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CONSEQUENCES OF THE CHERNOBYL ACCIDENT FOR THE NATURAL AND HUMAN ENVIRONMENTS

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ABSTRACT

In the ten years since the Chernobyl accident, an enormous amount of work has been done to assess the consequences to the natural and human environment. Although it is very difficult to summarize such a large and varied field, some general conclusions can be drawn. This Background Paper includes the main findings concerning the direct impacts of radiation on the flora and fauna; the general advances of knowledge in the cycling of radionuclides in natural, seminatural and agricultural environments; some evaluation of countermeasures that were used; and a summary of the human radiation doses resulting from the environmental contamination. Although open questions still remain, it can be concluded that: (1) At high radiation levels, the natural environment has shown short term impacts but any significant long term impacts remain to be seen; (2) Effective countermeasures can be taken to reduce the transfer of contamination from the environment to humans but these are highly site specific and must be evaluated in terms of practicality as well as population dose reduction; (3) The majority of the doses have already been received by the human population. If agricultural countermeasures are appropriately taken, the main source of future doses will be the gathering of food and recreational activities in natural and seminatural ecosystems.

1. ISSUES

In order to discuss the environmental consequences of the Chernobyl accident, we must first define 'consequences' and 'environment'. Consequences are the damages and benefits that can be observed and specifically attributed to the accident. There is still no unique scientific measure of damage to the environment, and in many cases it is determined by general consensus. In some cases, only changes to the pre-existing conditions can be noted and the designation of damage remains undetermined.

The term *environment* can be broadly defined as the combination of external conditions that affect the development, survival and reproduction of living things and in this respect can be considered to include conditions that affect human beings as well as flora and fauna. One of these external conditions is the natural background of radiation exposure, which was significantly elevated following the Chernobyl accident, especially in the region around the reactor and in the first few months after the accident. Where the additional radiation doses were so high as to affect the survival and reproduction of certain plants and animals, this can be considered a direct environmental consequence of the radiation exposure caused by the accident. However, the external conditions were also altered because of human interventions for self-protection. The usual considerations in this context for the human environment are doses and health risks, the loss of agricultural products and loss of use of land. Protective actions also changed the conditions for development and survival of various plants and animals, and their knock-on impact can be considered an indirect environmental consequence.

The main issues to be addressed in this paper are:

- After ten years, what environmental consequences can be specifically attributed to radiation from the release and to the subsequent protective actions taken as a result of the accident ; and
- Will the affected regions be safe and normal in the future? And if so, when?

2. BACKGROUND

For decades, research in radioecology and radiation protection has addressed the potential impact of releases of radioactive material to the environment. Except in the few cases of previous accidental releases (such as the Kyshtym (USSR, 1957) and Windscale (United Kingdom, 1957) accidents), field and laboratory experiments have investigated (1) the direct effects of radiation from large controlled sources on plants and animals, or (2) the movement of relatively low levels of radionuclides in the environment arising from the production and use of nuclear materials (including atmospheric weapons testing and nuclear power production), in order to assess the doses to humans.

Direct effects of radiation on plants and animals

The sensitivity of plants and animals to ionizing radiation varies widely, depending on the species (e.g. Turner 1975, Whicker 1974). The significant effects of exposures range from impaired reproductive capacity at high doses to mortality at very high doses. Further effects include reduced growth and lower yields of plants. It should be noted that these attributes can also be influenced by environmental factors other than radiation. Because of the complexities involved, generalization is difficult. Approximately, however, sensitivity to direct effects of radiation may be ordered as follows (from most sensitive to least): mammals, birds, fish, amphibians, reptiles, crustaceans, molluscs, insects, bacteria, protozoa and viruses. Higher plants overlap the upper range of sensitivity of animals, and mosses, lichens and algae appear at the lower ranges of sensitivity. For all organisms, the earlier life stages are most sensitive. Thus, embryos, juvenile stages and in general, the active growth and development phases are the most sensitive to radiation damage.

The responses of organisms to radiation exposure may become manifest from the individual biomolecule to the ecosystem. The significance of any response depends on the criterion of damage adopted. For humans, ethical considerations present the individual as the principal object for protection. Humans display an enormous range of attitudes towards other species, but for the vast majority of organisms it is the populations that are considered important.

High doses of radiation received over short periods are able to induce changes in attributes such as mortality, fertility, growth rate, vigour and mutation rate in individual plants and animals. Lethal dose thresholds for certain radiosensitive plants (such as coniferous trees) are typically about 10 Gy, with severe growth inhibition at some 40%–50% of the lethal dose and failure to seed at perhaps 25% of the lethal dose. At doses less than 10% of the lethal dose, plants typically maintain normal appearance (Sparrow et al. 1971). Mammals are the most radiosensitive terrestrial animals – acute lethal doses are of the order of 6–16 Gy for small mammals and 4–8 Gy for larger animals and domesticated livestock (Bond et al. 1965). The reproductive rates of mammals may be depressed at doses that are 10% of those which lead to mortality.

For the effects of chronic low levels of radiation, if the dose rate is below about 1 mGy (milligray) per day measurable detrimental effects among populations of terrestrial animals and plants are unlikely. For aquatic organisms even higher dose rates, 10 mGy per day, give no significant effect at the population level (IAEA 1976, IAEA 1992).

Clearly, radiation effects at the population and community levels are manifest only if sufficient numbers of individuals are killed or their reproduction is affected. Genetic information that is altered by radiation in the cells of an individual organism is, for various reasons, extremely unlikely to be perpetuated at the population level (UNSCEAR 1986). Other species may be indirectly affected, for example, by loss of habitat or gain of a competitive advantage. It should be noted that radiation is but one of the sources of stress that can influence the equilibrium between communities and ecosystems, and it is often difficult to correctly attribute any observed response to a specific cause.

The human environment

The majority of research on radionuclides in the environment has been to gain a better understanding of their transfer via plants and animals to human beings (e.g. IAEA 1982, UNSCEAR 1982). Major programmes were in progress before 1986 to develop and improve complex models that evaluate the likely consequences of accidental releases to the environment in terms of human health effects and land areas affected by countermeasures (e.g. CEC 1985). Radionuclides such as ^{131}I were shown to often be important in the short term and ones such as ^{137}Cs and ^{90}Sr had been shown to be important in the long term.

The environmental models as developed by 1986 were adequate for the purposes of implementing guidance on countermeasures in the early phase of an accident to protect against the initial high dose rates. Less attention had been given to the possible need of employing complex countermeasures in the longer term. The models focused on the major components of the risk to humans, and it is fair to say that minor components and transfer in less common environments were less well studied. Thus, for example, at the time of the Chernobyl accident it was possible to predict fairly accurately the concentration in milk, cereals, meat and green vegetables and their change with time in the short term, and to incorporate these expectations into emergency plans and take effective actions. However, there were some environments, so called natural and seminatural ecosystems (e.g. forests, meadows), where the existing information was insufficient or inadequately synthesized to be able to predict accurately the behaviour of important radionuclides. Moreover, prior to the accident, it was usual practice to calculate average values of radionuclide concentrations and radiation exposure. Since the Chernobyl accident, more attention has been given to the evaluation of the variation in nuclide concentrations and radiation doses.

Radiation protection is concerned with the protection of the health of individual human beings against the effects of ionizing radiation. Traditionally it has been thought likely that the level of safety required to protect all human individuals will protect the other species in our environment, although not necessarily on individual members of those species.

It is fair to say that at the time of the accident, while ranges of intervention levels existed for the most relevant countermeasures (such as sheltering, evacuation, distribution of stable iodine tablets, relocation and food countermeasures), there remained some confusion in their application. In particular, there were misconceptions on the role of dose limits, the need to consider avertable doses in assessing the benefits of a countermeasure, and finally in the need for realism in the application of action levels (ICP, RISO-R-716). Since then there have been major efforts by several international organizations to elucidate these issues, particularly by ICRP, OECD/NEA, EC, FAO, WHO and IAEA (see for example, BSS, SS109, FAO countermeasures).

Another impact on humans besides the doses received are the loss of amenity of the environment for the present and future generations because of increased levels of radiation or contamination above action levels. The loss of commercial use of the land, restricted or condemned recreational areas, and restrictions or bans on consumption of wild and home grown foods can have a serious impact on the living conditions of the local populations. In some cases, the perception of the contamination rather than its physical effect can be a very significant factor.

3. DISCUSSION

Release and dispersion of radioactive material

In June 1986, Soviet experts reported their evaluation of the amount of radionuclides released as a result of the accident ('source term') (INSAG 1986). At that time, contours had been drawn around the evaluations of deposited radioactive material and the release estimates were based on the material estimated to be inside the contours. These contours did not extend outside the ex-USSR. Since that time several studies have been carried out to refine the original estimates, taking into account among other things deposition outside the USSR (Belyaev et al. 1991, Borovoi 1992; Buzulukov and Dobrynin, 1993, OECD 1995). Revised estimates are presented in Table I and broad agreement exists between them. Notably, the revised estimates of the total release of ^{131}I and ^{137}Cs are approximately twice those of the original June 1986 evaluation.

The releases that occurred at the time of the explosion and during the 10 days that followed can be divided into several stages. The total daily releases (decay corrected to 6 May 1986) as presented in [INSAG 1] are reproduced in Fig. 1 (IAEA 1986). Some short isolated releases in the 20 days after the accident were inferred from measurements (Buzulukov and Dobrynin 1993), but the interpretation of these data is still debated and in any case they do not significantly affect the overall picture.

The pattern of atmospheric dispersion and deposition of the radionuclides was complex owing to large variations in the release fractions of the different radionuclides and changing meteorological conditions. An example of the trajectories of radioactive plumes originating from releases at Chernobyl at six different times is given in Borzilov and Klepikova (1993). Due to the time course of emissions the patterns indicated as '1' and '2' are expected to be relatively enriched in ^{131}I compared with ^{137}Cs. The distribution of fuel particles ('hot' particles) has been estimated as 0.3–0.5% of the core on site, 1.5–2% of the core in the near zone of 0–20 km, and 1–1.5 % of the core was deposited beyond 20 km (Rauret and Firsakova 1996). The volatile elements were more widely dispersed beyond 100 km. About 45% of the total ^{137}Cs was deposited within the former USSR (predominantly within Belarus, Ukraine and parts of Russia), 39% in Europe, 8% in Asia, 7% in the oceans and the remainder in the other regions of the northern hemisphere (UNSCEAR 1988). Essentially no material was detected in the southern hemisphere. More sophisticated analyses of the deposition pattern have recently been performed (Izrael et al. 1996) which may lead to a refinement of the UNSCEAR figures and source term estimates.

Approximately 85% of the total release was composed of radionuclides with half-lives less than about a month; another 13% was of radionuclides with half-lives of several months, about 1% had half-lives of about 30 a; while about 0.001% had half-lives greater than 50 a. Table II indicates how the total activity of the material remaining in the global environment falls away with time. Thus, due to radioactive decay the total amount of radioactive material still present in the environment after ten years has fallen to about 1% of the total amount released (80 PBq of long lived radionuclides, principally ^{137}Cs and ^{90}Sr).

Because of the short half-life of ^{131}I (8 days), adequate measurements of this radionuclide could not be made. Maps of deposition were developed on the basis of dose rate measurements, dispersion calculations, and limited information on the radionuclide mixes. These maps cannot be corroborated, which is unfortunate, because one of the more compelling potential radiogenic effects of radioiodine is an increased incidence of childhood thyroid cancer such as that in Belarus, Russia and Ukraine.

Caesium-137, an important contributor to human doses, has a longer half-life of 30 a, so efforts have continued to refine the ^{137}Cs deposition density maps (Figs 2–4). The most recent estimates of the area of land contaminated above certain levels in Belarus, Russia and Ukraine is presented in Table III. The deposition of significant levels of ^{90}Sr and plutonium was generally limited to the close-in areas. The overall patterns of contamination by these long lived radionuclides has remained essentially unchanged over the last ten years, with relatively little secondary transport of material. It should be noted that these environmental levels have been corroborated and will not change even if estimates of the total release change.

The contamination of freshwater systems early on was due to the direct deposition of material from the passing cloud of contaminated air. The radioactive material in freshwater aquatic systems is small compared to the total on land.

Direct effects on the natural environment

The assessment of the radiation exposure experienced by plants and animals presents a major problem, with the possibility of large temporal and spatial variations in the distribution of particularly beta and gamma emitting radionuclides, both within the organisms and in their external environment. However it is clear that the radioactive contamination in 1986 within parts of the 30 km zone typically reached several tens of MBq/m^2 (thousands of Ci/km^2) (Garger et al. 1996) and the external doses to plants and animals from the short lived radionuclides could correspondingly have been on the order of several tens of Gy (gray) in the first month (Likhtarev et al. 1996). During the summer and early autumn of 1986, the dose rate at the soil surface dropped by a factor 10–100 below the initial value. After ten years, caesium radionuclides are the major contribution to the low dose chronic irradiation. Direct radiation injury of plants and animals were reported only in localized areas within the 30 km zone in the first 3 years after the accident (IAEA 1996). Although the chronic dose rates in some of these areas theoretically may be at levels that might affect the fertility of some animal species, owing to animals' mobility their average exposure is lower; moreover, any impacts are difficult to separate from other confounding factors. Any effects on population numbers would be impossible to clearly attribute solely to radiation exposure.

Effects on plants

The accident at Chernobyl took place in the most radiosensitive period for plant communities, i.e. in the early growing season, when accelerated growth and formation of reproductive organs occurs. As was expected based on past research, coniferous forests were extremely sensitive to radiation. Within the first two weeks after the accident 500–600 hectares of trees in the vicinity of the reactor site received a dose of 80–100 Gy[1] resulting in death of pine trees (so-called 'red forest'), and partial destruction of the crowns of birch and alder trees. In the zone nearest to the reactor small sites of coniferous forests were almost completely destroyed. Over a larger area (approximately 3000 ha) where absorbed doses exceeded 8–10 Gy, 25–40% of coniferous forests died; 90–95% of pine trees showed significant damages to their reproductive tissues. However, by 1988–1989 most of the communities in this zone had recovered their reproductive functions. The death of these stands of trees amounted to less than 0.5 % of the forested area of the zone (Kozubov et al. 1991, IAEA 1996).

Morphological abnormalities in some vegetation, such as distorted and swollen bulges in stems, curled leaves, and dwarfish were reported. These same radiation conditions did not lead to any visible changes in growth or reproductive function of most species of herbaceous plants. Practically none of the species of herbaceous plants investigated revealed any significant relationship between the normal biological indicators and absorbed dose (IAEA 1996).

Effects on animals

Reported effects in animals at high initial doses range from severe direct effects to observable, but not necessarily significant, changes to the overall health of populations. In some cases, the populations returned to normal in a few years, and in others, the significance of the consequences are indeterminate. A series of reported examples are given below for illustration purposes.

Cows consuming contaminated pasture close to the reactor in the early phase after the accident received thyroid doses in the range of hundreds of Gy, resulting in atrophy and total necrosis of the thyroid (Alexakhin et al. 1992).

Arthropod (mainly insects and spiders) populations living in the 30 km zone, with increased accumulation of radionuclides in the forest litter and the top soil layer, showed

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[1] For perspective, doses of 10–1000 Gy are typically used to artificially induce mutations in seeds for plant breeding.

marked population decreases during 1986–1987 [Krivolutskii et al. 1990].[2] The current status of these populations are unknown.

Reports have been made of reduction in the reproductive capacity of brown frogs in the contaminated areas near the power plant, where for example, it was stated that in spring 1987 more than 33% of the eggs produced were completely or partially sterile, in comparison to a rate of 1.5% in a control population. This did not significantly improve the next year. However, no direct relationship between radioactive contamination of the area and cytogenetic changes had been determined. It is reported that chromosomal aberration rates in the red bone marrow cells of frogs increased by a factor of 3–10 during 1986–1989. Not unexpectedly, these later changes did not appear to be manifested as direct radiation effects on the population.

After the accident there were media reports on severe birth defects in agricultural animals outside of the 30 km zone in 1988/89. When this was studied further by a Ukrainian expert group, the frequency of these reported defects were similar in the highly contaminated and non-contaminated regions of Ukraine (Prister et al.), which indicates that the occurrence of the birth defects was not likely to have been related to the direct effects of radiation. Confounding factors include, for example, the possible excess use of fertilizers.

Despite considerable contamination, aquatic organisms have been shown to be tolerant of the radiation from the accident. Experiments in the Chernobyl power plant cooling pond have shown that fish continued to produce viable offspring at doses of up to 8 Gy (Sansone and Voitsekhovitch 1996). Benthic organisms, were found to be the most seriously affected. This is not surprising because the highest levels of radioactive contamination in aquatic systems are found in the sediment (in the cooling pond, the dose rate was 0.1–0.2 Gy per day from bottom sediments at the end of April 1986). The size of the mollusc population, Dreissena polymorpha, was observed to be in decline but, at the same time, no effects were noted for the 31 fish species examined. If there were any surprises, it was in the unusually high transfer of radiocaesium from the water into the fish (Sansone and Voitsekhovitch 1996).

Contamination of the environment

The seminatural environment

After the accident, it became clear that implementation of countermeasures in the settlements (urban environment) and agricultural environment were not sufficient in controlling all the main sources of exposure to humans in the affected areas. The seminatural environment was an important contributor to the production of food (e.g. grazing of cattle, collection of wild foods, etc.) and recreational activities in the lives of the population. Additional studies to further understand the transfer mechanisms were conducted in order to apply the best possible countermeasures.

[2] In the laboratory, doses of 50–170 Gy are typically used in artificially sterilizing insects in pest control programmes.

Meadows

By impeding the migration of the radionuclides into meadow grass, the main mechanisms for transfer of radionuclides into the food produced from grazing animals could be controlled. It was found that the movement of the soil contamination is very dependent on the fallout characteristics (physicochemical form, heterogeneity of deposition) and the concentration of the exchangeable radionuclides in the rooting zone (0–10 cm) of the plants. The downward migration of ^{137}Cs and ^{90}Sr in different types of meadows has been slow, leaving most of the contamination available for root uptake in the top 5 cm of soil. During the first 2–3 years after deposition, the migration of the ^{137}Cs was more rapid, but with time, these processes have slowed down by up to a factor of 6 (Belli and Tikhomirov 1996). Since 1991, the amount of ^{137}Cs transferred annually from soil to plant remains fairly constant with time.

Clay and organic content of the soil, and soil moisture are the dominant factors influencing this process (Prister et al. 1996, Fesenko et al. 1996). Migration is faster in high moisture and high organic soil (meadows that flood and have peaty soil). Based on data from the 30 km zone and the three countries, the time it will take for half of the ^{137}Cs to move out of the top 10 cm of soil (and therefore decrease the concentration in the grass) ranges from 55–73 years (19–21 years effective half-time) for sandy and sandy loamy soils to 99–143 years (23–25 years effective half-time) for light loamy and heavy loamy soils in a dry meadow. For peat, it is dramatically faster, typically 15–20 years (10–12 years effective half-life).

The transfer of ^{90}Sr is faster than for ^{137}Cs, with observed effective half-times of 7–12 years. The variation with soil type follows the same trends as with ^{137}Cs (Shutov et al. 1993).

Forests

More than 30000 km^2 of forests received ^{137}Cs deposition of more than 37 kBq/m^2 and about 1000 km^2 received deposition of more than 1.5 MBq/m^2 (Belli and Tikhomirov 1996). Forest canopies were very efficient in the initial interception of radioactive material from the contaminated air as it passed. Due to weathering and shedding of leaves, the contamination dropped to the forest floor after which the roots transported it back to the leaves (Tikhomirov and Shcheglov 1994, Sombre et al. 1994). After 10 years, 90–97% of the contamination is still found in the top 5–10 cm of the forest soil (leaving 3–10% on the bark, wood and leaves/needles of the tree). Of the total activity in trees, only 0.3–2.6% is found in the wood itself and the highest concentrations are found in the most recent annual rings of the trunk. However, since fresh rings grow each year, the mean concentration in the wood is slowly increasing.

Forests are important for the commercial use of the wood, as a source of food (berries, mushrooms, game etc.), and for recreation of the local population. In many local forests doses to forest workers are typically elevated but not hazardous. Temporary permissible levels for the ^{137}Cs concentration in wood have been set at a national level, and in many local forests it may be exceeded unless some changes are made to the production process for lumber. One countermeasure is to remove the most contaminated outer rings of the log before processing. Another is to use the wood for pulp, since the concentration of ^{137}Cs in bleached pulp is only about 1% of the original concentration in the raw wood. Some care has to be taken with the waste products from pulp mills (and also power/heating plants), which have concentrated levels of ^{137}Cs (Ravila and Holm 1996). In many local forests doses to forest workers are typically elevated but not hazardous.

Mushrooms growing wild in forests have traditionally been a staple ingredient in the Belorussian, Russian and Ukrainian diet, and mushroom picking is an important recreational activity. In many areas, the long term consumption of mushrooms and wild berries is a significant exposure pathway due to the relatively high concentrations of ^{137}Cs in these natural foods (VAMP 1996, Kenigsberg et al. 1996) (see Fig. 5). Restricting access to the forest and public advisory campaigns can help to control doses. Another possible countermeasure that has been found is the simple boiling of mushrooms for 5–10 minutes to decrease the ^{137}Cs level by more than 50% (VAMP 1992, Perepelyatnikova et al. 1996).

Animals that graze seminatural pastures, forests or mountain areas (such as sheep, goats, reindeer, moose, roe deer and wild boar[3]) often produce meat above nationally adopted food limits in many areas of Belarus, Ukraine, and Russia, as well as in Nordic countries and the United Kingdom (Aslanoglou et al. 1996, Eriksson et al. 1996). Countermeasures that cause minor impacts to the environment, such as hunting bans, changing slaughtering times, the use of impregnated 'Prussian Blue' salt licks and boli, and relocation of animals to uncontaminated pastures, have been employed effectively in these areas. These types of countermeasures will be needed for long periods of time because the effective half-life for caesium in these ecosystems are relatively long. For example, sheep grazing in the mountains in Norway show an effective half-life for ^{137}Cs in their meat of about 20 years (VAMP 1996).

Another potential risk from forests would be in the event of a fire. Evaluation of this situation has shown that the doses from direct inhalation of resuspended particles would only add a small component to total exposure. Resuspended particles from forest fires could give rise to a transient increase in the ^{137}Cs levels in milk and meat of animals grazing on adjacent lands (Besnus et al., 1996).

The agricultural environment

Although the type of agricultural production varies, it is the predominant use of the land in the contaminated areas. The control of 'contamination' in the commercial production of food was a top priority, so monitoring and agricultural countermeasures were implemented. 'Contamination' is a generic term used by radiation protection specialists and radioecologists to describe any finite levels of artificially produced radioactive material. Action levels of activity in foodstuffs were established by national authorities in the three most affected republics and by international organizations. Above the Action Levels, some countermeasure needed to be considered in order to reduce the concentration (or 'contamination') to below the Action Level. Table IV shows a selection of these action levels for ^{137}Cs in foodstuffs and drinking water. The levels in use in the three affected countries and the EU are in general below those established by the FAO/WHO Codex Alimentarius Commission and those adopted by the EU for use after any future accident.

The local conditions (level of contamination, type of soils, soil moisture) and crop type had a profound effect on the reduction of contamination in food achieved by application of the

[3] For example, ^{137}Cs activity in the meat of wild boar hunted in the district of Bragin in 1994–1995 was 8000–60000 Bq/kg.

countermeasures. For example, depending on the type of soil, the transfer factor between pasture and milk varies by several hundred (Strand et al. 1996).

Immediately after the accident, the priority was to control of the consumption of ^{131}I contaminated foods, largely milk and milk products. At this time there were strict restrictions on pasturing cows and harvesting of grasslands. After June 1986, owing to the short half-life of ^{131}I, the focus shifted to the longer term control of ^{137}Cs in crops to keep the food below permissible ('action') levels.

Since 1987, agricultural countermeasures have fallen into the categories of (Jacob and Meckback 1990, Shutov 1992):

- Organizational: monitoring of land and agricultural products; change in crops; changes in land use.
- Agrotechnical: ploughing.
- Agrochemical: liming of acid soils; application of potassium fertilizers; supplementing soils with natural sorbents.
- Veterinary and zootechnical: changes in diet; changes ins feed before slaughter; removal from pasture; use of caesium binders.
- Food processing: commercial or domestic home processing that removes radioactive contamination.

Figure 6 shows the measured decline of ^{137}Cs in the food grown in the Bryansk region of Russia for milk and dairy products, meat and vegetables, as percentages of the total food that exceeded the national temporary permissible levels from 1986 to 1995. For milk, meat, potatoes and cereals, the countermeasures applied can account for about 60% of the decrease in the ^{137}Cs contamination (Fesenko et al. 1995). The effectiveness and some of the costs for some of the countermeasures applied are shown in Table V.

The aquatic environment

After releases of radioactive material to atmosphere, aquatic ecosystems are usually much less important human exposure pathways than terrestrial systems. During the first week or so direct deposition occurred onto rivers, lakes and seas. Within 1 month the concentrations in the surface waters had fallen dramatically (IAEA 1996), and then the material that had been deposited on land slowly moved into aquatic systems, where it was ultimately concentrated in sediments or migrated down to the sea. One of the most contaminated rivers was the River Pripyat, close to the Chernobyl plant. On 1 May 1986, the ^{131}I and ^{137}Cs concentrations in the river water at Chernobyl were 2100 Bq/L and 250 Bq/L respectively. By 16 July, the levels had fallen to below the detection level for ^{131}I and to 7 Bq/L for ^{137}Cs, and by 1989 the ^{137}Cs concentration in water was about 0.4 Bq/L (Vakulovsky et al. 1994, Voitsekhovich et al. 1990). The early countermeasures to protect water supplies have been criticized as diverting significant national resources with extremely limited results (Voitsekhovitch et al. 1996)

Similar patterns have been observed in other rivers. The effective half-life of ^{90}Sr (about 6 years) in the River Pripyat fell more slowly than for ^{137}Cs owing to the inflow of radioactive materials from contaminated land in the watershed area (peat bogs at times of flooding). The sediment of the Rivers Dnieper and Pripyat sediments contain caesium, strontium, plutonium and americium (90% of the activity is due to ^{137}Cs), and after storms and floods ^{137}Cs and ^{90}Sr

are detected in elevated amounts owing to the resuspension and transport of solids. Although fluctuations may be seen in the future, the lifetime effective dose to a person using the River Pripyat directly for drinking water is estimated at 0.4 mSv (Vakulovsky), which poses no hazard.

Of particular public concern has been the drinking water reservoir for Kiev city. The ^{137}Cs concentration in the reservoirs during 1989–1990 was only about 0.004–0.04 Bq/L. Levels of ^{90}Sr was about an order of magnitude higher. These levels are well below any safety criteria, even for normal non-accident conditions (see Table IV). Although the dose is trivial, the public perception of a significant hazard remains (Sansome and Voisekhovitch 1996).

Even though the risk from drinking water from contaminated surface water bodies is very small, the ^{137}Cs concentrations in fish from some rivers and lakes (e.g. Koyanovskoe Lake, (Ryabov et al. 1996)) can exceed action levels owing to bioaccumulation, and hence countermeasures, such as fishing restrictions, may be needed. This is particularly important in low nutrient lakes common in northern Scandinavia, where conditions favour extremely high uptake of radiocaesium into fish. The observed ratio of ^{137}Cs concentration in freshwater fish and in the water of these lakes was on the order of 10000 in 1991–1993 (Saxen 1994).

The surface water concentration of ^{137}Cs in the Baltic Sea in 1990 gave an activity of about 0.1 Bq/L (Carlson and Snoeijs 1994) and fish contained about 15 Bq/kg (Aarkrog et al. 1995) These levels are falling very slowly with time because of the long mean residence time (20–30 years) of water in the Baltic Sea (NRPB/CEA 1979). In the Black Sea, concentrations in 1990 were even lower, and the mean residence time of water is estimated to be 19 years (CEC 1994). At no time has there been or is there expected to be any hazard from contamination in the marine environment.

The human environment

The principal aim of any decontamination of residential and industrial areas is cost effectively to reduce the overall external dose of inhabitants or workers, taking into account any negative effects on the environment. A significant part of the external dose received by people within and around their dwellings and places of work, is from activity in the soil or on other ground surfaces. As mentioned earlier, five years after the accident, about 90% of the caesium radionuclides was in the top 5–10 cm of undisturbed soil. Additionally, since the dose rate in air is derived from activity in the ground from up to a few tens of metres away, decontamination must be extended to 10–30 metres around the plot in order to significantly reduce exposures..

Large scale decontamination was performed in 1986–1989 on about one thousand settlements, tens of thousands of residential and social buildings, and more than one thousand farms. This operation was performed primarily by military personnel and included washing buildings with water or special cleaning solutions, cleaning residential areas, removal of contaminated soil, cleaning and washing of roads and decontamination of open water supplies (Balonov et al. 1991). Depending on the decontamination technique, the dose rate over different plots was reduced by a factor of 1.5–15. However, the effectiveness in terms of total external dose reduction was typically up to only 30% reduction. Observations of decontaminated plots after five years showed that there has been no significant recontamination (Balonov et al. 1991).

Decontamination was focused on removing the upper soil layer from populated areas, which is expensive. The volume of contaminated soil removed during decontamination of one village (population of several hundred persons), for example, was approximately 1000 cubic metres. The total amount of settlement decontamination waste exceeded one million cubic metres. These volumes were usually transported to remote areas and buried in special storage far from settlements (Balonov et al. 1991). It is notable that more cost effective approaches to decontamination, such as ploughing of soil to 30 cm or the use of so-called 'skim and burial' or 'triple digging' ploughs which bury the upper soil layer at a depth of 30–40 cm in situ were not used extensively, partly owing to lack of equipment. Studies performed in 1994–1995 have demonstrated that these methods produce practically the same dose reduction as topsoil removal with far less costs and no large scale waste disposal problems (Roed 1996).

Large scale decontamination, if carried out in a cost effective manner, can realistically achieve a reduction of 30% in annual external dose. Simple 'triple digging' techniques have not been carried out on a large scale but show potential for further dose reductions. Even a 30% reduction in annual dose is equivalent to waiting about ten years for natural weathering and decay processes to reduce the annual dose to the same level.

Resulting human exposure

Estimates of human exposure have been presented in terms of collective doses in Background Papers 2 and 3 for the purposes of discussing human health impacts. This section discusses the important pathways and nuclides contributing to human doses, how they change with time, and their relevance to countermeasures policy. Since the policy decisions made several years ago depended on the dose assessments carried out at that time, it is instructive also to review the influence that improvements in assessment methodology could have had on policy.

Several assessments have been made of the effective dose to the human population in the first year (UNSCEAR 1988, Zvonova and Balonov 1993, Golikov et al. 1993, Shuthov et al. 1993) and while they vary in the details according to the availability of local information, they all concur that the contribution to the effective dose from inhalation and external gamma radiation from material in the initial plume were small. Moreover, they agree that the most important pathways of exposure to humans were the ingestion of contamination in milk and other foods (^{131}I, ^{134}Cs and ^{137}Cs), and external exposure from radioactive deposits (^{131}I, ^{103,106}Ru, and ^{134,137}Cs).

In the first few months, because of the significant release of the short lived ^{131}I, the thyroid was the most exposed organ. The dominant route of exposure for thyroid dose was the pasture–cow–milk pathway, with a secondary component from inhalation (Likhtarev et al. 1993, 1994a). The contribution from inhalation of short lived radioiodines other than ^{131}I is deemed to be small, especially for young children, for whom the milk pathway is extremely important (Zvonova and Balonov 1993). The external dose pathway also makes only a small contribution to the thyroid dose. An accurate determination of individual human thyroid dose is highly dependent on whether or not countermeasures such as stable iodine prophylactics and/or food bans were effectively employed.

In the next nine years, the principal pathways by which humans were exposed were ingestion of ^{134}Cs and ^{137}Cs in food (especially milk), and the external dose (UNSCEAR

1988). Inhalation of resuspended material is negligible for the general population (Jacob et al. 1996). The doses from ingestion of food depend on consumption habits, and on the contamination levels in local foods (which, as discussed above, varies significantly according to soil type, as well as with time and according to the agricultural countermeasures that were applied). This has led to considerable variations in the transfer factors from ground contamination to internal dose between different locations.

Levels of contamination in milk and meat, cereal and potatoes have fallen by an order of magnitude between 1987 and 1991–1992. However, the caesium content of many species of natural food products, such as berries and mushrooms, have not shown any significant reduction since 1986 other than physical radioactive decay (Balonov et al. 1996). This component of the diet has a significant effect on the internal dose (see Fig. 7). The external doses for a given level of soil contamination are much less variable between settlements, although there can be a significant variation between individuals' exposures within a settlement (up to a factor of three from the average) (Likhtarev 1996, Erkin and Lebedev 1993).

The revised dose calculations used about 1 million whole body measurements (Balonov et al. 1996) which were carried out in more than 300 higher contamination settlements of Ukraine, Belarus and Russia (up to 1994). In the lower contamination areas, measurements of radionuclide concentrations in foodstuffs were applied in improving the dose assessment methodology.

An estimate of the breakdown of the total effective doses per unit ground contamination according to the external and internal components in urban and rural populations are presented by time period in Table VI. The top set of model estimates for internal dose do not take into account the countermeasures (which would have reduced the doses) or the consumption of wild food (which would have increased the doses). The bottom estimates are based on whole body measurement taken in the affected areas. It should be noted about 60% of the total external doses and over 90% of the total internal doses have already been received by now.

A more realistic prognosis of the doses a rural adult population would receive living in an area that was 0.6 MBq/m^2 (15 Ci/km^2) of ^{137}Cs in 1986 is given in Table VII. This takes into account both external and internal exposure from ingestion of ^{137}Cs and ^{90}Sr, inhalation of Pu radionuclides and ^{241}Am with relatively high and permanent resuspension of soil. It is assumed that agricultural countermeasures and delivery of non-contaminated food will be discontinued in 1996. Because of the low content of ^{90}Sr in the Chernobyl release and fallout outside of the 30 km zone, its contribution to the internal lifetime dose is less than 5–10% (Shuthov et al. 1993). Natural food products contribute significantly to internal dose, but inhalation of transuranium radionuclides is not important, even for agricultural workers who breathe resuspended particles. (Inhalation of resuspended $^{238,239,240}Pu$ contributes less than 1% to the internal dose even for outdoor workers (Ivanova et al. 1995).)

It is notable that the present estimates for the average external doses are significantly lower than both the official Soviet estimates (129–156 mSv per MBq/m^2) and the ICP estimates (105 mSv per MBq/m^2) of 1990 (ICP 1991).

The estimated total collective effective doses (reported in Background Paper 3) for 1986–1995 for persons living in areas with deposition density of > 15 Ci/km^2 (> 555 kBq/m^2) and 1–

15 Ci/km^2 (37–555 kBq/m^2) are 10000–20000 man.Sv and 20000–60000 man.Sv, respectively (populations are 270000 and 3.7 million respectively).

The global effective collective dose commitment resulting from the Chernobyl accident has been estimated to be about 600000 man.Sv. This dose is 2% of the total long term global collective effective dose received from all the nuclear weapons testing carried out in the atmosphere (30 million man.Sv) and 0.5% of the annual collective dose from natural background (120 million man.Sv/year (based on 2.4 mSv annual effective dose for 5 billion people)) (UNSCEAR 1988, 1993). If only the short term global collective effective dose from weapons tests is considered (without the long term dose of 5 million man.Sv from ^{14}C), the percentage increases to about 12%.

Radiation protection policy

In 1990 the ICP concluded that the foodstuff restrictions should probably have been less restrictive on radiological grounds, but recognized that there were many social and political factors to take into consideration – this is clear from the figures in Table IV. Several authors (Crick 1991, Hedemann Jensen 1995) have shown that the area of land and the length of time that an area of land requires agricultural countermeasures or other restrictions is quite sensitive to the chosen action level. The collective dose saved by a countermeasure on the other hand is much less sensitive (Fig. 8). Thus pessimism rapidly results in unnecessarily high costs, particularly concerning long lived radionuclides. For example, it is clear that the overestimation of the doses by a factor of two (due to conservative assumptions) leads to the extension of countermeasures by a period equal to the effective half-life (typically 10–20 years for ^{137}Cs).

4. FUTURE PROSPECTS

Looking to the future, the presence of long lived radionuclides ^{137}Cs, ^{90}Sr, plutonium radionuclides and ^{241}Am in the soil will continue to contribute to dose to humans. Table II shows the radioactive releases, activity levels prevailing today and projected activity in 60 years for key radionuclides. Caesium-137 is and will continue to be the dominant contributor to both external and internal exposure in the next several decades. Strontium-90 will continue to cycle in the food chain, and plutonium and americium will contribute mainly through inhalation and to a less extent by ingestion pathways. In general, until today the population in the most affected areas has received up to 60–80% of their total lifetime doses from the accident (IAEA 1996, Balonov et al. 1996).

The current annual exposure of people in the areas with ^{137}Cs soil contamination of 0.6 MBq/m^2 and above is in the range of 1–4 mSv, and is predicted to fall to an average annual dose of about 0.5–1 mSv 60 years in the future (Balonov 1992). These future doses may not warrant the continuation of active intervention, except in the more extreme case of consumption of local natural foods. This supports the real need to continue to evaluate the benefit of countermeasures as time passes.

From the studies of global fallout from the atmospheric weapons tests we can expect a long term reduction of the ^{137}Cs and ^{90}Sr content in agricultural food products at a rate of 3–7% per year depending on soil properties (UNSCEAR 1982). This will most likely not be the case with food derived from natural and seminatural environments, where loss of activity is expected to be 2 or 3 times slower (Shutov and Bruk, G., in press). It is expected that the intake of ^{137}Cs by inhabitants of contaminated areas not consuming local wild food will decrease on the average at 3–7% per year compared with the annual intake in 1993–1995, so it is expected that the committed internal dose for the time period of 1996–2056 will be a factor of 15–30 times the annual dose in 1996.

Subsequent decrease of the ^{137}Cs intake by consumers of local wild food is expected with the rate of 2–4% per year; however, in areas where food restrictions were strictly followed in the past, there may be increase in doses due to the relaxation of adherence to the rules or economic difficulties (Strand et al. 1996). The slow migration of ^{137}Cs into soil and physical decay will result in a decline in the external exposures at the rate of 3–6 % per year (Balonov et al. 1996).

A prognosis of the doses a rural adult population would receive living in an area that was 0.6 MBq/m^2 (15 Ci/km^2) of ^{137}Cs in 1986 is given in Table VII. The prognosis takes into account both external and internal exposure from ingestion of ^{137}Cs and ^{90}Sr inhalation of Pu radionuclides and ^{241}Am with relatively high and permanent resuspension of soil. It was assumed that agricultural countermeasures and delivery of 'non-contaminated' food would be discontinued in 1996. Natural food products contribute significantly to internal dose but inhalation of transuranium radionuclides is not important, even for agricultural workers who breathe resuspended particles. In this population the external dose remains dominant for the average person but for a higher exposed individual, the internal and external doses are about equal. The hypothetical doses presented in Table VI showed lower internal doses because it was assumed that wild foods were not consumed, and the highest transfer factor soil (peat) is not represented.

To generalize, in the projections for future doses over the long term (1996–2056), the collective dose to the most impacted population is expected to increase by a further approximately 50% of the dose received over the last 9 years (1986–1995); or in other words 66% of the total dose has already been received.

The prognosis for the 30 km zone remains to be seen. Generally, the natural environment seems to be recovering from the initial high exposures that caused some damage immediately after the accident. There is no evidence that the high chronic exposures to radiation in some specific localities are altering the local flora and fauna populations and they do not seem to have inhibited survival. If anything, the absence of people from the area has probably had more significant effects, such as the increased growth of some populations (IAEA 1996). Before rehabilitation of the area can be considered, more must be understood about the physical and chemical forms of the fuel particles as they disintegrate and migrate into the environment. The potential problems of the buried waste at the Chernobyl site must also be considered in terms of local groundwater contamination.

The uncertainty regarding the future of the destroyed reactor and the sarcophagus leaves open questions regarding the possibility and magnitude of future environmental releases. These issues are currently being discussed in international forums.

5. ISSUES ATTRACTING PARTICULAR PUBLIC INTEREST

Immediately after the accident and for the subsequent years, some key concerns about the contamination in the environment have been voiced in the media or by the public. The following issues were identified as having evoked a high level of public interest and are addressed below.

It has been claimed that aquatic ecosystems such as the Black Sea and the Kiev Reservoir could be significant hazard to humans. Generally speaking, aquatic ecosystems contribute typically 10–100 times less to the dose from releases to the atmosphere (such as in the Chernobyl accident) than terrestrial ecosystems. Furthermore, the actual doses from consumption of fish from the Black Sea (CEC 1994, CEC 1990) and from drinking water from the Kiev reservoir are well below even normal permissible levels and continue to fall (Voytsekhovich et al. 1990).

Radioactive contamination of agricultural lands result in birth defects among animals. In mass media sources in 1988–1989, it was reported that there was a marked increase in birth defects (abnormal number of legs, anomaly of different organs, etc.) among newborn animals. Investigation revealed that the number of such anomalies in agricultural animals in non-contaminated areas does not differ statistically from analogous values for contaminated areas affected by the Chernobyl accident (Prister et al.).

Importance of strontium and plutonium deposition. Due to the low content of ^{90}Sr in the Chernobyl release and fallout outside the 30 km zone its contribution to the internal lifetime effective dose does not exceed 5–10%, according to intake calculation and direct measurements of ^{90}Sr in human bones (autopsy samples). Similarly the contribution from the inhalation of $^{238,239,240}Pu$ does not exceed 1% even for outdoor workers (Ivanova et al. 1995).

Importance of resuspension. Resuspension of radioactive materials from the most contaminated zone does not lead to their significant transfer downwind. After 10 years, a conservative maximum of about 6% increase of contamination at a distance of 100 m away from the contaminated area is seen. This excess declines to 2% at a distance of 10 km. (Holländer and Garger 1996).

Importance of 'hot particles'. The key concerns regarding hot particles were the potential increased health impacts in exposed populations. To address this an IAEA Co-ordinated Research project to investigate the radiobiological impact of hot beta particles from the Chernobyl accident was organized. In the report from the second meeting, the majority of the participants "agreed that no evidence was presented that supported the concept that hot particles represented a greater risk than the same quantity of radioactivity uniformly distributed in the exposed tissue, whether it be lungs or skin" (IAEA 1994). The long term environmental transfer of the radionuclides in the hot particles is still being investigated.

6. CONCLUSIONS

The initial release to the environment

Broad agreement has been reached among various estimates concerning the initial release (source term) due to the accident. The majority of the release was of radionuclides with relatively short half-lives. Releases to the environment of some radiologically important radionuclides ^{131}I, ^{134}Cs, and ^{137}Cs are estimated now at a factor of 2–3 higher than in 1986 and namely 1.2–1.8 EBq, 46 PBq and 85 PBq, respectively. However, the reassessment of the source term has had no impact on the assessment of individual doses, which were based on the environmental or whole body measurements made in the affected areas.

The total amount of radioactive material still present in the environment after ten years has decayed to about 80 PBq of long lived radionuclides, principally ^{137}Cs and ^{90}Sr, or about 1% of the total amount released. The overall patterns of contamination by these long lived radionuclides has remained essentially unchanged over the last ten years, with relatively little secondary transport of material.

Direct effects on plants and animals

The highest doses to plants and animals in the environment occurred immediately after the accident in parts of the 30 km zone. Contamination levels typically reached several tens of MBq/m^2 (thousands of Ci/km^2) in some localities and external doses would correspondingly have been of the order of several tens of Gy in the first month from the short lived radionuclides. In summer and early autumn of 1986, the dose rate at the soil surface dropped by a factor 10–100 of the initial value.

Direct radiation injury of plants and animals was reported only in localized areas within the 30 km exclusion zone and within the first one to three years after the accident. Different organisms in the natural environment were exposed to high doses of irradiation, and the lethal doses for some radiosensitive ecosystems were reached. These lethal effects were seen in the coniferous forests in the nearest area around the plant and for some small mammals in this zone (5–30 km). For other ecosystems and individual plants and animals, no lethal effects were observed (including 30 km zone). By 1988–1989, in the 3000 ha around the plant, damaged conifers had recovered their reproductive functions, and are likely to recover fully. Herbaceous plants were little affected anywhere, but some abnormalities in other vegetation were found.

Severe direct effects of radiation at high doses were observed in some animals, but were not necessarily significant in changing the overall health of populations. For example, cows consuming contaminated pasture close to the reactor in the early phase after the accident received thyroid doses in the range of hundreds of Gy, resulting in atrophy and total necrosis of the thyroid of individual cows.

In some cases, the impacted populations returned to normal in a few years, and in others, the significance of the consequences are still unknown. Chronic dose rates in some areas within the 30 km exclusion zone may have reduced the fertility of animals of some species but it appears that other affected animal populations have already recovered. The overall significance of the observed changes in specific populations is difficult to determine.

There were media reports of severe birth defects in agricultural animals outside the 30 km zone in 1988–1989; however, the frequency of these reported defects was shown to be similar in highly contaminated and non-contaminated regions of Ukraine. No more severe effects observed in farm animals have been reported.

Long term effects on the natural populations would be as a result of damages to the reproductive processes and hereditary damage. There has been some evidence that supports recovery, however there is no general consensus at this time and it is difficult to come to a conclusion on the long term hereditary or ecological impacts. After ten years, caesium radionuclides are the major contribution to the low dose chronic irradiation; external doses in some isolated spots can still be of the order of 1 mGy per day; however, even in the 30 km zone, the natural environment seems to be recovering. Owing to the relocation of people from the 30 km zone, there have been some changes in the numbers and variety of animal and plant communities, but these changes have resulted disuse of the land, not from radiation effects. Some natural populations have thrived as a result of the cessation of human interference.

Contamination of the environment

The seminatural environment

This category of the environment, between natural environments and agricultural land that is managed, may have a dominant influence on the future doses to the human population.

Key factors controlling the migration of radionuclides from topsoil into plants in meadow ecosystems are the clay and organic content of the soil and soil moisture. In general, the current migration rate is slow and steady. This is expected to continue over the coming decades, even as the level of radioactive material in the soil declines. The transfer of ^{90}Sr, with observed effective half-times of 7–12 years, is higher than those obtained for ^{137}Cs, but the influence of different soil types is similar. This rate of transfer will be a central consideration in the long term use of meadows for the pasturing of cows.

Today, nearly all the contamination in forest ecosystems is found in the topsoil. The radiocaesium in trees is concentrated in the new growth rings owing to the soil–root transfer pathway. Forest fires have been identified as potential sources of the resuspension and redistribution of the ^{137}Cs currently remaining in the forests. Exposures to workers from heavily contaminated wood and wood processing waste will need to be monitored. No cost effective countermeasures that would reduce the transfer of contamination into the trees have been found.

Food products from animals that graze in seminatural pastures, forests or mountain areas and wild foods (game, berries, mushrooms) gathered by the population will continue to show high ^{137}Cs levels over the next decades and are likely to be an important source of future internal doses. These foods may still be contaminated above the strict nationally adopted limits in areas of Belarus, Ukraine and Russia; and also in Nordic countries and the United Kingdom. The effective half-time of ^{137}Cs in these areas is considerably longer than in agricultural areas. Sheep grazing on mountains in Norway, for example, show an effective half-time for ^{137}Cs in their meat of about 20 years.

The agricultural environment

In 1986, the concentration of some radionuclides (^{131}I, ^{137}Cs) in foods exceeded Action Levels adopted by international organizations (WHO, etc.). It was found that effective application of agricultural countermeasures could result in a significant reduction in the uptake of Cs and Sr into food. The reduction of contamination in food achieved by the countermeasures depended strongly on local conditions (level of contamination, type of soil, soil moisture) and crop type. For example, depending on the type of soil, the transfer factor between pasture and milk varies by several hundred. The proper application of these actions were very site specific.

Safe and relatively simple and cheap measures include, for example: deep ploughing of surface contaminated soils; addition of fertilizers or other chemicals to agricultural lands; change in crop type; changing feeding regimes and slaughtering times of cattle; the use of impregnated 'Prussian Blue' salt licks and boli to reduce the transfer of caesium to cattle; and relocation of animals to uncontaminated pastures.

Monitoring and countermeasures will need to be continued for agricultural land since food products are a main contributor of the dose to humans. However, much has been learned about the cost effectiveness of different countermeasures. It is expected that, with continued implementation, only very small amounts of food products with ^{137}Cs levels above the national temporary permissible levels will be found in the future.

The aquatic environment

Aquatic ecosystems have been shown to be tolerant of radioactive contamination, which gradually concentrates in sediments. Even in the Chernobyl plant's cooling pond, populations of only certain organisms were affected and no long term direct effects of radiation have been documented.

The radioactive material in freshwater aquatic systems is small compared to the total deposited. The activity in surface waters had fallen dramatically within one month of the accident. The public's perception notwithstanding, contamination levels in reservoirs are well below the criteria that would indicate a long term degradation in water quality. A lifetime effective dose due to using the River Pripyat directly for drinking water is estimated at 0.4 mSv. Fish, however, may accumulate radionuclides and countermeasures may be necessary even in countries outside of the areas of highest contamination, such as Sweden.

Most of the radioactive material released was deposited over land; however, the marine environment may have received as much as 10–20 PBq of the total ^{137}Cs. The highest concentrations were seen in the Baltic Sea and the Black Seas, which received about 3–5 PBq and 2–3 PBq of ^{137}Cs, respectively. However, the marine environment has never presented any contamination hazard.

Human exposure

Overall, the effective dose to the human population in the first year from inhalation and external gamma radiation from the initial plume was small. In subsequent years, the principal exposure pathways were external exposure from the deposited material and ingestion of radiocaesium in food (especially milk and potatoes). The transfer of contamination to different food products has varied by factors of several hundred because it is highly dependent on local soil type, time after the accident and the countermeasures applied in the specific area. The local ingestion doses have been affected accordingly. The dose assessment methodologies previously used have been improved, resulting in the downward revision of doses from the Soviet and the ICP estimates reported in 1990.

The majority of the total external doses and nearly all of the internal doses to be expected from the accident have already been received. Residual average annual doses due to the accident in the three most affected countries are now comparable with the variation in doses due to the natural background of radiation. Certain population groups are more at risk, such as young children.

There is evidence that doses in some areas from internal exposure have increased in recent years, thought to be due to the relaxation of restriction and increased consumption of mushrooms and natural foods. There are a number of different estimates (many cases for specific regions) of the doses expected to be accrued during the next 60 years. In most cases the internal and external doses are expected to decrease, but the degree of reduction depends on the level of contamination and on the effectiveness of the countermeasures applied in each area. It seems probable that during the next 50 years, foods from the forest ecosystem, particularly mushrooms, will remain a main source of internal dose and it will be necessary to continue to discourage consumption.

The global effect of the accident has been to have added very slightly to the natural background of radiation, with a collective dose of about 2% of that contributed by atmospheric weapons testing. Significant continuing consequences outside the three most affected countries are limited to high concentrations of caesium radionuclides in grazing animals in some habitats and in fish in certain types of lakes, which can significantly contribute to the dose of the populations that depend on those food sources.

Decontaminated urban areas

No significant recontamination of decontaminated urban areas has been observed. The effectiveness in terms of the reduction in total external dose was typically up to only 30%. Nevertheless, a 30% reduction in annual dose is equivalent to the effect of about ten years of natural weathering and decay processes. Other decontamination methods, such as simple 'triple digging' techniques, might be worthwhile in reducing doses in marginal territories to allow them to meet national criteria for rehabilitation.

Decontamination procedures do cause the need for adequate disposal of the waste that is generated. The total amount of waste that was created from decontamination procedures after the Chernobyl accident exceeded 1 million cubic metres. This does not present an external exposure problem but migration from the numerous small waste disposal sites could potentially cause a secondary problem of groundwater contamination. The removal of large volumes of soil from decontaminated settlements was probably not the most cost effective means of large scale decontamination.

Radiation protection policy

The results over the last 10 years have shown that the correct implementation of countermeasures can significantly help to control the doses to the human population. However, as the ICP concluded in 1990 (ICP 1991), the foodstuff restrictions should probably have been less restrictive on radiological grounds alone. The area of land and time for which agricultural countermeasures or other restrictions are required is quite sensitive to the choice of a national action level. The collective dose saved by a countermeasure, however, is more independent of this choice. Thus very restrictive levels can lead to unnecessarily high costs, particularly in relation to long lived radionuclides. For example, overestimation of doses by a factor of two leads to the extension of countermeasures by an effective half-life (typically 10–20 years for ^{137}Cs).

The loss of use of land is expected to be gradually reversed over years or decades. Based on the current dose assessments and the evaluation of successful implementation of countermeasures, the feasibility of rehabilitating evacuated areas in the future is being discussed. Resettlement of some evacuated lands could begin already, maintaining localized restrictions, but other active intervention measures may not be necessary in the future. Resettlement may be impeded by the fears and lack of trust of the population. One of the main lessons learned after the Chernobyl accident is the long lasting societal impact of the initial actions taken after an accident and the enforcement of overly strict radiation protection policies in the longer term, however, this is a topic for another session in this conference.

Future work

It can be concluded that: (1) at high radiation levels the natural environment have shown short term impacts but existence of significant long term impacts remains to be seen; (2) effective countermeasures can be taken to reduce the transfer of contamination from the environment to the human population but these are highly site specific and must be evaluated in terms of practicality as well as population dose reduction; and (3) the majority of the doses have already been received by the human population. If agricultural countermeasures are appropriately implemented, the main source of future doses will be due to the gathering of food and recreational activities in natural and seminatural ecosystems.

To improve the ability to assess the present and prospective doses to the population and apply the best radiation protection countermeasures, these issues should be addressed:

- Further study and monitoring of the migration into groundwater around the numerous small radioactive waste burial sites would protect drinking water supplies in the future and help allay the fears of the public.
- Why are dose estimates for internal doses from radiocaesium based either on diet measurements or on whole body counting often in disagreement? In general, the diet estimates have given doses which are 2–10 times higher that those obtained from whole body estimates.
- Would the processing of wild foods control contamination levels sufficiently to allow for the removal of restrictions
- Can natural and seminatural ecosystems be completely rehabilitated during the next generation? Are there countermeasures that may help facilitate the long term return of forests to lower contamination levels?
- Continued work on the cost effectiveness of countermeasures and long term intervention is needed to provide the necessary information for decision makers.

Although the answers to many questions have been found over the last ten years, further research issues that remain to be resolved that will allow for better preparation for any future accident are:

- Lack of movement of radionuclides from the top surface of the soil warrants further study.
- What will be the behaviour of 'hot particles' in the environment in the future as they continue to degrade?
- What is the role of fungi in the dynamics of radionuclides in forest soil, particularly for soil–plant transfer? Would this aid in development of better models for forest ecosystems?

- Our knowledge on the processes of migration of long lived radionuclides such as ^{90}Sr, ^{137}Cs, ^{239}Pu via food chains on a long term basis (for periods of some tens of years and a more prolonged time) is incomplete.

The loss of amenity of the land in the affected regions has been difficult to evaluate owing to the changing economic and social conditions in Belarus, Ukraine and Russia over the last 10 years. The work to evaluate the impacts to society from the commercial economic losses and change in quality of life due to the loss of recreational use of the natural environment should continue.

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TABLE I. ESTIMATES OF ACTIVITIES OF THE PRINCIPAL RADIONUCLIDES RELEASED IN THE CHERNOBYL ACCIDENT

Element group	Radionuclide	Activity of release (PBq)[a]	Activity of release (PBq)[b]
Inert gases	Kr-85		33
	Xe-133	6500	6500
Volatile elements	Te-129m		240
	Te-132	~1150	1000
	I-131	~1760	1200–1700
	I-133		2500
	Cs-134	~54	44–48
	Cs-136		36
	Cs-137	~85	74–85
Intermediate	Sr-89	~115	81
	Sr-90	~10	8
	Ru-103	>168	170
	Ru-106	>73	30
	Ba-140	~240	170
Refractory (including fuel particles)	Zr-95	196	170
	Mo-99	>168	210
	Ce-141	196	200
	Ce-144	~116	140
	Np-239	945	1700
	Pu-238	0.035	0.03
	Pu-239	0.03	0.03
	Pu-240	0.042	0.044
	Pu-241	~6	5.9
	Pu-242		0.00009
	Cm-242	~0.9	0.93
Total (excluding noble gases)		5300	8000[c]

[a] Estimate of total release during the course of the accident (OECD/NEA 96 based on Devell et al. 95)
[b] Estimate of release decay corrected back to 26 April 1986 based on Borovoy 92, Buzulukov, Dobrynin 93.
[c] The total release reported by the USSR (IAEA 1986) was 1.9 EBq (excluding noble gases) decay corrected to 6 May 1986. The estimates here are decay corrected back to 26 April 1986. Thus the present release estimate appears considerably higher, about 8 EBq, because it includes more short lived radionuclides. However, it should be considered a probable overestimate for the total release since many of these radionuclides would have decayed inside the damaged core before any release to the atmosphere.

TABLE II. RESIDUAL RADIOACTIVE MATERIAL IN THE GLOBAL ENVIRONMENT DUE TO THE CHERNOBYL ACCIDENT

Significant radionuclide	Released in 1986 (PBq[a])	Remaining in 1996 (PBq)	Remaining in 2056 (PBq)
I-131	1200–1700	0	0
Sr-90	8	6	1.5
Cs-134	44–48	1.6	0
Cs-137	74–85	68	17
Pu-238	0.03	0.03	0.02
Pu-239	0.03	0.03	0.03
Pu-240	0.044	0.044	0.03
Pu-241	5.9	3.6	0.2
Am-241[b]	0.005	0.08	0.2

[a] Estimate of release decay corrected back to 26 April 1986 based on (Borovoy 92, Buzulukov, Dobrynin 93).

[b] The activity of ^{241}Am in 1996 has increased since 1986 as it is a daughter product of ^{241}Pu (half-life 14 a). This increase has to be considered in any radiological prognosis; however, the doses from ^{241}Am will not exceed the present doses from other radionuclides.

TABLE III. AREA OF 137Cs CONTAMINATION OF THE TERRITORIES OF BELARUS, RUSSIA AND UKRAINE (THOUSAND km^2) (IAEA 1996)

Country	Surface deposition			
	1–5 Ci/km^2 37–185 kBq/m^2	5–15 Ci/km^2 185–555 kBq/m^2	15–40 Ci/km^2 555–1480 kBq/m^2	> 40 Ci/km^2 > 1480 kBq/m^2
	Control zone (x 000 km^2)	Voluntary evacuation zone (x 000 km^2)	Obligatory evacuation zones (x 000 km^2)	
Belarus	29.92	10.17	4.21	2.15
Russia	48.8	5.72	2.1	0.31
Ukraine	37.21	3.18	0.88	0.57
Total	115.93	19.07	7.19	3.03

TABLE IV. SELECTED ACTION LEVELS FOR ^{137}Cs IN DRINKING WATER AND FOOD PRODUCTS (Bq/L or Bq/kg)

Product	TPL 1993[a]	GALs FAO/WHO[b]	EC 1986[c]	MPLs (EC)[d]
Drinking water	20	1000[e]		1000
Milk	370[f]	1000	370	1000
Meat, meat products	600[g]	1000	600	1250
Grain, flour, cereals	370	1000	600	1250
Baby food	185	1000	370	400
Mushrooms	600	1000	600	–

[a] Temporary permissible levels adopted in Belarus, Ukraine and Russia in 1993 (with some exceptions) (RISO-94)

[b] Generic Action Levels for foodstuff moving in international trade (BSS 1995).

[c] EC adopted levels of May 1986, extended several times since (Lukyx 1990).

[d] EC Maximum permitted levels for future accidents (Lukyx 1990).

[e] WHO Guideline for drinking water quality (WHO 1993).

[f] In Belarus, 185 Bq/L is used.

[g] In Gomel region of Belarus, 370 Bq/kg is used.

TABLE V. EFFECTIVENESS OF DIFFERENT TYPES OF COUNTERMEASURES, AND COSTS WHERE AVAILABLE (REF FROM ALEXAHKIN)

Agrochemical countermeasures	Soil category	Crop type	Reduction in root uptake transfer	Total costs 1994 US$/ha
Liming	Soddy-podzolic sandy, sandy loam	Barley, winter rye, oats, maize silage, potato, beet roots, vegetable	1.8–2.3	10.2–55.8
Increased application of P-K fertilizers	Soddy-podzolic sandy, sandy loam	Barley, winter rye, oats, maize silage, potato, beet roots, vegetable	1.2–2.2	
Application of organic fertilizers	Soddy-podzolic sandy, sandy loam	Barley, winter rye, oats, maize silage, potato, beet roots, vegetables	1.3–1.6	44.4–54.9
Application of clay minerals	Soddy-podzolic sandy, sandy loam	Barley, winter rye, oats, maize silage, potato, beet roots, vegetables	Dubious effect, in light soils results mainly in 1.5–3.0 fold decrease of radionuclide accumulation in plants	
Combined application of lime, organic and mineral fertilizers	Soddy-podzolic sandy, sandy loam	Barley, winter rye, oats, maize silage, potato, beet roots, vegetable	2.5–3.5	84.6–111
Treatment type	**Product**	**Dosage**	**Reduction factor**	
Ferrocine	Milk	3–6 g per day	4– 8	
	Meat (cows)	3–6 g per day	3–5	
	Meat (sheep)	0.5–1 g per day	3–7	
Ferrocine boli	Milk	2–3 boli per 3	3–5	
Bifezh ferrocin	Milk	40 g per day	3–4	
Type of processing	**Product**		**Reduction factor (Bq/kg)/(Bq/kg)**	
Milk to butter	Milk		0.05–0.4	
Milk to cheese			0.5–2.5	
Milk to cream			0.6–2.2	
Milk to skimmed milk			0.9–1.0	
Soaking	Meat		0.5–0.7	
Salting			0.3–0.5	

TABLE VI. RECONSTRUCTION AND PROGNOSIS OF THE AVERAGE EFFECTIVE DOSE DUE TO EXTERNAL AND INTERNAL EXPOSURE OF THE ADULT POPULATION IN THE INTERMEDIATE TO FAR ZONE (100 km < D < 1000 km) DUE TO CONTAMINATION AS A RESULT OF THE CHERNOBYL ACCIDENT

Exposure	Population group	Soil type	Effective dose per unit deposition of ^{137}Cs mSv per MBq/m^2			
			1st year	1986–1995	1996–2056	1986–2056
External	Urban		8	23	17	40
	Rural		13	36	28	64
Internal[a]	Rural[b]	Turf-podzol	90/40	170/90	14	184/104
		Black soil	28/15	30/23	1/2	31/25

[a]The numerator is based on the current intake models (Balonov et al. 1996), while the denominator is derived from whole body measurements (IAEA 1996).
[b]The internal doses in the numerator do not take into account the effects of countermeasures and assume that no wild foods are consumed.

TABLE VII. PROGNOSIS FOR ADULT EXPOSURE (1996–2056) FOR LIVING IN A RURAL AREA WITH ACTIVITY DENSITY LEVELS FOR ^{137}Cs OF 15 Ci/km^2 [a]

Exposure pathway	Population group	
	Total (mSv)	Critical[b] (mSv)
External	20	27
Ingestion	10	33
Inhalation	0.1	0.3
Total	30	60

[a] Soil surface contamination in 1986: 0.6 MBq/m^2; dominating soil type: turf podzol sandy soil.

[b] Outdoor workers living in wooden houses and consuming both agricultural and natural food. (Balonov et al. 1992).

Note: Assuming that agricultural countermeasures and delivery of non-contaminated food be discontinued in 1996.

FIG. 1. The daily release of radioactive substances to the atmosphere (not including noble gases) due to the Chernobyl accident. The values shown are calculated from dynamics of the releases from the burning reactor and decay corrected to 6 May 1996 [INSAG-1]

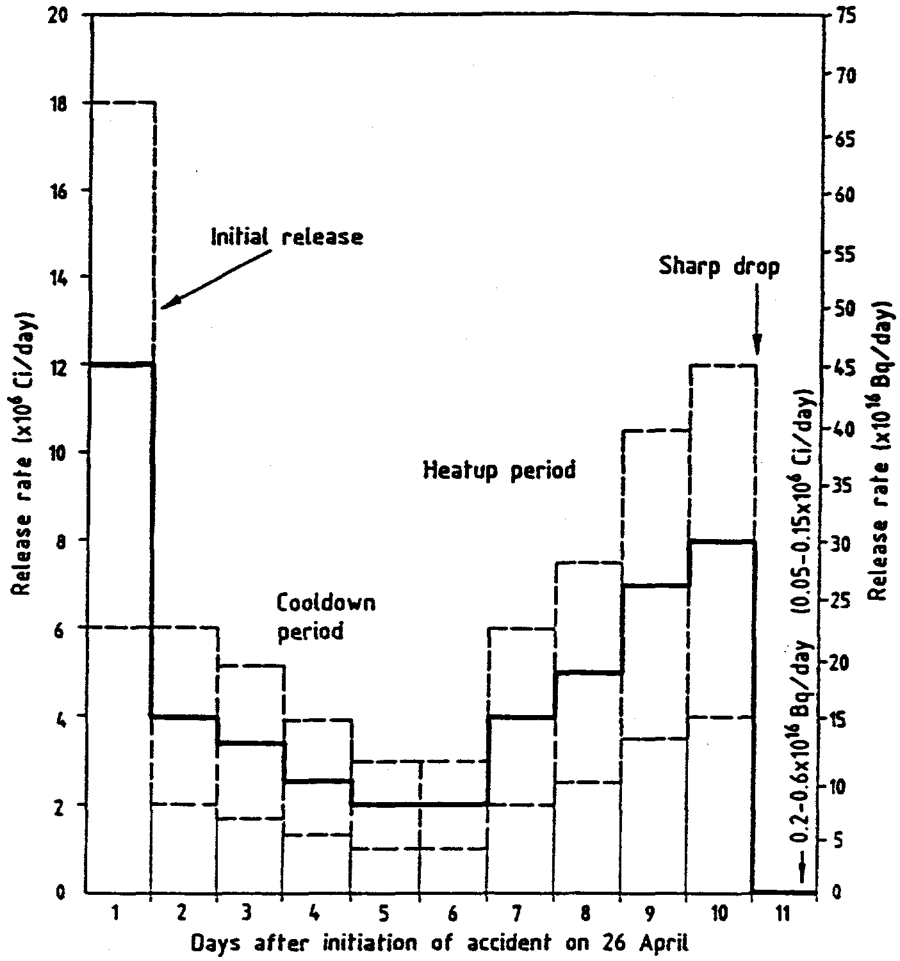


FIG. 2. Map of ^{137}Cs contamination in the western part of the Russian Federation as restored for 10 May 1986 (kBq/m^2).

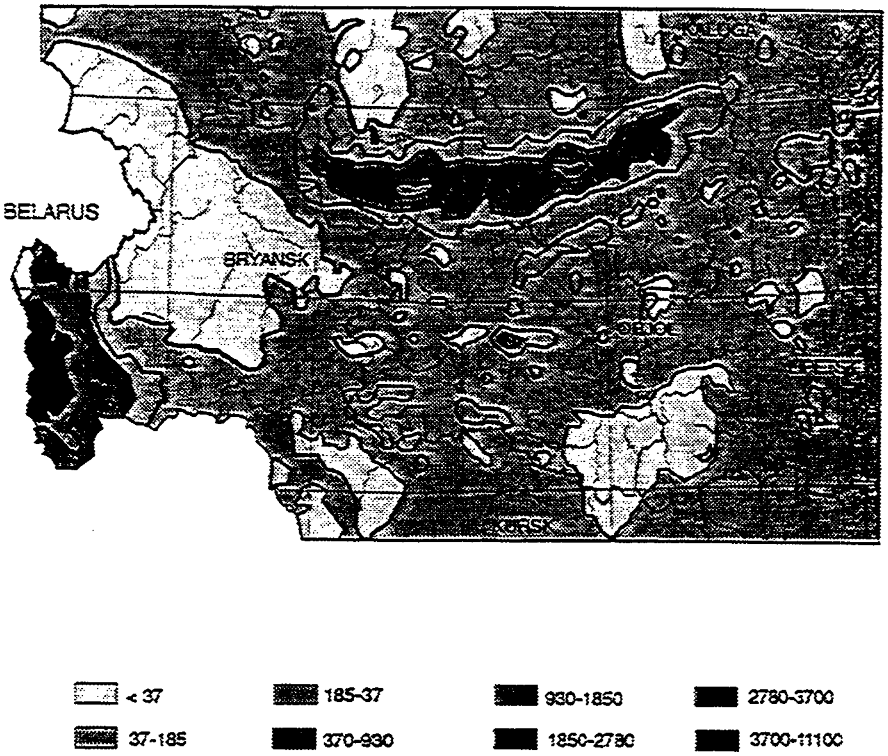


FIG. 3. Map of ^{137}Cs contamination in Belarus as restored for 1 January 1995 (kBq/m^2).

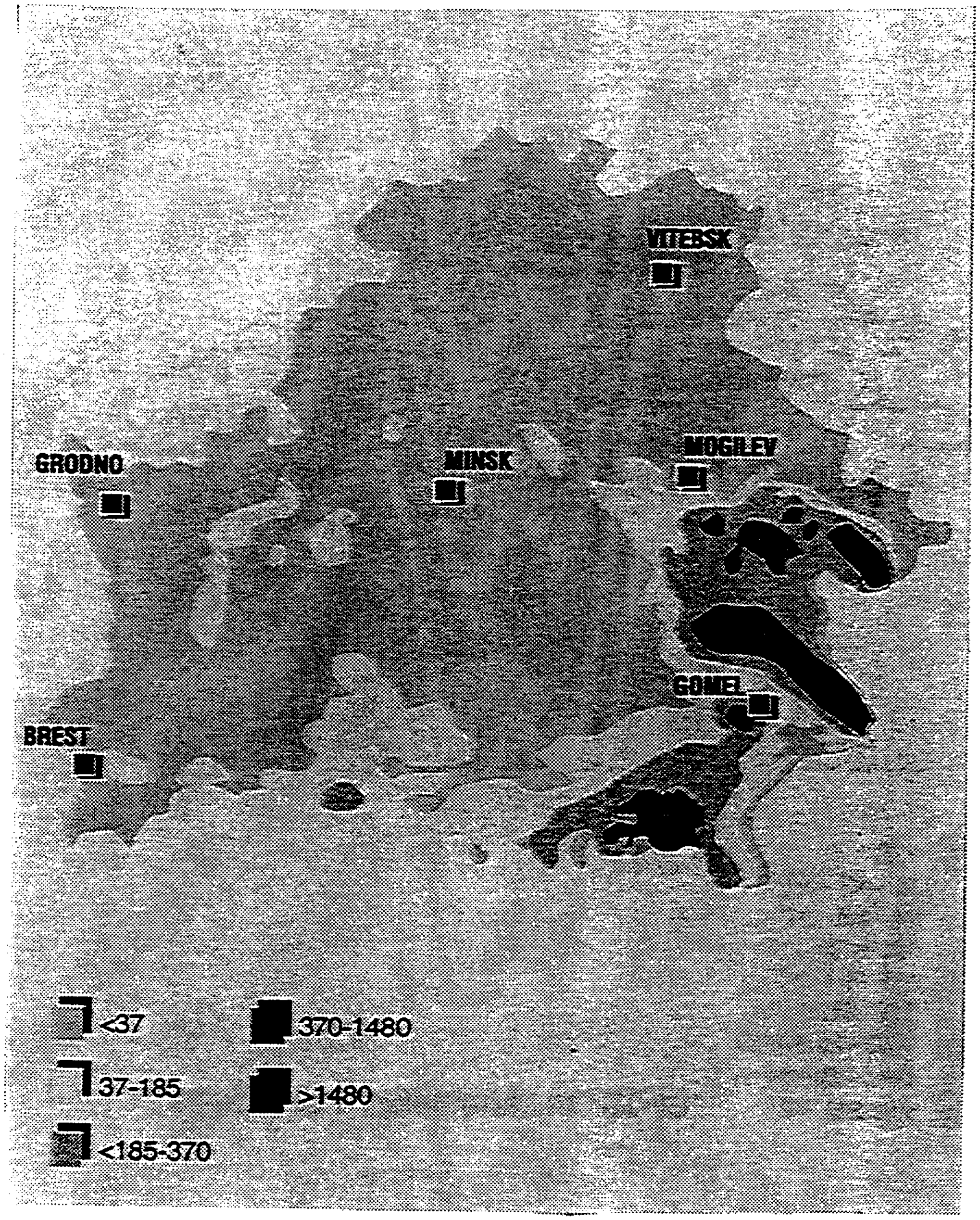


FIG. 4. Map of ^{137}Cs contamination in Ukraine (1996) (kBq/m^2).

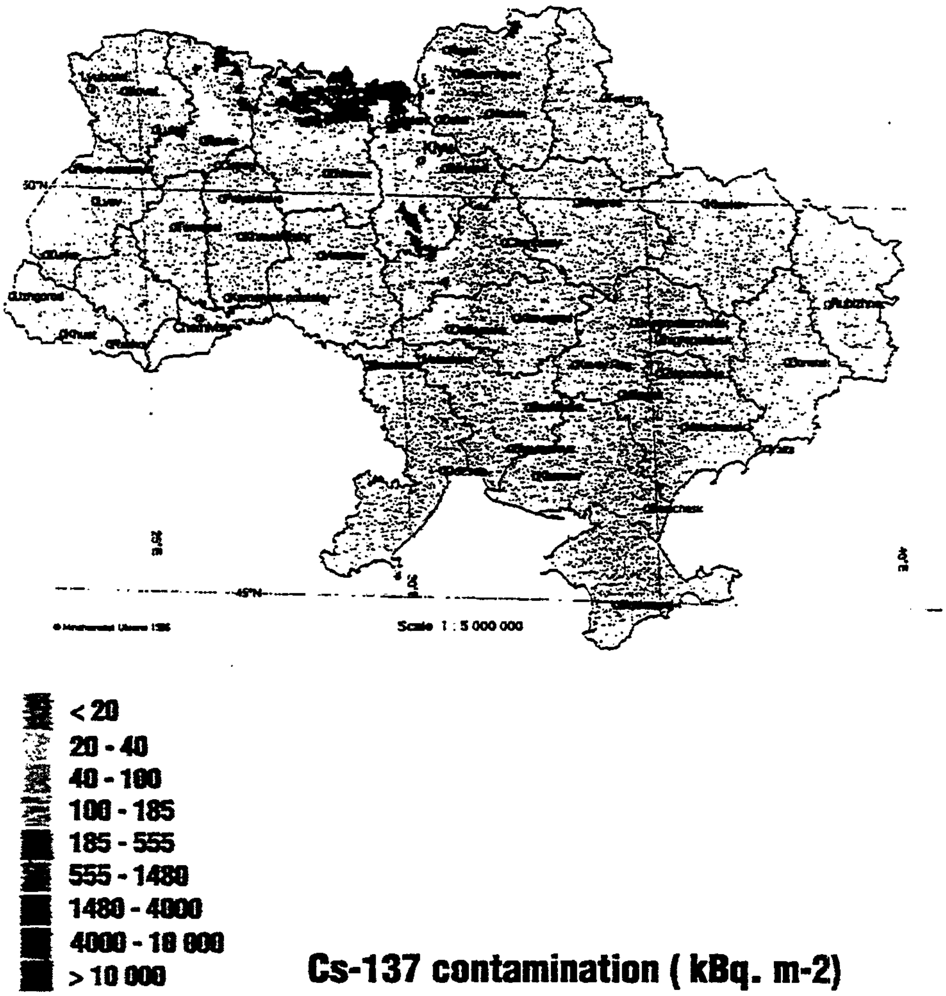


FIG. 5. Transfer of ^{137}Cs from soil to milk and mushrooms (boletus luteus) in Bryansk region as a function of time (Shutov and Bruk).

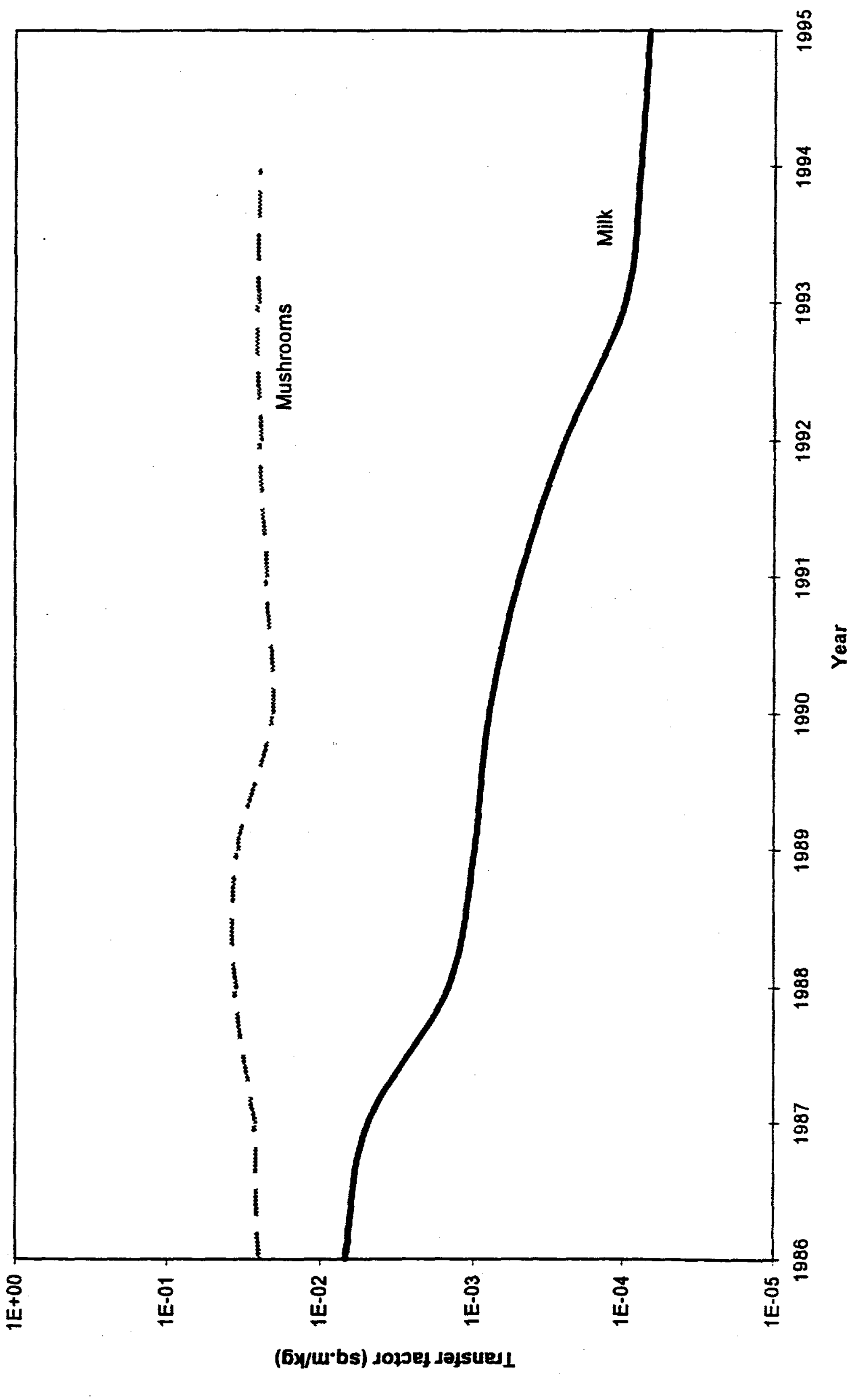
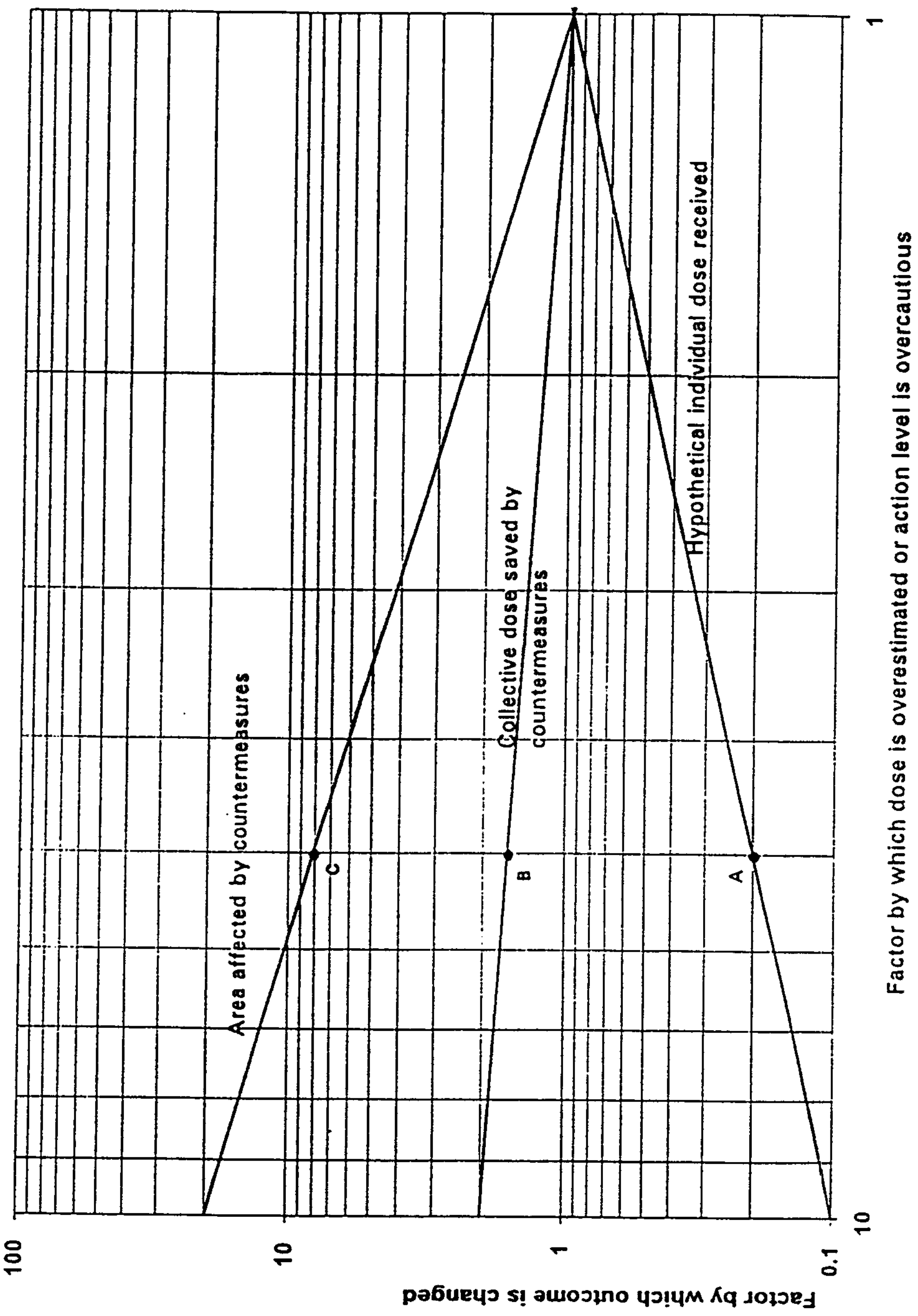


FIG. 8. The additional area of land for which countermeasures are needed and the collective dose saved for changes to an action level (Crick 91, RISO-R-742(EN).



Overestimating doses or selecting overcautious intervention levels has significant effects on the size of the areas for which countermeasures are needed, but much less of an impact on the collective dose saved. For example, if an intervention level is five times more cautious than need be, or if doses are overestimated by a factor of five, the hypothetical individual dose received by the 'critical person' is clearly a factor of five lower (A). The collective dose saved by using the tighter criterion is about 60% greater (B). However, the size of the area for which countermeasures are needed increases by a factor of about eight.