4. ENVIRONMENTAL COUNTERMEASURES AND REMEDIATION

The need for the application of urgent protective actions became evident very soon after the Chernobyl accident occurred. A wide range of countermeasures was applied for protecting the public from radiation, from urgent evacuation in 1986 of the inhabitants from the area of highest radioactive contamination to long term monitoring of radionuclides in foodstuffs in many European countries. The whole spectrum of the applied countermeasures and their effectiveness have been considered in a number of international reports [4.1–4.7].

The main subject of this section is the countermeasures that have been applied to the environment in order to reduce the radiological impact on humans. At the time of the Chernobyl accident the philosophy of radiation protection of non-human species had not been sufficiently developed to be practically applied for the purposes of justifying appropriate countermeasures. Such policies are currently still under development [4.8].

This section does not consider specifically the emergency and mitigatory actions applied to the damaged reactor aimed at reducing radioactive releases to the environment; these aspects have been covered elsewhere [4.2].

Environmental countermeasures have been applied since 1986 to urban, agricultural, forest and aquatic ecosystems. Most of these countermeasures were driven by relevant international and national radiological criteria.

4.1. RADIOLOGICAL CRITERIA

Countermeasures, termed protective actions at the emergency stage and remedial 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 optimize 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. However, 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.

4.1.1. International radiological criteria and standards

At the time of the Chernobyl accident in 1986, the relevant international radiation protection standards for protection of the public and workers were contained in International Commission on Radiological Protection (ICRP) Publication 26 [4.9]. Specific recommendations on the protection of the public in the event of a major radiation accident were given in ICRP Publication 40 [4.10]. The corresponding IAEA Basic Safety Standards, based on ICRP recommendations, were issued in 1982 [4.11]. The basic principles of modern radiation protection — justification, optimization and dose limitation — and the clear distinction between protection in normal and intervention situations were contained in these documents. At that time, the annual limit for occupational exposure was 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 the event of nuclear or radiological emergencies were not specifically established in these documents, and instead it was recommended:

(a) By almost all means to reduce human accidental exposure below doses that may result in deterministic health effects (acute radiation syndrome, radiation damage to particular organs or tissues);
(b) To intervene (i.e. to apply and subsequently withdraw countermeasures aimed at reducing stochastic health effects (cancer, genetic anomalies)) based on an optimization assessment taking into account both the collective dose reduction achieved by the application of the countermeasures and the associated economic and social intervention costs.

The most relevant ICRP guidance [4.10] recommended some generic two level criteria for intervention in the early accident phase — for sheltering, 5–50 mSv of whole body dose or 50–500 mSv to particular organs; for administration of stable iodine aimed at thyroid protection against intake of radioiodines, 50–500 mSv to the thyroid; for evacuation, 50–500 mSv of whole body dose or 500–5000 mSv to particular organs. For the intermediate accident phase, the generic criteria of 5–50 mSv of whole body dose or 50–500 mSv to particular organs were recommended for control of foodstuff contamination with radionuclides, and 50–500 mSv of whole body dose for relocation.

Afterwards, in connection with public concerns over the radiological consequences of the Chernobyl accident, new additional international regulations were 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 (see Table 4.1) [4.12].

New international basic radiation protection standards for the protection of the public and workers were developed by the ICRP in 1990 after research data had shown that radiation risk coefficients for stochastic human health effects were substantially higher than previously thought. The annual limits of exposure were substantially (by a factor of 2.5–5) reduced and established equal to 20 mSv for workers and 1 mSv for members of the general public [4.13]. The latter value is currently perceived as a safe level of human exposure.

Special limits for public protection in the event of nuclear or radiological emergencies were not established in these documents. Appropriate specific recommendations were developed later on intervention for the protection of the public in a radiological emergency [4.14]. In this guidance the optimization concept was confirmed as the basic one applicable in the event of an emergency and further elaborated with regard to dose averted as the consequence of intervention (see Fig. 4.1). The ICRP discarded the previous two level intervention criteria and recommended instead some intervention levels (in terms of averted effective dose) — 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/a for the control of foodstuffs.

A more recent ICRP publication (Publication 82) [4.15] 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 retaining the optimization principle, but also suggests generic radiological criteria for making decisions on countermeasure application. In particular, it proposes the value of the 'existing annual dose', including external and internal doses from natural and human-made radionuclides, of 10 mSv as the generic dose below which intervention is not usually expedient. This does not exclude intervention at lower doses if site specific optimization analysis proves it to be expedient. Inter alia, the ICRP recommended a generic intervention exemption level for radionuclides in commodities dominating human exposure equal to 1 mSv/a. This criterion could be applied for justification of the reference levels for radionuclides in food.

TABLE 4.1. GUIDELINE LEVELS FOR RADIONUCLIDES IN FOOD FOLLOWING ACCIDENTAL NUCLEAR CONTAMINATION, FOR USE IN INTERNATIONAL TRADE [4.12]

	Food for general consumption (Bq/g)	Milk and infant food (Bq/g)
Caesium-134, 137	1	1
Iodine-131	1	0.1
Strontium-90	0.1	0.1
Plutonium-239, americium-241	0.01	0.001

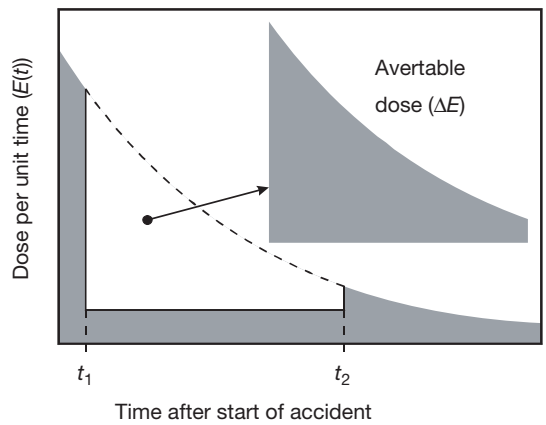

FIG. 4.1. Avertable dose and effective dose accumulated per unit time as a function of time when the protective measure is introduced at time t_1 and lifted again at time t_2.

4.1.2. National radiological criteria and standards

Limitations on human exposure caused by the Chernobyl accident, including standards for radionuclides in food, drinking water, timber, etc., were introduced soon after the accident, first by the USSR but also by many other European countries (i.e. Nordic countries, EU countries and eastern European countries [4.1]).

In accordance with the Standards of Radiation Safety [4.16] 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 until 26 April 1987), then 30 mSv for the second year and 25 mSv in each of 1988 and 1989 [4.3]. In all, until 1 January 1990, a dose to the general public not exceeding 173 mSv was allowed from the radioactive fallout of the Chernobyl accident.

In order to limit the internal exposure of members of the population, temporary permissible levels (TPLs) of radionuclide content in food products and drinking water were introduced in the USSR. Table 4.2 presents the TPLs for the main food products [4.3, 4.17]. The first TPL set approved

TABLE 4.2. TEMPORARY PERMISSIBLE LEVELS (Bq/kg) OF RADIONUCLIDE CONTENT IN FOOD PRODUCTS AND DRINKING WATER ESTABLISHED IN THE USSR (1986–1991) AFTER THE CHERNOBYL ACCIDENT [4.3, 4.17]

	TPL				
	4104–88	129–252	TPL-88	TPL-91	
Date of adoption	6 May 1986	30 May 1986	15 December 1987	22 January 1991	
Radionuclide	Iodine-131	Beta emitters	Caesium-134 and caesium-137	Caesium-134 and caesium-137	Strontium-90
Drinking water	3700	370	18.5	18.5	3.7
Milk	370–3700	370–3700	370	370	37
Dairy products	18 500–74 000	3700–18 500	370–1850	370–1850	37–185
Meat and meat products	—	3700	1850–3000	740	—
Fish	37 000	3700	1850	740	—
Eggs	—	37 000	1850	740	—
Vegetables, fruit, potato, root crops	—	3700	740	600	37
Bread, flour, cereals	—	370	370	370	37

TABLE 4.3. ACTION LEVELS (Bq/kg) FOR CAESIUM RADIONUCLIDES IN FOOD PRODUCTS ESTABLISHED AFTER THE CHERNOBYL ACCIDENT [4.3, 4.5]

	Action level				
	Codex Alimentarius Commission	EU	Belarus	Russian Federation	Ukraine
Year of adoption	1989	1986	1999	2001	1997
Milk	1000	370	100	100	100
Infant food	1000	370	37	40–60	40
Dairy products	1000	600	50–200	100–500	100
Meat and meat products	1000	600	180–500	160	200
Fish	1000	600	150	130	150
Eggs	1000	600	—	80	6 Bq/egg
Vegetables, fruit, potato, root crops	1000	600	40–100	40–120	40–70
Bread, flour, cereals	1000	600	40	40–60	20

by the USSR Ministry of Health on 6 May 1986 concerned ^{131}I activity concentrations in foodstuffs and was aimed at limiting the thyroid dose to children to 300 mGy. The next TPL set, adopted on 30 May 1986, concerned the content of all beta emitters in food products caused by surface contamination, with particular attention given to ecologically mobile and long lived caesium radionuclides. Later, TPLs introduced in 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, if all components contained caesium radionuclides at the level of TPL-86, would cause an internal dose of less than 50 mSv (at TPL-88 it would be less than 8 mSv and at TPL-91 it would be less than 5 mSv).

Action levels for ^{131}I in food established in some European countries in May 1986 varied within the range of 500–5000 Bq/kg. Later, the authorities of the EU established two values for caesium radionuclides in imported food, one for milk and infant food and another for all other food products (see Table 4.3) [4.3, 4.5]. Similar values were introduced in Nordic countries, with an exception for wild foods (reindeer meat, game, freshwater fish, forest berries, fungi and nuts), which 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 for ^{131}I and 10 kBq/kg for ^{137}Cs in imported food; for domestic foods the respective values were 2 and 1 kBq/kg. In the middle of May, action levels of 300 Bq/kg for ^{137}Cs in all food and 2 kBq/kg 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 in different countries and time periods.

Along with the standards for food products, standards were introduced by the USSR for agricultural raw material, wood (see Section 4.3) and herbs, and for beta contamination of different surfaces [4.3].

The general policy of the USSR, and later of the authorities in the separate republics, was to reduce both the radiological criteria and the TPLs along with the 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 decrease radionuclide content in products in order to limit associated human exposure. The TPLs were established by experts balancing the desire to reduce internal dose in populations with the need to maintain profitable agricultural production and forestry in the controlled areas. Different reference levels for numerous groups of food products were established with the aim of not restricting consumption of any foods unless the dose criterion might be exceeded.

By the end of 1991 the USSR had split into separate countries, among them Belarus, the Russian Federation and Ukraine, which had been strongly affected by the Chernobyl accident. Afterwards each country implemented its own policy of radiation protection of the public. Owing to the acceptance by the ICRP in 1990 of the annual effective dose limit for the public in regulated situations (practices) 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 the introduction of countermeasures, including long term remediation measures.

Current national TPLs for food products, drinking water and wood in the three countries are comparable with each other (see Table 4.3), and all of them are substantially lower than both the EU maximum permissible levels for import [4.5] and the Codex Alimentarius Commission's guidance levels for radionuclides in food moving in international trade [4.12].

The State authorities in the three countries have struggled to meet the established TPLs for products and the dose criteria by implementation of environmental countermeasures as described below and by inspection of foods throughout each country.

4.2. URBAN DECONTAMINATION

Decontamination of settlements was one of the main countermeasures applied to reduce external exposure of the public and cleanup 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 sources of external exposure 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 such as asphalt and concrete and to

a small extent on building walls and roofs. This is why most effective decontamination technologies involved removal of the upper soil layer.

The decontamination efficiency can be characterized by means of the following parameters: the dose rate reduction factor (DRRF), which is the relative reduction of dose rate above a surface following decontamination, and the dose reduction factor (DRF), which is the reduction of the effective external dose to an individual from gamma emitting radionuclides deposited in the environment.

4.2.1. Decontamination research

In order to ensure high decontamination effectiveness and to keep the associated costs low, several research projects have been implemented aimed at determining the values of the DRRF and DRF for particular decontamination technologies applied to different surfaces and artefacts in the human environment [4.18–4.20]. Reports from these experimental and theoretical studies contain validated models of urban decontamination and sets of model parameters and practical recommendations for cleanup 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 to optimize its implementation.

According to these and other studies, the contributions of different urban surfaces to the human external dose and the associated opportunities for dose reduction are determined by settlement and house design, construction material, the habits of the populations, the mode of radionuclide deposition (dry or wet), the radionuclide and physicochemical composition of the fallout, and time (see Section 3.2).

Following dry deposition, street cleaning, removal of trees and shrubs and ploughing of gardens are efficient and inexpensive means of achieving very significant reductions in dose and would rate highly in a list of short term remediation priorities. Roofs are important contributors to dose, but the cost of cleaning roofs is high and this countermeasure 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, gardens and lawns, both in the short term and the long term, would be given first priority, because a considerable reduction in dose (~60%) can be achieved at relatively low cost. Street cleaning would also be of benefit.

While planning decontamination for the long term, it is important to take into account the contribution of external dose to the total (external and internal) dose. In areas dominated by clay soils, transfer of caesium radionuclides in the food chain and the associated internal doses are low. In these areas the relative decrease in the 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 in the total dose due to village decontamination is expected to be less significant.

4.2.2. Chernobyl experience

Large scale decontamination was performed between 1986 and 1989 in the cities and villages of the USSR most contaminated after the Chernobyl accident. This activity was performed usually by military personnel and included washing of buildings 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 cleaning tens of thousands of residences and public buildings and more than a thousand agricultural farms [4.18, 4.21, 4.22].

In the early period following the accident, inhalation of resuspended radioactive particles of soil and nuclear fuel could contribute significantly 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 at the Chernobyl nuclear power plant and in the CEZ during the spring and summer of 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 [4.23] daily washing of streets in Kiev decreased the collective external dose to its three million inhabitants by 3000 man Sv, and decontamination of schools and school areas saved another 600 man Sv.

Depending on the decontamination technologies used, the dose rate over different measured plots was reduced by a factor of 1.5–15. However, the high cost of these activities hindered their comprehensive application on contaminated areas. Due to these limitations, the actual effectiveness of the decrease in annual external dose was 10–20% for the average population and ranged from about 30% for children visiting kindergartens 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 region of the Russian Federation [4.18].

Regular monitoring of decontaminated plots in settlements over five years showed that after 1986 there was no significant recontamination and that the exposure rate was decreasing over the long term, as described in Section 5.1 of this report. The averted collective external dose to 90 000 inhabitants of the 93 most contaminated settlements of the Bryansk region was estimated to be about 1000 man Sv [4.18].

Since 1990 large scale decontamination in the countries of the former USSR has been stopped, but particular contaminated plots and buildings with measured high contamination levels have been specifically cleaned. Some decontamination activities still continue in Belarus, aimed mostly at public buildings and areas: hospitals, schools, recreation areas, etc. However, in some contaminated Belarusian villages, cleanup of dwellings and farms has also been performed [4.22].

Another area of continuing decontamination activity is the cleanup 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 [4.22].

4.2.3. Recommended decontamination technologies

In accordance with present radiation protection methodology, a decision on intervention (decontamination) and selection of optimal decontamination technologies should be made giving consideration to the costs of all actions and to social factors. The calculated cost should address the various decontamination technologies for which an assessment of the averted dose has been made. The benefit (averted collective effective dose) and detriment (expenses, collective dose to decontamination workers) are to be compared for each decontamination technology by means of a cost–benefit analysis [4.9] or multiattribute analysis [4.24], 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 can be recommended for the long term:

(a) 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. The clean soil from the holes should be used to cover the decontaminated areas. Such a technology excludes the formation of special burial sites for radioactive waste.
(b) Private fruit gardens should be treated by deep ploughing or removal of the upper 5–10 cm layer of soil. By now, vegetable gardens have been ploughed many times, and the activity distribution in soil will be uniform in a layer 20–30 cm deep.
(c) 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 (a)).
(d) Cleaning or replacement of roofs.

These procedures can be applied both for decontaminating single private gardens and houses and for decontaminating settlements as a whole. It is evident that, in the latter case, the influence of the decontamination on further reduction in external radiation dose will be greater. Achievable decontamination factors for various urban surfaces are presented in Table 4.4. Detailed data on the efficiency, technology, necessary equipment, cost and time expenses, quantity of radioactive waste, and other parameters of decontamination procedures are contained in Ref. [4.25].

Radioactive waste generated from urban decontamination should be disposed of in

accordance with established regulatory requirements. In the event of large scale decontamination, temporary storage should be arranged in special isolated areas from which future activity release into the environment will be negligible. The site should be marked by the international symbol of radiation hazard.

4.3. AGRICULTURAL COUNTERMEASURES

The implementation of agricultural countermeasures after the Chernobyl accident has been extensive, both in the most severely affected countries of the former USSR and in western Europe. The main aim of agricultural countermeasures was the production of food products with radionuclide activity concentrations below action levels[2]. The application of countermeasures in intensive agricultural production systems was largely confined to Belarus, the Russian Federation 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 private food production from unimproved meadows, where high ^{137}Cs activity concentrations have persisted for many years [4.3, 4.4, 4.7].

High and persistent transfer of ^{137}Cs has also occurred in many contaminated areas of western Europe. In these countries countermeasures have largely been focused on animal food products, for example for grazing animals on unimproved pastures.

4.3.1. Early phase

From 2–5 May 1986 about 50 000 cattle, 13 000 pigs, 3300 sheep and 700 horses were evacuated from the CEZ together with the people [4.26]. In the CEZ more than 20 000 agricultural and domestic animals, including cats and dogs, remaining after the evacuation were killed and buried. Due to a lack of forage for the evacuated animals and difficulties in managing the large number of animals in the territories to which they had been moved, many of the evacuated animals were also slaughtered [4.27, 4.28]. In the acute period after the accident it was not possible to differentiate between the different levels of contamination of animals, and in the period May–July 1986 the total number of slaughtered animals reached 95 500 cattle and 23 000 pigs.

Many carcasses were buried and some were stored in refrigerators, but this produced great hygiene, practical and economic difficulties. Condemnation of meat was an immediately available and effective countermeasure to reduce ingested dose from animal products and was widely used 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 from entering the food chain. The following were recommendations [4.29] on how to achieve this:

[2] Referred to in the countries of the former USSR as temporary permissible levels (TPLs).

TABLE 4.4. ACHIEVABLE DECONTAMINATION FACTORS (DIMENSIONLESS) FOR VARIOUS URBAN SURFACES [4.25]

	Technique	DRRF
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	Cutting back or removal	~10
Streets	Sweeping and vacuum cleaning	1–50
Streets (asphalt)	Lining	>100

(a) Exclusion of contaminated pasture grasses from the animals' diet by changing from pasture to indoor feeding of uncontaminated feed;
(b) Radiation monitoring at processing plants and subsequent rejection of milk in which ^{131}I activity concentrations were above the action level (3700 Bq/L at that time);
(c) 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 towards milk from collective farms, and few private farmers were involved. Information on countermeasures for milk was only given to managers and local authorities and was not distributed to the private farming system of the rural population. This resulted in limited application of countermeasures, especially for privately produced milk in rural settlements, 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 a period of 1–2 months. However, this countermeasure was not in widespread use at this stage, partly due to a lack of availability 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 the contamination of pasture and identification of where contaminated milk would be found.

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 that 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 [4.3, 4.30].

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, the Russian Federation and Ukraine, respectively [4.31].

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

(i) 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);
(ii) Minimizing external exposure and formation of contaminated dust by omitting some procedures normally used in crop production;
(iii) Limiting the use of contaminated manure for fertilization;
(iv) Preparation of silage from maize instead of hay;
(v) Restricting the consumption of milk produced in the private sector;
(vi) Obligatory radiological monitoring of agricultural products;
(vii) 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 the farming of private plots [4.3, 4.30].

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

Sweden received some of the highest levels of deposition outside of the countries of the former USSR. Initially, Sweden imposed action levels on ^{131}I and ^{137}Cs activities in imported and domestic foods (see Section 4.1.2). A range of other responses were applied: (a) cattle were not put on to pasture if the ground deposition exceeded 10 kBq/m^2 of ^{131}I and 3 kBq/m^2 of radiocaesium; (b) advice was given not to consume fresh leafy vegetables and to wash other fresh vegetables; (c) restrictions were placed on the use of sewage sludge as fertilizer for soil; (d) deep ploughing was recommended; and (e) a higher cutting level for harvesting of grass was advised.

In Norway, crops in fields were monitored after harvesting, and those with radiocaesium above 600 Bq/kg fresh weight were discarded and ploughed in. Also, hay and silage harvested in June were monitored, and that with activity concentrations exceeding the 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 for the regulation of 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 [4.32].

4.3.2. Late phase

Radiological surveys of agricultural products showed that by the end of 1986 four regions of the Russian Federation (Bryansk, Tula, Kaluga and Orel), five regions of Ukraine (Kiev, Zhytomyr, Rovno, Volyn and Chernigov) and three regions of Belarus (Gomel, Mogilev and Brest) had food products that exceeded the action levels for radiocaesium. In the more contaminated areas of the Gomel, Mogilev, Bryansk, Kiev and Zhytomyr regions in the first year after the accident, the proportion of grain and milk exceeding the action levels was about 80% [4.3, 4.7, 4.26].

Additionally, in the early 1990s in Ukraine, 101 285 ha of agricultural land was withdrawn from agricultural use (about 30% of this area had a ^{137}Cs contamination level above 555 kBq/m^2). Privately owned cattle were moved with the people from some settlements. Provision of 'clean' foodstuffs produced in the collective sector or imported from 'clean' regions was organized for those residents not resettled.

In the Russian Federation in 1987–1988 further evacuations of agricultural animals were carried out, but on a more elective basis 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. Of 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 were 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 some settlements were moved. In total, 470 settlements were moved. In all these resettlements, the agricultural animals accompanied their owners to the new locations where possible.

Application of countermeasures in contaminated areas had two major radiation protection aims. The first was to guarantee foodstuff production corresponding to the action levels and to ensure an annual effective dose to local inhabitants of less than 1 mSv. The second was to minimize the total flux of radionuclides in agricultural production. Generally, the earlier agricultural countermeasures were applied, the more cost effective they were [4.33].

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 with acceptably low radiocaesium levels. In the second year after the accident the radiocaesium activity concentration in grain was much lower than in the first year, and countermeasure application ensured that most grain was below the action levels. By 1991 less than 0.1% of grain in all three countries had radiocaesium contents above 370 Bq/kg.

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 the action levels in all three countries. Changes with time of milk and meat exceeding the action levels can be seen in Fig. 4.2. It should be noted that the values of the 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. 4.3.

The differences in the time trend shown in Fig. 4.2 among the countries mainly relate to changes in the action levels 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

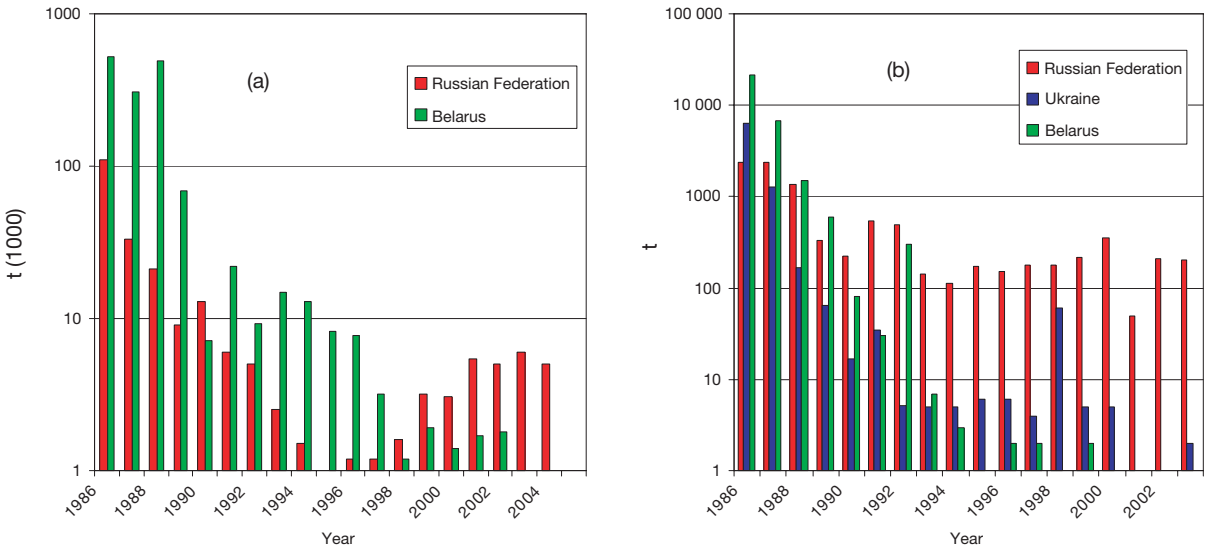

FIG. 4.2. Amounts of milk (a) and meat (b) exceeding the action levels in the Russian Federation (collective and private), Ukraine and Belarus (only milk and meat entering processing plants) after the Chernobyl accident [4.26].

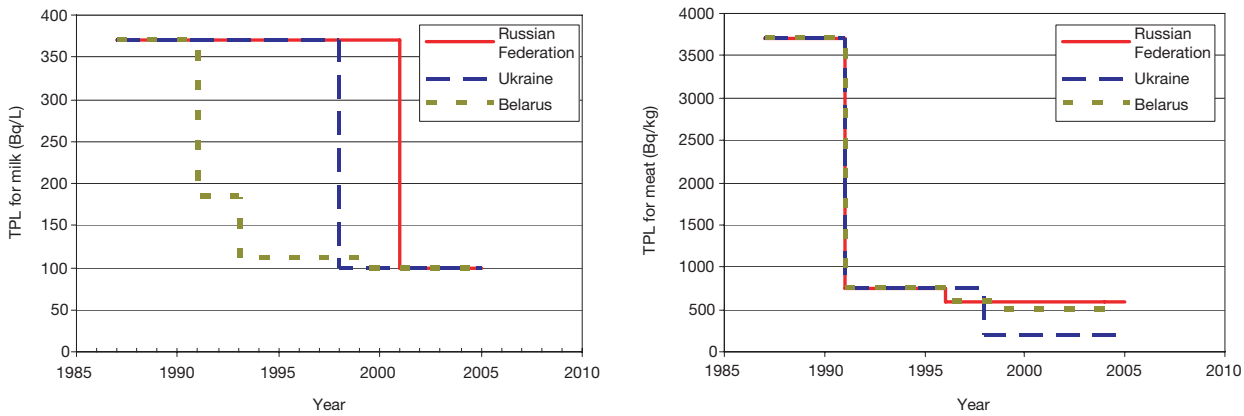

FIG. 4.3. Changes with time in the action levels (TPL) in the USSR and later in the three independent countries [4.34].

reduction in the amounts of meat above the action levels in Ukraine and Belarus is because animals are monitored before slaughter to ensure that the meat is below the required level. In the Russian Federation, where animals are also monitored before slaughter, the concentration data are higher, because they refer to both privately and collectively produced meat.

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. 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. 4.2).

4.3.3. Countermeasures in intensive agricultural production

The main countermeasures used in the USSR, and later in the three independent countries, are briefly described below. The priority was 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 countermeasures have been repeatedly revised and updated [4.35–4.37].

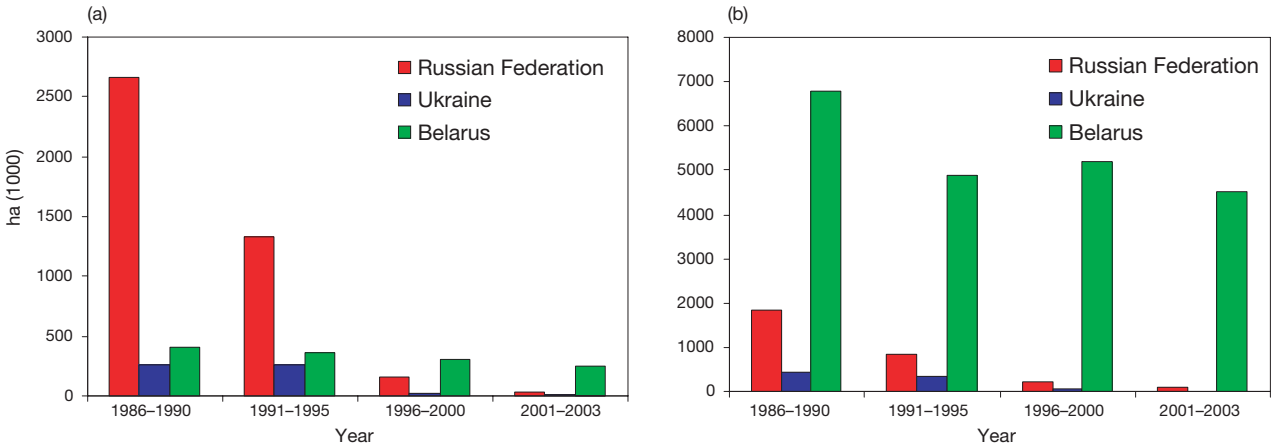

FIG. 4.4. Changes in the extent of agricultural areas treated with liming (a) and mineral fertilizers (b) in the countries most affected by the Chernobyl accident [4.34].

4.3.3.1. 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) fertilizers and lime. Ploughing dilutes the radioactive contamination 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 fertilizers increases plant production, thereby diluting the radioactivity in the plant. In addition, the use of fertilizers reduces root uptake into plants by decreasing the Cs:K ratio in the soil solution [4.30].

When soil treatment includes all the above measures it is commonly called radical improvement; this has been found to be the most efficient and 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 fertilization rates. Commonly, high value legumes and cereal grasses were grown on the treated land. The nature of the treatment and the efficiency of the radical improvement of hay meadows and pastures strongly depend on the type of meadow and the soil properties. Traditional surface improvement, involving soil discing, fertilization and surface liming, was less effective. Some marshy plots were drained, deep ploughed, improved and used as grassland. In the 1990s 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 fertilization 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 [4.30, 4.38].

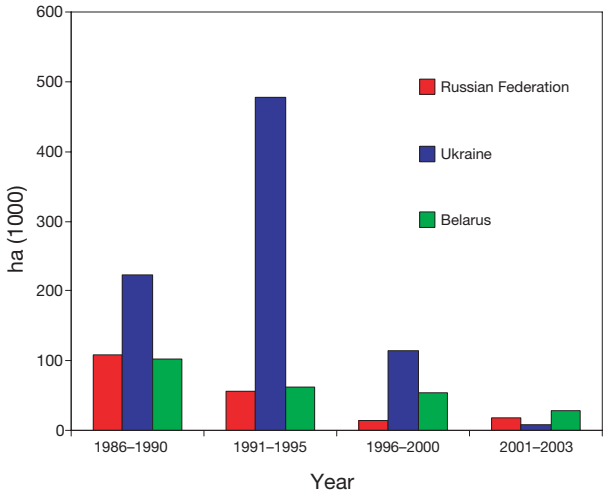

FIG. 4.5. Areas of radical improvement in the countries most affected by the Chernobyl accident [4.34].

Areas that received additional fertilizers in each of the three most affected countries are shown in Fig. 4.4; areas receiving radical improvement are shown in Fig. 4.5. The average amount of additional potassium fertilizers added was about 60 kg/ha of K_2O annually between 1986 and 1994. In the mid-1990s 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 the Russian

Federation this halted the previous decrease in the amounts of milk and meat exceeding the radiation safety standards (see Fig. 4.2); for example, in the more contaminated areas, such as Novozybkov (Bryansk region), because of insufficient use of potassium fertilizers, ^{137}Cs activity concentrations in agricultural products in 1995–1996 increased by more than 50% compared with the period of optimal countermeasure application (1991–1992).

The effectiveness of soil treatment is influenced by soil type, nutrient status and pH, and also by the plant species selected for reseeding. In addition, the application rates of NPK fertilizers and lime affect the reduction achieved. Several studies have shown that the reduction factors achieved for soil–plant transfer of radiocaesium following radical improvement, liming and fertilization were in the range of two to four for poor, sandy soils and three to six for more organic soils. An added benefit was the reduction in external dose rate by a factor of two to three due to the dilution of the surface contamination layer after ploughing.

Even though the radiological problems associated with ^{90}Sr are less acute than those of ^{137}Cs, some countermeasures have been developed and a reduction of two to four in the soil–plant transfer of radiostrontium following discing, ploughing and reseeding has been achieved.

Despite these countermeasures, in the more highly contaminated areas of the Bryansk region the radiocaesium contamination of 20% of the pasture and hay on farms still exceeded the action levels in 1997–2000. Concentrations of ^{137}Cs in hay varied between 650 and 66 000 Bq/kg dry weight.

4.3.3.2. 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 until 2002 (Fig. 4.6). The extent of the difference is considerable, and fodder crops such as lupin, peas, buckwheat and clover, which accumulate high amounts of radiocaesium, were completely or partly excluded from cultivation.

In Belarus rapeseed is grown on contaminated areas with the aim of producing two products: edible oil and protein cake for animal fodder. Varieties of rapeseed are grown that have a twofold to threefold lower ^{137}Cs and ^{90}Sr uptake rate than other varieties. When the rapeseed is grown, additional fertilizers (liming with 6 t/ha 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 that is used for the protein cake. During processing of the rapeseed, 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 past decade the area under rapeseed cultivation has increased fourfold to 22 000 ha [4.40].

4.3.3.3. Clean feeding

The provision of uncontaminated feed or pasture to previously contaminated animals for an appropriate period before slaughter or milking ('clean' feeding) effectively reduces radionuclide contamination, respectively, in meat and milk at a rate that depends on the animal's biological half-life for each radionuclide. Radiocaesium activity concentration in milk responds rapidly to changes in diet, as the biological half-life is a few days. For meat the response time is longer, due to the longer biological half-life in muscle [4.28].

Clean feeding reduces uptake of the contaminating radionuclides; it has been one of the most important and frequently used countermeasures after the Chernobyl accident for meat from agricultural animals in both the countries of the former USSR and western Europe. Official estimates of the number of cattle treated are between 5000 and 20 000 annually in the Russian Federation and 20 000 in Ukraine (supported by the government up

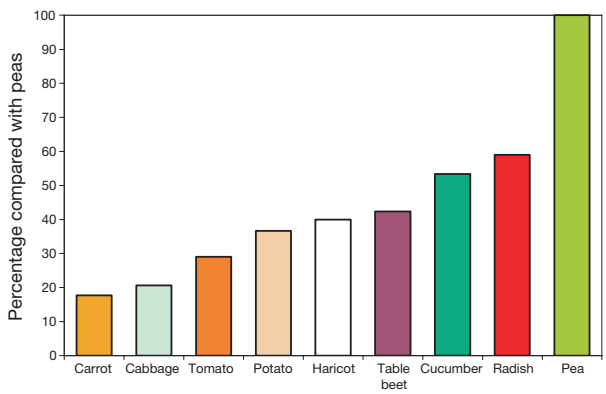

FIG. 4.6. Comparison of ^{137}Cs uptake in different crops, normalized to that of peas [4.39].

to 1996) [4.3]. Clean feeding is routinely used in all three countries for meat production and is combined with live monitoring of animals, so that if animal flesh is above the action levels the animals can be returned to the farm for further clean feeding.

4.3.3.4. Administration of caesium binders

Hexacyanoferrate compounds (commonly referred to as Prussian blue) are highly effective radiocaesium binders. They 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, partly 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 ten [4.41].

Prussian blue can be added to the diet of animals as a powder, incorporated into pelleted feed during manufacturing or mixed with sawdust. A locally manufactured hexacyanoferrate called ferrocyn (a mixture of 5% $KFe[Fe(CN)_6]$ and 95% $Fe_4[Fe(CN)_6]$) has been developed in the Russian Federation. It has been administered as 98% pure powder, salt licks (10% ferrocyn) and in sawdust with 10% adsorbed ferrocyn (called bifege) [4.42].

The number of cattle treated annually with Prussian blue in each of the three countries is shown in Fig. 4.7. In addition, slow release boli containing hexacyanoferrate have been developed that are introduced into the animals' rumen and gradually release the caesium binder over a few months. The boli, originally developed in Norway, consist of a compressed mixture of 15% hexacyanoferrate, 10% beeswax and 75% barite [4.43].

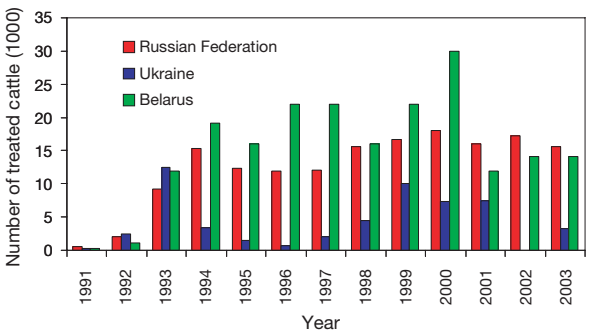

FIG. 4.7. Changes with time in the use of Prussian blue in the three countries of the former USSR (provided by Forum participants from official national sources).

Prussian blue has been used to reduce the ^{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 [4.44]. In Belarus a special concentrate with Prussian blue is distributed at a rate of 0.5 kg of concentrate per cow daily, and an average value of three for the reduction factor for milk has been achieved.

Prussian blue has not been used as extensively in Ukraine as in the Russian Federation and Belarus, and its use was confined to the early 1990s. This is because in Ukraine no local source of Prussian blue is available and the cost of purchasing it from western Europe was considered to be too high. Therefore, instead, locally available clay mineral binders have been used on a small scale. These were cheaper but somewhat less effective than Prussian blue.

4.3.4. Summary of countermeasure effectiveness in intensive production

The effectiveness of the different agricultural countermeasures in use on farms is summarized in Table 4.5. The reduction factors (ratio of radiocaesium activity concentration in the product before and after countermeasure application) achieved by each measure are given.

4.3.5. Countermeasures in extensive production

Extensive production in the three countries of the former USSR is largely confined to the grazing of privately owned cattle on poor, unimproved meadows. Owing to the poor productivity of these areas, radiocaesium uptake is relatively high compared with land used by collective farms. Radical improvement of meadows used by privately owned cattle 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 the Russian Federation and Belarus. In the Russian Federation all three Prussian blue delivery systems are used, according to availability and preference [4.46].

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 changes in slaughter times. Many of these countermeasures were still in use in 2004. The application of long term countermeasures has been most extensive in Norway and Sweden, but long term countermeasures have also been applied in the UK and Ireland.

AFCF is a highly effective hexacyanoferrate compound achieving up to a fivefold reduction in radiocaesium in lamb and reindeer meat and up to a threefold reduction in cow's milk and a fivefold reduction in goat's milk. The use of AFCF has been temporarily authorized in the EU and in some other countries. AFCF as a caesium binder is effective in extensive production systems, in contrast to many other countermeasures whose 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 [4.47]. It has been estimated that the use of boli as a countermeasure for sheep was 2.5 times as cost effective as feeding with uncontaminated feed [4.48]. Salt licks containing AFCF have also been used, but are less effective [4.49].

Management regimes have been modified for some animals in contaminated areas; for example, 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 are restricted in some areas. The animals are monitored to ensure that their ^{137}Cs activity concentrations are below the action level 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. 4.8, which shows the number of reindeer in Sweden that had radiocaesium activity concentrations above the action level and the number of slaughtered animals. The

TABLE 4.5. SUMMARY OF THE REDUCTION FACTORS ACHIEVED WITH THE DIFFERENT COUNTERMEASURES USED IN THE THREE COUNTRIES OF THE FORMER USSR [4.30, 4.34, 4.40, 4.45]

	Caesium-137	Strontium-90
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 fertilizers	1.5–3.0	0.8–2.0
Application of organic fertilizers	1.5–2.0	1.2–1.5
Radical improvement:		
First application	1.5–9.0[a]	1.5–3.5
Further applications	2.0–3.0	1.5–2.0
Surface improvement:		
First application	2.0–3.0[a]	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 caesium binders	2–5	—
Processing milk to butter	4–6	5–10
Processing rapeseed to oil	250	600

[a] For wet peat, up to 15 with drainage.

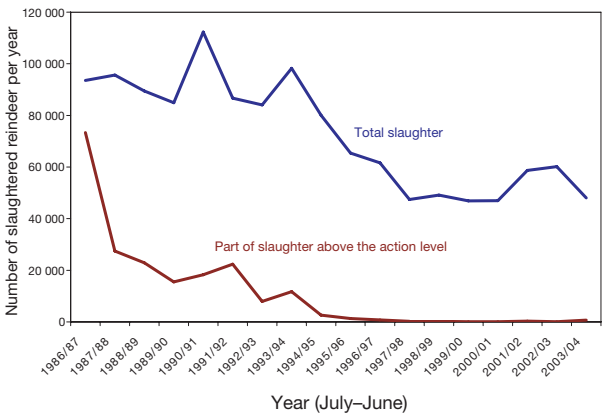

FIG. 4.8. Change with time in the number of reindeer in Sweden with radiocaesium activity concentrations above the action level and in the number of slaughtered animals [4.50].

high number of slaughtered animals in the first year was in part due to the low action level of 300 Bq/kg fresh weight, which was subsequently increased to 1500 Bq/kg from 1987. The decline has been achieved partly by extensive use of countermeasures, including clean feeding and change of slaughter time.

4.3.6. Current status of agricultural countermeasures

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

In Belarus, fertilization with phosphorus–potassium is used on collective farms, and milk above the action level from the farms is processed into butter. Radical improvement is used on private farms together with Prussian blue for milk. Rapeseed 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 the action level is used within the settlements, partially to feed pigs. All other countermeasures are directed at private farmers. These countermeasures currently comprise the radical improvement of meadows and the use of clay mineral caesium binders for privately produced milk.

In the Russian Federation, fertilizers (largely potassium) are supplied to intensive farms. For private farms, Prussian blue is provided for privately produced milk and, on request, for privately produced meat intended for market.

In all contaminated settlements a service for the monitoring of local produce exists, 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 are still restricted in certain areas of the UK.

4.3.7. A wider perspective on remediation, including socioeconomic issues

Experience after the Chernobyl accident has shown that account has to be taken, in developing restoration strategies, of a wide range of different issues to ensure the long term sustainability of large and varied types of contaminated areas [4.51]. The selection of robust and practicable restoration strategies should take into account not only radiological criteria but also: (a) practicability, including effectiveness, technical feasibility and the acceptability of the countermeasure; (b) cost–benefit; (c) ethical and environmental considerations; (d) requirements for effective public communication; (e) the spatial variation in many of these factors; and (f) the contrasting needs of people in urban, rural and industrial environments [4.52]. When not only radiological factors but also social and economic factors are taken into account, better acceptability of countermeasures by the public can be achieved.

A number of European Commission (EC) and United Nations projects have applied some of the above considerations in trying to provide appropriate information to, and interaction with, people in contaminated territories and in involving them in making decisions about responding to enhanced radiation doses and about ways of living 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 ETHOS project [4.53, 4.54] identified the dissemination of a practical radiological culture within all segments of the population as a prerequisite, especially for professionals in charge of public health. The EC Tacis Programme ENVREG project [4.55] in Belarus and Ukraine sought to minimize the environmental and secondary medical effects resulting from the

Chernobyl accident by improving the public perception and awareness of these effects.

Most recently, the EC CORE project [4.56] was initiated to address long term rehabilitation and sustainable development in the Bragin, Chechersk, Slavgorod and Stolin areas of Belarus. CORE community projects include health care, radiological safety, information and education. In addition, critical socioeconomic constraints are being addressed, specifically using a crediting system for small businesses and farmers, the cost effective production of 'clean' products, the creation of a rural entrepreneurs' centre and the promotion of community economic initiatives.

The Chernobyl debate is increasingly about socioeconomic issues and the communication of technical information in an understandable way. The ETHOS, ENVREG and CORE projects all have a strong community focus and target Chernobyl affected communities and other local stakeholders. Feedback from the communities should indicate which approaches are proving successful and to what extent. The holistic philosophy of these projects of considering both environmental and social problems is in line with the recent United Nations initiative known as Strategy for Recovery [4.57].

4.3.8. Current status and future of abandoned land

In this section the extent of recovery of abandoned land is summarized for each of the three countries of the former USSR. In 2004, 16 100, 11 000 and 6095 ha of previously abandoned land in Belarus, the Russian Federation and Ukraine, respectively, were returned to economic use [4.26]. In general, there is currently little effort being devoted to any further rehabilitation of abandoned areas.

4.3.8.1. Exclusion and resettlement zones in Belarus

The CEZ covers a total of 215 000 ha in Belarus. The people who used to reside there were evacuated in 1986. Since May 1986, lands in the CEZ have been taken out of agricultural and other production. The Polessye State Radioecological Reserve (PSRR) was set up by a government decree in 1988 and comprises mainly the CEZ but also includes some other areas with high levels of transuranium radionuclide contamination. Access to the PSRR is forbidden and very few, mostly old, people are currently present, without permission, in the area. Pursuant to the law on the legal regime of territories contaminated as a result of the Chernobyl nuclear power plant accident [4.58], most of the land in the CEZ cannot be brought back into economic production within a millennium, because of contamination with long lived transuranium radionuclides. In the CEZ only activities related to ensuring radiation safety, fighting forest fires, preventing the transfer of radioactive substances, protecting the environment and scientific research and experimental work are permitted.

While the CEZ (i.e. the Bragin, Khoiniki and Narovlya areas of the Gomel region) is the most contaminated area adjacent to the Chernobyl nuclear power plant, a further resettlement zone was identified in the early 1990s from which more people were evacuated; this zone covers a total area of 450 000 ha.

A total area of agricultural land of 265 000 ha received a deposition of ^{137}Cs at levels in excess of 1480 kBq/m^2 and/or of ^{90}Sr in excess of 111 kBq/m^2 and/or of plutonium isotopes in excess of 3.7 kBq/m^2. All this land is excluded from agricultural use.

The remaining abandoned agricultural land in the resettlement zone could be used for agriculture in the future. The present state of the ecosystems and the economic infrastructure of the resettlement zone are 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 led to an increase in the number of perennial weeds and shrubs. Unlike in the CEZ, in the resettlement zone limited access for certain maintenance activities, such as the activities needed to maintain roads, electricity transmission lines, etc., is permitted.

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, surveys of former agricultural lands were conducted to determine whether it is possible to rehabilitate the land for agricultural use. This was based on radiological considerations only.

By 2001 a total of 14 600 ha of previously withdrawn land had been returned to use [4.34], and recently this has been 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 has been used to minimize radiocaesium and radiostrontium uptake based on official guidelines [4.37].

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 by Bogdevitch et al. [4.59], a total of about 35 000 ha of the 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, feeding of Prussian blue to cattle, liming and fertilization.

Methodologies for the rehabilitation of abandoned land are being developed and improved, in particular with respect to economic evaluation. The main obstacles to the renewed agricultural use of abandoned land are the destroyed infrastructure, the high production cost and the low market demand for the agricultural 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.

4.3.8.2. Rehabilitation of contaminated lands in Ukraine

The first priority was the rehabilitation of land on which people are living. Consideration has since been given to the potential rehabilitation of abandoned areas. Such areas can be rehabilitated if this procedure is expedient with respect to economic and social criteria. The main condition for human occupancy of such areas without restrictions is that the additional annual effective dose should not exceed 1 mSv.

The efficiency of countermeasures is determined by the following criteria:

(a) Radiological: reduction of radionuclide content in local products and in the associated individual and collective dose.
(b) Economic: increased product market value.
(c) Social and psychological: public opinion on a given countermeasure.

In 2004, on the basis of 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 that could be rehabilitated declines (see Table 4.6). Table 4.6

TABLE 4.6. REHABILITATION OF ZONES OF OBLIGATORY RESETTLEMENT (OUTSIDE THE CHERNOBYL EXCLUSION ZONE)[a]

Area	Abandoned land (ha)	Can be rehabilitated judged on radiological, economic and social criteria (ha)
Kiev region		
1998–2000 (done)	—	3475
2001–2005	—	4720
Total	29 342	8205
Zhytomyr region		
1998–2000 (done)	—	2620
2001–2005	—	4960
Total	71 943	7580

[a] Provided by Forum participants from official national sources.

shows a scheme for rehabilitation based on technical criteria over a seven year period The first phase, from 1998 until 2000, was implemented, but that for the second phase was not, due to changing economic and social conditions.

In the CEZ, the limiting radionuclide is now ^{90}Sr rather than ^{137}Cs. On the basis of radiological considerations, the south-west 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.

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

Some people have returned to abandoned areas to live, and others live outside them but use the land for agricultural activities such as hay production. Countermeasures are not being applied in the abandoned areas, but there is sanitary and regulatory control of these activities.

4.3.8.3. Abandoned zones in the Russian Federation

Areas in the Russian Federation with high levels of radioactive soil contamination were abandoned in stages from 1986 until 1989, and in total 17 000 ha of agricultural land was excluded from economic use. 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 keep the highly contaminated areas in economic use, and hence most of the abandoned areas were subjected to intensive countermeasure application. However, these efforts were only partially successful, and the land was gradually abandoned; in the 1990s, the intensity of countermeasure application was reduced. Overall, about 11 000 ha was returned to agricultural use by 1995. These decisions to return land to agricultural use 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 governing the quality of agricultural products (TPL-93) [4.60].

Between 1995 and 2004 there was 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 to 3500 kBq/m^2, has been proposed by the 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 optimized. During the first planned stage, up to 2015, it is proposed to produce grain and potatoes using agricultural workers who live elsewhere but would come into the contaminated area as necessary. Soil based countermeasures (liming, potassium 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 is planned, and from 2025 the re-establishment of populated settlements could commence. Thus by 2045 all abandoned land could be used once more, although application of different countermeasures would be needed up to 2055 to ensure that annual doses to the local inhabitants were less than 1 mSv.

4.4. FOREST COUNTERMEASURES

Countermeasures for forested areas contaminated with radionuclides are only likely to be implemented if they can be accepted by foresters or landowners on a practical basis (i.e. actions are 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 cannot be implemented quickly and must be planned carefully. They are likely to be long term activities and their beneficial effects take time to be realized.

4.4.1. Studies on forest countermeasures

Generally, prior to the Chernobyl accident countermeasures to offset doses due to large scale contamination of forests had not been given very much attention. Several international projects in the 1990s gave rise to a number of publications in which suggestions and recommendations were made for possible countermeasures to be applied in forests [4.61–4.64]. However, in the three countries of the former USSR, actions had already been taken to restrict activities in the more contaminated zones, which included significant areas of forestry [4.65]. 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 the ideas developed by researchers will remain as theoretical possibilities rather than as methods that can be applied in real forests on a realistic scale. The following section describes some of the more feasible countermeasures that have been devised for forests contaminated with radiocaesium. This is illustrated in Section 4.4.3 by studies in which countermeasures have actually been put into practice.

4.4.2. Countermeasures for forests contaminated with radiocaesium

There are several categories of countermeasure that are, in principle, applicable to forest ecosystems [4.66, 4.67]. A selection of these is shown in Table 4.7. These can be broadly categorized into (a) management based and (b) technology based countermeasures.

4.4.2.1. Management based countermeasures

Under the broad heading of management based countermeasures, the principal remedial methods applied after the Chernobyl accident involved restrictions on 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 in the three independent countries [4.65]. These restrictions can be categorized as follows:

(a) Restricted access, including restrictions on public and forestworker access. This has been assisted by the provision of information from local monitoring programmes and education on issues such as food preparation [4.65].
(b) 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 former USSR, mushrooms are particularly important and can often be severely contaminated.
(c) 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 fertilizer.
(d) Alteration of hunting practices. The consumption of fungi by animals such as roe deer leads to strong seasonal trends in their body content of radiocaesium (see Section 3.3). Thus excessive exposures can be avoided by eating the meat only in seasons in which fungi are not available as a food source for the animals.
(e) Fire prevention is a fundamentally important part of forest management under any circumstances, but it is also important after a large scale deposition to avoid secondary contamination of the environment, which 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 3.3). One of the ways in which forest fires can be avoided is by minimizing human presence in the forest, so this countermeasure is strongly linked to restricting access, as described above.

4.4.2.2. Technology based countermeasures

This category of countermeasures includes the use of machinery and/or chemical treatments to alter the distribution or transfer of radiocaesium in forests. Many mechanical operations are carried out as part of normal forestry practice; examples of these have been described by Hubbard et al. [4.69] with reference to their use as countermeasures. Similarly, applications of fertilizers and pesticides may be made at different times in the forest

TABLE 4.7. SELECTED COUNTERMEASURES THAT HAVE BEEN CONSIDERED FOR APPLICATION IN CONTAMINATED FORESTS [4.68]

Countermeasure	Category	Caveat	Benefit	Cost
Normal operation	Management	—	No loss of productivity or amenity	No dose reduction and 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 and costs for hunting
Delayed cutting of mature trees	Management/ agrotechnical	Marginal feasibility	Reduced contamination of wood due to: — Radioactive decay — Fixation of caesium in soil — Loss from soil and wood	Delay in revenue
Early clear cutting and replanting or self-regeneration	Management/ agrotechnical	Must consider tree age at time of contamination; possibly in combination with soil mixing	Reduced tree contamination: — Lower soil–tree transfer — Delayed harvest time — Alternative tree crop	Higher dose to workers during replanting and operational costs
Soil improvement: harrowing after thinning or clear cutting	Agrotechnical	Cost effectiveness is dependent on the area to be treated; possibly in combination with fertilizer application	Improved tree growth, therefore growth dilution; dilutes radionuclide activity concentrations in the soil surface layer and decreases them in mushrooms, berries and understorey game	Operational costs, worker doses and environmental or ecological costs (e.g. nitrate and other nutrients lost)
Application of phosphorus–potassium fertilizer and/or liming	Agrotechnical	Phosphorus–potassium: may only be effective for caesium, especially effective for younger stands Lime: particularly useful for ^{90}Sr	Reduction of uptake to trees, herbs, etc., maybe better growth and dilution effect and higher fixation	Cost of fertilizer, worker dose and negative ecological effects
Limiting public access	Management	Note: people normally residing in forests not considered	Reduction in dose, possible increase in public confidence	Loss of amenity/social value, loss of food and negative social impacts
Salt licks	Agrotechnical	—	Reduction in caesium uptake by grazing animals	Continuing cost of providing licks
Ban on hunting	Management	—	Reduction in dose due to ingestion of game	Need to find alternative supply of meat
Ban on mushroom collection	Management	—	Reduction in internal dose	Need to find alternative mushroom supply

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 [4.68]. 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 are 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 fertilizers. 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 that involve technological intervention, application of chemicals or altering the harvesting patterns in forests are unlikely to be used in practice.

4.4.3. Examples of forest countermeasures

Case studies in which forest countermeasures, particularly technology 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 fertilizers, in particular, has been used with some success (see Section 4.3). In practice, restrictive countermeasures were applied in the USSR, and later in the three independent countries, as well as in a limited number of other countries, such as Sweden.

In the Bryansk region of the Russian Federation, individual restrictions on forestry and on the population living near forests were recommended according to the level of ^{137}Cs deposition. For forests receiving depositions 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 for collection of forest plants, was prohibited. In forests receiving depositions between 555 and 1480 kBq/m^2, collection of forest products was also prohibited, but limited forestry activities continued. At deposition levels between 185 and 555 kBq/m^2, harvesting of trees was continued on the basis of radiological surveys that 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 deposition levels less than 74 kBq/m^2.

One of the major effects of the restrictions that 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 region. 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 that inevitably lead to a disturbance of normal societal behaviour patterns. Furthermore, wood production is still under the official control of local forest authorities [4.65]; the currently applicable permissible levels for contamination of wood and forest products in the Russian Federation are shown in Table 4.8. Similar restrictions and permissible levels have been implemented in different regions of Belarus, notably the Gomel and Mogilev regions.

The use of caesium binders, particularly Prussian blue, in domestic animals has been one of the more effective techniques used to reduce doses from contaminated forests in the three countries of the former USSR. The principles underlying this method are described in Section 4.3; 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 five in milk and a factor of three in meat can be achieved at optimum dosage [4.65].

One example of intervention in normal forest related practices in countries outside the former USSR 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 in the Gävle area. The intervention level for such foodstuffs in Sweden is 1500 Bq/kg. 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; 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 [4.71].

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 [4.72].

4.5. AQUATIC COUNTERMEASURES

There are a number of different intervention measures that can be employed following fallout of radioactive material to reduce doses to the public via the surface water pathway. These actions may be grouped into two main categories: those aimed at reducing doses from radionuclides in drinking water and those aimed at reducing doses from the consumption of contaminated aquatic foodstuffs.

TABLE 4.8. TEMPORARY PERMISSIBLE LEVELS FOR CAESIUM-137 IN WOOD AND FOREST FOOD PRODUCTS IN THE RUSSIAN FEDERATION [4.70]

	TPL (Bq/kg)
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 use 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 material	7 400
Seeds of trees and bushes	7 400

In the context of the atmospheric fallout of radionuclides to both terrestrial and aquatic systems, it has been shown [4.73–4.75] that doses from terrestrial foodstuffs are in general much more significant than doses from drinking water and aquatic foodstuffs. However, in the Dnieper River system, the river water transported radionuclides to areas that were not significantly contaminated by atmospheric fallout. This created a significant stress in the population and a demand to reduce radionuclide fluxes from the affected 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 of workers implementing these countermeasures were high.

Measures to reduce doses via drinking water may, however, be required, particularly in the short term (timescale of weeks) after fallout, when activity concentrations in surface waters are relatively high. Owing to the importance of short lived radionuclides, early intervention measures, particularly the changing of supplies, can significantly reduce radiation doses to the population. Measures to reduce doses due to freshwater foodstuffs may be required over longer timescales as a result of the bioaccumulation of radionuclides in the aquatic food chain.

Reviews of aquatic countermeasures (e.g. Refs [4.76–4.79]) have considered both direct (restrictions) and indirect intervention measures to reduce doses:

(a) Restrictions on water use or changing to alternative supplies;
(b) Restrictions on fish consumption;
(c) Water flow control measures (e.g. dykes and drainage systems);
(d) Reduction of uptake by fish and aquatic foodstuffs from contaminated water;
(e) Preparation of fish prior to consumption.

There is no evidence that countermeasures were required, or applied, in marine systems after the Chernobyl accident.

4.5.1. 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 Kiev 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 Refs [4.76, 4.79].

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 [4.80, 4.81]. The average removal of these radionuclides from water (dissolved phase) was up to a factor of two.

After the accident, the upper gates of the Kiev 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 the inflowing rivers, which was believed to be highly contaminated. In fact, because of direct atmospheric deposition to the reservoir surface, the surface waters in the reservoir were much more contaminated than the deep waters. As noted by Voitsekhovitch et al. [4.80], "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 on 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 radioactive material downstream [4.82]. In the Dnieper River, the time it takes for water to travel from the Kiev reservoir to the Black Sea varies between three and ten months. Over the time that the water takes to travel downstream, radioactive pollution is reduced by decay of short lived radionuclides and transfers to reservoir bed sediments (particularly of radiocaesium) [4.82].

4.5.2. Measures to reduce direct and secondary contamination of surface waters

Standard antisoil erosion measures can be used to reduce runoff of radionuclides attached to 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 should also be 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 in the Pripyat River [4.79]. These canal bed traps were found to be highly inefficient for two reasons: (a) the flow rates were too high to trap the small suspended particles carrying much of the radionuclide contamination; and (b) a significant proportion of the radionuclide activity (and most of the 'available' activity) was in dissolved form 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 was adsorbed by these zeolite barriers [4.80]. In addition, the rivers and streams on which they were placed were later found to contribute only a few per cent to the total radionuclide load in the Pripyat–Dnieper system.

After the Chernobyl accident, spring flooding of the highly contaminated Pripyat floodplain resulted in increases in ^{90}Sr activity concentrations in the Pripyat River from annual average activity concentrations of around 1 Bq/L to a maximum of around 8 Bq/L for a flood event covering an approximately two week period [4.83]. In 1993 a dyke was constructed around the highly contaminated floodplain 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 [4.80]. A second dyke was constructed on the right bank of the Pripyat in 1999. The annual average ^{90}Sr activity concentration in Kiev reservoir

water, however, was below 1 Bq/L in all years from 1987 onwards. The radiological significance of the ^{90}Sr activity concentrations in Kiev reservoir water, even during the short flood events, is therefore very low, although 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 the introduction of a strongly sorbing material such as a zeolite or an (uncontaminated) mineral soil. This method has not been tested. Using a model for the removal of radiocaesium from lakes by settling of suspended particles, Smith et al. [4.78] identified two problems with this method: (a) large, deep lakes would require extremely large amounts of sorbent; and (b) secondary contamination of the lake by remobilization of activity from the catchment and/or bottom sediments would require repeat applications in most systems.

4.5.3. Measures to reduce uptake by fish and aquatic foodstuffs

Bans on the consumption of freshwater fish have been applied in the limited zones affected by the Chernobyl accident [4.84]. 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 [4.85]. Farmed fish could be used as an alternative source of freshwater fish in areas affected by fishing bans, since farmed fish fed with uncontaminated food do not accumulate radionuclides significantly [4.86].

The addition of lime to reduce radionuclide levels in fish was tested in 18 Swedish lakes [4.87]. The results of the experiments showed that liming had no significant effect on the uptake of ^{137}Cs in fish in comparison with control lakes. Although the uptake of ^{90}Sr was not studied in these experiments, it is expected that increased calcium concentration in lakes may have an effect on the ^{90}Sr concentration in fish. Experience of lake liming, in conjunction with artificial feeding of fish in Ukraine, has been summarized by Voitsekhovitch [4.79].

It is known that the concentration factor for 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 [4.87]. 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 et al. [4.88, 4.89] added potassium chloride fertilizer on to the frozen lake surface. Results showed a significant (factor of three) 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 two to three 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 that have water abstracted for drinking.

Manipulation of the aquatic food web by intensive fishing was carried out in four lakes in Sweden [4.87], and as a complementary measure in an additional three lakes. This resulted in a reduction of the fish population by about 5–10 kg/ha. The species reduced were mainly pike, perch and roach. No effect of intensive fishing on ^{137}Cs concentrations in fish was observed. Fertilization was carried out in two Swedish lakes using Osmocoat (5% phosphorus and 15% nitrogen). The concentrations of total phosphorus generally showed no change in the long term mean value: it appears that the fertilization treatment was not carried out 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 [4.90]. Ryabov suggested bans on the consumption of smoked and dried fish, because these processes increase concentrations of radionuclides (per unit of weight consumed) [4.84]. Other preparation processes may reduce radionuclide levels in fish by approximately a factor of two. An effective measure to reduce the consumption of radiostrontium is to remove the bony parts of fish prior to cooking, since strontium is mainly concentrated in the bones and skin. Various other food preparation methods are discussed in Ref. [4.91].

4.5.4. Countermeasures for groundwater

There is no evidence that measures have ever been taken 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 [4.77] has shown 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 radioactive waste sites in the CEZ. These measures 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 precipitation from entering the shelter, and drainage of rainwater collected in the bottom rooms of the shelter, have also to be considered as preventive measures to reduce groundwater contamination around the Chernobyl nuclear power plant industrial site.

4.5.5. Countermeasures for irrigation water

As discussed previously, irrigation did not add significantly to the radionuclide contamination of crops that had previously been affected by the atmospheric deposition of radionuclides. Thus, in practice, no countermeasures were directly applied to irrigation waters. However, the experience described in Ref. [4.79] shows that the change from sprinkling to drainage irrigation of agricultural plants (e.g. vegetables) can reduce the transfer of radionuclides from water to crops by several times. This, in combination with improved fertilization of irrigated lands, can effectively reduce radionuclide levels in crops irrigated with water from reservoirs affected by radioactive pollution.

4.6. CONCLUSIONS AND RECOMMENDATIONS

The Chernobyl accident prompted the introduction of an extensive set of short and long term environmental countermeasures by the authorities in the most affected countries to reduce its negative consequences. Unfortunately, there was not always openness and transparency towards the public, and information was withheld. This can, in part, explain some of the problems experienced later in communication with the public, and the mistrust of the competent authorities. Similar behaviour in many other countries outside Belarus, the Russian Federation and Ukraine led to a distrust in authority that, in many countries, has prompted investigations on how to deal with such major accidents in an open and transparent way and on how the affected people can be involved in decision making processes.

The unique experience of countermeasure application after the Chernobyl accident has already been widely used both at the national and international levels in order to improve preparedness against future nuclear and radiological emergencies [4.12, 4.14, 4.41, 4.91, 4.92].

4.6.1. Conclusions

(a) The Chernobyl accident prompted the introduction of an extensive set of short and long term environmental countermeasures by the USSR and, later, independent country authorities, aimed at reducing the accident's negative consequences. The countermeasures involved large amounts of human, economic and scientific resources.

(b) When social and economic factors along with the radiological factors are taken into account during the planning and application of countermeasures, better acceptability of these measures by the public is achieved.

(c) The unprecedented scale and long term consequences of the Chernobyl accident required the development of some additional national and international radiation safety standards, to take account of changes of radiation exposure conditions.

(d) 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.

(e) 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 implications.

(f) 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, which enabled most farming practices to continue in affected areas.

(g) Decontamination of settlements was widely applied in contaminated regions of the USSR during the first years after the Chernobyl accident as a means of reducing the external exposure of the public; this was cost effective with regard to external dose reduction when its planning and implementation were preceded by a remediation assessment based on cost–benefit considerations and external dosimetry data.

(h) The decontamination of urban environments has produced a considerable amount of low level radioactive waste, which creates a problem of disposal. However, secondary contamination of cleaned up plots has not been observed.

(i) 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 the use of forest products:

  (i) Restrictions on public and forest worker access as a countermeasure against external exposure.

  (ii) Restricted harvesting by the public of food products such as game, berries and mushrooms contributed to a reduction in internal dose. In the affected countries mushrooms are a common dietary component, and therefore this restriction has been particularly important.

  (iii) 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 fertilizer.

  (iv) Alteration of hunting practices, aimed at avoiding consumption of meat with high seasonal levels of radiocaesium.

  (v) Fire prevention, especially in areas with large scale radionuclide deposition, in order to avoid secondary contamination of the environment.

(j) Experience has shown that forest restrictions can 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 explain the purpose of the suggested changes in the use of some forest areas.

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

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

(m) The most effective countermeasure for aquatic pathways was the early restriction of drinking water abstraction and the change to alternative supplies. Restrictions on the consumption of freshwater fish have proved effective in Scandinavia and Germany; however, in Belarus, the Russian Federation and Ukraine such restrictions may not always have been adhered to.

(n) 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 the consumption of fish will be retained in a few cases (in closed lakes) for several more decades.

4.6.2. Recommendations

4.6.2.1. Countries affected by the Chernobyl accident

(a) Long term remediation measures and countermeasures in the areas contaminated with radionuclides should be applied if they are radiologically justified and optimized.

(b) Authorities and the general public should be particularly informed on radiation risk factors

and the technological possibilities to reduce them in the long term by means of remediation and countermeasures. Local authorities and the public should be involved in related discussions and decision making.

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

(d) Particular attention must be given to private farms in several hundred settlements and to about 50 intensive farms in Belarus, the Russian Federation and Ukraine, where radionuclide concentrations in milk still exceed the national action levels.

(e) Emphasis should be on the most efficient long term remediation measures; these are the radical improvement of pastures and grasslands and the draining of wet peaty areas. The most efficient regular agricultural countermeasures are the pre-slaughter clean feeding of animals accompanied by in vivo monitoring, the application of Prussian blue to cattle and the enhanced application of mineral fertilizers in plant cultivation.

(f) 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 the national action levels.

(g) Advice should continue to be given on individual diets, as a way of reducing consumption of highly contaminated wild food products, and on simple cooking procedures to remove radioactive caesium.

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

4.6.2.2. Worldwide

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

(a) Practically all the long term agricultural countermeasures implemented on a large scale on contaminated lands of the three most affected countries can be recommended for use in the event of future accidents. However, the effectiveness of soil based countermeasures varies at each site. Analysis of soil properties and agricultural practices before application is therefore of great importance.

(b) Recommendations on the decontamination of the urban environment in the event of large scale radioactive contamination should be distributed to the owners and operators of nuclear facilities that have the potential for substantial accidental radioactive release (nuclear power plants and reprocessing plants) and to authorities in adjacent regions.

4.6.2.3. Research

(a) Generally, the physical and chemical processes involved in environmental countermeasures and remediation technologies, both of a mechanical nature (radionuclide removal, mixing with soil, etc.) and a chemical nature (soil liming, fertilization, etc.), are understood sufficiently to be modelled and applied in similar circumstances worldwide. Much less understood are the biological processes that could be used in environmental remediation (e.g. reprofiling of agricultural production, bioremediation, etc.). These processes require more research.

(b) An important issue that requires more sociological research is the perception by the public of the introduction, performance and withdrawal of countermeasures in the event of an emergency, as well as the development of social measures aimed at involving the public in these processes at all stages, beginning with the decision making process.

(c) There is still substantial diversity in the international and national radiological criteria and safety standards applicable to the remediation of areas affected by environmental contamination with radionuclides. The experience of radiological protection of the public after the Chernobyl accident has clearly shown the need for further international harmonization of appropriate radiological criteria and safety standards.

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5. HUMAN EXPOSURE LEVELS

5.1. INTRODUCTION

5.1.1. Populations and areas of concern

Following the Chernobyl accident, both workers and the general public were affected by radiation that carried a risk of adverse health effects. UNSCEAR selected the following three categories of exposed populations: (a) workers involved in the accident, either during the emergency period or during the cleanup phase; (b) the inhabitants of contaminated areas who were evacuated in 1986; and (c) the inhabitants of contaminated areas who were not evacuated [5.1].

In this section consideration is given primarily to members of the general public exposed to radionuclides deposited in the environment. The workers 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 nuclear power plant and in the CEZ) are not considered here. For information on Chernobyl worker populations, the reader is referred to the comprehensive material provided by UNSCEAR [5.1, 5.2] and by the Chernobyl Forum in its report considering the human health effects [5.3].

Information on the radiation doses received by members of the general public, both those evacuated from the accident area and those who live permanently in contaminated areas, is required for the following health related purposes:

(a) Substantiation of countermeasures and remediation programmes;
(b) Forecast of expected adverse health effects and justification of corresponding health protection measures;
(c) Information for the public and the authorities;
(d) Epidemiological and other medical studies of radiation caused adverse health effects.

In this section the methodologies and data specifically required for the estimation of mean doses to population groups living in particular settlements and selected by the factors influencing either external or internal dose or both are presented. These factors are usually age, sex, occupation, food habits, etc. Dose distributions among group members and collective doses are also considered. Individual doses to members of the public, used mainly in analytical epidemiological studies, are presented in the Chernobyl Forum report on the health consequences of the Chernobyl accident [5.3]. On these subjects, substantial progress has been achieved since publication of the comprehensive UNSCEAR report in 2000 [5.1].

As mentioned in Section 3.1, atlases have been prepared that show the deposition of ^{137}Cs and other radionuclides throughout the former USSR and other countries of Europe [5.4, 5.5]. These indicate that the most affected countries are Belarus, the Russian Federation and Ukraine. In addition, the countries of Austria, Bulgaria, Finland, Greece, Italy, Norway, Republic of Moldova, Slovenia, Sweden and Switzerland 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 3.2).

5.1.2. Exposure pathways

Following the Chernobyl accident there were several pathways by which humans were exposed to radioactive material (Fig. 5.1). The main pathways are listed below in the approximate time sequence in which the doses were received:

(a) External dose from cloud passage;
(b) Internal dose from inhalation during cloud passage and of resuspended material;
(c) External dose from radionuclides deposited upon soil and other surfaces;

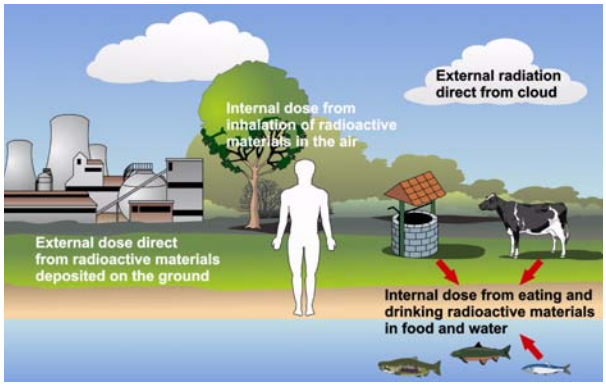

FIG. 5.1. Pathways of exposure of humans to environmental releases of radioactive material.

(d) Internal dose from the consumption of contaminated food and water.

Under most exposure conditions for members of the general public the two most important pathways are dose from radiation from the decay of radionuclides deposited upon the soil and other surfaces and dose from the ingestion of contaminated food and water. If persons are evacuated quickly after passage of the initial cloud, then the most important pathways are the first two in the list, because the latter two pathways have been prevented.

5.1.3. Concepts of dose

Methods of calculating radiation dose have been refined over the years, and specific concepts have evolved [5.1, 5.6]. The fundamental measure of radiation dose to an organ or tissue is the 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. Since this is a rather large amount of dose, it is common to use units of mGy (one thousandth of a gray) or μGy (one millionth of a gray).

Since many organs and tissues were exposed as a result of the Chernobyl accident, it has been very common to use an additional concept, 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 ionizations 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. Where many individuals have been exposed to an event, such as happened following the Chernobyl accident, an additional concept, the collective dose, can be used. The collective dose is the sum of the doses to all individuals within a particular group, which may 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 the collective effective dose is the man-Sv.

Finally, UNSCEAR has employed the concept of dose commitment to examine the long term consequences of a practice or accident [5.1]; 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 material. This is true even though it will take many years for the doses to be received by the persons alive at that time and by persons not yet born or conceived.

5.1.4. 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 5.1 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 5.2 shows the annual individual effective

TABLE 5.1. RADIATION DOSES FROM NATURAL SOURCES [5.1]

	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

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 areas of Belarus, the Russian Federation and Ukraine.

5.1.5. Decrease of dose rate with time

To calculate the radiation 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 the radioactive decay of the radionuclides. Additional DRRFs 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, which results in increased absorption of the emitted radiations within the soil. Typically, a two component exponential function describes this process [5.7, 5.8].

The availability of ^{137}Cs for ingestion also decreases with time at a rate faster than radioactive decay. This additional long term decrease is due mainly to the adsorption 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 [5.9, 5.10].

5.1.6. Critical groups

In all situations that involve the exposure of large segments of the population to natural or human-made radioactive material, there is always a significant spread in the radiation dose received by various members of the population living within the same geographical 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 may 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; also, 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 is often 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 radiocaesium, 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.

TABLE 5.2. EFFECTIVE DOSES IN 2000 FROM NATURAL AND HUMAN SOURCES [5.1]

	Worldwide average annual per caput effective dose (mSv)	Range or trend in exposure
Natural background	2.4	Typical range of 1–10 mSv
Diagnostic medical examinations	0.4	Ranges from 0.04 to 1 mSv at the 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 the northern hemisphere
Chernobyl accident	0.002	Has decreased from a maximum of 0.04 mSv in 1986 (in the northern hemisphere); higher at locations nearer the accident site
Nuclear power production	0.0002	Has increased with expansion of the nuclear programme but decreased with improved practice

5.2. EXTERNAL EXPOSURE

5.2.1. Formulation of the model for 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 doses:

(a) Parameters that describe the external gamma radiation field;
(b) Parameters describing human behaviour in this field;
(c) Conversion factors from dose in air to organ or effective dose.

The basic model for human external exposure in the event 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 a 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 and their radioactive decay, vertical migration of long lived radionuclides and the presence of snow cover.

Radiation exposure is influenced by altered or disturbed environments. In models this factor is taken into account by using location factors. The location factor LF_i is 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 [5.11]. Human behaviour in the radiation field is described by occupancy factors OF_{ik}, which represent the fraction of time spent by individuals of the kth population group at the ith point of the settlement of interest. The third type of data necessary for assessment of the effective external dose are conversion factors CF_k, which convert measured values (the absorbed dose in air) to a parameter that can be directly related to health effects — the effective dose to the kth population group.

On this basis, a deterministic model for the assessment of the effective external dose rate E_k for representatives of the kth population group is represented in Fig. 5.2.

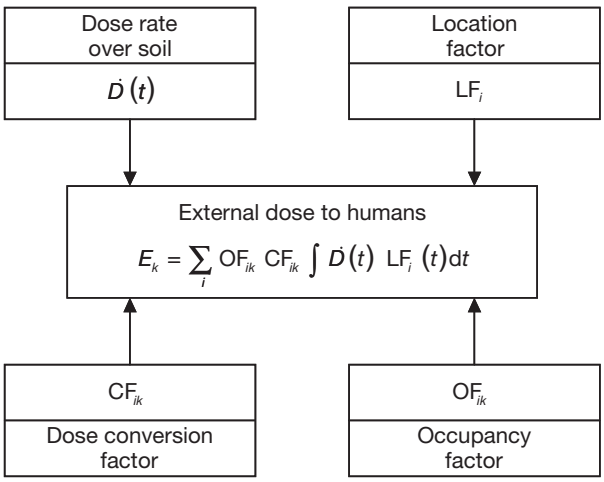

FIG. 5.2. Model of external exposure of the kth population group (i is a location index) [5.9].

5.2.2. Input data for the estimation of effective external dose

Numeric values for the parameters listed above have been determined from long term dosimetric investigations in the most highly contaminated regions after the Chernobyl accident.

5.2.2.1. 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 nuclear power plant boundaries the initial dose rate over lawns and meadows ranged between 3 and 10 μGy/h in areas contaminated at about 37 kBq/m^2 (1 Ci/km^2) of ^{137}Cs and up to 10 000 μGy/h within the CEZ with higher deposition levels. Exposure rates decreased rapidly, due to the radioactive decay of short lived radionuclides, as shown in Fig. 5.3.

Owing to different isotopic compositions of radionuclide fallout in different geographical areas [5.8, 5.13, 5.14], the contribution of short lived radionuclides to the overall dose rate was highly variable. In the CEZ, ^{132}Te + ^{132}I, ^{131}I and ^{140}Ba + ^{140}La dominated during the first month and then ^{95}Zr + ^{95}Nb for another half year before ^{137}Cs and ^{134}Cs became dominant (Fig. 5.4). In contrast, in the far zone only the radioiodine isotopes dominated during the first month; afterwards ^{137}Cs and ^{134}Cs dominated, with a moderate contribution from ^{103}Ru and ^{106}Ru (Fig. 5.5). Since 1987 more than

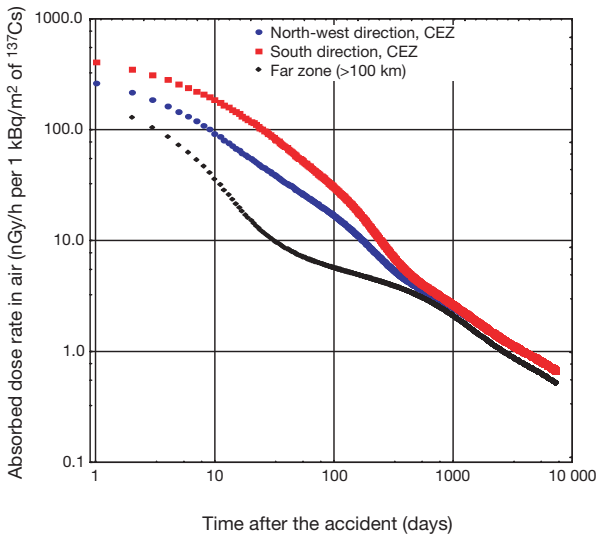

FIG. 5.3. Dynamics of standardized dose rate in air over undisturbed soil after the Chernobyl accident in different geographical areas [5.12].

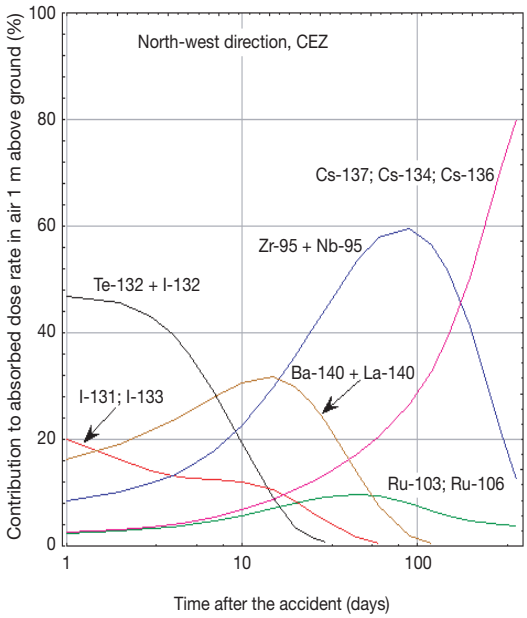

FIG. 5.4. Relative contribution of gamma radiation from individual radionuclides to the external gamma dose rate in air during the first year after the Chernobyl accident (north-west direction, CEZ) [5.12].

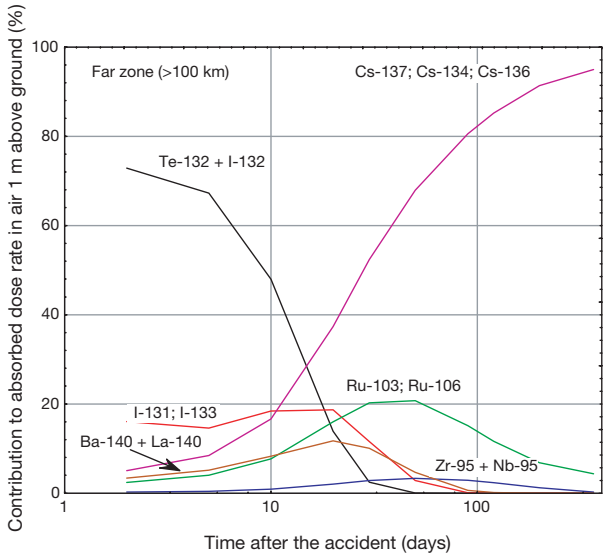

FIG. 5.5. Relative contribution of gamma radiation from individual radionuclides to the external gamma dose rate in air during the first year after the Chernobyl accident (far zone — more than 100 km from the Chernobyl nuclear power plant) [5.12].

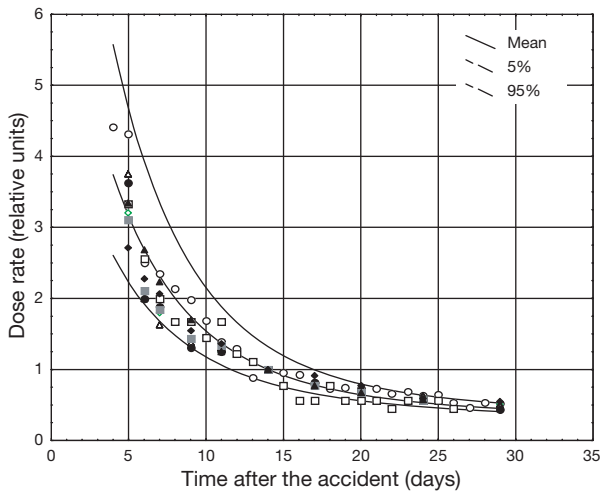

FIG. 5.6. Dose rate in air during the first days after the accident in several rural settlements in the Bryansk and Tula regions of the Russian Federation (normalized to the dose rate on 10 May 1986). Points indicate dose rate measurements and curves represent calculated values according to the isotopic composition [5.7].

90% of the dose rate in air has come from the gamma radiation of long lived ^{137}Cs and ^{134}Cs. Thus the radionuclide composition of the deposited activity was a major factor in determining the external exposure of the population in the early period of time after the accident. Model estimates of the 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. 5.6).

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), the Russian

Federation, Sweden and Ukraine [5.7, 5.8, 5.15]. 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 [5.16], and in Bavaria (Germany), where contamination was due to global fallout. The last two data sets were obtained 20 to 30 years after deposition; this allows for long term predictions to be applied to the Chernobyl depositions. The measurement sites were considered to be representative of reference sites (i.e. open, undisturbed fields).

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. 5.3). At that time, the dose rate was mainly determined by gamma radiation of caesium radionuclides (i.e. ^{137}Cs (half-life 30 years) and ^{134}Cs (half-life 2.1 years), and later, one decade and more after the accident, mainly 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. [5.7] and Likhtarev et al. [5.8] 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. 5.7. The latter value is rather uncertain. It corresponds to an effective half-life that takes into account both the radioactive decay of ^{137}Cs and its gradual deepening in soil after 17–19 years.

5.2.2.2. 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 to people from deposited activity. These differences are attributable to varying source distributions as a result of deposition, runoff, weathering and shielding. All such effects can be summarized by the term 'location factors'.

Location factors for typical western European buildings have been assessed [5.11, 5.17, 5.18]. Gamma spectrometric measurements performed in Germany and Sweden [5.19–5.22] allowed the determination of location factors in urban environments and their variation with time over 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 Belarus, the Russian Federation and Ukraine began two to three years after the accident. The results of one such later investigation in Novozybkov (in the Bryansk region of the Russian Federation) are presented in Fig. 3.12 (Section 3).

5.2.2.3. 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 have 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 5.3, where values of occupancy factors for the summer period are presented for different groups of the rural populations of Belarus, the Russian Federation and Ukraine [5.15].

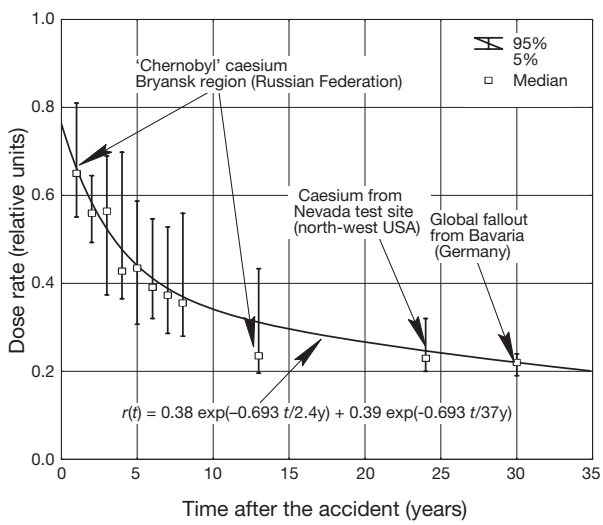

FIG. 5.7. Reduction of the ^{137}Cs gamma dose rate in air due to caesium migration in undisturbed soil relative to the dose rate caused by a plane source on the air–soil interface (from Ref. [5.7]).

5.2.2.4. 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 [5.15] and Monte Carlo calculations [5.23]. The values were 0.75 Sv/Gy for adults, 0.80 Sv/Gy for schoolchildren (7–17 years) and 0.90 Sv/Gy for pre-school children (0–7 years). For the calculation of effective doses, conversion factors CF_k were used that are independent of the location and time after the accident.

5.2.3. Results

5.2.3.1. Dynamics of external effective dose

Shortly after the deposition of the fallout the gamma radiation field was dominated by emissions from short lived radionuclides, as discussed above (see Figs 5.4 and 5.5). As the mixtures at different locations varied widely, the radionuclide composition of the deposited activity was a major factor in determining the external exposure of the population during the early period after the accident.

Another relevant parameter in the midterm 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 decreased during the first years after deposition, with an exponential half-life of one to two years (see Fig. 3.12). In the five to seven 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 30, mainly due to radioactive decay of short lived radionuclides (see Fig. 5.8). During the following decade the external dose rate decreased because of the radioactive decay of ^{134}Cs and ^{137}Cs and the migration of radiocaesium into the soil. Afterwards, the external dose rate was mainly due to ^{137}Cs. In the long term, radiocaesium becomes 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. 5.8) [5.7].

5.2.3.2. Measurement of individual external dose with thermoluminescent dosimeters

In general, before the Chernobyl accident, individual external doses were measured only for occupational exposures. After the Chernobyl accident, individual external doses to members of the population were also measured. For this purpose thermoluminescent dosimeters were distributed to the inhabitants of the more contaminated areas of Belarus, the Russian Federation and Ukraine [5.24–5.28]. Inhabitants wore thermoluminescent dosimeters for about one month in the spring and summer periods. Examples of such results are presented in Figs 5.9 and 5.10 for rural and urban areas, respectively. According to these results it can be concluded that the urban

TABLE 5.3. VALUES OF OCCUPANCY FACTORS FOR THE SUMMER PERIOD FOR DIFFERENT GROUPS OF THE RURAL POPULATIONS OF THE RUSSIAN FEDERATION, BELARUS AND UKRAINE[a] [5.15]

Location	Indoor workers	Outdoor workers	Pensioners	Schoolchildren	Pre-school 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 settlements	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

[a] The first number corresponds to data for the Russian Federation, the second is for Belarus and the third is for Ukraine [5.15].

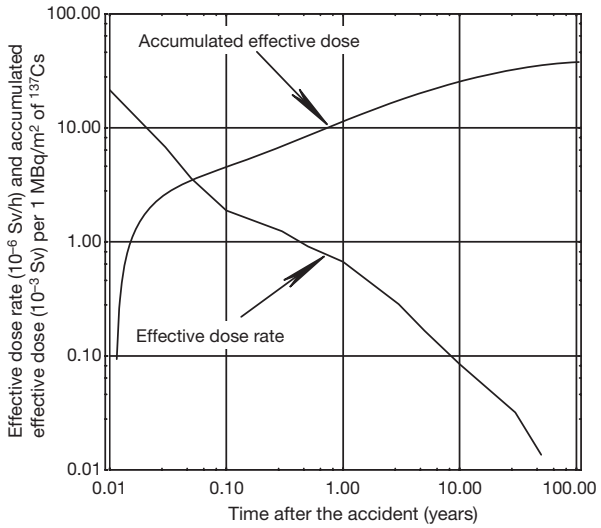

FIG. 5.8. Model prediction of the time dependence of the external effective gamma dose rate and the accumulated external effective dose to the urban population of the Bryansk region of the Russian Federation [5.7].

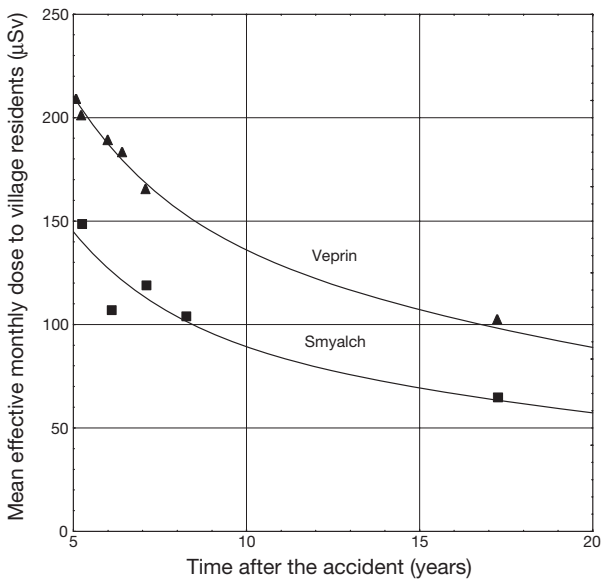

FIG. 5.9. Results of thermoluminescence measurements of mean monthly doses among inhabitants living in wooden houses in the Veprin and Smyalch villages (Bryansk region of the Russian Federation) in different time periods following deposition [5.28].

population has been exposed to a lower dose by a factor of 1.5–2 compared with 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.

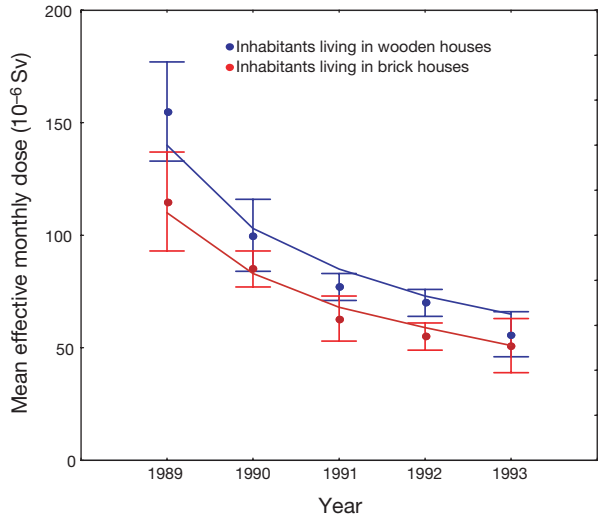

FIG. 5.10. Effective dose rates for indoor workers in Novozybkov (Russian Federation). The points with error bars represent average values and 95% confidence intervals (+/– two standard errors) of thermoluminescence measurements [5.28].

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. At present, the average external dose to 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), herders (factor 1.6) and field crop workers (factor 1.3) living in one-storey wooden houses [5.9, 5.15].

Analysis of the results of measurements of inhabitants of settlements showed that the distribution of individual doses can be described by a log-normal function [5.7]. Figure 5.11 presents a comparison of model calculations with individual thermoluminescence measurements performed in 1993 in four villages of the Bryansk region (565 measurements). The distributions of the ratio of individual external doses to the mean value of measured doses in each of the villages are 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.

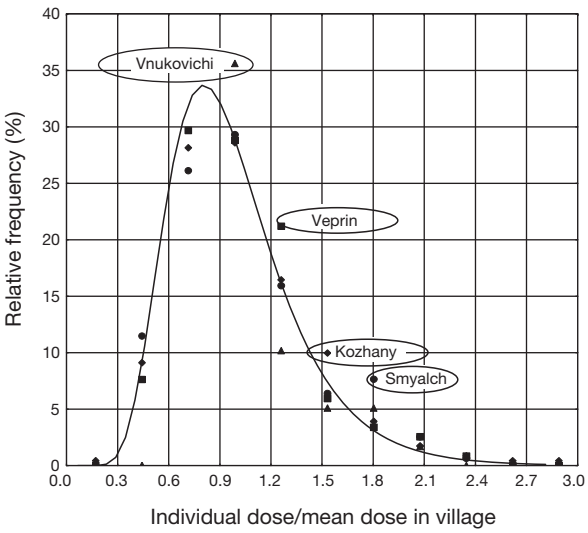

FIG. 5.11. Frequency distributions of monthly effective external doses to individual persons as measured in the summer of 1993 with thermoluminescent dosimeters in four villages of the Bryansk region of the Russian Federation (points) and calculated by the stochastic model (curve). Doses are normalized to the arithmetic mean of the individual doses determined for each of the villages (from Ref. [5.7]).

5.2.3.3. Levels of external exposure

To illustrate actual levels of external exposure and differences in the level of exposure among various population groups, Table 5.4 presents calculated values of effective external doses during different time intervals for rural and urban populations in the Russian Federation and Ukraine, and Table 5.5 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.

At present, the average annual external dose to residents of a rural settlement with a current ^{137}Cs soil deposition of ~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 under normal conditions. The external dose due to Chernobyl deposition accumulated to the present time is 70–75% of the total lifetime dose (70 years) for persons born in 1986 and living all the time in contaminated areas.

TABLE 5.4. AVERAGE NORMALIZED EFFECTIVE EXTERNAL DOSE TO THE ADULT POPULATION IN THE INTERMEDIATE (100 km < DISTANCE < 1000 km) ZONE OF CHERNOBYL CONTAMINATION

	Population	E/σ_{137} (μSv · kBq^{-1} · m^{-2} of ^{137}Cs)[a]				
		1986	1987–1995	1996–2005	2006–2056	1986–2056
Russian Federation	Rural	14	25	10	19	68
[5.7, 5.28]	Urban	9	14	5	9	37
Ukraine [5.8]	Rural	24	36	13	14	88
	Urban	17	25	9	10	61

[a] σ_{137} is given as for 1986.

TABLE 5.5. RATIO OF AVERAGE EFFECTIVE EXTERNAL DOSES IN SEPARATE POPULATION GROUPS TO THE MEAN DOSE IN A SETTLEMENT [5.9]

Type of dwelling	Indoor workers	Outdoor workers	Herders, foresters	Schoolchildren
Wooden	0.8	1.2	1.7	0.8
One to two storey, brick	0.7	1.0	1.5	0.9
Multistorey	0.6	0.8	1.3	0.7

5.3. INTERNAL DOSE

5.3.1. Model for internal dose

The general form of models used to calculate internal dose is shown in Fig. 5.12 [5.9]. 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}.

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 [5.2, 5.10] or from special surveys of the affected populations [5.29, 5.30]. Other data needed for dosimetric calculations are taken from publications of the International Commission on Radiological Protection for age specific inhalation rates [5.31] and for age specific dose coefficients [5.32]. 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.

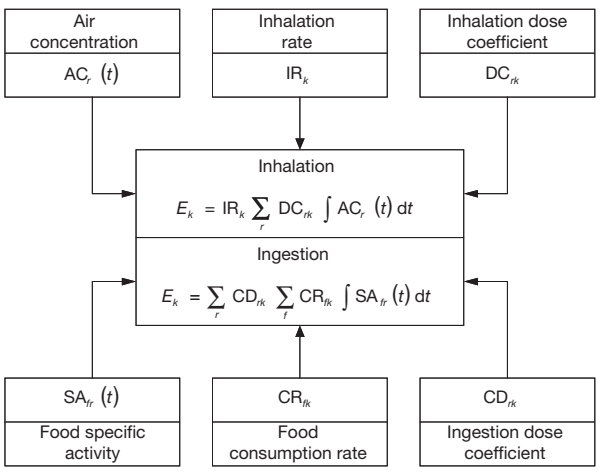

FIG. 5.12. Model for calculation of internal exposure for persons exposed to Chernobyl fallout [5.9].

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 [5.33–5.35] and for ^{137}Cs (e.g. Refs [5.10, 5.36]). Especially for the thyroid, direct measurements are not sufficient 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 into the body must be made in order 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 [5.1]. Also, enough time has now passed since the Chernobyl accident 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. [5.10] 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 with a half-life of 2.9 ± 0.3 years and 10% with 15 ± 7.6 years. The second value is very uncertain, due to the short time of observation compared with the radiological half-life of ^{137}Cs of 30 years. These data are in general agreement with those observed in the Russian Federation [5.37, 5.38].

5.3.2. Monitoring data as input for the assessment of internal dose

A unique feature of Chernobyl related monitoring of human internal exposure was the extensive application of whole body measurements of radionuclide content in the human body and its organs (mainly thyroid); these measurements were performed along with regular measurements of radionuclides in food, drinking water and other components of the environment. This combination of various kinds of monitoring data allowed substantial improvement in the precision of the 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 the concentration of radionuclides in air was relatively high. Later, the assessment of doses via inhalation was needed in relation to 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 the 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, the Russian Federation and Ukraine). Gamma spectroscopy for ^{131}I and ^{134,137}Cs and radiochemical analyses for ^{90}Sr have been the main types of measurement. In some laboratories beta spectroscopy was successfully applied to determine different radionuclides in samples; when radionuclide composition was well known, total beta activity measurements were also made. In most of the measurements, ^{137}Cs in raw animal products (milk, meat, etc.) was determined; the number of these measurements performed since 1986 and available for dose estimation comprises a few million. Generic data on radionuclide measurements in food are presented in the sections pertaining to the terrestrial environment.

Activity concentrations of soluble radionuclides (mainly ^{131}I, ^{134,137}Cs, ^{90}Sr) in drinking water were determined in 1986 both in surface and underground sources (see Section 3.5). Later, these activity concentrations declined to relatively low levels, and their contribution to internal dose was usually negligible compared with that associated with the intake of food.

In May–June 1986, ^{131}I activities were measured in the 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 most 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, data on large scale measurements were used as the main basis for the reconstruction of thyroid dose.

Most of the 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 which were performed in the three most affected countries. The measurement data were widely used both for model validation concerning radionuclide intake and evaluation of the effectiveness of countermeasures. In the most contaminated regions of Belarus, the Russian Federation and Ukraine, whole body measurement data were used to obtain 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 that is readily detectable by whole body counters, have been measured in excreta samples, and, since the 1990s, in samples taken at autopsy. Several hundred samples of human bone tissue have been 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 [5.39, 5.40].

Reduced monitoring programmes for radiation protection purposes, and specifically for the justification of remediation efforts, are continuing in the affected areas.

5.3.3. Avoidance of dose by human behaviour

In addition to the countermeasures 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 doses 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 [5.10, 5.41]. 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 gradual reduction of the public's caution over wild food products, such self-imposed restrictions became less widespread.

5.3.4. Results for doses to individuals

5.3.4.1. 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 [5.2]. Attention has been paid to thyroid dose reconstruction since the early 1990s, when an increase of thyroid cancer morbidity was discovered in children and adolescents residing in areas of Belarus, the Russian Federation and Ukraine contaminated with Chernobyl fallout [5.1, 5.3, 5.42].

In association with radioepidemiological studies, the main patterns of thyroid dose formation were clarified and published in the 1990s [5.33–5.35] and summarized in Ref. [5.1]. Nevertheless, important new work in this area has appeared recently [5.43–5.45]. The general approach to internal dose reconstruction has been elaborated in Ref. [5.46].

Methodologies of thyroid dose reconstruction for the Chernobyl affected populations developed in parallel in Belarus, the Russian Federation and Ukraine, with the participation of 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 many tens of thousands of ^{131}I thyroid measurements available, although of different quality, that are used as the basic data for thyroid dose reconstruction. In the Russian Federation, additionally, data on ^{131}I in milk were used. Owing to the use of human and environmental ^{131}I measurements, the reconstructed doses are realistic rather than 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 used for dose reconstruction. The subsidiary quantities used for dose reconstruction in the settlements where historical ^{131}I measurements are not available are ^{137}Cs soil deposition values as indicators of radioactive contamination in the area.

However, the methodologies for 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. [5.45] developed a model with linear dependence of thyroid dose on ^{137}Cs soil deposition. In Belarus, where both dry and wet deposition of radioiodines occurred, a semiempirical model based on non-linear dependence of thyroid dose on ^{137}Cs soil deposition was developed by Gavrilin et al. [5.35] and widely applied. In another recently published paper devoted to the same problem, a comprehensive radioecological model of radioiodine environmental transfer was developed and successfully applied for thyroid dose reconstruction [5.44]. In the Russian Federation, where wet deposition of radioiodines dominated, a linear semiempirical model of dependence of ^{131}I activity concentration in milk and of thyroid dose on ^{137}Cs soil deposition of more than 37 kBq/m^2 was developed [5.43] and applied [5.47]. Despite differences in the applied methodological approaches, the general agreement, except for areas of low contamination, is satisfactory [5.3].

The thyroid doses resulting from the Chernobyl accident comprise four contributions: (a) internal dose from intakes of ^{131}I; (b) 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); (c) external dose from the deposition of radionuclides on the ground; and (d) 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 a lesser extent, of green vegetables; children on average received a dose that was much higher 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.

An example of age and sex dependence of the mean thyroid dose to inhabitants of a settlement, based on 60 000 measurements of ^{131}I in thyroids performed in Ukraine in May 1986, is presented in Fig. 5.13 [5.48]. The mean thyroid dose to infants is larger by a factor of about seven than to young adults (19–30 years) residing in the same rural or urban settlements; this ratio decreases monotonically with age as an exponential function, with some deviation in adolescents. Differences in age

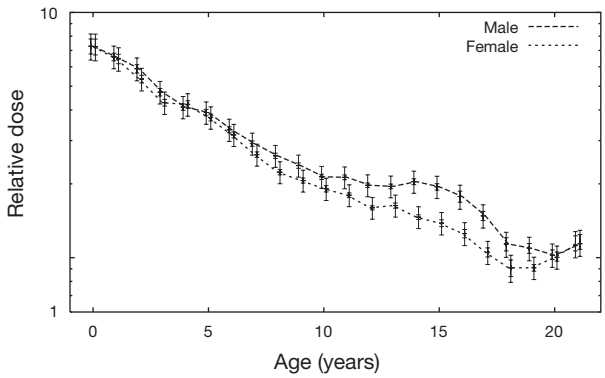

FIG. 5.13. Age–sex dependence of the mean thyroid dose to inhabitants of a settlement standardized to the mean dose to adults from the same settlement [5.48].

dependence between males and females seem to be insignificant. Similar patterns were revealed both from Belarusian and Russian measurements of ^{131}I in the thyroid [5.34, 5.35].

As the rural population living in contaminated areas depends more on local agricultural production than does an urban population, thyroid doses caused predominantly by the consumption of contaminated milk and dairy products are higher in rural than in urban populations by a factor of about two [5.1].

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 (^{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 was exposed to radioiodines via inhalation only and was evacuated about 1.5 days after the accident. Analysis of direct thyroid and lung spectrometric measurements performed on 65 Pripyat evacuees has shown that the contribution of short lived radionuclides to thyroid dose 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 [5.49]. 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 [5.49, 5.50].

The distribution of individual thyroid doses is illustrated in Table 5.6 for children and adolescents residing in the northern regions of Ukraine (i.e. the Kiev, Zhytomyr and Chernigov regions) most affected by radiation after the Chernobyl accident [5.45]. The dose distributions presented in Table 5.6 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 five to nine years, and less than 0.01% of adolescents. Doses to adults are lower by a factor of about 1.5 than those to adolescents (see Fig. 5.13). In all age groups presented in Table 5.6, and especially in the younger ones, doses were high enough to cause both short term functional thyroid changes and thyroid cancer in some individuals [5.1, 5.3, 5.42].

Similar data for Belarus and the Russian Federation are available [5.35, 5.47]. Substantially more detail on the calculation of thyroid doses to individuals is provided in the dosimetry section of the Chernobyl Forum report on health effects [5.3].

Generally, it can be stated that adequate methodologies for thyroid dose reconstruction for people who resided in the spring of 1986 in the contaminated areas of Belarus, the Russian Federation and Ukraine have been developed and published. These estimates of both individual and collective doses are being widely used by research scientists and national health authorities both in forecasts of thyroid morbidity and in radioepidemiological studies.

5.3.4.2. 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 [5.39], inhalation of plutonium radionuclides and ^{241}Am does not significantly contribute to human dose in this context.

Generic dose conversion parameters have been developed to reconstruct the past, assess the current and forecast future average effective internal doses. 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 the Russian Federation and Ukraine are given in Table 5.7 [5.9, 5.10, 5.15]. 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}) ($\mu Sv \cdot kBq^{-1} \cdot m^{-2}$).

In a series of experimental whole body measurements and associated annual internal dose calculations it was found that long term doses to children caused by ingestion of food containing caesium radionuclides are usually lower by a factor of about 1.1 to 1.5 than those to adults and adolescents (see, for example, Refs [5.51, 5.52]).

The mean internal doses to residents of rural settlements strongly depend on soil properties. For assessment purposes, soils are classified into three

TABLE 5.6. DISTRIBUTION OF INDIVIDUAL THYROID DOSES FOR AGE GROUPS OF CHILDREN AND ADOLESCENTS FROM THE KIEV, ZHYTOMYR AND CHERNIGOV REGIONS OF UKRAINE, BASED ON IODINE-131 IN THYROID MEASUREMENTS [5.45]

Category and age group	Number of measurements	Per cent 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	9 119	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	5 147	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	1 475	30	45	22	2.7	1.0
5–9 years	2 432	55	36	8.4	0.6	0.08
10–18 years	4 732	73	23	3.6	0.13	0.02

TABLE 5.7. RECONSTRUCTION AND PROGNOSIS OF THE AVERAGE EFFECTIVE INTERNAL DOSE TO THE ADULT RURAL POPULATION IN THE INTERMEDIATE (100 km < DISTANCE < 1000 km) ZONE OF CHERNOBYL CONTAMINATION

	Soil type	E/σ_{137} ($\mu Sv \cdot kBq^{-1} \cdot m^{-2}$ of ^{137}Cs)[a]				
		1986	1987–1995	1996–2005	2006–2056	1986–2056
Russian Federation [5.9]	Soddy podzolic sandy	90	60	12	16	180
	Black	10	5	1	1	17
Ukraine [5.10, 5.15]	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

[a] σ_{137} is given as for 1986.

major soil types: (a) black or chernozem soil; (b) podzol soil (including both podzol sandy and podzol loam soils); and (c) peat bog or peat soil. Due to the environmental behaviour of ^{137}Cs, internal exposure exceeds external dose in areas with peaty soil. Contributions due to internal and external exposure are comparable in areas with light sandy soil, and the contribution of internal exposure to the total (external and internal) dose does not exceed 10% in areas with dominantly black soil. 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 5.7). Some of these discrepancies can be explained by the different meteorological conditions (mainly dry deposition in Ukraine and wet deposition in the Russian Federation) that occurred in different parts of the Chernobyl affected areas and by different food consumption habits.

Multiplication of the parameters presented in Table 5.7 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 the Russian Federation, also from ^{90}Sr and ^{89}Sr) but not from radioiodines. Dose estimates are given on the assumption that countermeasures against internal exposure were not applied. In broad terms, the most 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 with settlement average values varies by a factor of about three for internal exposure. The group most subjected to internal exposure from ^{137}Cs is adults consuming both locally produced agricultural animal foods (e.g. milk, dairy products, etc.) and natural foods (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 up to 0.004 mSv/a from ingestion of local foods in black soil areas, up to 0.04 mSv/a in sandy soil areas and about 0.1 mSv/a in villages located in peaty soil areas. In the period 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 1–2 mSv in villages located in peaty soil areas.

To avoid the 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 5.8 and 5.9. The ^{137}Cs soil deposition is subdivided into two ranges: 0.04–0.6 MBq/m^2 (1–15 Ci/km^2) and above 0.6 MBq/m^2 (i.e. 0.6–4 MBq/m^2 (15–100 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

TABLE 5.8. PAST (1986–2000) AND FUTURE (2001–2056) MEAN CHERNOBYL RELATED EFFECTIVE INTERNAL DOSES (mSv) TO ADULT RESIDENTS OF AREAS WITH CAESIUM-137 SOIL DEPOSITION ABOVE 0.04 MBq/m^2 (1 Ci/km^2) IN 1986 [5.53]

Population	Caesium-137 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

0.6 MBq/m^2, application of active countermeasures (i.e. agricultural restrictions, decontamination measures, recommendations to restrict consumption of locally gathered natural foods (forest mushrooms and berries, lake fish, etc.)) has been mandatory.

Dosimetric models 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 (see Table 5.8). In 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 soil and up to 30 mSv for peat soil). In the more contaminated podzol soil areas, an effective dose of up to 50 mSv can still be expected.

As can be seen from Table 5.9, the more elevated internal doses in some of the settlements are above the national action level of 1 mSv/a. For some population groups in contaminated areas, wild foods (forest mushrooms, game, forest berries, fish) can make an important contribution to dose [5.9, 5.15, 5.30]. Studies of ^{137}Cs intake of the rural population in the Bryansk region of the Russian Federation indicated that natural foods contributed about 20% of total uptake in 1987, but up to 80% in 1994–1999 [5.29]. The relative contribution of wild foods to internal dose has risen gradually because of the substantial reduction of radionuclide content in agricultural foods derived from vegetables and animals, combined with a much slower decrease in the contamination of wild foods. In the latter period, the highest contributions to ^{137}Cs intake (and, by inference, internal dose) came from forest mushrooms, followed by forest berries, game and lake fish.

Similar trends were found in residents of Kozhany (Bryansk region), located on the coast of a highly contaminated lake, where natural foods contributed an average of 50–80% of ^{137}Cs intake [5.30]. Men were more likely to eat natural foods than women, and there was a positive correlation between consumption of mushrooms and fish that indicated a liking by 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.

5.3.4.3. 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 related exposures was attributed to aquatic pathways [5.53]. Although these doses were, in fact, estimated to be very low, there was an inadequate appreciation by the local population of the risks of using water from contaminated aquatic systems. This created an (unexpected) stress in the population concerning the safety of the water supply 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 [5.53]:

(a) Consumption of drinking water from rivers, lakes, reservoirs and wells in the contaminated areas. The most significant exposures via consumption of drinking water resulted from the use of water from the Dnieper River basin, and, in particular, the reservoirs of the

TABLE 5.9. ANNUAL (2001) MEAN CHERNOBYL RELATED EFFECTIVE INTERNAL DOSES (mSv) TO ADULT RESIDENTS OF AREAS WITH CAESIUM-137 SOIL DEPOSITION ABOVE 0.04 MBq/m^2 (1 Ci/km^2) IN 1986 [5.53]

Population	Caesium-137 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

Dnieper River system. The Dnieper cascade is a source of drinking water for more than eight million people. The main consumers of drinking water from the Dnieper River live in the Dnipropetrovsk and Donetsk regions. In Kiev, water from the Dnieper and Desna Rivers is used by about 750 000 people. The remaining part of the population use water mainly derived from groundwater sources.

(b) Consumption of fish. The Dnieper River reservoirs are used intensively for commercial fishing. The annual catch is more than 25 000 t. There was no significant decrease in fishing in most of these reservoirs during the first decade after the accident. During the first two to three years, however, restrictions were placed on the consumption of fish from the Kiev reservoir. In some smaller lakes, both in the former USSR and in parts of western Europe, fishing was prohibited during the first months and even years after the accident.

(c) Consumption of agricultural products grown on land irrigated with water from the Dnieper reservoirs. In the Dnieper River basin there is more than 1.8×10^6 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 in irrigated fields can take place because of root uptake of the radionuclides introduced with irrigation water and due to direct incorporation of radionuclides through leaves following sprinkler irrigation. However, recent studies have shown that, in the case of irrigated land in southern Ukraine, irrigation water did not add significant amounts of radioactive material 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 that consumes fish from local rivers and lakes, however, 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. Owing to 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 [5.53].

Contaminated rivers could potentially have led to significant doses in the first months after the accident through consumption of drinking water, mainly through contamination with 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 Kiev during the first few weeks after the Chernobyl accident [5.53].

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/a. Thus long term doses via the drinking water pathway were small in comparison with doses (mainly from short lived radionuclides) in the early phase [5.53].

The contribution of different exposure pathways to dose is shown in Fig. 5.14 for the village of Svetilovichy in the Gomel region 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.

5.4. TOTAL (EXTERNAL AND INTERNAL) EXPOSURE

The generalized data for both external and internal (not including dose to the thyroid)

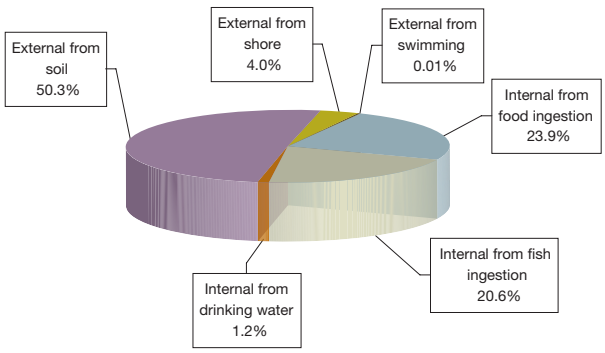

FIG. 5.14. Contributions of different pathways to the effective dose to the critical group of the population of Svetilovichy in the Gomel region of Belarus [5.53, 5.54].

exposures of the general public presented in Tables 5.4 and 5.9, respectively, have been summarized in Table 5.10 in order to estimate broadly the mean individual total (external and internal) effective doses accumulated by residents of radioactively contaminated areas during 1986–2000 and to forecast doses for 2001–2056. Table 5.11 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, the Russian Federation and Ukraine, separately for rural and urban populations and for different soil types, with no account taken of current countermeasures. Both accumulated and current annual total doses are presented for adults, since, 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 occupancy (see Tables 5.3 and 5.5), food habits and metabolic features.

As can be seen from Table 5.10, both accumulated and predicted mean doses in settlement residents vary over two orders of magnitude depending on the radioactive contamination of the area, soil type and settlement type. Thus in 1986–2000 the dose range was from 2 mSv in towns located in black soil areas, up to 300 mSv in villages located in areas with podzol sandy soil. According to the forecast, the doses expected in 2001–2056 are substantially lower than the doses already received (i.e. in the range of 1–100 mSv). In total, if countermeasures were not applied, the populations of some of the more contaminated villages in Belarus and the Russian Federation would receive lifetime effective doses of 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 70–700 mSv in various regions.

Based on local demographic data [5.51], ^{137}Cs soil deposition maps (see Section 3.1) and the

TABLE 5.10. PAST (1986–2000) AND FUTURE (2001–2056) MEAN CHERNOBYL RELATED TOTAL EFFECTIVE DOSES (mSv) TO ADULT RESIDENTS OF AREAS WITH CAESIUM-137 SOIL DEPOSITION ABOVE 0.04 MBq/m^2 (1 Ci/km^2) IN 1986 [5.53]

Population	Caesium-137 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 5.11. ANNUAL (2001) MEAN CHERNOBYL RELATED TOTAL EFFECTIVE DOSES (mSv) TO ADULT RESIDENTS OF AREAS WITH CAESIUM-137 SOIL DEPOSITION ABOVE 0.04 MBq/m^2 (1 Ci/km^2) IN 1986 [5.53]

Population	Caesium-137 in soil (MBq/m^2)	Soil type		
		Black	Podzol	Peat
Rural	0.04–0.6	0.05–0.8	0.1–1	0.2–2
	0.6–4	—	1–5	—
Urban	0.04–0.6	0.03–0.4	0.05–0.6	0.1–1

current level of countermeasure application (see Section 4), the vast majority of the five million people currently residing in the contaminated areas of Belarus, the Russian Federation and Ukraine (see Table 3.2) — that is, in the early 2000s — receive annual effective doses of less than 1 mSv (i.e. less than the national action levels in the three countries). For comparison, a worldwide average annual dose from natural background radiation is about 2.4 mSv, with a typical range of 1–10 mSv in various regions [5.1].

The number of residents of the contaminated areas in the three most affected countries that currently receive more than 1 mSv annually can be estimated to be about 100 000 persons. As the future reduction of both the external dose rate and radionuclide (mainly ^{137}Cs) activity concentrations in food will be rather slow (see Sections 5.2 and 3.3–3.5), the reduction of human exposure levels is expected to be slow (i.e. about 3–5%/a with currently applied countermeasures).

5.5. COLLECTIVE DOSES

5.5.1. Thyroid

A summary of the collective doses to the thyroid for the three most contaminated countries, based on the thyroid dose reconstruction techniques described in Section 5.3.4.1, is shown in Table 5.12. The total thyroid collective dose is 1.6×10^6 man Gy, with nearly half received by the group of persons exposed in Ukraine. The present estimate of the collective thyroid dose does not differ from that made in Ref. [5.1].

5.5.2. Total (external and internal) dose from terrestrial pathways

Estimates of collective dose accumulated in 1986–2005 via the terrestrial pathways of external irradiation and ingestion of contaminated foods are given in Table 5.13 for the three most affected countries. The total collective dose was estimated to be 43 000 man Sv in 1986–1995, including 24 000 man Sv from external exposure and 19 000 man Sv from internal exposure, according to UNSCEAR [5.1], annex J, table 34. According to the models of exposure dynamics presented above [5.7], the estimated collective effective external doses in 1986–2005 are about a factor of 1.2 higher, and collective effective internal doses are higher by a factor of 1.1–1.5 (depending on soil properties and applied countermeasures), than those obtained in 1986–1995. In total, the collective dose increased by 9000 man Sv, or by 21%, during the second decade, compared with the first decade, after the accident,

TABLE 5.12. COLLECTIVE THYROID DOSES IN THE THREE COUNTRIES MOST CONTAMINATED BY THE CHERNOBYL ACCIDENT [5.1]

	Collective thyroid dose (10^3 man Gy)
Russian Federation	300
Belarus	550
Ukraine	740
Total	1600

TABLE 5.13. ESTIMATED COLLECTIVE EFFECTIVE DOSES IN 1986–2005 TO THE POPULATIONS OF THE CONTAMINATED AREAS OF BELARUS, THE RUSSIAN FEDERATION AND UKRAINE (CAESIUM-137 SOIL DEPOSITION IN 1986 MORE THAN 37 kBq/m^2)[a]

	Population (millions of people)	Collective dose (10^3 man Sv)		
		External	Internal	Total
Belarus	1.9	11.9	6.8	18.7
Russian Federation	2.0	10.5	6.0	16.5
Ukraine	1.3	7.6	9.2	16.8
Total	5.2	30	22	52

[a] Excluding thyroid dose. (Modified from Ref. [5.1], annex J, table 34, using dosimetric models presented in this report.)

and reached 52 000 man Sv. This is in good agreement with the predictions made by UNSCEAR in 1988 [5.2].

The recent estimate of collective dose based on both human and environmental measurements implicitly accounts for substantial, but not specified, amounts of collective dose saved by the institution of countermeasures that included evacuation, relocation, prohibition on the use of foodstuffs and longer term remediation of contaminated areas.

5.5.3. Internal dose from aquatic pathways

The most important aquatic system (the Dnieper River basin) occupies a large area with a population of about 32 million people who use the water for drinking, fishing and irrigation. Estimates have been 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) [5.55, 5.56]. A long term hydrological scenario has been analysed using a computer model [5.57]. 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 [5.58]. The results of the calculations are given in Table 5.14.

Dose estimates for the Dnieper River 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 Section 4) carried out during 1992–1993 on the left bank floodplain of the Pripyat River decreased exposure by approximately 700 man Sv. Other protective measures on the right bank in the CEZ (during 1999–2001) will further reduce collective doses by 200–300 man Sv [5.59].

5.6. CONCLUSIONS AND RECOMMENDATIONS

5.6.1. Conclusions

(a) The collective effective dose (not including dose to the thyroid) received by about five million residents living in the areas of Belarus, the Russian Federation and Ukraine contaminated by the Chernobyl accident (^{137}Cs deposition on soil of >37 kBq/m^2) was approximately 40 000 man Sv during the period 1986–1995. The groups of exposed

TABLE 5.14. COLLECTIVE DOSE COMMITMENT (CDC_{70}) DUE TO STRONTIUM-90 AND CAESIUM-137 FLOWING FROM THE PRIPYAT RIVER TO THE DNIEPER RIVER AND DOWNSTREAM [5.56, 5.58]

Region	Population (millions of people)	Strontium-90 CDC_{70} (man-Sv)	Caesium-137 CDC_{70} (man-Sv)	Ratio ^{90}Sr CDC_{70}/^{137}Cs CDC_{70}
Chernigov	1.4	4	2	2
Kiev	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
Zaporozhe	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

persons within each country received an approximately equal collective dose. The additional amount of collective effective dose projected to be received between 1996 and 2006 is about 9000 man Sv.

(b) The collective dose to the thyroid was nearly 2×10^6 man Gy, with nearly half received by persons exposed in Ukraine.

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

(d) 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 thyroid cancer effects in some individuals.

(e) The internal thyroid dose from the intake of ^{131}I was mainly due to the consumption of fresh cow's milk and, to a lesser extent, of green vegetables; children, on average, received a dose that was much higher than that received by adults, because of their small thyroid masses and consumption rates of fresh cow's milk that were similar to those of adults.

(f) 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), since short lived radioiodines decayed during transport of the radioiodines along the food chains. The highest relative contribution (20–50%) to the thyroid dose to the public from short lived radionuclides was received by the residents of Pripyat through inhalation; these residents were evacuated before they could consume contaminated food.

(g) Both measurement and modelling data show that the urban population was exposed to a lower external dose by a factor of 1.5–2 compared with 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 foods than the rural population, both effective and thyroid internal doses caused predominantly by ingestion were lower by a factor of two to three in the urban than in the rural population.

(h) The initial high rates of exposure declined rapidly due to the decay of short lived radionuclides and to the movement of radiocaesium 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, which reduces the availability of caesium to plants and thus to the human food chain.

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

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

(k) 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 soil, and the contribution of internal exposure to the total (external and internal) dose does not exceed 10% in areas with predominantly black soil. The contribution of ^{90}Sr to the internal dose, regardless of natural conditions, is usually less than 5%.

(l) The long term internal doses to children caused by ingestion of food containing caesium radionuclides are usually lower by a factor of about 1.1–1.5 than those in adults and adolescents.

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

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

(o) The vast majority of the approximately five million people residing in the contaminated areas of Belarus, the Russian Federation and Ukraine currently receive annual effective doses of less than 1 mSv (equal to the national action levels in the three countries). For comparison, a worldwide average annual dose from natural background radiation is about 2.4 mSv, with a typical range of 1–10 mSv in various regions of the world.

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

(q) Based upon available information, it does not appear that the doses associated with hot particles were significant.

(r) The assessment of the Chernobyl Forum agrees with that of UNSCEAR [5.1] in terms of the dose received by the populations of the three most affected countries: Belarus, the Russian Federation and Ukraine.

5.6.2. Recommendations

(a) Large scale monitoring of foodstuffs, whole body counting of individuals and provision of thermoluminescent dosimeters to members of the general public are no longer necessary. The 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 with dosimeters for external dose and with whole body counting for internal dose.

(b) Sentinel or marker individuals in more highly contaminated areas not scheduled for further remediation might be identified for continued periodic whole body counting and monitoring of 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.

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6. RADIATION INDUCED EFFECTS ON PLANTS AND ANIMALS

6.1. PRIOR KNOWLEDGE OF RADIATION EFFECTS ON BIOTA

The effects of radiation on plants and animals have long been of interest to scientists; in fact, much of the information on the effects on humans has evolved from studies on plants and animals. Additional research followed the development of nuclear energy and concerns about the possible impacts of increased, but authorized, discharges of waste radionuclides into the terrestrial and aquatic environments. The magnitude of these authorized releases has always been controlled on the basis of the limitation of human exposure, but it has been recognized 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 ionizing radiation on plants and animals that several authoritative reviews had been compiled to summarize the findings [6.1–6.4].

Some broad generalizations about the effects of radiation exposure can be gleaned from the research that has been conducted over the past 100 years. Foremost are the relatively large differences in doses required to cause lethality among various taxonomic groups (Fig. 6.1). 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.

Figure 6.2 summarizes information on the doses required to be delivered over a short time period 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 [6.8]. He showed that characteristics such as large chromosomes, normal (rather than diffuse) centromeres, small chromosome number, uninucleated cells, diploid or haploid cells, sexual reproduction, long intermitotic time 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 6.1).

Scientific reviews (e.g. Ref. [6.3]) have indicated that mammals are the most sensitive organisms and that reproduction is a more sensitive endpoint than mortality. For acute exposures of

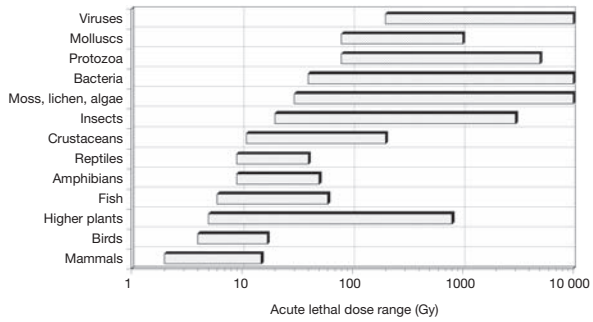

FIG. 6.1. Acute dose ranges that result in 100% mortality in various taxonomic groups. Humans are among the most sensitive mammals, and therefore among the most sensitive organisms [6.5].

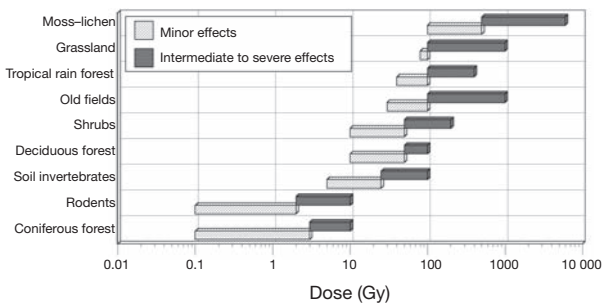

FIG. 6.2. Range of short term radiation doses (delivered over 5–60 d) that produced effects in various plant communities, rodents and soil invertebrates. Minor effects include chromosomal damage, changes in productivity, reproduction and physiology. Intermediate effects include changes in species composition and diversity through selective mortality. Severe effects (massive mortality) begin at the upper range of intermediate effects [6.6, 6.7].

mammals, mortality generally occurs at doses above 3 Gy, while reproduction is affected at doses below 0.3 Gy. Chronic exposures alter the responses, with mortality occurring at greater than 0.1 Gy/d and reproduction effected at less than 0.01 Gy/d. 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 soil is a sink for most radioactive contamination.

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 in its organs and tissues. Owing to their particular use of the habitat, plants and animals within a contaminated area may receive radiation doses that can be substantially higher 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 [6.3]).

Although all exposures to ionizing radiation 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 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 life stage (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:

(a) To terrestrial and aquatic plant populations, and aquatic animal populations at chronic dose rates of less than 10 mGy/d; or
(b) To terrestrial animal populations at dose rates of less than 1 mGy/d.

It should be emphasized, 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 biological responses investigated, indicated little likelihood of any

TABLE 6.1. PRINCIPAL NUCLEAR CHARACTERISTICS AND FACTORS INFLUENCING THE SENSITIVITY OF PLANTS TO RADIATION [6.5, 6.8]

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 nuclei	High polypolid
Sexual reproduction	Asexual reproduction
Long intermitotic time	Short intermitotic time
Long dormant period	Short or no dormant period
Slow meiosis	Fast meiosis

significant response. The above dose rates are with reference to population level effects, not to impacts on individual organisms.

More recent reviews of the effects of radiation exposure on individual organisms carried out in the framework of two EC projects, FASSET (Framework for the Assessment of Environmental Impact) and EPIC (Environmental Protection from Ionising Contaminants in the Arctic), have produced broadly consistent conclusions [6.9–6.11]. 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 (2.4 mGy/d). Detrimental responses then increase progressively with increasing dose rate and usually become clear at greater than 1 mGy/h (24 mGy/d) given over a large fraction of the lifespan. The significance of the minor morbidity and cytogenetic effects on the individual, or on populations more generally, seen at dose rates of less than 2.4 mGy/d has yet to be determined [6.11].

The recently compiled EPIC database covers a very wide range of radiation dose rates (from below 10^{-5} Gy/d up to more than 1 Gy/d) to wild flora and fauna observed in northern parts of the Russian Federation and in the Chernobyl contaminated areas [6.10]. 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 for chronic low linear energy transfer 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.

6.2. TEMPORAL DYNAMICS OF RADIATION EXPOSURE FOLLOWING 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 [6.4]. 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/^{132}I, ^{133}Xe, ^{131}I and ^{140}Ba/^{140}La). Most of these short lived, highly radioactive nuclides were deposited on to plant and ground surfaces, resulting in the accumulation of large doses that measurably affected biota. High exposures of the thyroids of vertebrate animals also occurred during the first days and weeks following the accident from the inhalation and ingestion of radioactive iodine isotopes or their radioactive precursors.

The measured exposure rates on the day of the accident in the immediate vicinity of the damaged reactor are shown in Fig. 6.3. These exposure rates were mainly due to gamma radiation from deposited radionuclides and range up to about 20 Gy/d. 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 beta 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 20–30 days can be generally characterized as an acute exposure regime that had pronounced effects on biota.

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

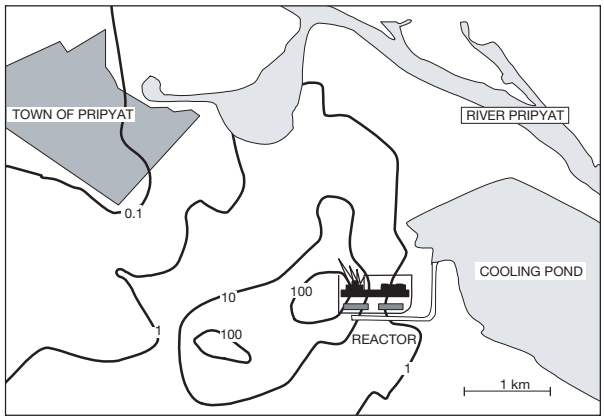

FIG. 6.3. Measured exposure rates in air on 26 April 1986 in the local area of the Chernobyl reactor. Units of isolines are R/h (1 R/h is approximately 0.2 Gy/d) [6.12].

processes. Dominant transport processes included rain induced transfer of radionuclides from plant surfaces on to 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. 5.3), damaging total doses were still accumulated. The modifying effect of radionuclide wash-off by rain on radiation damage of conifers is shown in Fig. 6.4.

In general, approximately 80% of the total radiation dose accumulated by plants and animals was received within three months of the accident, and over 95% of this was due to beta radiation [6.4]. This finding agrees with earlier studies on the importance of beta radiation, relative to the gamma component, to the total dose from radioactive fallout; for example, when a 10 h old mixture of fresh fission products was experimentally deposited on cereal plants at differing stages of growth, at a density of 7 GBq/m^2, the ratio of resulting beta and gamma dose rates, measured with thermoluminescent dosimeters, varied from 1 to 130 [6.13].

FIG. 6.4. A small conifer, the upper portion damaged by the initial deposition of radiation on to the crown of the plant, and the lower part of the plant damaged by the irradiation of surface deposited material subsequently washed from the plant's crown, leaving the middle section of the plant unaffected (photograph courtesy of T. Hinton, 1991).

Measurements made with thermoluminescent dosimeters on the soil surface at sites within the CEZ indicated that the ratio of beta dose to gamma dose was about 26:1 (i.e. 96% of the total dose was from beta radiation). For a gamma dose rate of 0.01 mGy/h at the soil surface, 15 days after the accident, the total cumulative dose in the first month from beta and gamma 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 [6.14].

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 beta and gamma 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 (i.e. the soil). Aside from the spatial heterogeneity in the 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 CEZ and reproduction of those plants and animals that are present means that new animals and plants are constantly being introduced into the radioactively contaminated conditions that exist around Chernobyl today. The current conditions are presented in Section 6.8.

6.3. RADIATION 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 beta emitting contamination on critical plant tissues resulted in their receiving a significantly larger dose than animals living in the same environment [6.13, 6.15]. Large apparent inconsistencies in dose–response observations occurred when the beta radiation component was not appropriately accounted for [6.16].

Within the CEZ, 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 the productivity of some species [6.15]. 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 0.1–2.6 GBq/m^2 and with estimated dose rates initially received by plants reaching 300 mGy/d. Spot necroses on leaves, withered tips of leaves and inhibition of photosynthesis, transpiration and metabolite synthesis were detected, as well as an increased incidence of chromosome aberrations in meristem cells [6.17]. The frequency of various anomalies in winter wheat exceeded 40% in 1986–1987, with some abnormalities apparent for several years afterwards [6.18].

Coniferous trees were already known to be among the more radiosensitive plants, and pine forests 1.5–2 km west of the Chernobyl nuclear power plant received a sufficient dose (>80 Gy) to cause mortality [6.19] at dose rates that exceeded 20 Gy/d [6.12]. The first signs of radiation injury in pine trees in close proximity to the reactor were yellowing and needle death, which appeared within two to three weeks. During the summer of 1986 the area of radiation damage expanded in the north-west 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 [6.19] and Arkhipov et al. [6.20] 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 more than 1 Gy, growth rates were reduced and morphological damage occurred; and at more than 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 6.2 shows the variation in activity concentration and dose among pine trees within the CEZ. The radiosensitivity of spruce trees was observed to be greater than that of pines. At absorbed doses as low as 0.7–1 Gy, spruce trees had malformed needles, buds and shoot growth [6.22].

Of the absorbed dose to critical parts of trees, 90% was due to beta radiation from the deposited radionuclides and 10% to gamma radiation. 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 in reclamation efforts [6.20]. 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

TABLE 6.2. RADIOACTIVE CONTAMINATION (kBq/kg) OF CONIFEROUS TREES AS A FUNCTION OF DISTANCE FROM THE CHERNOBYL REACTOR (AZIMUTH 205–260°), WITH CORRESPONDING ESTIMATES OF THE AIR DOSE RATE (mGy/h) IN OCTOBER 1987 AND THE ACCUMULATED EXTERNAL DOSE (Gy) [6.21]

Distance from the Chernobyl nuclear power plant (km)	Air dose rate (mGy/h)[a]	External dose (Gy)[a]	Activity concentration in needles (kBq/kg)					
			Caesium-144	Ruthenium-106	Zirconium-95	Niobium-95	Caesium-134	Caesium-137
2.0	2.2	126	13 400	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

[a] Dose rate and dose of gamma radiation at 1 m height from the soil surface.

induced damage to conifers were discernable (Table 6.3).

6.4. RADIATION EFFECTS ON SOIL INVERTEBRATES

Although between 60% and 90% of the initial fallout was captured by the forest canopy and other plants [6.19], 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 3.4 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, since the timing of the accident coincided with their most radiosensitive life stages: reproduction and moulting following their winter dormancy.

Within two months of the accident, the number of invertebrates in the litter layer of forests 3–7 km from the nuclear reactor was reduced by a factor of 30 [6.14], and reproduction was strongly affected (larvae and nymphs were absent). Doses of approximately 30 Gy (estimated from thermoluminescent dosimeters 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, was no different from control sites; however, species diversity remained markedly lower [6.14].

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 ten soil cores taken from pine stands in July 1986 at a distance of 3 km from the Chernobyl nuclear power plant, compared with 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 ten cores taken from the control site, whereas no species was found in all ten cores from the 3 km location [6.23]. The number of invertebrate species found at the heavily contaminated sites was only half that of controls in 1993, and complete species diversity did not recover until 1995, almost ten years after the accident [6.14].

Compared with invertebrates within the forest litter layer, those residing in arable soil were not affected so much. A fourfold reduction in earthworm numbers was found in arable soils, but no catastrophic mortality of 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 radiation, the main contributor (94%) to the total dose. The dose to invertebrates in forest litter was threefold to tenfold higher than that to those residing in surface soil [6.14].

Although researchers are unclear if sterility of invertebrates occurred in the heavily contaminated

TABLE 6.3. ZONES AND CORRESPONDING DAMAGE TO CONIFEROUS FORESTS IN THE AREA AROUND THE CHERNOBYL NUCLEAR POWER PLANT [6.22]

Zone and classification	External gamma dose[a] (Gy)	Air dose rate[a] (mGy/h)	Internal dose to needles (Gy)
Conifer death (4 km^2): complete death of pines, partial damage to deciduous trees	>80–100	>4	>100
Sublethal (38 km^2): death of most growth points, death of some 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

[a] Dose rate and dose of gamma radiation at 1 m height from the soil surface.

sites at Chernobyl [6.14], 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 of less than 200 Gy [6.24], 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 [6.8], radiosensitivity of insects is related to the average interphase nuclear volume [6.24].

6.5. 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 the gut, the thyroid and other body organs. Injuries to cattle are a major fallout consequence for rural populations, because of livestock loss but also because of the associated social and psychological implications [6.25, 6.26].

In the period shortly after the accident, domestic livestock within the CEZ were exposed to high levels of radioactive iodine (^{131}I and ^{133}I, with half-lives of 8 d and 21 h, respectively); this resulted in significant internal and external doses from beta and gamma radiation (Table 6.4). A thyroid dose of 76 Gy from the two isotopes of iodine is sufficient to cause serious damage to the gland [6.27]. The 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 two to three times higher than normal [6.15]. 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 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 to 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 [6.28]. Similar effects were observed in cattle evacuated from the Belarusian portion of the CEZ [6.26].

Although most livestock were evacuated from the area after the accident, several hundred cattle were maintained in the more contaminated areas for a two to four month period. By the autumn of 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 had received a thyroid dose of more than 180 Gy [6.26]. The 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. Haematological parameters were normal for animals kept in areas with ^{137}Cs contamination of 0.2–1.4 MBq/m^2 (5–40 Ci/km^2) [6.28].

TABLE 6.4. DOSES TO CATTLE THAT STAYED IN THE 30 km ZONE OF CHERNOBYL FROM 26 APRIL TO 3 MAY 1986 [6.21]

Distance from the Chernobyl nuclear power plant (km)	Surface activity (10^8 Bq/m^2)	Absorbed dose (Gy)		
		Thyroid	Gastrointestinal 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

Chronic radiation damage was observed in over 2000 sheep and 300 horses (3–8 years old) removed from the highly contaminated Khoiniki area of Belarus 1.5 years after the accident [6.26]. 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 1.5–2 times higher 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, leukopaenia, thrombocytopaenia, eosinophilia and myelocytosis. Seventy per cent of the animals had thyroid hormone concentrations in blood serum that were lower than the detection levels of the assay methods [6.26].

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/a. Scientific evidence indicates that increased birth defects are not distinguishable from background frequencies at such low doses [6.25]. Additionally, data for 1989 show that livestock birth defects in the contaminated area of the Zhytomyr region were no higher than in the uncontaminated areas of the same region. Photographs 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 finished prior to the accident; this much publicized observation of teratogenesis was therefore caused by factors other than radiation from the Chernobyl accident.

6.6. RADIATION EFFECTS ON OTHER TERRESTRIAL ANIMALS

Four months after the accident, surveys and autopsies of wildlife and abandoned domestic animals that remained within the 10 km radius of the Chernobyl nuclear power plant were conducted [6.14]. Fifty species of birds were identified, including some rare species; 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 or in their ovaries.

During the autumn of 1986, the number of small rodents on highly contaminated research plots decreased by a factor of two to ten. 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 radiation. The numbers of animals were recovering by the spring of 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 twofold to threefold compared with controls. Resorption of embryos also increased markedly in rodents from the affected areas; however, the number of progeny per female did not differ from controls [6.29].

6.7. RADIATION EFFECTS ON AQUATIC ORGANISMS

Cooling water for the Chernobyl nuclear power plant was obtained from the 21.7 km^2 human-made reservoir located south-east of the plant site. The cooling reservoir became heavily contaminated following the accident (see Section 3.5 for details) with over $6.5 \pm 2.7 \times 10^{15}$ Bq of a mixture of radionuclides in the water and sediments [6.30]. Aquatic organisms were exposed to external exposure from radionuclides in water and contaminated bottom sediments and irradiation from contaminated aquatic plants. Internal exposure 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 Fig. 6.5.

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 60–80% of the dose. During the second week, the contribution of

short lived radionuclides to doses to aquatic organisms decreased by a factor of two. Maximum dose rates to fish were delayed (Fig. 6.5), due to the time required for their food webs to become contaminated with longer lived radionuclides (mostly $^{134,137}Cs$, $^{144}Ce/^{144}Pr$, $^{106}Ru/^{106}Rh$ and $^{90}Sr/^{90}Y$). Differences in dose rates among fish species occurred due to their trophic positions. Non-predatory fish (carp, goldfish, bleak) reached estimated peak dose rates of 3 mGy/d 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 [6.21]. Accumulated doses were highest for the first generation of fish born in 1986 and 1987. Bottom dwelling fish (goldfish, silver bream, bream, carp) that received significant exposure from the bottom sediments received accumulated total doses of approximately 10 Gy.

In 1990 the reproductive capacity of young silver carp was analysed [6.31]. 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 the 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 the controls (5–7%). In contrast, Pechkurenkov [6.32] reported that the number of cells with chromosome aberrations in 1986–1987 in carp, flat bream 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 the chronic effects of ionizing radiation on reproduction in fish, with the Chernobyl data included (Table 6.5), have been summarized.

6.8. GENETIC EFFECTS IN ANIMALS AND PLANTS

Quality data concerning the incidence of Chernobyl related induced mutations in plants and animals are relatively sparse. An increased mutation level was apparent in 1987 in the form of various morphological abnormalities observed in Canada flea-bane, common yarrow and mouse millet plants. Examples of abnormalities include unusual branching of stems, doubling of the number of racemes, abnormal colour and size of leaves and flowers, and development of 'witches' brooms' in pine trees. Similar effects in the 5 km radius circle near the reactor also appeared in deciduous trees (leaf gigantism, changes in leaf shape; see Fig. 6.6).

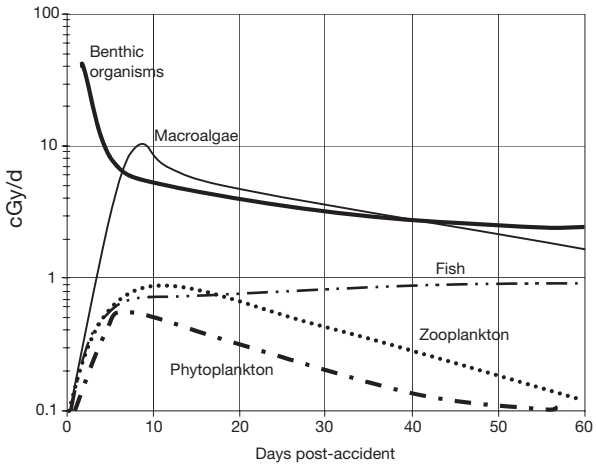

FIG. 6.5. The dynamics of absorbed dose rate (cGy/d) of organisms within the Chernobyl cooling pond during the first 60 days following the accident. Data are model results based on concentrations of radionuclides in the water column and lake sediments [6.21].

FIG. 6.6. Typical morphological abnormality seen on conifer trees. Such enhancement of vegetative growth and gigantism of some plant parts were not uncommon (photograph courtesy of T. Hinton, 1991).

Morphological changes were observed at an initial gamma close rate of 0.2–0.3 mGy/h. At 0.7–1.3 mGy/h enhancement of vegetative reproduction (heather) and gigantism of some plant species were observed [6.19, 6.20, 6.34, 6.35].

Cytogenetic analysis of cells from the root meristem of winter rye and wheat germ of the 1986 harvest demonstrated a dose dependence 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 [6.36]. 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 was higher than in the first.

From 1986 to 1992 mutation dynamics were studied in populations of *Arabidopsis thaliana Heynh.* (L.) within the CEZ [6.37]. On all study plots in the first two to three 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 four to eight 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 one.

Zainullin et al. [6.38] 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. Mutation levels were increased in 1986–1987 in flies inhabiting contaminated areas with initial exposure rates of 2 mGy/h and more. In the subsequent two years mutation frequencies gradually returned to normal.

Studies of adverse genetic effects in wild mice have been reported by Shevchenko et al. [6.39] and Pomerantseva et al. [6.40]. 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 at a site in the Bryansk region of the Russian Federation. The

TABLE 6.5. CHRONIC EFFECTS OF IONIZING RADIATION ON REPRODUCTION IN FISH, DERIVED FROM THE FASSET DATABASE [6.33]

Dose rate (μGy/h)	Dose rate (mGy/d)	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
1000–1999	24–48	Mean lifetime fecundity decreased, early onset of infertility
2000–4999	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
5000–9999	120–240	Reduction in number of fish surviving to one month of age
		Increased vertebral abnormalities
>10 000	>240	Interbrood time tending 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 four years, female salmon had significantly reduced fecundity

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 of 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 for mice sampled at the most contaminated site. At dose rates of about 2 mGy/h, 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 harbouring recessive lethal mutations decreased with time post-accident [6.40]. 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 (ESTRs). These are repeat DNA sequences that are distributed throughout the germline and that have a high background (spontaneous) mutation rate. At present, ESTRs are considered to have no function, although this is a matter of much interest and discussion [6.41, 6.42]. Minisatellite mutations have only rarely been associated with recognizable genetic disease [6.43].

Although laboratory examination of mutations in mouse ESTR loci shows clear evidence of a mutational dose response [6.44, 6.45], no convincing data on elevated levels of minisatellite mutations in plants or animals residing in the Chernobyl affected areas appear to have been published so far in the 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 [6.42]. This is an area of science that requires additional research.

6.9. SECONDARY IMPACTS AND CURRENT CONDITIONS

Prior to the accident, much of the area around Chernobyl was covered in 30–40 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.

Exposure to ionizing radiation is an environmental stress, in many ways similar to other environmental stresses such as pollution by metals or the destruction caused by 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 towards 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 and the abundance and gross physiological conditions of small mammal populations in the CEZ appear to be similar to background locations in other parts of Ukraine [6.46–6.48]. Reports on the current genetic conditions of rodents within the zone are contradictory; for example, Shevchenko et al. [6.39] found significant disorders in spermatogenesis, while Baker et al. [6.46] found no reproductive inhibition or germ line mutations.

Layered on top of the impacts of the radiation exposure was the abrupt and drastic change that occurred when humans were removed from the CEZ. 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 CEZ 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 cleanup workers, including substantial road construction.

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 CEZ (e.g. a ban on hunting), tended to compensate for the adverse biological effects of the radiation and promoted an increase in the populations of wild animals. Significant population increases of game mammals (wild boar, roe deer, red deer, elk, wolves, foxes, hares, beavers, etc. (Fig. 6.7)) and bird species (black grouse, ducks, etc.) were observed soon after the Chernobyl accident [6.49, 6.50].

More than 400 species of vertebrate animals, including 67 ichthyoids, 11 amphibians, 7 reptiles, 251 birds and 73 mammals, inhabit the territory of the evacuated town of Pripyat and its vicinity; more than 50 of them belong to a list of those protected according to national Ukrainian and European Red Books. The CEZ has become a breeding area for white tailed eagles, spotted eagles, eagle owls, cranes and black storks (Fig. 6.8) [6.51].

In the Pripyat River floodplain a developed system of artificial drainage channels now supports about a hundred families of beavers. Recognizing the value of the abandoned land around Chernobyl, 28 endangered Przhevalsky wild horses were introduced in 1998. After six years their number had doubled [6.51]. In both the Ukrainian and Belarusian parts of the CEZ, 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 US National Parks such as Yellowstone and the Grand Tetons, and at large US 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.

(a)

(b)

FIG. 6.7. Wild boar (a) and wolves (b) inhabiting the CEZ are not afraid of people because of long term hunt prohibition (photographs courtesy of S. Gaschak, 2004).

FIG. 6.8. A white tailed eagle chick observed recently in the CEZ. Before 1986 these rare predatory birds had rarely been found in this area (photograph courtesy of S. Gaschak, 2004).

On the other hand, the absence of forest management, and the associated increase in forest fires, have substantial impacts on 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 of 1992, when the area of burnt forest amounted to 170 km^2 (i.e. about one sixth of the woodlands) [6.52].

Without a permanent residence of humans for 20 years, the ecosystems around the Chernobyl site are now flourishing. The CEZ has become a wildlife sanctuary [6.47], and it looks like the nature park it has become.

6.10. CONCLUSIONS AND RECOMMENDATIONS

6.10.1. Conclusions

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

(b) The environmental response to the Chernobyl accident was a complex interaction among radiation dose, dose rate and its temporal and spatial variations, and 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:
   (i) Increased mortality of coniferous plants, soil invertebrates and mammals;
   (ii) Reproductive losses in plants and animals;
   (iii) Chronic radiation syndrome in animals (mammals, birds, etc.).

   No adverse radiation induced effects have 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.

(c) Following the natural reduction of exposure levels due to radionuclide decay and migration, populations have been recovering from the acute radiation effects. By the next growing season after the accident, the 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 the major radiation induced adverse effects in plants and animals.

(d) 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 ionizing radiation. Thus rapidly developing cell systems, such as meristems of plants and insect larvae, were predominantly affected by radiation. At the organism level, young plants and animals were found to be the most sensitive to the acute effects of radiation.

(e) Genetic effects of radiation, in both somatic and germ cells, were observed in plants and animals in the CEZ during the first few years after the accident. Both in the CEZ 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.

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

6.10.2. Recommendations for future research

(a) In order to develop a system of environmental protection against radiation, the long term impact of radiation on plant and animal populations should be further investigated in the CEZ; this is a globally unique area for

radioecological and radiobiological research in an otherwise natural setting.

(b) In particular, multigenerational studies of radiation effects on the genetic structure of plant and animal populations might bring fundamentally new scientific information.

(c) There is a need to develop standardized methods for biota–dose reconstruction, for example in the form of a unified dosimetric protocol.

6.10.3. Recommendations for countermeasures and remediation

(a) Protective actions for farm animals in the event of a nuclear or radiological emergency should be developed and internationally harmonized based on modern radiobiological data, including the experience gained in the CEZ.

(b) It is likely that any technology based remediation actions aimed at improving the radiological conditions for plants and animals in the CEZ would have adverse impacts on biota.

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7. ENVIRONMENTAL AND RADIOACTIVE WASTE MANAGEMENT ASPECTS OF THE DISMANTLING OF THE CHERNOBYL SHELTER

The destruction of the unit 4 reactor at the Chernobyl nuclear power plant created radioactive contamination and radioactive waste[3] in the unit, the Chernobyl nuclear power plant site and the surrounding area (further referred to as the CEZ). The future development of the CEZ depends on the strategy for the conversion of unit 4 into an ecologically safe system (i.e. the development of the NSC, the dismantlement of the current shelter, removal of FCM, and eventual decommissioning of the reactor site).

In particular, the long term strategy for unit 4 involves implementation of the NSC concept to cover the unstable shelter and the related radioactive waste management activities at the Chernobyl nuclear power plant site and the CEZ. Currently units 1, 2 and 3 (1000 MW RBMK reactors) are shut down awaiting decommissioning; 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 of unit 4 and the existing and future environmental impact associated with it and the management of the radioactive waste from the accident at the Chernobyl nuclear power plant site and in the CEZ.

7.1. CURRENT STATUS AND THE FUTURE OF UNIT 4 AND THE SHELTER

7.1.1. Unit 4 of the Chernobyl nuclear power plant after the accident

In the course of the 1986 accident, a small part of the nuclear fuel (3.5% according to past estimates [7.1] or 1.5% according to recent estimates [7.2]) and a substantial fraction of volatile radionuclides (see Section 3.1) were released from the damaged unit 4. The remainder of the damaged nuclear fuel, more than 95% of the fuel mass at the moment of the accident, i.e. about 180 t, was left in the remains of the reactor [7.1]. The uncertainty in this estimate is discussed in Section 7.1.5.

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 [7.1] (see Fig. 7.1). The total amount of material dumped on the reactor was approximately 5000 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 [7.1].

At the Chernobyl nuclear power plant site in mid-May 1986 there were high levels of air radiation dose rate and air activity concentration, caused by relatively uniform contamination of the area with finely dispersed nuclear fuel and aerosols of short lived radionuclides, and also the presence of dispersed nuclear fuel particles or fragments. These fragments consisted of discrete and non-uniform material from the reactor core, reactor constructional material and graphite.

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 metres of radioactive waste generated by this work were disposed of in the pioneer wall and the cascade wall. Construction of walls around the damaged reactor reduced the radiation dose rates by a factor of 10–20 [7.3]. The completion of the pioneer wall and the cascade wall and a significant reduction in radiation levels allowed the shelter to be constructed.

The shelter, which was intended to provide the environmental containment of the damaged reactor, was erected within an extremely short period of time, between May and November 1986, under conditions of high radiation exposure of the personnel. The steps taken to save time and cost during the construction, and the high dose rates inside the structure, resulted in a lack of reliable and comprehensive data on the stability of the damaged older structures, a need for remote control

[3] The radioactive waste in the CEZ does not include the waste associated with the decommissioning of Chernobyl nuclear power plant units 1, 2 and 3.

FIG. 7.1. The destroyed reactor after the accident in 1986.

concreting and the impossibility of carrying out welding in some specific situations.

7.1.2. Current status of the damaged unit 4 and the shelter

The shelter [7.4] was 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. 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 on the 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 20 years since the accident.

The shelter has approximately 1000 m^2 of openings in its surface. These openings allow approximately 2000 m^3/a of precipitation to percolate through the radioactively contaminated debris and eventually to pool in rooms in the lower levels of unit 4 (see Fig. 7.2) [7.5]. Condensation within unit 4 of approximately 1650 m^3/a of water and the residues from periodic spraying of 180 m^3/a of liquid dust suppressant contribute 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 plutonium and 6 mg/L of uranium. About 2100 m^3/a of the collected water evaporates, and about 1300 m^3/a leaks through the foundation into the soil beneath unit 4 [7.6]. The existing Chernobyl nuclear power plant radioactive waste management system is not capable of treating liquid radioactive waste that contains transuranic elements.

The inside conditions of unit 4 (Fig. 7.3) are hazardous and present significant risks to workers and the environment. General area radiation dose rates range from 2 μSv/h to 0.1 Sv/h inside the

FIG. 7.2. Opening in the shelter allowing infiltration of atmospheric water.

FIG. 7.3. Reactor room of unit 4 after the accident.

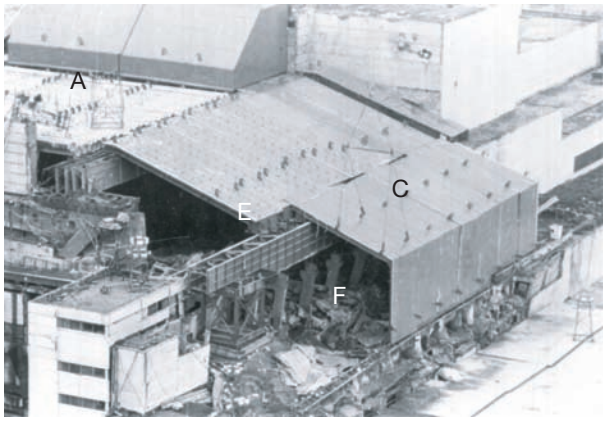


FIG. 7.4. Major structural components of the Chernobyl shelter: (A) pipe roofing; (B) southern panels; (C) southern hockey sticks; (D) B1/B2 beams; (E) mammoth beam; (F) octopus beam.

shelter [7.5]. Individual occupational radiation exposures during current operations at unit 4 are controlled so that they do not exceed the dose limit of 20 mSv/a [7.7].

Unit 4 is ventilated during current activities through a monitored exhaust above the reactor room. The unfiltered exhaust air is normally below the permitted limits for atmospheric discharge, and a filtration system exists for use should the exhaust air levels approach the permitted discharge limits. The ventilation system is zoned so that air flows from outside the shelter through spaces with increasing levels of contamination.

Unit 4 and the associated cascade walls have an accumulation of FCM, including large core fragments that could conceivably lead to criticality under flooded conditions. Such a criticality accident is considered unlikely; however, if criticality should occur, it might lead to the exposure of some workers inside unit 4 to an external dose of only a few millisieverts, because workers tend to avoid the spaces with criticality risk. It has been estimated that, in such a case, there would be no significant consequences inside and outside the CEZ [7.5, 7.8, 7.9].

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. 7.4); improvement of the physical protection and access control system; design of an integrated automated control system (control of building structure conditions, seismic control, nuclear safety control and radiation control); modernization of the dust suppression system; and additional structural stabilization. Computerized control systems were installed in the shelter [7.9] 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 from Ukraine and donor countries[4].

The magnitude and importance of possible future radioactive releases from the shelter (in the event of its collapse) significantly depend on the radiological and physicochemical 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 [7.5, 7.10] shows that the typical size of these particles (activity median aerodynamic diameter) is from 1 to 10 μm. Hence most of the material is expected to be respirable, which increases its potential inhalation hazard. The potential for inhalation hazard is increased by the winds that may be generated if the shelter roof were to collapse.

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 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 CEZ [7.11, 7.12].

Another concern related to the FCM is its possible transport out of the shelter into groundwater through the accumulated water. The potential for FCM to dissolve in 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 [7.3]; 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 of mobile radionuclides such as ^{90}Sr migrating and reaching the Pripyat River was expected to be very low [7.9]. The expected significance of this phenomenon is not known, and therefore monitoring of the evolving groundwater situation 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 may still be rising. This effect is considered to have occurred 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 the Kiev reservoir from potential contamination through groundwater [7.9].

The main potential hazard associated with the shelter is a possible collapse of its top structures and the release of radioactive dust into the environment; therefore a dust suppression system was installed beneath the shelter roof that periodically sprays dust suppression solutions and fixatives. The system has operated since January 1990, and more than 1000 t of dust suppressant has been sprayed during this period.

7.1.3. Long term strategy for the shelter and the new safe confinement

In order to avoid a collapse of the shelter, some measures have been implemented and additional measures are planned to strengthen unstable parts of the shelter and to extend their stability from 15 to 40 years [7.13]. In addition, the NSC is planned to be built as a cover over the existing shelter as a longer term solution (see Fig. 7.5). The Ukrainian Government supports 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 a 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 the eventual decommissioning of the reactor.

The specific operational aspects related to the construction and operation of the NSC, including maintenance in the long term, have not yet been identified. It is important to note that the NSC

[4] Contributors to the work of the Chernobyl Shelter Fund include Austria, Belgium, Canada, Denmark, the European Commission, Finland, France, Germany, Greece, Ireland, Italy, Kuwait, Luxembourg, the Netherlands, Norway, Poland, Spain, Sweden, Switzerland, Ukraine, the UK and the USA. Additional donors to the Fund include Iceland, Israel, Korea, Portugal, Slovakia and Slovenia.

design is based on the current plans for removal of the FCM that depend on the availability of a final geological disposal facility about 50 years from now [7.13]. This extended dormancy period could result in the dispersal of the special human resources needed to remove and dispose of the FCM safely. Accordingly, there are good reasons for removing the FCM and structural material as soon as possible after the construction of the NSC.

7.1.4. Environmental aspects

7.1.4.1. Current status of the shelter

At present, the environmental contamination around the Chernobyl nuclear power plant site is due to 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 activities carried out in the CEZ. The main dose contributing radionuclides within the CEZ around the Chernobyl nuclear power plant site are ^{137}Cs, ^{90}Sr, ^{241}Am and ^{239,240}Pu (see also Section 3); the distribution of these radionuclides is shown in Fig. 7.6 [7.2].

7.1.4.2. 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 4–10 GBq/a, which is many times lower than the regulatory limit of 90 GBq/a [7.9]. The uncontrolled releases depend on the locations and areas of the openings in the external structures and the air transfer rate through them, which depends on many factors, such as temperature, barometric pressure, humidity and wind speed and direction.

As a result, the air in the immediate vicinity of the shelter contains finely dispersed fuel particles with concentrations of up to 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 emitters 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. If a person (worker) were to spend an entire year adjacent to the shelter, a recent inhalation dose assessment indicates that releases would result in an annual dose of about 0.5 mSv, which would decrease to about 0.0002–0.0005 mSv beyond a distance of 10 km [7.14]. Inhalation doses from the ongoing releases outside the CEZ are significantly less than the dose limits for the population [7.7].

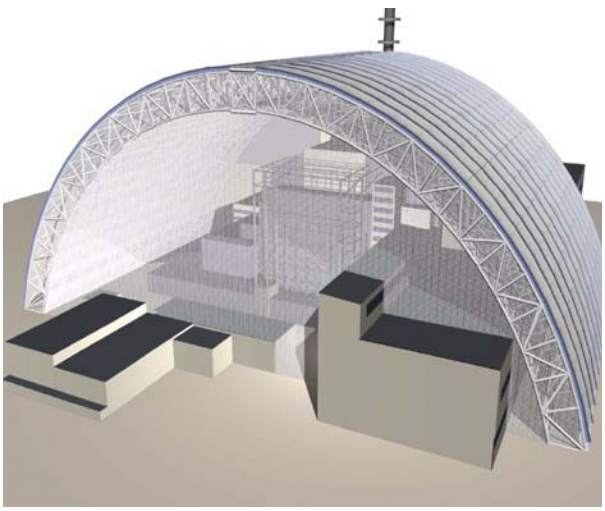


FIG. 7.5. Planned NSC.

7.1.4.3. Impact on surface water

The average concentrations of radionuclides in surface water bodies are declining. In the Pripyat River in 2003, for example, concentrations were observed to be 0.05 (max. 0.12) Bq/L for ^{137}Cs and 0.15 (max. 0.35) Bq/L for ^{90}Sr [7.15]. The main sources of radionuclides in the rivers in the CEZ during ordinary and high water seasons continue to be runoff from the watersheds situated outside the immediate Chernobyl nuclear power plant area, infiltrating waters from the Chernobyl nuclear power plant cooling pond and old water reclamation systems in the heavily contaminated territories. During winter and low water seasons, the radionuclide fluxes from regional groundwater contribute the majority of the radionuclide migration to the Pripyat River from this area. However, the values of radionuclide flux from all groundwater into surface water are still relatively low, and the contribution of groundwater contamination plumes from the temporary

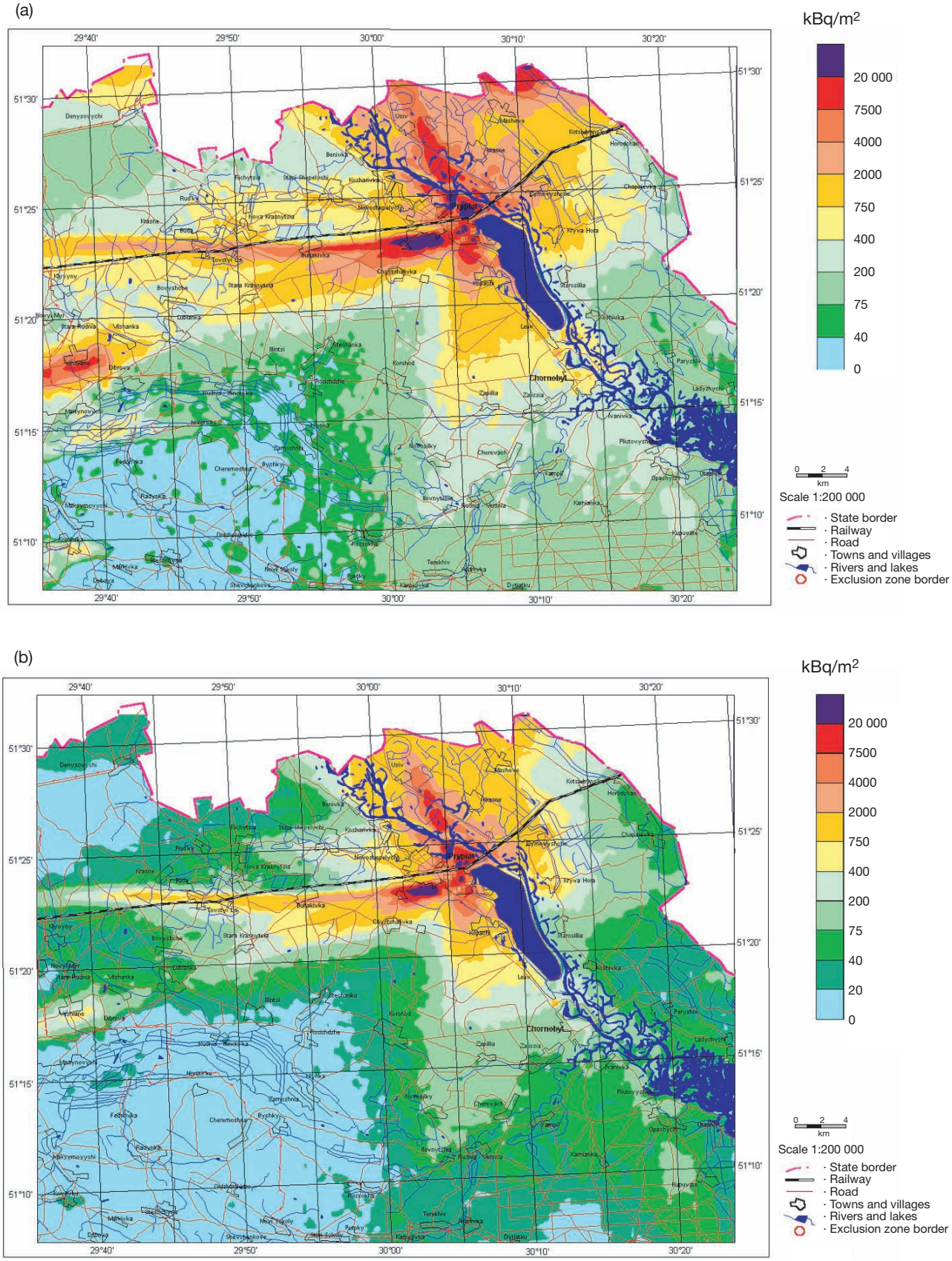
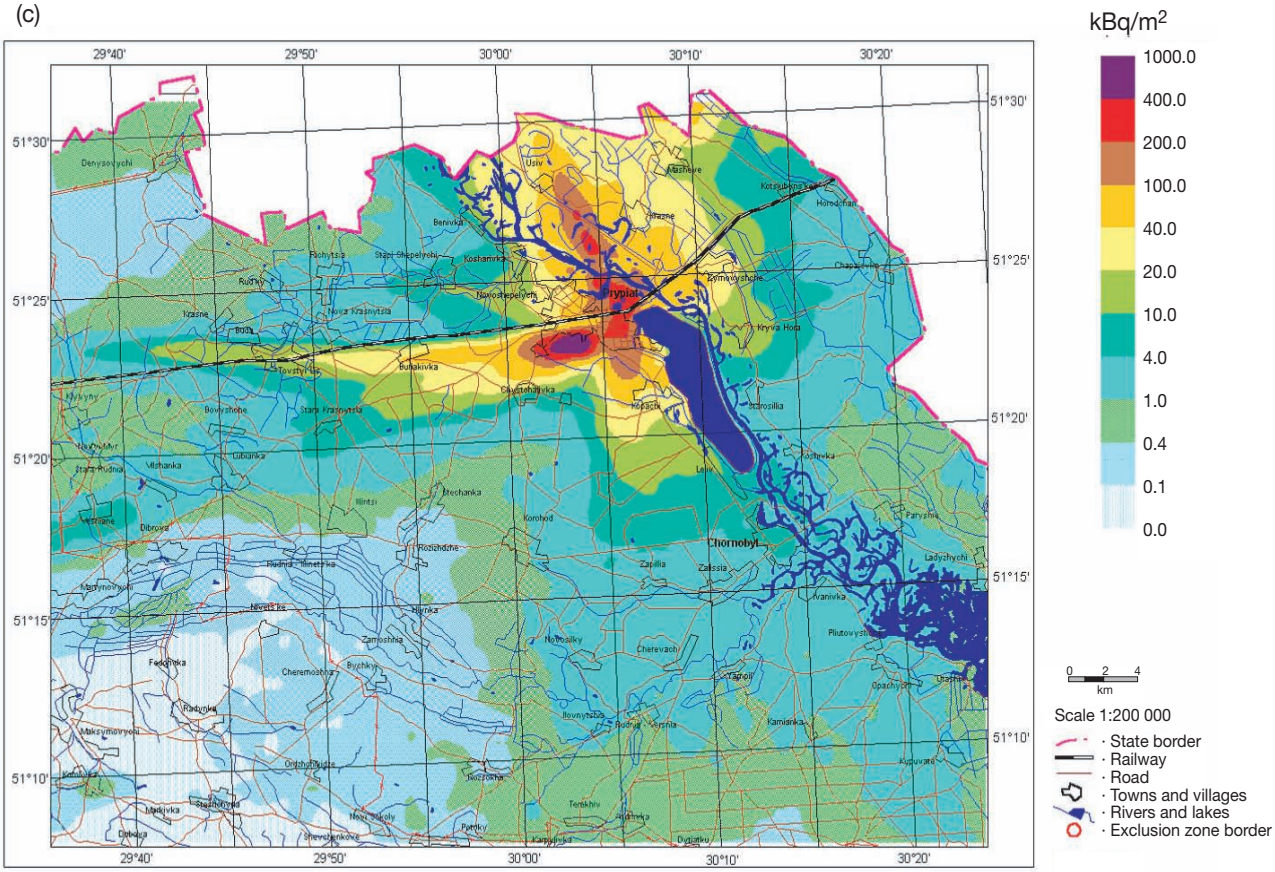


FIG. 7.6. Surface contamination by radioactive fallout within the CEZ [7.2]. (a) Caesium-137 in soils of the CEZ in 1997 (kBq/m^2); (b) ^{90}Sr in soils of the CEZ in 1997 (kBq/m^2); (c) ^{241}Am in soils of the CEZ in 2000 (kBq/m^2); (d) ^{239,240}Pu in soils of the CEZ in 2000 (kBq/m^2).

(c)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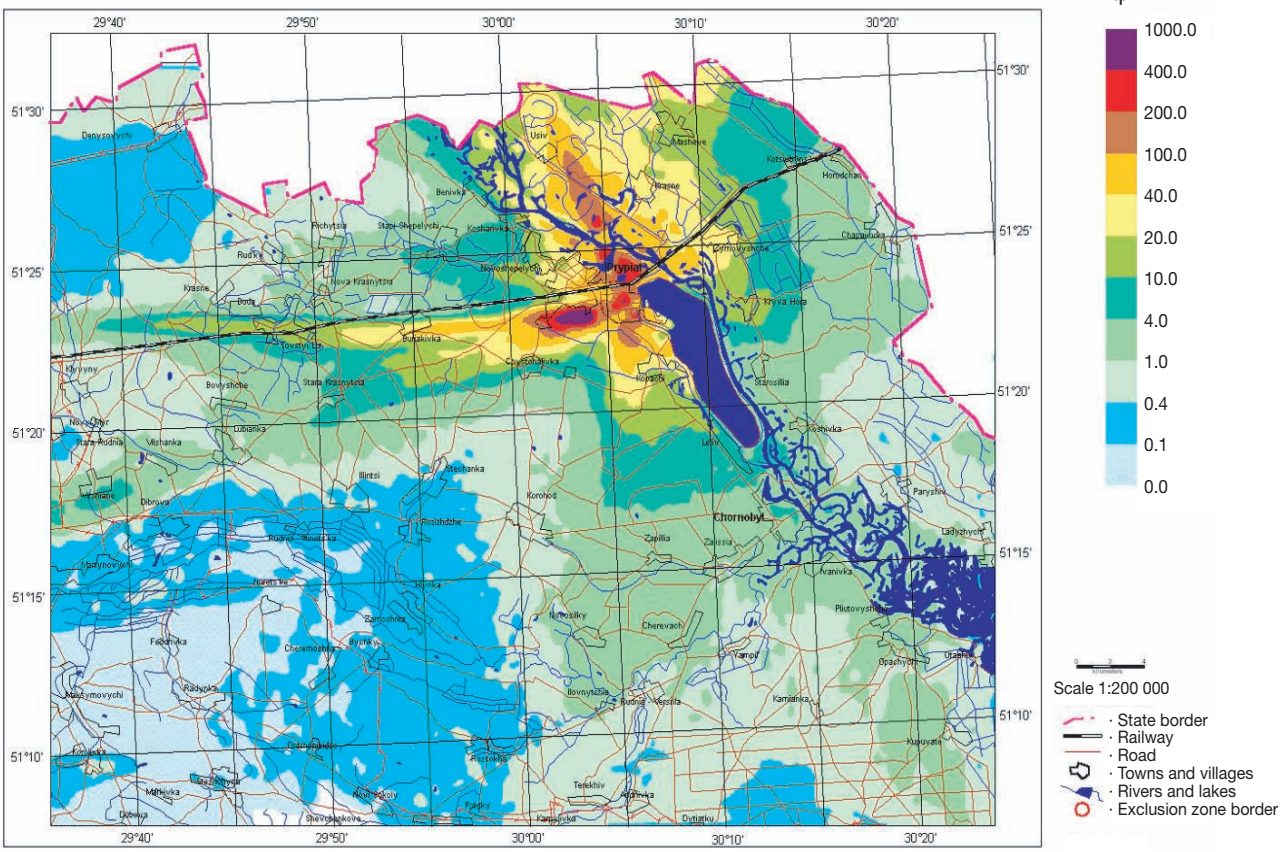
(d)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· Towns and villages
· Rivers and lakes
· Exclusion zone border

radioactive waste facilities and the shelter area has been identified as about 3–10% [7.15] of the annual migration of radionuclides into the Pripyat–Dnieper River system from the CEZ (see also Section 3.5).

7.1.4.4. Impact on groundwater

Surface contamination around the Chernobyl nuclear power plant site is the cause of groundwater contamination, with groundwater 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 recent studies, the primary source term for radionuclide contamination of the groundwater is considered to be water accumulating inside the 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 nuclear power plant site.

In some places, ^{137}Cs in groundwater in the subsurface horizons near the shelter reaches 100 Bq/L and even 3000–5000 Bq/L. However, in the majority of the shelter area, ^{137}Cs concentrations in groundwater are more or less similar and vary from 1 to 10 Bq/L. Typical concentrations of ^{90}Sr in groundwater around the shelter site are in the range from 2 to 160 Bq/L, with maximum concentrations observed during the past five years ranging from 1000 to 3000 Bq/L. Estimated concentrations of transuranic elements in the groundwater of this area also vary over a wide range, from 0.003 to 3–6 Bq/L for ^{238}Pu and ^{239,241}Pu, and from 0.001 to 8–10 Bq/L for ^{241}Am [7.16, 7.17].

7.1.4.5. Impacts of shelter collapse without the new safe confinement

Owing to 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/a [7.5, 7.18], and therefore an analysis (summarized from Ref. [7.6]) of the potential impacts of a shelter collapse has been performed for scenarios without and with the NSC in place.

(a) 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 an activity of about 1.6×10^{13} Bq. This could lead to an additional annual inhalation dose of up to 0.4 Sv near the shelter. The estimated annual doses outside the CEZ could reach 2 mSv [7.6], which would exceed the established dose limits for the public in Ukraine [7.7].

Within the boundaries of the CEZ, 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 Fig. 7.7 [7.8]. The highest relative increase in soil contamination would occur if the wind were to blow the plume from a shelter collapse to the south-west towards 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 CEZ boundaries, at 50 km from the shelter, additional surface contamination with ^{137}Cs, ^{90}Sr and ^{238,239,240}Pu due to a shelter collapse would contribute from a few to 10%.

(b) Impact on surface water

In the event of a shelter collapse, additional radioactive material could also be deposited in and near the rivers.

As shown in Fig. 7.7, radionuclide depositions into the Pripyat River could be as high as 1.1×10^{12} Bq of ^{90}Sr, 2.4×10^{12} Bq of ^{137}Cs, 1.6×10^{10} Bq of ^{238}Pu, 4.0×10^{10} Bq of ^{239,240}Pu and 5.0×10^{10} Bq of ^{241}Am. Estimates of the maximum possible concentrations of these radionuclides in the Dnieper reservoirs show that the peak concentration of ^{90}Sr can be expected in the Kiev 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 Kakhovka 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 [7.7]) would not be exceeded if an accident leading to the maximum impacts were to occur at the shelter.

The maximum possible concentrations of ^{137}Cs that can be expected in the Kiev and Kanev reservoir water, even in the worst simulated scenarios, are three to ten times lower than the limits for potable water. Such a collapse would not affect the concentrations of ^{238}Pu, ^{239,240}Pu and ^{241}Am in the Pripyat and Dnieper Rivers [7.6].

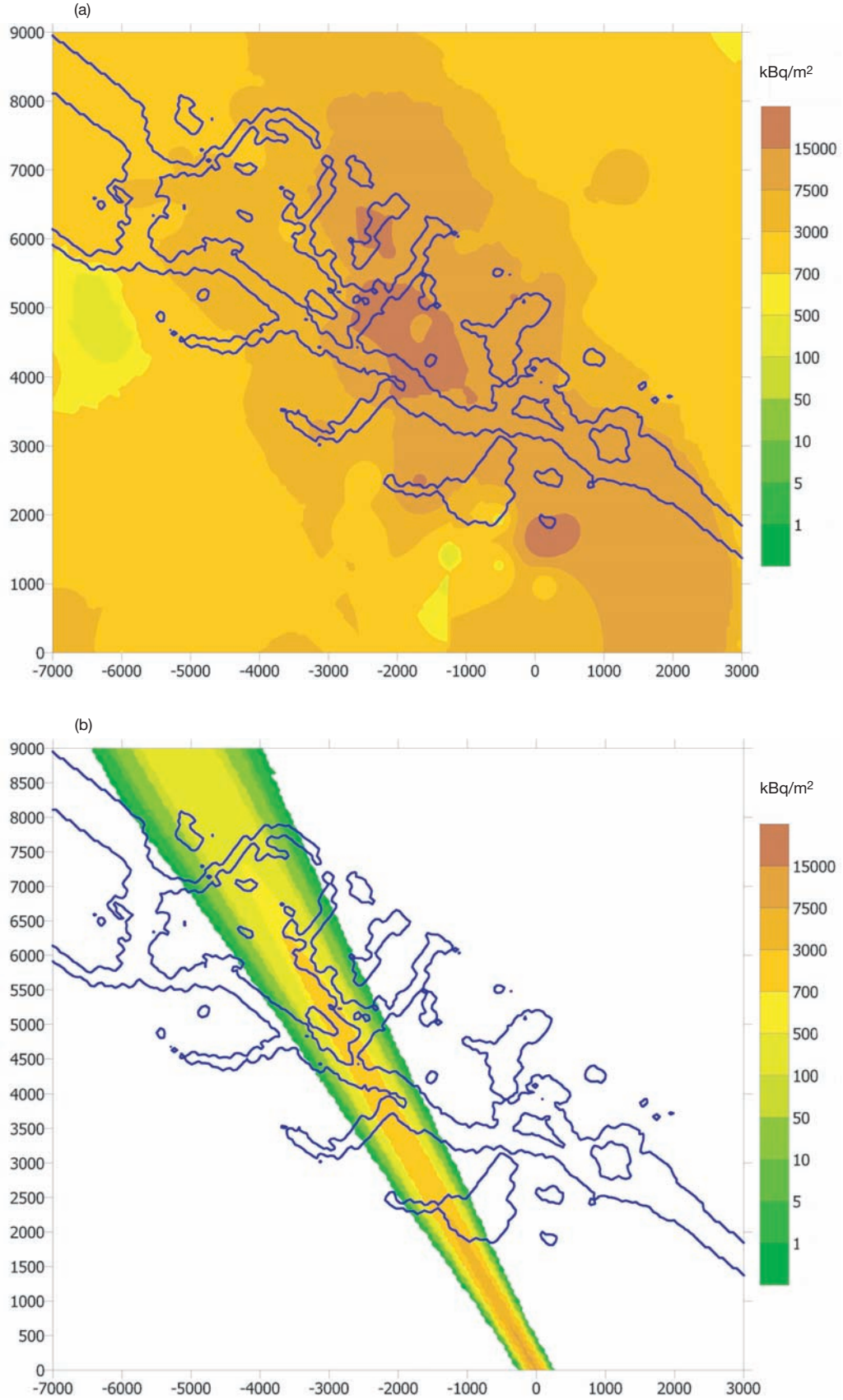

FIG. 7.7. Strontium-90 soil density of Chernobyl fallout (a) upstream of Yanov bridge, 1999, and (b) predicted from a shelter collapse [7.6]. The distance from the shelter is given on both axes.

Releases from a shelter collapse could lead to some increased exposure of people living downstream of the most impacted area in the CEZ and who consume water and fish from the reservoirs. Radiation doses to individuals are discussed in Ref. [7.6], in which the highest values are predicted to be for professional fishermen and typical consumers.

(c) Impact on groundwater

Rainwater infiltration and condensation within the existing shelter have also been studied [7.6]. The radiological significance of the large pool of water in the basement of the shelter was confirmed. 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 an assessment of groundwater contamination without the 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 on the Pripyat River.

7.1.4.6. Impacts of shelter collapse within the new safe confinement

(a) Impact on air

Placement of the NSC over the shelter is expected to reduce the release of dust on to the site resulting from a collapse, thus reducing the magnitude of 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 the confinement capability designed into the NSC. The doses are expected to be reduced by factors of seven to 70 compared with the estimated doses in the event of a shelter collapse without the NSC, depending on the capacity of the NSC ventilation system [7.6]. This leads to an expected decrease of outdoor workers' exposure by a factor of two in comparison with the scenario of a shelter collapse without the NSC. However, some workers might be inside the NSC at the time of the collapse; doses to these workers might be increased because of containment of the dust.

For the small number of individuals who have chosen to reside within the CEZ, inhalation doses are expected to be reduced by factors of 50 to 500, to no more than 1 or 2 mSv [7.6]. Even assuming the worst (95th percentile) meteorological conditions and also assuming that the dust cloud passes over one of the larger cities, such as Slavutych, an increase of latent fatal cancer risk projected for the population in the event of a collapse with the NSC is not expected.

Very minor additions to soil contamination would be caused by the discharge and deposition of airborne radionuclides should the shelter collapse inside the NSC. Within the boundaries of the CEZ, the radionuclide deposition 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 south-west towards 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.

(b) Impact on surface water

Emplacement of the NSC would ensure that, in the event of a collapse, additional deposition of radionuclides on surface water would be minimal. The depositions illustrated in Fig. 7.7 would be reduced by factors of 50 to 500 [7.6], and the resulting concentrations in downstream waters would not exceed the Ukrainian norms.

(c) Impact on groundwater

The dynamics of radionuclide migration into the groundwater with the presence of the NSC was evaluated assuming that the water level is reduced to zero in the basement one and a half years after the NSC is constructed. After NSC construction, the precipitation fluxes are expected to be minimized and evaporation fluxes will be higher than fluxes from dust suppression and condensation. This means that the water level in the basement is expected to diminish due to seepage through the walls, and the room would therefore be empty in less than two years.

7.1.5. Issues and areas for improvement

7.1.5.1. Influence of the source term uncertainty on environmental decisions

There is still considerable uncertainty regarding the amount of nuclear fuel remaining in unit 4. One estimate [7.1, 7.19] is that inside unit 4 there is approximately 95% of the 190 t of nuclear fuel (as uranium) that was in the reactor at the time of the accident. Another estimate [7.12] is that 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). The estimated radioactivity inside the shelter in 1995 was approximately 7×10^{17} Bq [7.3]. Despite these and other 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 the safety and environmental consequences of unit 4 and the shelter evolution, as well as for the selection of adequate solutions for the long term management of the associated radioactive waste.

7.1.5.2. Characterization of fuel-containing material

The physical state of the 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 an unknown oxidation rate, particle size and behaviour. Another related important uncertainty is on the dust distribution in the shelter, and more specifically in the NSC atmosphere over long term operation of the facility. As estimates of the environmental impacts (e.g. transport and inhalation calculations) of long term shelter development are sensitive to the assumptions about these source term parameters, it is necessary that these parameters be further investigated. This would contribute to increased confidence in the safety assessment results and the selection of appropriate protective measures for workers, the public and the environment.

7.1.5.3. Removal of fuel-containing material concurrent with development of a geological disposal facility

The stabilization 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. A long term management strategy for the FCM and long lived radioactive waste therefore needs to be developed to ensure safe management of this waste.

It can 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 the unstable structures of the shelter, continuing with radioactive waste predisposal management and temporary storage on the Chernobyl nuclear power plant site until the geological disposal facility becomes available. Due to the high content of long lived radionuclides there is also no significant worker dose benefit to be obtained from 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.

7.2. MANAGEMENT OF RADIOACTIVE WASTE FROM THE ACCIDENT

In the course of remediation activities, both at the Chernobyl nuclear power plant site and in its vicinity, large volumes of radioactive waste were generated and placed in temporary near surface waste storage facilities located in the CEZ (Fig. 7.8) at distances of 0.5–15 km from the nuclear power plant site. Sites for temporary waste storage of the

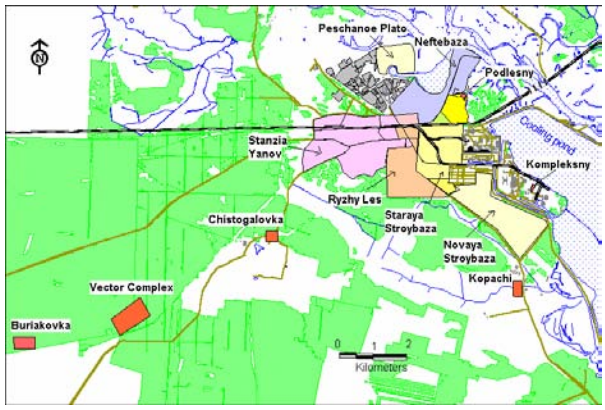

FIG. 7.8. Temporary radioactive waste disposal facilities in the territory of the CEZ.

TABLE 7.1. UKRAINIAN SOLID RADIOACTIVE WASTE CATEGORIZATION [7.21]

	Range of specific activity (kBq/kg)			
	Group 1[a]	Group 2[a]	Group 3[a]	Group 4[a]
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	$>10^{5}$	$>10^{6}$	$>10^{7}$	$>10^{8}$

[a] Group 1: transuranic alpha radionuclides; group 2: alpha radionuclides (excepting transuranium); group 3: beta and gamma radionuclides (excluding those in group 4); group 4: ^{3}H, ^{14}C, ^{36}Cl, ^{45}Cf, ^{53}Mn, ^{55}Fe, ^{59}Ni, ^{63}Ni, ^{93m}Nb, ^{99}Tc, ^{109}Cd, ^{135}Cs, ^{147}Pm, ^{151}Sm, ^{171}Tm and ^{204}Tl.

trench and landfill type were created from 1986 to 1987, intended for radioactive waste generated after the accident as a result of the cleanup of contaminated areas, to avoid dust spread, reduce radiation levels and provide better working conditions at unit 4 and its surroundings. These facilities were established without proper design documentation, engineered barriers or hydrogeological investigations, which are required by 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 the management of existing radioactive waste. However, as reported in some Ukrainian studies [7.20], to date a broadly accepted strategy for radioactive waste management at the Chernobyl nuclear power plant site and the CEZ, and especially for high level and long lived waste, has not been developed. Some of the reasons are the large number of 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 in the documented radioactive waste inventories (volume, activity, etc.).

The existing radioactive waste from the accident and potential radioactive waste to be generated during the NSC construction, shelter dismantling, FCM removal and decommissioning of unit 4 can be categorized as:

(a) Radioactive waste from the shelter and the nuclear power plant site that will be created by construction of infrastructure and the NSC;
(b) Accident generated transuranic waste that has been mixed with radioactive waste from operations at Chernobyl nuclear power plant units 1, 2 and 3;

TABLE 7.2. CLASSIFICATION OF RADIOACTIVE WASTE WITH UNKNOWN SPECIFIC ACTIVITY USING THE DOSE RATE AT 0.1 m DISTANCE [7.21]

	Dose rate (μGy/h)
Low activity	1–100
Intermediate activity	100–10 000
High activity	>10 000

(c) Radioactive waste in temporary radioactive waste facilities located throughout the CEZ;
(d) Radioactive waste in existing radioactive waste disposal facilities.

The safety and environmental issues related to each of these categories of radioactive waste are presented in this section. The radioactive waste expected to be generated during the decommissioning of the Chernobyl nuclear power plant units 1, 2 and 3 represents an additional category that is not the subject of this report.

The current Ukrainian legislation applies a categorization of radioactive waste in accordance with its specific activity and radiotoxicity, specified in Table 7.1 [7.21].

For waste contaminated with unspecified mixtures of radionuclides emitting gamma radiation, use of the classification ‘low’, ‘intermediate’ and ‘high’ activity is allowed using the air dose rate at a distance of 0.1 m, as specified in Table 7.2 [7.21].

Current radioactive waste management practice in Ukraine does not fully comply with the above classification; measures are therefore being taken to bring it into conformity with the new regulations [7.22].

7.2.1. Current status of radioactive waste from the accident

7.2.1.1. Radioactive waste associated with the shelter

The shelter is considered to be "the destroyed unit 4 after a radiological accident" and "a near surface storage facility for unconditioned radioactive waste at a stage of stabilization and reconstruction" [7.22, 7.23]. The amount and type of waste, debris and other radioactive material inside the shelter is presented in Table 7.3.

In addition, soil that was heavily contaminated by the deposition of fuel fragments and with radionuclides and debris from the accident (metal pieces, concrete rubble, etc.) was also collected and stored in the vicinity of unit 4:

(a) Three pioneer walls (west, north and south of the shelter), where contaminated soil, concrete and containers are stored and which contain an estimated 1700–4900 m^3 of high level waste[5] and up to 72 000 m^3 of low and intermediate level waste [7.25, 7.26].
(b) The cascade wall north of the shelter, where core fragments, metal, concrete, core pit equipment and accident cover material are stored (16 600 m^3 of high level waste, 117 t of reactor core elements and 53 400 m^3 of low and intermediate level waste) [7.25].
(c) The industrial site around the shelter, where concrete, gravel, sand, clay and contaminated soil are stored that contain 7000 m^3 of high level waste and 286 000 m^3 of low and intermediate level waste [7.27]. Other studies show that fuel, graphite, etc., are located in the contaminated soil [7.26].

The radioactive waste inside the pioneer and cascade walls was later covered with concrete. This material is considered to be high level waste that is not acceptable to be disposed of in near surface disposal facilities. Since 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.

[5] High level waste falls into two subcategories: low temperature waste with a heating rate of less than 2 kW/m^3 and heat generating waste with a heating rate higher than 2 kW/m^3 [7.24].

TABLE 7.3. ESTIMATED INVENTORY IN THE SHELTER [7.25]

	Type of radioactive waste and criteria of assessment	Category of radioactive waste	Amount
FCM	Fresh fuel assemblies, spent fuel assemblies, lava type material, fuel fragments, radioactive dust	High level	About 190–200 t, 700 t of graphite
Solid radioactive waste with less than 1% nuclear fuel (mass)	Fragment of the core with a dose rate at 10 cm of more than 10 mSv/h		
Liquid radioactive waste	Changing inventory based on precipitation (e.g. pulp, oils, suspensions with soluble uranium salts)	Low level (up to 3.7×10^5 Bq/L)	2500–5000 m^3
		Intermediate level (more than 3.7×10^5 Bq/L)	500–1000 m^3
Solid radioactive waste	Metal equipment and building material, for example concrete, dust, non-metal material (organic, plastic)	High level	38 000 m^3 (building material), 22 240 t (metal constructions)
		Low and intermediate level	300 000 m^3 (building material and dust), 5000 m^3 (non-metal)

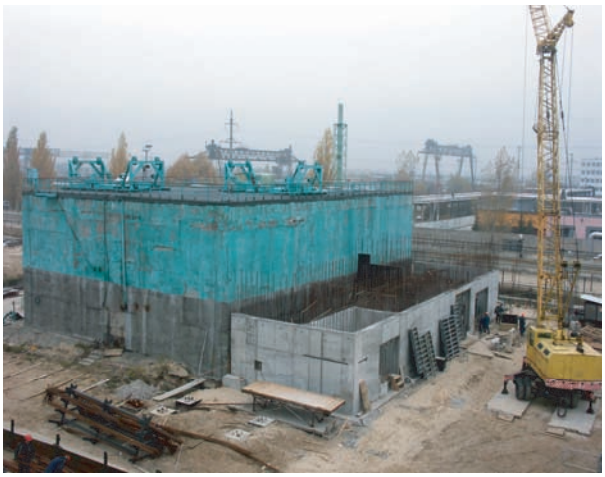

FIG. 7.9. Existing above ground storage facility for solid radioactive waste at the Chernobyl nuclear power plant site.

It is estimated 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. FCM) according to Ukrainian legislation [7.28, 7.36].

7.2.1.2. Mixing of accident related waste with operational radioactive waste

During 1986–1993, some low and intermediate level radioactive waste and high level waste with transuranic elements were stored together with some operational radioactive waste from units 1, 2 and 3 in an above ground storage facility (see Fig. 7.9) at the Chernobyl nuclear power plant site.

This waste amounts to about 2500 m^3, with a total radioactivity of about 131 TBq [7.19], and is stored unconditioned. Once filled, the storage facility was backfilled with concrete grout and covered with 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 present, this facility is being extended and is intended to be used for the disposal of radioactive waste produced during the decommissioning of units 1, 2 and 3.

7.2.1.3. Temporary radioactive waste storage facilities

The largest volumes of radioactive waste generated by unit 4 remediation activities are located in the CEZ (see Fig. 7.8). Sites for temporary storage of radioactive waste, of the trench and landfill type, were constructed shortly after the accident at distances of 0.5–15 km from the nuclear power plant site. They were created from 1986 to 1987 and intended for radioactive waste generated after the accident as a result of the cleanup of contaminated areas to avoid dust spread, reduce radiation levels and provide 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 the total volume of disposed radioactive waste estimated to be over 10^6 m^3. The main inventories of activity are concentrated in the Stroibaza and Ryzhy Les temporary radioactive waste facilities along the western trace of the Chernobyl fallout (see Fig. 7.8). The specific activity of the radioactive waste in the temporary radioactive waste facility at Ryzhy Les is 10^5–10^6 Bq/kg of ^{90}Sr and ^{137}Cs and 10^3–10^4 Bq/kg of plutonium isotopes (total).

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 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 setting, in which waste was stacked and covered with a layer of soil from the nearby environment. These facilities are therefore very variable with regard to their potential for release, which depends 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 have been studied (see Table 7.4) [7.19, 7.29].

There are also many other temporary radioactive waste facilities, estimated to comprise about 800 trench facilities each with waste disposal volumes in the range of 8×10^2 to 2×10^6 m^3 [7.29, 7.30]. The inventories of these facilities are known for about half of them. The facilities are not under regulatory control. Estimates made for a few sites show that their radioactive contents can be high (10–1000 TBq), sometimes of an order of magnitude comparable with the total radioactivity present in soil in the CEZ (about 7000 TBq) [7.30].

7.2.1.4. 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 the disposal of higher level accident waste than the radioactive waste located in the temporary radioactive waste facilities [7.19].

Buriakovka, built in 1987, is the only disposal facility currently in operation in the CEZ. It comprises 30 trenches covered with a 1 m clay layer and is located on 23.8 ha. Up to 652 800 m^3 of radioactive waste has been disposed of. 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. It consists of metal, soil, sand, concrete and wood contaminated with ^{90}Sr, ^{137}Cs, ^{134}Cs, $^{238,239,240}Pu$, $^{154,155}Eu$ and ^{241}Am. Radioactive waste with dose rates at 10 cm from the surface in the range of 0.003–10 mGy/h was accepted in this facility.

The Podlesny vault type disposal facility was commissioned in December 1986 and closed in 1988. The facility was designed for the disposal of high level waste with a dose rate 10 cm from the surface in the range of 0.05–2.5 Gy/h. Material with dose rates above this was also disposed of in the facility. The total radioactive waste volume of 11 000 m^3 of building material, metal debris, sand, soil, concrete and wood was placed in two vaults. The disposal facility was covered with concrete at its

TABLE 7.4. STATUS OF TEMPORARY RADIOACTIVE WASTE FACILITIES [7.19, 7.29]

	Size (ha)	Number of trenches	Number of landfills	Radioactive waste type	Radioactive waste volume (10^3 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[a] 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	$>4 \times 10^{13}$
Ryzhy Les	227	Estimated at more than 61	Estimated at more than 8	Mainly soil, some construction and domestic material	500	Up to 4×10^{14}
Staraya Stroibaza	130	More than 100	—	Soil, metal, concrete and wood	171	1×10^{15}
Novaya Stroibaza	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 work clothes	160	4×10^{12}
Kopachi	125	—	—	Construction waste from demolition	110	3×10^{13}

[a] According to Ukrainian legislation, short lived waste is radioactive waste whose release from regulatory control is achieved earlier than 300 years after disposal; long lived waste is radioactive waste whose release from regulatory control is achieved later than 300 years after disposal [7.21].

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 may be higher than initially estimated, and a need for a re-estimation of the current inventory was identified. Due to the uncertainties in the inventory, it is assumed that various types of waste were disposed of, including FCM.

The Kompleksny vault type facility was based on reconstructed facilities of the unfinished units 5 and 6 at the Chernobyl nuclear power plant site. Kompleksny was in operation from October 1986 until 1988 and was designed for low and intermediate level waste corresponding to dose rates up to 0.01 Gy/h at 10 cm from the surface of the waste container. More than 26 200 m^3 of solid waste with a total activity of 4×10^{14} Bq was disposed of in 18 000 containers and later covered with sand and clay. This waste is mainly sand, concrete, metal, construction material and bricks. Due to the high level of groundwater at different periods of the year, the facility is flooded 0.5–0.7 m above its bottom. Significant uncertainties exist associated with the radionuclide inventory because of the lack of data about the radioactive waste disposed of at the site.

At present, a new near surface facility, the Vektor complex, for low and intermediate level radioactive waste processing, storage and disposal, is under development. This complex will include [7.19]:

(a) An engineering facility for the processing of all types of solid radioactive waste (capacity of 3500 m^3/a);
(b) A disposal facility for short lived solid radioactive waste (55 000 m^3 total capacity);
(c) A storage facility for long lived solid radioactive material;
(d) A storage facility for FCM;
(e) Intermediate storage for high level conditioned radioactive waste to be prepared for final disposal at the deep geological disposal facility.

7.2.2. Radioactive waste management strategy

At present, no further dismantlement and cleanup 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 liquid [7.6]. It also requires the removal of 100 000 m^3 of contaminated soil around unit 4, which may still contain fuel fragments. Preliminary studies for the dismantlement of the shelter superstructure predict that about 1200 t of steel, with an estimated volume of radioactive waste of 1800 m^3 [7.14], mainly metal and large concrete pieces, will be removed. This waste is planned to be sorted based upon its radiation level. High level waste, which is expected to be only a small part, is planned to be placed in containers and stored within the NSC.

According to Ukrainian legislation [7.31], all industrial radioactive waste is classified according to the scheme shown in Fig. 7.10: 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 a near surface disposal facility. Following these criteria, a strategy for the 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 level waste are to sort the waste according to its physical characteristics (soil, concrete, metal, etc.) and, possibly, to 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 [7.19] or the Vektor disposal site.

The long lived waste is planned to be placed in interim storage. Different storage options are being considered at the Chernobyl nuclear power plant or the Vektor site, and a decision has not yet been made. After construction of the NSC, decommissioning of

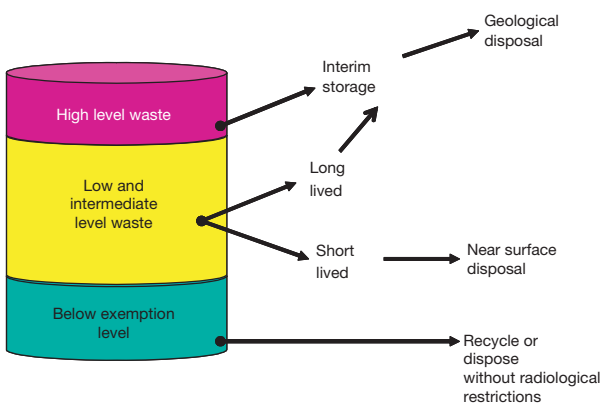

FIG. 7.10. Planned management of radioactive waste at the Chernobyl nuclear power plant site [7.19].

the shelter facilities is envisaged, including shelter dismantlement and further removal of FCM. High level radioactive waste will be partially processed in place and stored at a temporary storage site until a deep geological disposal site is ready. At present, this strategy is considered as the preferred option for high level radioactive waste and FCM [7.19]. To implement this strategy, it is planned to organize a system for processing and temporary storage of the high level and long lived radioactive waste at the Vektor facility complex that 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 in the comprehensive programme on radioactive waste management that was approved by the Ukrainian Government [7.25]. Prior to elaboration of such a programme, special field studies and geological investigations must be carried out in the CEZ and its surrounding area, and in particular in the areas with crystalline rocks that have a depth of more than 500 m. According to Ref. [7.25], it is considered reasonable to begin an 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 to be performed at the new liquid radioactive waste treatment plant at the Chernobyl nuclear power plant site. However, the management of liquid waste containing transuranic elements remains an issue to be resolved.

In addition, in the strategy for management of radioactive waste from the accident, account should be taken of the management of other storage sites containing about 2000 pieces of contaminated equipment (transport vehicles, helicopters, tanks, etc.) that were used in the first months after the accident and for which the final disposition has not been determined.

7.2.3. Environmental aspects

Safety concerns in relation to most of the temporary radioactive waste facilities in the CEZ need to be viewed within the context that most of these facilities are located in very contaminated areas, with surface levels of ^{90}Sr in the range of 400–20 000 kBq/m^2, 700–20 000 kBq/m^2 for ^{137}Cs and 40–1000 kBq/m^2 for ^{239,241}Pu. In this same territory, the temporary radioactive waste facilities occupy a relatively small volume covered with several metres 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 water sources used as water supplies. The measurements reported in the French–German initiative [7.32] (see Table 7.5) clearly show that some temporary radioactive waste facilities have a significant influence on groundwater. In particular, flooded and partially flooded trenches are giving rise to enhanced migration, due to the absence of engineered safety features. More favourable settings, such 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

TABLE 7.5. CONTAMINATION OF GROUNDWATER NEAR SELECTED TEMPORARY RADIOACTIVE WASTE STORAGE FACILITIES IN 1994–1995 [7.16] AND IN 1999 [7.23]

	Strontium-90 (Bq/L)		Caesium-137 (Bq/L)		Plutonium-239, 240 (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
Stroibaza	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

the bottom of the disposal facility [7.19]. The degree of contamination is monitored in these disposal sites using a monitoring system, established in 1986–1989, that needs upgrading.

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 [7.19, 7.33]. The highest levels of contamination are detected at the northern part of the Chernobyl nuclear power plant site, groundwater from which also runs into the Pripyat River. Therefore, actual and potential impacts from radionuclides exist for those radioactive waste facilities located immediately next to the riverside in alluvial soils and which might be at regular risk of flooding during high water periods [7.20, 7.32]. These types of disposal facility have been studied during the past five years and continue to be studied as a basis for their step by step removal and relocation to the properly established disposal facilities.

As mentioned above, the rate of radionuclide migration with groundwater 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 adsorption capacity of the soils and geological media surrounding the temporary radioactive waste facilities, several studies have shown that a significant fraction of ^{90}Sr is still associated with the fuel matrix, which delays its release to the pore water in the soil for many years. As a result, radionuclide concentrations in groundwater, even for such mobile radionuclides as ^{90}Sr, are very low. Plutonium isotopes (and ^{241}Am associated with them) have not yet been adequately studied; however, it is well known that their migration beyond the temporary radioactive waste sites is negligible (Fig. 7.11).

Studies of the vertical and lateral transfer rates of radionuclides demonstrated that, for the local soil, there is a low risk of radionuclide contamination in groundwater and therefore a proportionately low risk of significant contamination of the Pripyat River in the future, as discussed also in Section 3.5 (see Fig. 3.58). It has been shown that the leading edge of contaminated groundwater from most of the significant temporary radioactive waste facilities may reach regional surface water within 100 or more years, making this issue of minimal importance in terms of radiological impact for populations living downstream of the Pripyat River system [7.17, 7.34]. However, for the CEZ, groundwater is still an important potential source for radionuclide migration in the environment, and therefore the waste facilities have to be under regular monitoring and institutional control.

The long term strategy for the temporary facilities is related to the management of the

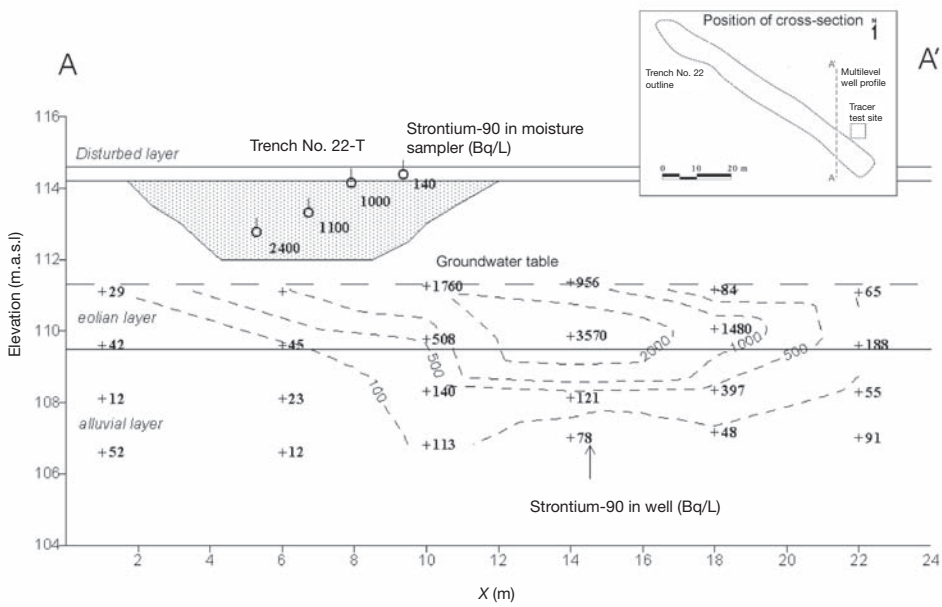

FIG. 7.11. Spatial distribution of ^{90}Sr (Bq/L) in the groundwater near surface trench No. 22 of the Ryzhy Les facility in October 1998 [7.34].

associated radiological risks. The ultimate goal should be that waste is disposed of 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 on potential critical groups. For those temporary radioactive waste facilities located near the banks of the Pripyat River that could be inundated by floodwaters, the preferred strategy is to remove and relocate the waste into the properly established disposal facilities.

For the temporary radioactive waste facilities in the CEZ whose inventory is not known, and for which the potential for future contamination of surrounding groundwater and surface water is uncertain, safety assessments need to be performed, taking into account radioactive decay and natural attenuation. There is clearly a need to assess, with an increased level of confidence, the migration of contamination plumes and their interface with major water resources and supply areas (aquifers, rivers, reservoirs, local supplies for the nuclear power plant and the CEZ). Such assessments need to consider all sources of release likely to affect these water resources.

The safety assessment results will guide decisions 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 CEZ occurs after some hundreds of years. It is obvious that institutional control at such disposal sites will need to be maintained for a period of a few hundred years to allow ^{137}Cs and ^{90}Sr activities to decay to insignificant levels. This will require significant resources for monitoring, implementation of recovery actions and probably major restrictions on resettlement. However, long term institutional controls should not be considered as an alternative to recovery actions to improve overall safety in the CEZ.

7.2.4. Issues and areas of improvement

7.2.4.1. Radioactive waste management programme for the exclusion zone and the Chernobyl nuclear power plant

A comprehensive programme for radioactive waste management has not yet been established for further cleanup of contaminated areas or temporary radioactive waste facilities at the Chernobyl nuclear power plant and within the CEZ. As mentioned earlier, the ongoing strategy is to monitor the temporary waste sites with the highest radiological risk to the environment, so as to assess whether cleanup or environmental protection actions are needed. In addition, options for the long term processing, storage and disposal of long lived and high level waste from the Chernobyl nuclear power plant and the CEZ, 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 ensure the consistent and coordinated long term management of all types of accident waste and hence provide protection of workers, the public and the environment.

7.2.4.2. Decommissioning of unit 4

Two main factors need to be addressed within the strategy for dismantlement of the shelter and decommissioning of unit 4: the safety implications of the 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 of 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 consideration given to the capacity of these facilities and also the possibility of using the existing facilities for the decommissioning of the Chernobyl nuclear power plant. Particular attention needs to be given to the establishment of an adequate infrastructure and facilities for the management of long lived waste (in particular large quantities of soil, transuranic liquid waste and contaminated metal) and high level waste (i.e. FCM) and their subsequent disposal.

7.2.4.3. Waste acceptance criteria

The waste management programme being implemented does include criteria for the categorization of accident radioactive waste, which are needed for the selection of an appropriate management option for individual radioactive waste streams. Adoption of criteria for waste management, based on ^{137}Cs and alpha specific radioactivity levels in the waste, is under

development. Although such criteria are more adapted to appraising the potential for waste to be accepted in near surface facilities, the question of estimating the acceptable 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 ensure protection of workers and the environment, as well as the public, in the long term.

7.2.4.4. Long term safety assessment of existing radioactive waste storage sites

There is a need to identify the remaining temporary radioactive waste storage facilities and to 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 prioritize the needs for safety assessment. These assessments should evaluate safety in the present conditions and with consideration of possible future resettlement. Consideration should be given to the need to restrict the number of sites that are flooded or will, in the future, need extensive control over some hundreds of years.

In order to select those facilities with higher radiological risk it is important to improve methods for assessing the radioactive content of the waste 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 estimates of the potential impact of individual facilities on the environment will be reduced, and a consistent assessment that takes into account all existing and potential sources of contamination in the CEZ will become feasible.

7.2.4.5. Potential recovery of temporary waste storage facilities located in the Chernobyl exclusion zone

Work is under way on the development of a strategy for the management of temporary waste storage facilities; this envisages three options for the different facilities, depending on their status and radiological hazard to the environment [7.19, 7.29]:

(a) Retrievability of waste and disposal in the short term in order to minimize environmental consequences and improve the safety of workers; for example, industrial sites, the shelter, the flooded temporary storage facilities and the Kompleksny disposal facility.
(b) Possible temporary storage of waste under institutional control in accordance with radiation protection requirements with a view to future disposal; for example, the Podlesny disposal facility and contaminated equipment from the activities aimed at mitigation of the Chernobyl accident.
(c) Investigation of facilities that need to be studied in order to decide on adequate intervention measures; for example, temporary radioactive waste facilities and soil from the construction of the NCF.

7.3. FUTURE OF THE CHERNOBYL EXCLUSION ZONE

The long term development of the CEZ is an important and complex task that must consider various technical, economic, social and other factors; various options have been considered for the evolution of this zone. According to Likhtarev et al. [7.35], after 2015 about 55% of the territory around the Chernobyl nuclear power plant could be considered for release from radiological limitations according to Ukrainian legislation. However, the final decision on permitting people to return 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 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 CEZ is to recover the affected areas of the CEZ, redefine the CEZ 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 growing of food crops and cattle grazing and the use of only clean feed for cattle. Accordingly, these resettled areas are best suited for an industrial site rather than for a residential area.

For the reasons given above, the activities focused on decontamination and dismantlement of the

shelter and on radioactive waste management in this territory are expected to continue, which requires optimal management of this area. The new concept foresees division of the CEZ into different sections:

(a) The industrial zone is planned to include the most contaminated areas, where the Chernobyl nuclear power plant, 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 the infrastructure for NSC construction, new roads, shipping yards, railways and other support structures are planned. The town of Chernobyl has been considered as an option for such infrastructure development [7.6]. If the CEZ is selected as the site for construction of the geological repository for high activity and long lived radioactive waste, a significant amount of drilling and mining work will have to be performed, which will also require specific development of the engineering infrastructure.
(b) The sanitary restricted zone is considered to be a buffer area between industrial and nature reserve areas.
(c) The nature reserve areas are planned to be located where most industrial and human activities are prohibited, with the aim of preservation of the basic natural landscapes and biodiversity of the region.

The rehabilitation of the CEZ is expected to create optimal conditions for industrial activity and environmental protection for a long period of time; for example, the NSC is expected to be operational for at least 100 years. Different types of radioactive storage facility must provide safe storage for 300 or more years. A possible activity in 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 repository from different parts of Ukraine.

Continued monitoring and support studies in the CEZ 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, the need for additional engineering barriers and the implementation of environmental remediation technologies.

In summary, the future of the CEZ for the next hundred years and more is envisaged to be associated with the following activities:

(i) Construction and operation of the NSC and relevant engineering infrastructure;
(ii) De-fuelling, decommissioning and dismantling of units 1, 2 and 3 of the Chernobyl nuclear power plant and shelter;
(iii) Construction of facilities for the processing and management of radioactive waste, in particular a deep geological repository for high activity and long lived radioactive material;
(iv) Development of nature reserves in the area that remains closed to habitation;
(v) Maintenance of environmental monitoring and research activities.

7.4. CONCLUSIONS AND RECOMMENDATIONS

7.4.1. Conclusions

It can 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, the public and the environment in the future. Therefore, continuation of the stabilization measures at the shelter and construction of the NSC are expected to improve safety and prevent or mitigate accident scenarios that would be expected to have consequences outside the CEZ.

It is also required that prompt solutions be found for the safe predisposal and disposal management of the radioactive waste to be generated during this period, in particular for the management of long lived and high level waste. Planning and evaluation of safety for the decommissioning of unit 4 after the construction of the NSC is needed in order to develop appropriate measures and to allocate necessary resources for the conversion of the shelter into a safe environmental system.

The 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 Chernobyl nuclear power plant and the CEZ. A comprehensive strategy for the management of all waste streams is needed to ensure adequate infrastructure and capabilities for the processing, storage and disposal of this waste. Such a strategy will also need to take into account 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 present an unacceptable hazard to the public; however, an assessment of their long term impact on the public and the environment is needed. This should be done taking into account the remaining sources of radioactive contamination in the CEZ, and particularly those facilities that are flooded and represent higher risks.

For the less known and less studied waste facilities it will be necessary to reduce the uncertainties associated with the waste inventories and facility characteristics, assess their long term safety, 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 CEZ and the conversion of the zone into a safe environmental system.

7.4.2. Recommendations

Recognizing the ongoing effort on improving safety and addressing the aforementioned uncertainties in the existing input data, the following main recommendations are made regarding the dismantling of the shelter and the management of the radioactive waste generated as a result of the accident.

(a) Since individual safety and environmental assessments have been performed only for individual facilities at and around the Chernobyl nuclear power plant, a comprehensive safety and environmental impact assessment, in accordance with international standards and recommendations, that encompasses all activities inside the entire CEZ, should be performed.

(b) During the preparation and construction of the NSC and soil removal, special monitoring wells are expected to be destroyed. Therefore, it is important to maintain and improve the environmental monitoring strategies, methods, equipment and staff qualification needed for the adequate performance of monitoring of the conditions at the Chernobyl nuclear power plant site and the CEZ.

(c) The dismantling of the shelter after a delay of 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. This long term strategy raises concerns related to the potential loss of the most experienced personnel at the Chernobyl nuclear power plant and the maintenance of a stable workforce necessary for the safe operation of the NSC. It is reasonable, therefore, to begin retrieving FCM soon after dismantling the unstable structures of the shelter rather than waiting for the availability of a geological disposal facility.

(d) Development of an integrated radioactive waste management programme for the shelter, the Chernobyl nuclear power plant site and the CEZ is needed to ensure application of consistent management approaches and sufficient facility capacity for all waste types. Specific emphasis needs to be given to the characterization and classification of waste (in particular waste with transuranic elements) from all remediation and decommissioning activities, as well as to the establishment of sufficient infrastructure for the safe long term management of long lived and high level waste. Therefore, development of an appropriate waste management infrastructure is needed in order to ensure sufficient waste storage capacity; at present, the rate and continuity of remediation activities at the Chernobyl nuclear power plant site and in the CEZ are being limited.

(e) A coherent and comprehensive strategy for the rehabilitation of the CEZ is needed, with particular focus on improving the safety of the existing waste storage and disposal facilities. This will require development of a prioritization approach for remediation of the sites, based on safety assessment results, aimed at making decisions about those sites at which waste will be retrieved and disposed of and

those sites at which the waste will be allowed to decay in situ.

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The explosion on 26 April 1986 at the Chernobyl nuclear power plant and the consequent reactor fire resulted in an unprecedented release of radioactive material from a nuclear reactor and adverse consequences for the public and the environment. Although the accident occurred nearly two decades ago, controversy still surrounds the real impact of the disaster. Therefore the IAEA, in cooperation with the Food and Agriculture Organization of the United Nations, the United Nations Development Programme, the United Nations Environment Programme, the United Nations Office for the Coordination of Humanitarian Affairs, the United Nations Scientific Committee on the Effects of Atomic Radiation, the World Health Organization 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 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 in which further research is required. This report presents the findings and recommendations of the Chernobyl Forum concerning the environmental effects of the Chernobyl accident.

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