the Chernobyl Exclusion Zone (Kashparov et al. 2003).

6.1.4.2. Impact on air

Currently, radioactive aerosol releases into the atmosphere from the Shelter are considered to result from two main sources: controlled releases from the Central Hall of Unit 4 into the environment through the exhaust-ventilation system and ventilation stack No. 2, and uncontrolled releases through the leaks in the roof and walls. Ventilation stack No. 2 releases from 4 to 10 GBq y^{-1}, which is many times lower than the regulatory limit of 90 GBq y^{-1} (NEA 2002). The uncontrolled releases depend on the location and area of openings in the external structures and the air-transfer rate through them, which depends on many conditions such as temperature, barometric pressure, humidity, wind speed and direction.

As a result, the air in the immediate vicinity of the Shelter contains finely dispersed fuel particles with maximal concentrations of 40 mBq m^{-3} of ^{137}Cs at distances less than 1 km and 2 mBq m^{-3} at about 3 km from the Shelter. The aerosol particles have radioactive compositions similar to those of the fuel; the primary beta components are ^{90}Sr and ^{137}Cs, while the alpha emitters are mostly plutonium and ^{241}Am. Inhalation doses to individuals outside the Shelter result from a combination of the ongoing Shelter releases and resuspended material from the initial accident. And if a person (worker) were to spend an entire year adjacent to the Shelter, a recent inhalation-dose assessment (Schmieman et al. 2004) indicates that releases would result in annual doses of about 0.5 mSv, which would decrease to about 0.0002 to 0.0005 mSv beyond a distance of 10 km. Inhalation doses from the ongoing releases outside of the Exclusion Zone are significantly less than dose limits for the population (NRSU 1998).

6.1.4.3. Impact on surface water

It can be noted that the average concentrations of radionuclides in surface-water bodies are declining and, for instance in the Pripyat River in 2003, concentrations were observed to be 0.05 (max 0.12) Bq L^{-1} for ^{137}Cs and 0.15 (max 0.35) Bq L^{-1} for ^{90}Sr (IAEA 2005). The main sources of radionuclides into the rivers in the Exclusion Zone during ordinary and high water seasons continue to be runoff from the watersheds situated out of the immediate ChNPP area, infiltrating waters from the ChNPP cooling pond, and old water-reclamation systems in the heavily contaminated territories.

During wintertime and low water seasons, the radionuclide fluxes from regional groundwater contribute to the majority of radionuclide migration to the Pripyat River from this area. However, the absolute values of radionuclide flux from all groundwater into surface water are still relatively low, and the contribution of groundwater-contamination plumes from the temporary radioactive waste facilities and the Shelter area has been identified as about 3-10 % (IAEA 2005) of the annual migration of radionuclides into the Pripyat-Dnieper River system from the Exclusion Zone (see also Section 2.5).

6.1.4.4. Impact on groundwater – see Section 2.5.5.1 for details

Surface contamination around the ChNPP site is the cause of groundwater contamination, with contamination levels of 100-1000 Bq m^{-3} for ^{90}Sr and about 10-100 Bq m^{-3} for ^{137}Cs. Radionuclide contamination of the groundwater at the Shelter site is much higher. In more recent studies, the primary source term for radionuclide contamination of the groundwater is considered to be water accumulating inside of underground rooms of Unit 4 (as a result of precipitation), groundwater accumulated near the Pioneer Wall (because of absence of a drainage system), and other water infiltrating from the NPP site.

In some places, ^{137}Cs in groundwater in the sub-surface horizons near the Shelter reach 100 Bq L^{-1} and even 3000-5000 Bq L^{-1}. However, in the majority of the Shelter area, ^{137}Cs concentrations in groundwater are more or less similar and vary in the range from 1 to 10 Bq L^{-1}. Typical amounts of ^{90}Sr in groundwater around the Shelter site are in the range from 2 Bq L^{-1} to 160 Bq L^{-1} with maximal concentrations observed during the most recent 5 years ranging from 1000-3000 Bq L^{-1}. Estimated concentrations of transuranic elements in the groundwater of this area also vary over a wide range, from 0.003 Bq L^{-1} to 3-6 Bq L^{-1} for ^{238}Pu and $^{239+241}$Pu and from 0.001 to 8-10 Bq L^{-1} for ^{241}Am (Bugai et al. 1996; Shestopalov 2002).

6.1.4.5. Impacts of Collapse without New Safe Confinement

Because of concerns about the long-term stability of the Shelter, estimates have been made of the probability of its collapse. Depending on the mechanisms considered, the probability ranges from about 0.001 to 0.1 per year (Vargo et al. 2000; Borovoy 2001) and therefore, an analysis (summarized from (EIA 2003)) of the potential impacts of the Shelter collapse has been performed without and with the NSC in place.

Impact on air

Collapse of the Shelter could raise a large cloud of fine dust (up to 500-2000 kg) containing 8-50 kg of nuclear fuel particles with activity of about 1.6×10^{13} Bq. This could lead to an increase of annual inhalation dose up to 0.4 Sv near the Shelter. The estimated annual doses outside the Exclusion Zone could reach 2 mSv (EIA 2003), which would exceed the established dose limits for public in Ukraine (NRSU 1998).

Within the boundaries of the Exclusion Zone, the depositions of radionuclides from such a collapse would be in all cases a small fraction of the existing contamination levels caused by the original Chernobyl accident. Typical results are shown in Figure 6.8 (EIA 2003) The highest relative increase in soil contamination would occur if the wind were to blow the plume from a Shelter collapse to the southwest toward the area that received the least impact from the original accident. In this case, the additional deposition might add about 10% to the existing soil-contamination levels. Outside the boundaries, at 50 km distance from the Shelter, additional surface contamination of ^{137}Cs, ^{90}Sr and $^{238+239/40}$Pu due to the Shelter collapse would contribute from a few to 10 percent.

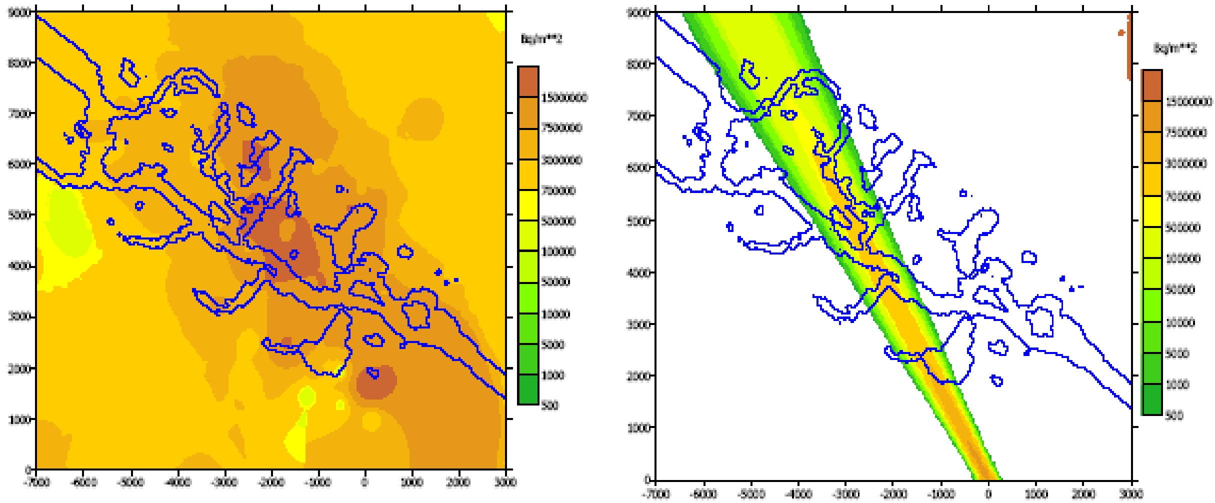
Impact on surface water

In case of the Shelter collapse, additional radioactivity could also be deposited in and near the rivers.

As shown in Fig. 6.8, radionuclide depositions into the Pripyat River could be as high as: ^{90}Sr - 1.1×10^{12} Bq, ^{137}Cs - 2.4×10^{12} Bq, ^{238}Pu - 1.6×10^{10} Bq, ^{239,240}Pu – 4.0×10^{10} Bq and ^{241}Am- 5.0×10^{10} Bq. Estimates of the maximum possible concentrations of these radionuclides in the Dnieper reservoirs show that the peak concentration for ^{90}Sr can be expected in the Kyiv Reservoir on the 41st day after the accident happens, and would be about 700 Bq m^{-3}. The maximum concentration of ^{90}Sr in the Khahovka Reservoir would be about 200 Bq m^{-3} or less. This confirms that the normative values of ^{90}Sr in potable water (2000 Bq m^{-3}, NSRU 1998) would not be exceeded, if an accident leading to the maximum impacts happens at the Shelter.

The maximum possible concentration of ^{137}Cs that can be expected in the Kyiv and Kanev Reservoir water, even in the worst simulated scenarios, are 3-10 times lower than the limits for potable water. Such events would not affect the concentrations of ^{238}Pu, ^{239,240}Pu and ^{241}Am in the Pripyat and Dnieper Rivers (EIA 2003).

Releases from a Shelter collapse could lead to some increased exposure of people living downstream of the most impacted area in the Exclusion Zone and and who consume water and fish from the reservoirs. Radiation doses to individuals are discussed in (EIA 2003), where the highest values are predicted for professional fishermen and typical consumers.



(a) (b)

FIG. 6.8. ^{90}Sr soil density of Chernobyl fallout, (a) upstream of Yanov Bridge, 1999, and (b) predicted from a Shelter collapse.

Impact on groundwater

Rainwater infiltration and condensation within the existing Shelter was also studied. This study confirms the significance of a large pool of water in the basement of the Shelter. The leakage of the heavily contaminated water from this pool through the concrete walls and floor of the room is a main source of the contamination of the vadose zone and groundwater beneath the Shelter. Under existing conditions, a positive water balance exists and water collects in the basement rooms.

The results of evaluation of groundwater contamination without an NSC show that a concentration of ^{90}Sr in the groundwater of about 4×10^9 Bq m^{-3} is expected to occur at distances less than 100 m from the Shelter, and would decline to 100 Bq m^{-3} at 600 m from the Shelter. The contamination is predicted to reach the Pripyat River in 800 years. However, the infiltration fluxes of ^{90}Sr from the Shelter even without the NSC are not expected to cause significant impacts to the Pripyat River.

6.1.4.6. Impacts of Shelter Collapse within New Safe Confinement

Impact on air

Placement of the NSC over the Shelter is expected to reduce the release of dust onto the site from a collapse, thus reducing the magnitude of accidental inhalation doses. The dust would largely settle within the NSC and not be released to the environment except through normal ventilation pathways. The amount of transported dust would depend on the ventilation and confinement capability designed into the NSC. The doses are expected to be reduced by factors of 7 to 70 compared to the estimated doses in case of a Shelter collapse without a NSC, depending on the capacity of the NSC ventilation system (EIA 2003). This leads to an expected decrease of outdoor workers' exposure by a factor of two in comparison to the scenario of Shelter collapse without an NSC. However, some workers might be inside of the NSC at the time of the collapse; doses to these workers might be increased because of the containment of the dust.

For the small number of individuals who have chosen to reside within the 30-km zone, inhalation doses are expected to be reduced by factors of 50 to 500, to no more than 1 or 2 mSv (EIA 2003). Even assuming 95-th percentile meteorological conditions and also assuming that the dust cloud would pass over one of the larger cities, such as Slavutych, an increase of latent fatal cancer risk projected for the population for collapse with the NSC is not expected.

Very minor impacts to soil contamination would be caused by airborne radioactivity discharged and settled should the Shelter collapse inside the NSC. Within the boundaries of the Exclusion Zone, the radionuclide depositions would be in all cases a small fraction of the existing levels caused by the original Chernobyl accident - the highest relative increase would occur if the wind were to blow the plume from a Shelter collapse to the southwest toward the area that received the least impact from the original accident; in this case, the additional deposition might add less than 0.2% to the existing soil-contamination levels.

Impact on surface water

Emplacement of the NSC would ensure that, in the event of a collapse, additional deposition of radionuclides in surface water would be minimal. The depositions illustrated in Fig. 6.8 would be reduced by factors of 50 to 500 (EIA 2003), and resulting concentrations in downstream waters would not exceed Ukrainian norms.

Impact on groundwater

The dynamics of radionuclides migration into the groundwater with the presence of the NSC was evaluated considering the decrease of water level to zero in the basement one and a half year after the NSC is constructed. After NSC construction, the precipitation fluxes are expected to be minimised and evaporation will be higher than fluxes from the dust-suppression system and condensation. This means that the water level in the basement is expected to diminish due to the seepage through the walls, and the room would be empty in less than two years.

6.1.5. Issues and areas for improvement

6.1.5.1. Source-term uncertainty influence on environmental decisions

There is still considerable uncertainty regarding the amount of nuclear fuel remaining in Unit 4. One estimate (Integrated RAW Programme 2004; UNSCEAR 2000) states there is approximately 95% of the 190 t of nuclear fuel (as uranium) inside Unit 4, which was in the reactor at the time of the accident. Another estimate (STC 1998) states there is 60% of the original core plus the fuel in the decay pool and in the central room remaining in the facility (212 t total, less 80 t of FCM, less 6 t of blown out fuel = 126 t remaining); estimated radioactivity inside the Shelter in 1995 was approximately 7×10^{17} Bq (Borovoy et al. 1999). Despite a wide scope of studies, to date there is no comprehensive information regarding the amount and distribution of fuel inside the Shelter. This lack of knowledge is an important factor in the evaluation of safety and environmental consequences of Unit 4 and Shelter evolution, as well as for the selection of adequate solutions for long-term management of associated radioactive waste.

6.1.5.2. Characterization of fuel-containing material

The physical state of FCM appears to be changing with time. It appears that the FCM has begun to oxidize and may be decomposing into fine particulate matter with unknown oxidation rate and exact particle size and behavior. Another related important uncertainty is the dust distribution in the Shelter, and more specifically in the NSC atmosphere over long-term operation of the facility. As estimates of environmental impacts (e.g., transport and inhalation calculations) of long-term Shelter development are sensitive to the assumptions about these source-term parameters, it would be necessary that these parameters be further investigated. This would contribute to increased confidence in the safety-assessment results and selection of appropriate protective measures for the workers, the public and the environment.

6.1.5.3. Removal of FCM concurrent with development of a geological disposal facility

The stabilisation of the Shelter and construction of the NSC is expected to generate significant amounts of long-lived radioactive waste, some of which would contain FCM. However there are no plans for removal of the FCM until a geological disposal facility is constructed and commissioned. Therefore a long-term management strategy of FCM and long-lived radioactive waste needs to be developed to ensure safe management of this waste.

In addition, temporary facilities for storage of this waste are still to be developed. And, therefore, it could be concluded that there is no technical reason to delay removal of the FCM until a geological disposal facility is available. Removal of FCM could commence following dismantlement of unstable structures at the Shelter, continuing with radioactive waste predisposal management and temporary storage on the ChNPP site until the geological disposal facility is available. Due to the high content of long-lived radionuclides there is also no significant worker dose benefit in waiting for the availability of a geological disposal facility. Whether retrieved now or after 50 years, remote retrieval and radioactive waste-management techniques will be required to remove the FCM and restore the Unit 4 site.

6.2. Management of radioactive waste from the accident

In the course of remediation activities both at the Chernobyl NPP site and in its vicinity large volumes of radioactive waste were generated and placed in temporary near surface waste-storage facilities located in the Exclusion Zone (Fig. 6.9) at distances 0.5 to 15 km from the NPP site. Sites for temporary waste storage of trench and landfill type were created from 1986 to 1987 and intended for radioactive waste generated after the accident as a result of the clean up of contaminated areas to avoid dust spread, reduce the radiation levels, and enable better working conditions at Unit 4 and its surroundings. These facilities were established without proper design documentation, engineered barriers, or hydrogeological investigations according to contemporary waste-safety requirements.

During the years following the accident economic and human resources were expanded to provide a systematic analysis and an acceptable strategy for management of existing radioactive waste. However, as reported in some Ukrainian studies (Bondarenko et al. 2004), to date a broadly accepted strategy for radioactive waste management at the ChNPP site and the Exclusion Zone, and especially for high-level and long-lived waste, has not been developed. Some of the reasons to be mentioned are the large number and areas covered by the radioactive waste-storage and disposal facilities of which only half are well studied and inventoried. This results in large uncertainties of radioactive waste inventories (volume, activity, etc.) as presented in this section.

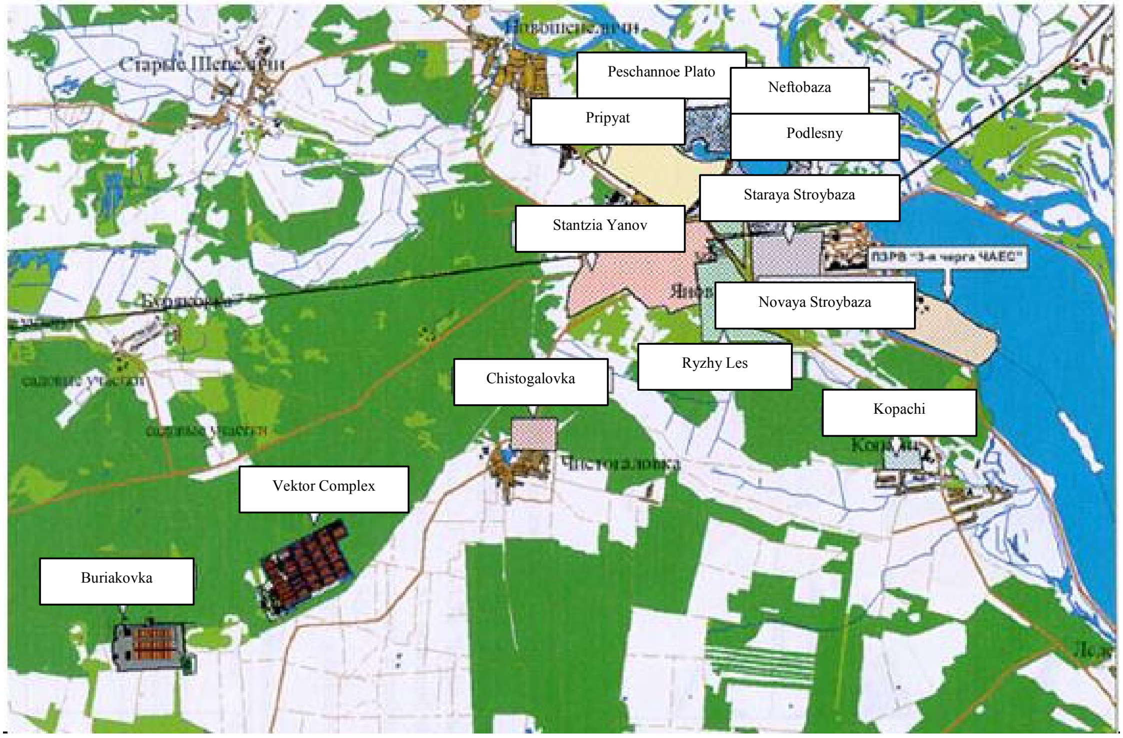


FIG. 6.9. Temporary radioactive waste-disposal facilities in the territory of the Chernobyl Exclusion Zone.

The existing radioactive waste from the accident and potential radioactive waste to be generated during NSC construction, Shelter dismantling, FCM removal and decommissioning of Unit 4 can be categorised as:

— Radioactive waste from the Shelter and the NPP site which will be created by construction of infrastructure and the NSC;

— Accident-generated transuranic waste, which has been mingled with radioactive waste from operations at ChNPP Units 1, 2 and 3;

— Radioactive waste in temporary radioactive waste facilities located throughout the Exclusion Zone; and

— Radioactive waste in existing radioactive waste-disposal facilities.

The safety and environmental issues related to each of these categories of radioactive waste is presented in this section. The radioactive waste expected to be generated during the decommissioning of the ChNPP Units 1, 2 and 3 represent an additional category that is not the subject of this report.

The current Ukrainian legislation applies the following categorisation of radioactive waste in accordance with its specific activity and radionuclide radiotoxicity as specified in Table 6.1 (Basic Sanitary Rules 2000).

For waste contaminated with unspecified mixtures of radionuclides emitting gamma radiation, the use of classification “low-”, “intermediate-”, “high-” activity is allowed with the criteria of the air-dose rate at a distance 0.1 m as specified in Table 6.2 (Basic Sanitary Rules 2000).

TABLE 6.1. SOLID RADIOACTIVE WASTE CATEGORIZATION

Radioactive waste categories	Range of specific activity, kBq kg^{-1}			
	Group 1*	Group 2*	Group 3*	Group 4*
Low activity	10^{-1} - 10^{1}	10^{0} - 10^{2}	10^{1} - 10^{3}	10^{3} – 10^{5}
Intermediate activity	10^{1} - 10^{5}	10^{2} -10^{6}	10^{3} - 10^{7}	10^{5} - 10^{8}
High activity	> 10^{5}	> 10^{6}	> 10^{7}	> 10^{8}

* Group 1 - *Transuranic* alpha radionuclides, Group 2- *Alpha* radionuclides (excepting transuranium); Group 3 *Beta, gamma* radionuclides (excluding those in group 4), and Group 4 - H-3, C-14, Cl-36, Cf-45, Mn-53, Fe-55, Ni-59, Ni-63, Nb-93m, Tc-99, Cd-109, Cs-135, Pm-147, Sm-151, Tm-171, Tl-204.

TABLE 6.2. CLASSIFICATION OF RADIOACTIVE WASTE WITH UNKNOWN SPECIFIC ACTIVITY WITH THE CRITERIA OF DOSE RATE AT 0.1 M DISTANCE.

Radioactive waste category	Dose rate, μGy h^{-1}
Low activity	1 - 100
Intermediate activity	100 - 10000
High activity	> 10000

TABLE 6.3. ESTIMATED INVENTORY IN THE SHELTER (ADAPTED FROM CONCEPT 1999)

	Type of radioactive waste and criteria of assessment	Category of radioactive waste	Amount
Fuel Containing Material	Fresh fuel assemblies (FFA), Spent fuel assemblies (SFA) Lava type material, fuel fragments, radioactive dust	High level waste	About 190-200 t 700 tonnes of graphite
Solid Radioactive Waste with less than 1% nuclear fuel (mass)	Fragment of the core with dose rate at 10 cm of more than 10 mSv/h		
Liquid radioactive waste	Changing inventory based on the precipitation (e.g. pulp, oils, suspensions, with soluble U salts)	Low level (up to 3.7x 10^{5} Bq/l)	2500 - 5000 m^3
		Intermediate (more than 3.7x10^{5} Bq/l)	500 - 1000 m^3
Solid radioactive waste	Metal equipment and building material e.g. concrete, dust, non-metal material (organic, plastic material)	High level waste	38,000 m^3 (building material) 22,240 t (metal constructions)
		Low and intermediate level waste	300,000 m^3 (building material and dust) 5 000 m^3 (non-metal)

Current radioactive waste-management practice in Ukraine does not fully comply with the above classification, therefore the measures are being taken to bring it into conformity with the new regulations (National Report 2003).

6.2.1. Current status of radioactive waste from the accident

6.2.1.1. Radioactive waste associated with the Shelter

The Shelter is considered as “a destroyed Unit 4 after a radiological accident” and as “a near surface storage facility for unconditioned radioactive waste at a stage of stabilization and reconstruction” (NRSU-97/D-2000). The amount and type of waste, debris and other radioactive material inside the Shelter can be presented as follows (see Table 6.3).

In addition, soils that were heavily contaminated by the deposition of fuel fragments and of radionuclides and debris from the accident (metal pieces, concrete rubble, etc.) were also collected and stored in the vicinity of Unit 4 (see also Fig. 6.2):

(1) **Three Pioneer Walls** (west, north and south of the Shelter, see Fig. 6.2), where contaminated soil, concrete and containers are stored and which contain an estimated 1700 m^3 - 4900 m^3 of high-level waste[6] (HLW) and up to 72,000 m^3 of low and intermediate level waste (LILW) (Concept 1999; EC 2001);

(2) **Cascade Wall** north of the Shelter (Fig. 6.2), where core fragments, metal, concrete, core-pit equipment, and accident-cover material is stored (16,600 m^3 of HLW, 117 t of reactor-core elements, and 53,400 m^3 of LILW); (Concept 1999)

(3) **Industrial Site** around the Shelter, where concrete, gravel, sand, clay, and contaminated soil are stored which contains 7000 m^3 of HLW and 286,000 m^3 of LILW (Panasuk et al. 2002). Studies (EC 2001) show that fuel, graphite, etc. are located in the contaminated soil.

The radioactive waste inside the Pioneer and Cascade Walls was later covered by concrete. This material is considered to be high-level waste that is not acceptable to be disposed in near surface-disposal facilities. Because it cannot be retrieved easily for conditioning, the radioactive waste recovered from these walls is to be part of a global strategy for the decommissioning of Unit 4.

It is estimated (Final Report 2001) that the current and expected radioactive waste from Unit 4 can be categorized as short-lived low and intermediate level waste (soil from the construction of the NSC, construction material, concrete, metal constructions, etc.) and high-level waste (e.g., from the construction of NSC) and high-level waste (e.g., FCM) according to Ukrainian legislation.

6.2.1.2. Mixing of accident-related waste with operational radioactive waste

During 1986 to 1993, some low and intermediate radioactive waste and high-level waste with transuranic elements were also stored together with some operational radioactive waste from Units 1, 2, and 3 in an above ground storage facility (see Fig. 6.10) at the ChNPP site.

[6] HLW falls into two subcategories, i.e., low-temperature waste below heating rate of 2 kW/cub.m. and heat-generating waste with heating rate higher than 2kW/cub.m. (IAEA 1994).

FIG. 6.10. Existing above ground storage facility for solid radioactive waste at the ChNPP Site.

This waste amounts to about 2500 m^3 with a total radioactivity of about 131 TBq (Integrated RAW Programme 2004) and is stored unconditioned. Once filled, the storage facility was backfilled with concrete grout and covered by a concrete roof to reduce radiation levels and water infiltration. Thus, the retrieval of the radioactive waste stored in this facility cannot be easily achieved and will require particular care. Plans for such retrieval are currently under study. At the moment, this facility is under extension and intended to be used for disposal of radioactive waste to be produced during decommissioning activity on Units 1, 2 and 3.

6.2.1.3. Temporary radioactive waste facilities

The largest volumes of radioactive waste generated by Unit 4 remediation activities are located in the Exclusion Zone (see Fig 6.9). Sites for temporary storage of radioactive waste of trench and landfill type were constructed shortly after the accident at distances 0.5 to 15 km from the NPP site. They were created from 1986 to 1987 and intended for radioactive waste generated after the accident as a result of the clean up of contaminated areas to avoid dust spread, reduce the radiation levels, and enable better working conditions at Unit 4. These facilities were established without design documentation, engineered barriers, or hydrogeological investigations.

The total area of temporary radioactive waste facilities is about 8 km^2 with a total volume of disposed radioactive waste estimated to be over 10^6 m^3. The main inventories of activity are concentrated in temporary radioactive waste facilities "Stroybaza" and "Ryzhy Les" along the western trace of Chernobyl fallout (see Fig. 6.9). The specific activity of radioactive waste in the temporary radioactive waste facility at "Ryzhy Les" of ^{90}Sr and ^{137}Cs is 10^5-10^6 Bq kg^{-1} and the sum of plutonium isotopes is estimated to be 10^3 -10^4 Bq kg^{-1}.

Most of the facilities are structured in the form of trenches 1.5 - 2.5 m deep in the local sandy soil. The radioactive material (soil, litter, wood material, and building debris) is overlain by a layer of alluvial sand 0.2-0.5 m thick. The majority of the temporary radioactive waste facilities consist of trenches in various types of geological settings, where waste was stacked and covered with a layer of soil from the nearby environment. These facilities are therefore very different with regard to their potential for releases, which depend on the total radioactivity stored, the waste form (in particular timber), the retention capacity of the substratum along migration pathways, and the location of the sites in hydrogeological settings. At least half of these temporary radioactive waste facilities (see Fig. 6.9) have been studied, and, according to available knowledge, they are located at nine sites as presented in Table 6.4 (Saversky et al. 2001; RAW in Ukraine 2003).

There are also many other temporary radioactive waste facilities, in total estimated to comprise about 800 trench facilities each with waste-disposal volumes in the range of 800 to 2 × 10^6 m^3 (IAEA 2001, Saversky et al. 2001). The number and inventory of these facilities is known for half of them, and the facilities are not under regulatory control. Estimations made for a few sites show that the radioactivity stored can be high (from 10 to 1000 TBq), sometimes on the order of magnitude comparable to the total radioactivity present in soil from the Exclusion Zone (about 7000 TBq) (IAEA 2001).

TABLE 6.4. STATUS OF TEMPORARY RADIOACTIVE WASTE FACILITIES

Title	Size, ha	Number of trenches	Number of landfills	Radioactive waste type	Radioactive waste volume, thousand m^3	Total activity, Bq
Sites with well known inventories						
Neftebaza	53	221	4	soil, plants, metal, concrete and bricks	104	4 × 10^{13}
Peschannoe Plato	78	2	82	short lived* low and intermediate level waste of soil, rubble, and concrete	57	7 × 10^{12}
Partially investigated sites						
Stantzia Yanov	128	known more than 36	-	soil, plants, metal, concrete and bricks	30	>4 × 10^{13}
Ryzhy Les	227	estimated more than 61	estimated more than 8	mainly soil, some construction and domestic material	500	Up to 4 × 10^{14}
Staraya Stroybaza	130	more than 100	-	soil, metal, concrete and wood	171	1 × 10^{15}
Novaya Stroybaza	122			soil, plants, metal, concrete and bricks	150	2 × 10^{14}
Pripyat	70			contaminated vehicles, machinery, wood and construction waste	16	3 × 10^{13} Bq (1990)
Chistogalovka	6			material from demolition of buildings, soil, wood, and working clothes	160	4 × 10^{12}
Kopachi	125			construction waste from demolition	110	3 × 10^{13}

* According to Ukrainian legislation, Short-lived waste – radioactive waste whose level of release from regulatory control is achieved earlier than 300 years after disposal; Long-lived – radioactive waste whose level of release from regulatory control is achieved later than 300 years after disposal (Basic Sanitary Rules 2000).

6.2.1.4. *Radioactive waste-disposal facilities*

The main radioactive waste-disposal facilities for accident waste are the Buriakovka, Podlesny, and Kompleksny Sites, which are under regulatory control. These three near surface disposal sites were established after the accident to dispose of radioactive waste from remediation actions carried out during the first year following the accident. These sites were chosen and designed for disposal of higher-level accident waste than the radioactive waste located in the temporary radioactive waste facilities as follows (RAW in Ukraine 2003):

Buriakovka – Built in 1987 is the only disposal facility currently in operation at the Exclusion Zone. It comprises 30 trenches with 1-m clay layer located on 23.8 ha. Up to 652,800 m^3 of radioactive waste have been disposed, and after in-situ compaction this was reduced to 530,000 m^3 with a total radioactivity of 2.5×10^{15} Bq of solid short-lived low and intermediate level waste (SL-LILW). It consists of metal, soil, sand, concrete and wood contaminated with ^{90}Sr, ^{137}Cs, ^{134}Cs, $^{238\text{-}240}Pu$, $^{154\text{-}155}Eu$, and ^{241}Am. Radioactive waste with dose rates at 10 cm from the surface in the range of 0.003–10 mGy h^{-1} were accepted in this facility.

Podlesny – This vault type disposal facility was commissioned in December 1986 and closed in 1988. The facility was designed for disposal of high-level waste with dose rate 10 cm from the surface in the range of 0.05–2.5 Gy h^{-1} and above were also disposed in the facility. The total radioactive waste volume of 11,000 m^3 of building material, metal debris, sand, soil, concrete, and wood was disposed in two vaults. The disposal facility was covered with concrete at its closure. In 1990 the estimated total radioactivity of the disposed waste was 2600 TBq. In 2002 a re-evaluation of the facility status showed reasons to believe that the total activity of waste disposed at this site might be higher than initially estimated, and a need for re-estimation of the current inventory was identified. Due to the uncertainties in inventory, it is estimated that various types of waste were disposed, including FCM.

Kompleksny – This vault-type facility was based on reconstructed facilities of unfinished Units 5 and 6 at the ChNPP site. Kompleksny was put in operation from October 1986 until 1988. It was designed for LILW corresponding to dose rates up to 0.01 Gy h^{-1} at 10 cm from the surface of the waste container. More than 26,200 m^3 of solid waste with a total radioactivity of 4×10^{14} Bq were disposed in 18,000 containers, and later covered with sand and clay. This waste was mainly sand, concrete, metal, construction material and bricks. Due to the high level of ground water at different periods of the year, the facility is flooded from 0.5 to 0.7 m from its bottom. Significant uncertainties exist associated with the radionuclide inventory, because of the lack of data about the radioactive waste disposed at the site.

At present, a new near surface Vektor complex, for low and intermediate radioactive waste processing, storage and disposal is under development. This complex will include (Integrated RAW Programme 2004):

- Engineering facility for processing of all types of solid radioactive waste (capacity of 3,500 m^3 per year);
- Disposal facility for short-lived solid radioactive waste (55,000 m^3 total capacity);
- Storage facility for long-lived solid radioactive material;
- Storage facility for fuel containing material; and
- Intermediate Storage for high-level conditioned radioactive waste, to be prepared for final disposal at the deep geological disposal facility.

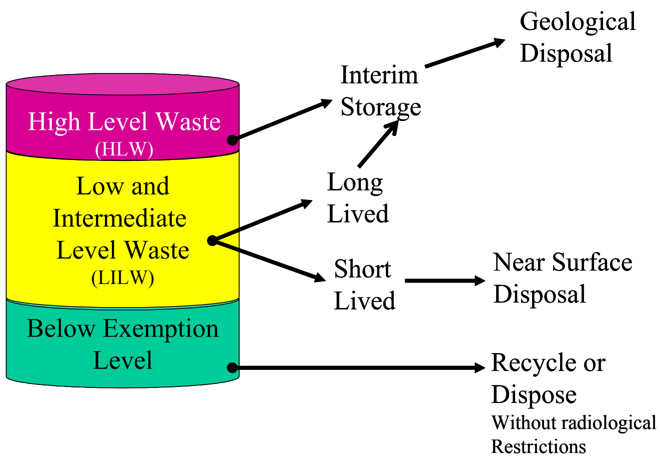


FIG. 6.11. Planned management of radioactive waste at the ChNPP Site (Integrated RAW programme 2004).

6.2.2. Strategy on radioactive waste management

At present no further dismantlement and clean up of Unit 4 is planned. However, estimates of the radioactive waste generation and subsequent management options have been performed for the construction of the NSC and the dismantlement phase of the unstable structures of the Shelter. The preparation phase is expected to generate about 390 t of solid radioactive waste and about 280 m^3 of liquids (EIA 2003). It also requires the removal of 100,000 m^3 of contaminated soil around Unit 4 that may still contain fuel fragments. Preliminary estimates for the dismantlement of the Shelter superstructure predict about 1,200 t of steel with an estimated volume of radioactive waste being 1,800 m^3 (Schmieman et al. 2004), mainly metal and large concrete pieces. This waste is planned to be sorted based upon its radiation levels. High-level waste, which is expected to be only a small quantity, is planned to be placed in containers and stored within the NSC.

According to Ukrainian legislation (Law 1995), all industrial radioactive waste produced in management practice is classified according to the scheme shown in Fig. 6.11 - high level and long-lived waste must be disposed of in a deep geological disposal facility; low- and intermediate-level and short-lived radioactive waste in near surface disposal facility. Following these criteria, the strategy for management of radioactive waste from the 1986 accident needs to be developed, and in particular for the management of high-level and long-lived radioactive waste.

The planned options for low radioactivity waste are to sort the waste according to its physical characteristics (e.g., soil, concrete, metal, etc.) and possibly decontaminate it and/or condition it for beneficial reuse (reuse of soil for NSC foundations, melting of metal pieces), or send it for disposal in a new extension of the Buriakovka disposal facility (RAW in Ukraine 2003), or to the Vektor disposal site.

The long-lived waste is planned to be placed into interim storage. Different storage options are being considered at the ChNPP or Vektor site, and a decision has not yet been made. After construction of the NSC decommissioning of the Shelter facilities is envisaged, including Shelter dismantlement and further removal of FCM. High-level radioactive waste is planned to be partially processed in place and then will be stored at the temporary storage site until a deep geological disposal site is ready for its final disposal. At the moment, this strategy is considered as the basic option for high-level radioactive waste and FCM (RAW in Ukraine 2003). To implement this strategy, it is planned to organise a system for processing and temporary storage of the high-level and long-lived radioactive waste at the Vektor facility complex, which is now being developed. When the Vektor facility is in full operation, it may be possible to begin the work of removing FCM and other radioactive waste from the Shelter under cover of the NSC.

Such a strategic approach is foreseen by the Comprehensive Programme on radioactive waste management that was approved by the Ukrainian Government in 1996, and then issued in the (Concept 2004). Prior to elaboration of such a program, special field studies and geological investigations must be carried out in the Exclusion Zone and its surrounding area, and in particular at the areas with crystalline rocks that have a depth of more than 500 m. According to (Concept 2004), it is considered reasonable to begin a specific investigation for exploring the most appropriate geological site in this area in 2006. Following such planning, the construction of a deep geological disposal facility might be completed before 2035-2040.

Management of future liquid radioactive waste from the Shelter is planned at the new liquid radioactive waste-treatment plant at the ChNPP site. However, the management of liquid waste containing transuranic elements remains an issue to be resolved.

In addition the strategy for management of radioactive waste from the accident also needs to consider management of other storage sites containing about 2000 pieces of contaminated equipment (transport vehicles, helicopters, tanks, etc.) at the sites (Rossoha-1/Rossoha-2) that was used in the first months after the accident, and for which the final disposition has not been determined.

6.2.3. Environmental aspects

Safety concerns for most of the temporary radioactive waste facilities in the Exclusion Zone need to be viewed within the context that most of these facilities are located in very contaminated areas, with surface levels for ^{90}Sr in the range of 400 to 20,000 kBq m^{-2}; for ^{137}Cs - 700 to 20,000 kBq m^{-2}, and for $^{239+241}$Pu - 40 to 1000 kBq m^{-2}. In this same territory, the temporary radioactive waste facilities occupy a relatively small volume covered with several meters of soil and other geological material.

The major concern is the risk of increased contamination of groundwater and the possibility, in the future, that such contamination reaches major sources used as water supplies. The measurements reported in the French-German initiative (2004) (see Table 6.5), clearly show that some temporary radioactive waste facilities have significant radiological impact on groundwater. In particular, flooded and partially flooded trenches are giving rise to enhanced migration due to the absence of engineered safety features, while more favorable settings, as the Buriakovka Site, lessen the radionuclide-release variations and maintain concentrations in groundwater at comparatively low-levels.

For a part of the year, some temporary radioactive waste facilities are very near or in the groundwater table, which can affect radionuclide dispersion. It is noticeable that water-table levels at the trenches and landfills vary from about 1 to 7 m depth and vary depending on the season. Part of the facilities at Stantzia Yanov and Neftebaza are constantly flooded. Flooding is also an important concern at the Kompleksny disposal site, where the waste containers are flooded from 0.5 to 0.7 m from the bottom of the disposal facility (RAW in Ukraine 2003). Therefore the degree of contamination is monitored in these disposal sites using a monitoring system, established in 1986-1989 period, which needs upgrading.

TABLE 6.5. CONTAMINATION OF GROUNDWATER NEAR SELECTED TEMPORARY RADIOACTIVE WASTE FACILITIES, BQ L^{-1}, IN 1994-1995 (BUGAI ET AL. 1996) AND IN 1999 (DEREVETS ET AL. 2000)

Waste Site	^{90}Sr, Bq/L		^{137}Cs, Bq/L		$^{239+240}Pu$, Bq/L
	1994-1995	1999	1994-1995	1999	1994-1999
Ryzhy Les	100 - 120 000	100-230	0.1 -100	0.1-2.5	0.4 –0.6
Stroybaza	3 - 200	30-50	1-20	0.02-0.004	No data
Peschannoe Plato	3 - 10	2-40	0.7 - 3	0.02-0.1	No data

Monitoring results of groundwater contamination around the temporary storage facilities indicated concentrations of ^{90}Sr in the range of 100–100 000 Bq m^{-3} (RAW in Ukraine 2003; Bulletin 2004). The highest levels of contamination are detected at the northern part of the ChNPP site that also runs into the Pripyat River. Therefore, actual and potential impacts from radionuclides reasonably exist for those radioactive waste facilities located in surface soils immediately next to the riverside in alluvial soils and which might be at regular risk of flooding during high water periods (French-German Initiative 2004, Bondarenko et al. 2004). These types of disposal facilities were studied during the last 5 years, and continue to be inventoried as a basis for their step-by-step removal and relocation to the stationary organized disposal facilities.

As mentioned above, the rate of radionuclide migration with groundwater fluxes is much lower than the hydraulic transport of the water itself. This means that, due to retardation factors and geochemical processes, the majority of radionuclides being released from the body of the temporary radioactive waste facilities are accumulating in the geological media. Taking into account the adsorbtion capacity of the soils and geological media surrounding the temporary radioactive waste facilities, several prior studies have shown that a significant fraction of ^{90}Sr is still associated with the fuel matrix, which makes its release to the pore water in the soil prolonged for many years. As a result, fluxes of radionuclides with ground water, even for such mobile radionuclides as ^{90}Sr, are very low. Plutonium isotopes (and ^{241}Am associated with it) have not been adequately studied yet. However, it is well known that its migration beyond the temporary radioactive waste sites is negligible (Fig. 6.12).

Studies of the vertical and lateral transfer rates of radionuclide fluxes demonstrated that for the local soil, there is a low risk of radionuclide contamination to groundwater and, therefore, proportionally low risk for significant contamination of the Pripyat River in the future, as discussed also in Section 2.5 of this report (see Fig. 2.59). Studies showed (Antropov et al. 2001, Shestopalov et al. 2002) that the leading edge of contaminated groundwater from most of the significant temporary radioactive waste facilities may reach regional surface water within 100 and more years, making this issue of minimal importance in terms of radiological risk assessment for populations living downstream of the Pripyat River system. However, for the Exclusion Zone groundwater is still an important potential source for radionuclide migration in the environment, and therefore these sources have to be under regular monitoring and institutional control.

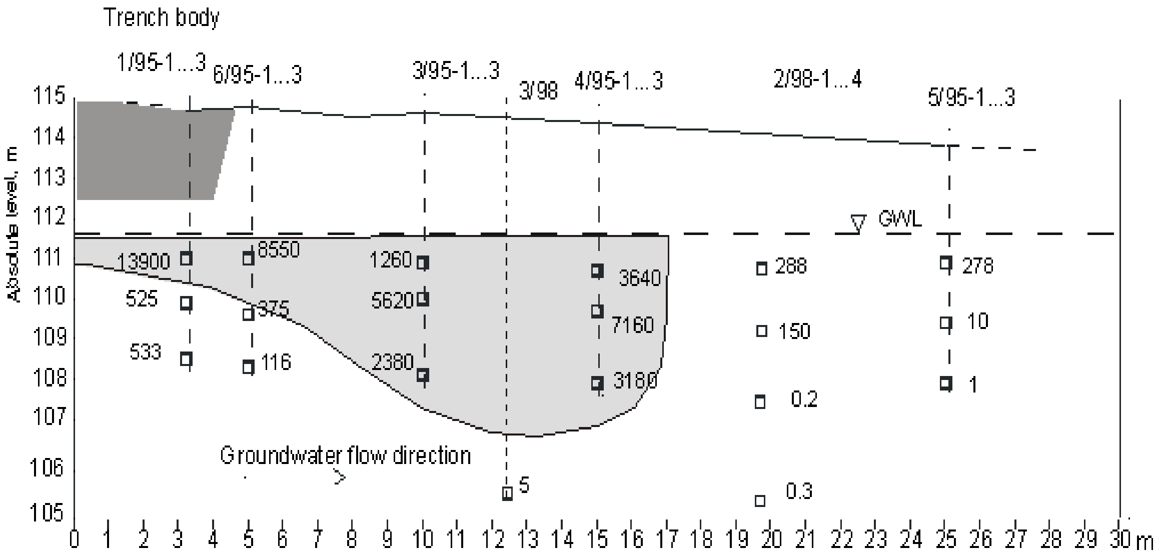


FIG. 6.12. Spatial distribution of ^{90}Sr (Bq L^{-1}) in the groundwater near surface trench No. 22 of the Ryzhy Les facility in October 1998 (Antropov et al. 2000).

The long-term strategy of the temporary facilities is related to the decision for managing the radiological risks associated with temporary radioactive waste facilities. The ultimate goal needs a priori to be that waste be disposed or left in temporary radioactive waste facilities that ensure sufficient confinement of ^{137}Cs and ^{90}Sr to allow their decay without having generated significant impacts to potential critical groups in the long-term. For those temporary radioactive waste facilities located by the banks of the Pripyat River that could be inundated by floodwaters, the preferred strategy is to remove and relocate the waste into the organized disposal facilities according to the waste hazards and waste categorization.

And for those temporary radioactive waste facilities in the Exclusion Zone that have not been inventoried, and for which the potential for undesirable future contamination of surrounding groundwater and surface water is uncertain, safety assessments need to be performed taking into account the radioactive decay and natural attenuation. There is clearly a need to assess with an increased level of confidence the migration of contamination plumes, their interface and radioactivity that may be released in major water resources and supply areas (aquifers, rivers, reservoirs, local supply for the NPP and Exclusion Zone). Such assessment needs to consider at the same time all sources of releases likely to affect these water resources.

The safety-assessment results will guide the decision on appropriate remediation or institutional control measures at the temporary sites. Operational waste-acceptance criteria (e.g., activity concentrations) also need to be established to ensure that potential exposures from various scenarios remain acceptable, on the hypothesis that resettlement in the Exclusion Zone has occurred after hundreds of years. It is obvious that such disposal sites will need to undergo institutional control for a period of a few hundred years to allow ^{137}Cs and ^{90}Sr contamination to decay to insignificant levels. This would require significant resources for monitoring, implementation of recovery actions to stabilize situations with secondary waste to manage, and probably major restrictions for resettlement. However, long-term institutional controls should not be considered as an alternative to recovery actions that can improve overall safety of the Exclusion Zone.

6.2.4. Issues and areas of improvement

6.2.4.1. Comprehensive Radioactive Waste-Management Programme for the Exclusion Zone and the ChNPP

There is no precise comprehensive programme for radioactive waste management established today for further cleanup of contaminated areas or temporary radioactive waste facilities at the ChNPP and within the Exclusion Zone. As mentioned before, the ongoing strategy is to monitor temporary waste sites with the highest radiological risk to the environment, so as to assess whether clean up or environmental protection actions are needed. In addition, options for long-term processing, storage and disposal of long-lived and high-level waste from the ChNPP and the Exclusion Zone, as well as management of liquid-waste contaminated with transuranic elements are to be selected, and the necessary facilities developed. Development of such a programme could assure the consistent and coordinated long-term management of all types of accident waste and hence protection of workers, public and the environment.

6.2.4.2. Ultimate Decommissioning of Unit 4 and Radioactive Waste Generation

The strategy for dismantlement of the Shelter and decommissioning of Unit 4 needs to consider two main factors: the safety implications of management of associated radioactive waste (in particular of the high-level waste) and the safety implications of delaying recovery operations. The strategy for the management of radioactive waste that cannot be disposed in near surface facilities needs to be developed. Specifically, there is a need for new waste-management facilities (e.g., storage of long-lived waste, geological disposal), with considerations on the capacity of these facilities and also the possibility to use the existing facilities dedicated to decommissioning of the ChNPP. Particular attention needs to be given to the establishment of an adequate infrastructure and facilities for management of long-lived waste (in particular large quantities of soil, transuranic liquid waste, and contaminated metal) and high-level waste (i.e., fuel containment material), and subsequent disposal.

6.2.4.3. Waste Acceptance Criteria

The waste management programme being implemented does not consider unified criteria for categorization of accident radioactive waste needed for selection of an appropriate management option for individual radioactive waste streams. Adopting unified criteria for waste management, based on ^{137}Cs and alpha specific radioactivity levels in waste, is under development. Though such criteria are more adapted for appraising the potential for waste to be accepted in near surface facilities, the question of estimating the specific activities for existing waste, especially in temporary radioactive waste facilities, remains difficult to solve. The development of waste-acceptance criteria is important in order to assure protection of workers and environment, as well as public in the long-term.

6.2.4.4. Long Term Safety Assessment of Existing Radioactive Waste Sites

There is a need to identify the remaining temporary radioactive waste facilities, and appropriately mark them to prevent inadvertent intrusion. The long-term impact of these facilities on the environment also needs to be evaluated in order to estimate the need where necessary, for implementation of upgrading or remedial actions.

Taking into account the large number of facilities, there is also a need to prioritise the needs for safety assessment and therefore establish categories of waste disposal and temporary facilities that allow appraising their hazard potential for the public and the environment.

These assessments should evaluate safety in present conditions and with consideration of possible future resettlement. Considerations on the need to restrict the number of sites that are flooded or will, in the future, need extensive control for over some hundreds of years.

In order to select those facilities with higher radiological risk it is important to improve methods for assessing waste radioactivity in the temporary facilities, especially of long-lived radionuclides. For pragmatic reasons, this assessment should be based on a limited number of parameters and measures. In this way uncertainties that affect present estimations of the potential impact of individual facilities to the environment will be reduced and a consistent assessment that takes into account all existing and potential sources of contamination in the Exclusion Zone will become feasible.

6.2.4.5. Potential Recovery of past waste facilities located in the Exclusion Zone

Work is underway on development of a strategy for the management of past waste facilities (Saversky et al. 2001, RAW in Ukraine 2003), which envisages three options for the different facilities, depending on their status and radiological hazard to the environment:

— Retrievability of waste and disposal in the short term in order to minimize environmental consequences and improve safety of workers, e.g., Industrial sites, the Shelter, the flooded temporary storage, and the Kompleksny disposal facility;

— Possible temporary storage of waste under institutional control in accordance with radiation-protection requirements with a view to future disposal – e.g., Podlesny disposal facility, contaminated equipment from the liquidation of the Chernobyl accident; and

— Investigation of facilities, which need to be studied in order to decide on adequate intervention measures, e.g., temporary radioactive waste facilities, soil from the construction of new confinement.

6.3. Future of the Chernobyl Exclusion Zone

Long-term development of the Exclusion Zone is an important and complex task, that takes into account various technical, economic, social and other factors; various options have been considered for the evolution of this zone. According to (Likhtarev et al. 2000) after 2015 about 55% of the territory around the Chernobyl NPP could be considered for release from radiological limitations according to Ukrainian legislation. However, the final decision about permission to return people to this zone must take into account the inhomogeneous character of the contaminated land, specific features of radionuclide migration and accumulation in different portions of the local landscape, and also the routine habits of the population living in this region (hunting, fishing, berry picking, mushroom gathering, etc.).

The overall plan for the development of the Exclusion Zone is to recover the affected areas of the Exclusion Zone, redefine the Exclusion Zone, and make the non-affected areas available for resettlement by the public. This will require well-defined administrative controls as to the nature of activities that may be performed in the resettled areas, prohibition of food crops and cattle grazing, and use of only clean feed for cattle. Accordingly, these resettled areas are best suited for an industrial site rather than a residential area.

For the reasons given above, the activity focused on decontamination and dismantlement of the Shelter facility and on radioactive waste management in this territory is expected to continue, which requires the optimal management of this area. The new concept foresees division of the Exclusion Zone into different sections:

— **The industrial zone** is planned to include the most contaminated areas, where the Chernobyl NPP, facilities for processing radioactive waste, and main radioactive waste-storage areas are situated. Primarily industrial activities are envisaged to be carried out here, specifically the construction of the NSC facilities. To provide infrastructure for NSC construction, new roads, shipping yards, railways, and other support structures are planned to be created. The town of Chernobyl has been considered as an option for such infrastructure development (EIA 2003). If the Exclusion Zone is selected as the site for construction of the geological depository for high-activity and long-lived radioactive waste, a significant amount of drilling and mining work has to be performed, which will also require specific development of the engineering infrastructure.

— **The sanitary restricted zone** is considered to be a buffer area between industrial and natural reserve areas.

— **The natural reserve areas** are planned to be located where most industrial and human activity are prohibited, with the aim of preservation of the basic natural landscapes and biodiversity of the region.

First order rehabilitation of land is expected to create optimal conditions for industrial activity and environmental protection for a long period of time. For instance, the NSC is expected to be operational for at least 100 years. Different types of radioactive storage facilities must provide safe storage for 300 or more years. A possible action at this area may be construction of the main geological disposal facility for radioactive waste. A National Engineering Centre may be established for processing all categories of radioactive material and waste to be delivered to the geological depository from different parts of Ukraine.

Continued monitoring and supporting studies in the Exclusion Zone are needed to form a basis for the review and optimization of the management strategy in the contaminated territories, and also for developing basic and practical knowledge about the dynamics and evolution of radionuclide migration, need of additional engineering barriers, or implementation of environmental remediation technologies.

In summary, it can be noted that the future of the Exclusion Zone for the next hundred years and more is envisaged to be associated with the following activities:

— Construction and operation of the NSC and relevant engineering infrastructure;

— De-fuelling, decommissioning and dismantling of Units 1, 2 and 3 of the ChNPP and the Shelter;

— Construction of facilities for processing and management of radioactive waste, in particular a deep geological repository for high-activity and long-lived radioactive material;

— Development of natural reserves in the area that remains closed to habitation; and

— Maintenance of environmental monitoring and research activities.

6.4. Conclusions and recommendations

6.4.1. Conclusions

It could be concluded that the existing uncertainties associated with the stability of the Shelter structures, the radioactive inventory, the insufficient confinement, the evolving characteristics of the FCM and the conditions inside and around the Shelter (e.g., groundwater conditions) create uncertain safety conditions from the point of view of protection of workers, public and environment in the future. Therefore, continuation of the stabilisation measures at the Shelter and construction of the NSC is expected to improve safety and prevent or mitigate accidental scenarios that would be expected to have consequences outside the Exclusion Zone.

However, these measures are foreseen for not longer than a hundred years and also require prompt solution of safe predisposal and disposal management of radioactive waste to be generated during this period, in particular for management of long-lived and high-level waste. Planning and evaluation of safety for decommissioning of Unit 4 after the construction of the NSC is needed in order to develop appropriate measures and allocate necessary resources for the conversion of the Shelter into a safe environmental system.

Decommissioning of Unit 4 will generate significant amounts of radioactive waste with a wide range of characteristics that will need to be safely managed as part of the decommissioning and waste-management activities at the ChNPP and the Exclusion Zone. A comprehensive strategy for management of all waste streams is needed to ensure adequate infrastructure and capabilities for processing, storage and disposal of this waste. The development and implementation of such a strategy will also need to consider the future development of underground and on-surface storage and disposal facilities, some of which are flooded.

At present studies show that the known waste facilities do not represent an unacceptable hazard to public; however, assessment of their long-term impact to the public and the environment in conjunction with the remaining sources of radioactive contamination in the Exclusion Zone is needed, and particularly for those facilities that are flooded and represent higher risk.

For less known and studied waste facilities it will be necessary to reduce the uncertainties associated with the waste inventories and facility characteristics, to assess the long-term safety of these facilities; to monitor the dynamics of radionuclide migration into the environment, and, where necessary, implement remediation measures. This is important for the successful implementation of waste-management activities in the Exclusion Zone and conversion of the zone into a safe environmental system.

6.4.2. Recommendations

Recognising the ongoing effort on improving safety and addressing the aforementioned uncertainties in the existing input data, the following main recommendations are made regarding the dismantlement of the Shelter and management of radioactive waste generated as a result of the accident:

(1) Because individual safety and environmental assessments have been performed only for individual facilities at and around the ChNPP, a comprehensive safety and environmental impact assessment according to the international standards and

recommendations that encompasses all activities inside the entire Exclusion Zone is to be performed.

(2) During the preparation and construction of the NSC and soil removal special monitoring wells are expected to be destroyed. Therefore, taking into account the changing hydrogeological conditions at the Shelter and potential exposure pathways (e.g., through groundwater), it is important to maintain and improve environmental monitoring strategies, methods, equipment and staff qualification needed for the adequate performance of monitoring the conditions at the ChNPP site and the Exclusion Zone.

(3) Dismantlement of the Shelter after a long period of time (about 50 years) does not seem to be a viable option, due to the need for long-term maintenance of structure stability and integrity, resources and knowledge. The long-term strategy raises concerns related to the potential loss of the most experienced personnel at the Chernobyl NPP, and maintaining stable manpower loading necessary for the safe operation of the NSC. It is reasonable to begin retrieving FCM soon after dismantling unstable structures of the Shelter rather than waiting for the availability of a geological disposal facility. However, in optimisation of the Shelter dismantlement, special attention should be paid to doses that will be received by the workers in different time periods.

(4) Development of an integrated radioactive waste-management programme for the Shelter, the ChNPP site and the Exclusion Zone is needed to assure application of consistent management approaches, and sufficient facility capacity for all waste types. Specific emphasis needs to be paid to the characterisation and classification of waste (in particular waste with transuranic elements) from all the remediation and decommissioning activities, as well as the establishment of sufficient infrastructure for safe long-term management of long-lived and high-level waste. Therefore, development of a necessary waste-management infrastructure is needed in order to assure sufficient waste capacity, which at present limits the rate and continuity of remediation activities at the Chernobyl NPP site and in the Exclusion Zone.

(5) A coherent and comprehensive strategy for rehabilitation of the Exclusion Zone is needed with particular focus on improving safety of the existing waste-storage and disposal facilities. This will require development of a prioritisation method for remediation of the sites, based on safety-assessment results, aiming at decisions on which sites from which waste will be retrieved and disposed, and at which sites the waste will be allowed to decay *in situ*.

REFERENCES TO SECTION 6

ANTROPOV, V.M., BUGAI, D.A., DUTTON, L.M.C., M.Y. et al. (2001) Review and Analysis of Solid Long-lived and High Level Radioactive Waste arising at the Chernobyl Nuclear Power Plant and the Restricted Zone// DG Environment Project No B7-5350/99/51983/MAR/C2, NNC, 370 p.

BASIC SANITARY RULES OF RADIATION SAFETY OF THE UKRAINE (2000) Ministry of Health of Ukraine, Kiev.

BOGATOV S. and BOROVOY A. (2000) Assessment of inventory and determination of features of dust contained in the Shelter. Preprint of ISTC Shelter, Chernobyl, 16 p. (In Russian).

BONDARENKO O.O., DROZD I.P., LOBACH G.O., TOKAREVSKIY V.V., SHIBETSKIY Y.O. (2004) Present problem of the radioactive waste management at the Chernobyl Exclusion Zone. Bulletin of Ecological State of the Chernobyl Exclusion Zone, No 23, pp. 36-40 (In Ukrainian).

BOROVOY A. (2001) Object Shelter Safety Analysis Report.

BOROVOY A., BOGATOV S. and PASUKHIN E. (1999) Current status of the Shelter object and its impact on the environment. Radiokhimiya, v. 41, Nr. 4, pp.368 – 378 (In Russian).

BUGAI, D.A., WATERS, R.D., DZHEPO, S.P. and SKALSKY, A.S. (1996) Risks from radionuclide migration to groundwater in the Chernobyl 30-km zone. Health Physics **71**, pp 9-18.

BULLETIN OF ECOLOGICAL STATE OF THE CHERNOBYL EXCLUSION ZONE (2004), No 21 (In Ukrainian). ***Specify ref. - MB***

CONCEPT OF RADIOACTIVE WASTE MANAGEMENT AT THE SHELTER (1999) Approved by State Commission on Issues on Complex Solution of Problems of ChNPP on 15.11.1999.

DEREVETS, V. et al. (2000) Radiation conditions of the Chernobyl Exclusion Zone in 1999. Bulletin of Ecological State of the Chernobyl Exclusion Zone, No 17, pp. 5-19 (In Ukrainian).

ENVIRONMENTAL IMPACT ASSESSMENT (2003) New Safe Confinement Conceptual Design: Chernobyl Nuclear Power Plant – Unit 4. State Specialized Enterprise Chernobyl Nuclear Power Plant, Kyiv.

EUROPEAN COMMISSION (2001) Characterisation of Radioactive Waste Located at “Shelter” Industrial Site”. Report EUR 19844.

FINAL REPORT (2001) Analysis of Current Safety of the Shelter and Assessment Prognosis of Situation Development. ***Specify ref. - MB***

FRENCH-GERMAN INITIATIVE FOR CHERNOBYL (2004) Final Report on Project n°2 "Radioecological Consequences of the Chernobyl Accident" Sub-Project "Waste Dumps and Waste Strategies Management ", Deliverable 18.

GMAL, B., MOSER, E. F., PRETZSCH, G. and QUADE, U. (1997) Criticality Behaviour of the Fuel-Containing Masses inside the Object „Shelter“ of the Chernobyl NPP, Unit 4, GRS Report A - 2414.

INTEGRATED RADIOACTIVE WASTE PROGRAMME AT THE STAGE OF SHUTDOWN OF THE CHERNOBYL NPP AND TRANSFORMATION OF SHELTER INTO ENVIRONMENTAL SAFE SYSTEM (2004) Approved 15.06.2004 by Ministry of Fuel and Energy, Administration of Exclusion Zone, State Committee of Nuclear Regulation, Ministry of Environment and Natural Resources, and Ministry oh Health of Ukraine.

INTERNATIONAL ATOMIC ENERGY AGENCY (1994) Classification of Radioactive Waste, Safety Series No. 111-G-1.1, IAEA, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (2001) Present and Future Environmental Impact of the Chernobyl Accident, IAEA TECDOC 1240, Vienna.

INTERNATIONAL ATOMIC ENERGY AGENCY (2005) Radiological Conditions of the Dnieper River Basin. Assessment by an IAEA Expert Team and Recommendations for an Action Plan, Vienna (in print).

KASHPAROV V.A., LUNDIN S.M., ZVARICH S.I., YOSCHENKO V.I., LEVTCHUK S.E., KHOMUTININ YU.V., MALOSHTAN I.N., PROTSAK V.P. (2003) Territory contamination with the radionuclides representing the fuel component of Chernobyl fallout. *The Science of The Total Environment*, **317:** 105-119.

LAW OF UKRAINE ON RADIOACTIVE WASTE MANAGEMENT of 30.06.1995 (Government Courier Paper).

LIKHTAREV I.A., KOVGAN L.M., BONDARENKO O.O. (2000) If there is future for exclusion zone and population of relocated territories? (Opinion of a radiologist). Bulletin of Ecological State of the Chernobyl Exclusion Zone, No 15, pp. 44-49 (In Ukrainian).

NATIONAL REPORT DEVELOPED IN COMPLIANCE WITH THE JOINT CONVENTION ON THE SAFETY OF SPENT FUEL MANAGEMENT AND ON THE SAFETY OF RADIOACTIVE WASTE MANAGEMENT (2003) Kyiv**.**

NOVOKSHCHENOV, V. (2002) The Chernobyl Problem. Civil Engineering 72:5.

NRSU (1998) RADIATION SAFETY STANDARDS OF UKRAINE (NRSU-97); Ministry of Health of Ukraine, Kiev, 135 p.

NUCLEAR ENERGY AGENCY, ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT (2002) Chernobyl: Assessment of Radiological and Health Impacts. Update of Chernobyl: Ten Years On.

PANASUK, N. PAVLUCHENKO, N., RUDKO. V. et al. (2002) Results of the Assessment of Contaminated Soil and Underground Water at the Shelter Site, Journal of Chernobyl Problems No. 7, p. 97.

PRETZSCH G. (1997) Radiological Consequences of the Hypothetical Accident "Roof Breakdown" of the Chernobyl Sarcophagus, In: One Decade after Chernobyl: Summing up the Consequences of the Accident, IAEA-TECDOC-964, v.2, pp. 591-597, IAEA, Vienna.

SAVERSKY, S, BUGAY, D., ANTROPOV, V. (2001) Experience and Perspectives of Radioactive Waste Management at the Exclusion Zone, Bulletin of Ecological State of the Chernobyl Exclusion Zone, No 17 (In Ukrainian).

SCHMIEMAN E.A, WRONA, M.W, CONVERT P.H, NEMCHINOV, Y.I, P, BELICARD, M. DURST, V. KULISHENK, O, and HOGG C.B (2004) Conceptual Design of the Chernobyl New Safe Confinement – an Overview. Bulletin of the Canadian Nuclear Society.

SCIENCE AND TECHNOLOGY CENTER IN UKRAINE (1998) Comprehensive Risk Assessment of the Consequences of the Chornobyl Accident, Kyiv.

SHESTOPALOV V.M., Ed. (2002) Chernobyl Disaster and Groundwater. A A Balkema Publisher, 289 p.

SIP PROJECT (2004) Shelter Transformation Safety. ***Specify ref. - MB***

STATEMENT ON POLICY OF REGULATION OF NUCLEAR AND RADIATION SAFETY OF THE SHELTER, CHERNOBYL NPP (1998) Minister Approval No. 49 of 08.04.1998.

UNSCEAR (2000) Sources and Effects of Ionizing Radiation. United Nations Scientific Committee on the Effects of Atomic Radiation, Report to the General Assembly, Vol. II, Annex J: Exposures and Effects of the Chernobyl Accident.

VARGO G.I., BARYAKHTAR V., KHOLOSHA V., KUKHAR V., LOS I., POYARKOV V., SHESTOPALOV V. (2000) The Chernobyl Accident: A Comprehensive Risk Assessment. Battelle Press, Columbus.

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Consultants Meetings

Vienna, Austria: 30 June – 4 July 2003
Vienna, Austria: 15–19 December 2003
Vienna, Austria: 26–30 January 2004
Vienna, Austria: 14–18 June 2004
Vienna, Austria: 18–22 October 2004
Vienna, Austria: 29 November – 3 December 2004
Vienna, Austria: 31 January – 4 February 2005