Chapter 5
Environmental Pathways of Radionuclides to Animal Products in Different Farming and Harvesting Systems

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This chapter briefly describes the NREs which released large amounts of radionuclides that had the potential to cause significant contamination of animals and animal products. It then describes the key environmental and metabolic pathways of animals and animal product contamination. The different methods used to quantify the transfer of radionuclides between relevant environmental pathways are also described. Radionuclide-specific information is provided in subsequent sections. Observed effects on agricultural and game animals after two NREs are also described.

5.1 Major Nuclear or Radiological Emergencies Causing Animal and Animal Product Contamination

There have been a range of different NREs that have contaminated animal and animal products. Animal products have been contaminated after all of the four largest NREs that have occurred from nuclear reactors or waste storage facilities. Estimated radionuclide releases from these four sources are listed in Table 5.1. Most of the radionuclides listed in Table 5.1 may be important contributors to internal exposure to humans via animal products after a NRE.

Although many different radionuclides can be released following a NRE, some are short-lived, and others do not readily transfer into food. Additional radionuclides, not listed above, of potential relevance for animal products after NREs include ^{3}H, ^{14}C, ^{35}S, ^{60}Co, ^{95}Nb, ^{99}Tc, ^{103}Ru, ^{106}Ru, ^{110}Ag, ^{129}I, ^{132}Te, ^{192}Ir, ^{235}U and ^{241}Am. The relative importance of these different radionuclides varies depending on the magnitude of the release and on environmental and agricultural husbandry

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Table 5.1 Estimated releases of selected radioisotopes for the four largest NREs which led to animal product contamination

Radioactive atmospheric releases (TBq)				
Isotopes	Kyshtym	Windscale	Chernobyl	Fukushima Daiichi
Reference source	Akleyev et al. (2017)	Garland and Wakeford (2007)	UNSCEAR (2011)	IAEA (2015)
^{131}I		1800	1,760,000	100,000–400,000
^{137}Cs	260	180	~85,000	7000–20,000
^{134}Cs		12	~47,000	8300–50,000
^{210}Po		42		
^{90}Sr	4000	0.75	10,000	3.3–140
Pu isotopes	1.5	0.02	46[a] and 2600[b]	0.0034–0.025[a] and 0.0003–1.2[b]
^{95}Zr	18,400	16	84,000	17
^{144}Ce and ^{141}Ce	48,700	13	134,000	29
^{106}Ru	2700	3	>73,000	0.002

[a]Pu alpha
[b] ^{241}Pu

characteristics. For animals and animal products, it also depends heavily on the extent to which the radioisotopes are accumulated by animal tissues – this issue is addressed in Sect. 5.4.3.

Examples of the features controlling the contamination of animal products and their consequences are given in this chapter based on information acquired after each of the four NREs.

5.2 Key Environmental Processes Controlling Animal Product Contamination

There are a large number of different environmental factors which affect the extent to which radionuclides, such as those listed in Table 5.1, will accumulate in animals and animal products in the human food chain. Some factors are more important in the emergency phase after a NRE whilst others are more relevant in the transition to recovery phases.

They include:

- Interception on, and loss from, plant surfaces
- Chemical form
- Soil fixation processes
- Rates of plant uptake
- Diet of food-producing animals
- Absorption rates in the gut of animals

- Transfer rates to tissues (including milk)
- Dynamic changes with time in tissue contamination
- Diet and habits of humans

Some of these processes are highly dependent on which radionuclides have been released (such as soil fixation and gut absorption), whereas others are not (such as interception and human dietary preferences).

There are definable situations where there is substantial transfer of radionuclides into food products caused by particular features of the release or the contaminated system. In such situations, the feature is considered to be radioecologically sensitive to that radionuclide (Howard 2000). A typical example is the presence of certain soil types which fail to permanently fix radiocaesium ions to soil particles, thereby allowing continued transfer into the soil solution and subsequent uptake by plants and then animals (Fig. 5.1).

Milk and meat products can become contaminated rapidly, especially if radionuclides are released to the atmosphere. Radionuclides in milk can be a major source of internal dose via the human food chain soon after a release. Radioiodine (especially ^{131}I), radiocaesium (^{134}Cs and ^{137}Cs) and ^{90}Sr are often key components of ingestion dose via animal products, potentially over decades for ^{137}Cs and ^{90}Sr. The radioactive contamination of animals and animal products impacts not only on farmers and consumers but also on agricultural and regulatory ministries and the

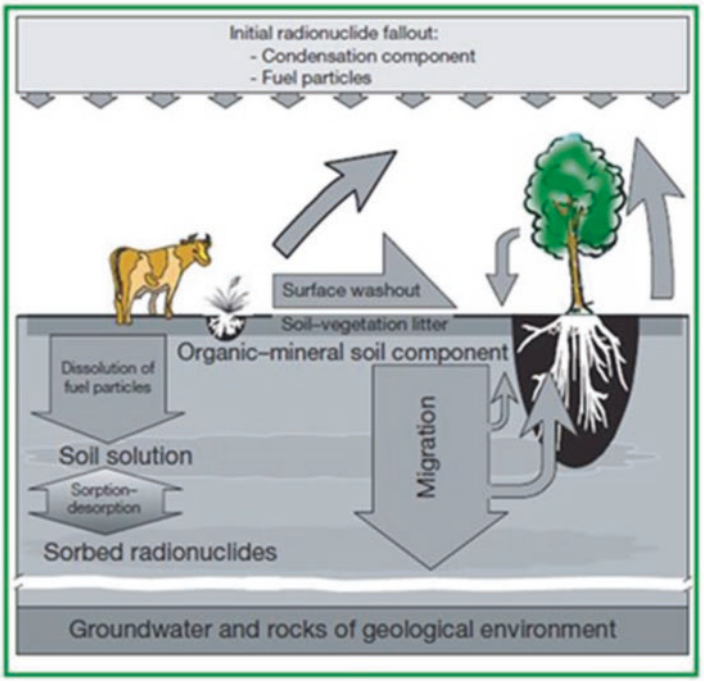

Fig. 5.1 Routes of radionuclide transfer in the environment (IAEA 2006a)

food industry. Professional groups that may be involved in the response to a NRE need to be informed about how animal products become contaminated and what controls the extent to which major radionuclides will be retained in, or lost from, animal tissues.

There are a number of different routes through which agricultural, free-ranging domesticated animals and game animals may become contaminated with radionuclides released from NREs. The key routes of contamination are:

- Inhalation into the lung of gaseous radionuclides or particulates present in the atmosphere, or of resuspended contaminated material such as windblown soil particles. Such pathways are only relevant for emergency stages after a NRE when radionuclides may be present in the air.
- Direct uptake of radionuclides in volatile, gaseous forms via plant stomata.
- Direct deposition of radionuclides onto external surfaces of plants (such as leaves, bark, grain and other edible parts) and animals (such as fur, feathers, skin).
- Ingestion of plants, fungi and soil contaminated with radionuclides by animals.
- Ingestion of contaminated water from sources such as water butts, surfaces of plants, puddles and streams by animals.

The relative importance of the above routes of contamination of animal products in the human food chain depends on the environmental pathways. The importance of these pathways depends on many factors such as the time of year that the NRE happened, the radionuclides released and the prevailing animal production or harvesting practices in affected areas.

Other characteristics that affect the extent of radionuclide contamination of animal products include the characteristics of the land used for production (such as the soil type and plant uptake rates), the extent of gastrointestinal absorption, the metabolic fate in the animal and the rate of loss from tissues (principally in urine, faeces and milk). These pathways are described in more detail below, focusing on aspects relating to the human food chain.

5.2.1 Vegetation Interception

The interception and retention of radionuclides by plants which are then consumed by grazing or browsing animals is a key process in the emergency phase after a NRE. It provides a fast and effective route for initial transfer of recently deposited radionuclides to animal products.

Once radionuclides are released into the air (or to water), various physical and chemical processes influence the extent to which they are transported and dispersed in the environment. The physical and chemical forms of the radionuclide, and the turbulence of the receiving medium (such as air movements and water flow), play an important role during the initial phase.

Other processes affect the transfer of radionuclides from the air (or the water column) to the receiving surface. Potential deposition mechanisms include:

- Aerosols washed from the atmosphere during precipitation
- Gravitational settling of suspended particulate material in the atmospheric or aquatic releases
- Impaction, whereby suspended particles come into contact with solid objects within an air or water stream
- Chemical sorption and exchange, dependent on both the chemical and physical form of the radionuclide and the interacting surface

Radionuclides interact with solid materials such as soil particles and sediments in many different ways including electrostatic attraction and the formation of chemical bonds. The radionuclide activity concentration per unit mass of solid is affected by the surface area available for adsorption per unit mass or volume and is, therefore, greater for smaller objects. In terrestrial areas, the interception of radionuclides by vegetation occurs for both wet and dry deposition.

Wet deposition occurs when radionuclides in air are washed out by precipitation. Vegetation surfaces retain a fraction of radionuclides deposited with the rain, with the remaining fractions falling onto the ground. The fraction of radionuclides in the air that is initially intercepted is an important quantity in radioecological models because direct deposition can lead to relatively high activity concentrations in pasture grazed by animals, and other feed crops.

Plants with a relatively high biomass per unit area will intercept more radionuclides in wet deposition, associated with a higher interception fraction. Other factors such as the capacity of the canopy to retain water, ionic form of the radionuclide, precipitation amount and intensity, vegetation maturity and leaf area index (LAI – upper-side green leaf area per unit ground surface area) can all influence the extent of interception of wet deposition (IAEA 2009). For example, the interception fraction of ^{137}Cs by grass was reported to decline with increasing intensity of rainfall from 0.1 to 0.2 for low rates of up to 1 mm of rainfall to an order of magnitude lower at higher rates of 11 mm of rainfall (Kinnersley et al. 1997).

Most of the intercepted radionuclides are gradually transferred to the soil and are only temporarily present on the surface of the vegetation. Radionuclide activity concentrations on vegetation may be reduced by various physical processes, including wash-off by rain or irrigation, surface abrasion, leaf bending from wind action, resuspension, tissue senescence, leaf fall, herbivore grazing, growth and evaporation.

Interception and retention of radionuclides on plant surfaces is a critical process in the emergency phase after a NRE. If a NRE occurs before the growing season, the likely transfer of radionuclides to grazing animals will be low, but may still occur if stored feed is not covered or animals are kept outdoors. Conversely, a NRE occurring at the height of the growing season with light rainfall when plant biomass is high and animals are outside grazing pasture may present an immediate problem to responding authorities. After the Chernobyl NRE, dairy cows in affected areas of the USSR were grazing pasture which had sufficient leaf mass in late April and early May to intercept significant amounts of radioiodine and radiocaesium.

Dry deposition is dependent on the characteristics of the intercepting surface, usually quantified using the surface roughness (Heinemann and Vogt 1980), which

generally increases as the plant canopy develops. The extent of interception for dry deposition depends on the standing biomass of plants, the chemical form and the particle size of the deposit. Interception is similar for small (up to a few micrometres diameter) particles, but as particle size increases, interception decreases probably because larger particles roll off the plant surface more easily than smaller ones. Furthermore, if vegetation is moist or wet, absorption increases possibly due to an enhanced stickiness. Particles with a diameter up to a few micrometres are relatively more important because larger particles from a radioactive cloud are rapidly depleted. As for wet deposits, the extent of interception of dry deposits depends on many factors including plant yield, particle size, the crop, the chemical form and whether the receiving surface characteristics are wet or dry.

Although relatively minor in comparison to the above routes, stored crops intended as fodder for animals may become contaminated by surface deposits of radionuclides if they are not covered outdoors.

Information is available on how to quantify interception in IAEA documents TECDOC 1616 and TRS 472 (IAEA 2009, 2010).

5.2.2 Chemical Form of the Released Radionuclides

The chemical form of the released radionuclides impacts on many different pathways, including the extent of interception, the rate at which radionuclides are released into the soil solution and are then available for plant uptake and the ability of the radionuclide to be absorbed in the animal's GI tract. Examples of the impact of chemical form will be given in the relevant sections below.

5.2.3 Radionuclide Behaviour in Soils

Plants take up nutrients and pollutants from the soil solution, so the radionuclide activity concentration in soil solution is a critical determining factor for plant uptake. The activity concentration of radionuclides in soil solution is determined by processes influencing the loss of radionuclides that are adsorbed onto soil components that move into the soil solution usually by competitive ion exchange (quantified as the cation exchange capacity). The concentration and composition of other elements present in the soil are important in determining radionuclide distribution between soil and soil solution. The amount and nature of clay minerals in soils and the concentrations of competitive major cations are often key factors in determining exchange mechanisms in soils of radionuclides, but other factors, such as microbial activity, may also affect radionuclide mobility.

In the emergency and transition phases of a NRE, radionuclide movement into the soil solution may be relatively high, leading to high initial contamination of plants via root uptake. With time the availability of radionuclides in soil solution

tends to reduce as radionuclides gradually adsorb to soil components. The rate of reduction varies with radionuclide and soil type.

Vertical migration of radionuclides down the soil column arises from various transport mechanisms including convection, dispersion, diffusion and biological mixing. Radionuclides can also migrate to deeper soil layers at faster rates when there is a high amount of rainfall over a short period of time, especially if there are surface cracks in dry soil or when soils contain a relatively large proportion of sand particles. Soil-dwelling animals can also relocate material both laterally and vertically during the construction of burrows, tunnels and chambers, and the roots of plants can cause a similar effect.

Large-scale lateral migration of radionuclides can also occur in catchments and is often associated with soil erosion or heavy rainfall events such as typhoons. The distribution of radionuclides in sediment or soil layers of the floodplain can be considerably altered by such events.

A high rate of radionuclide vertical migration in soil matter may be beneficial as it will remove radionuclides out of the rooting zone, thereby reducing external doses and plant uptake for surface routing species. However, for many undisturbed soils, most of the deposited radiocaesium is retained in the upper 10 cm layer.

5.2.4 Radionuclide Transfer from Soil to Crops

The uptake of radionuclides, as for other trace elements by plant roots, is a competitive physiological process (IAEA 2010). The processes influencing radionuclide transport from soil to plants vary with both radionuclide and soil type. The fraction of deposited radionuclides taken up by plant roots can differ by orders of magnitude between different elements and between different physico-chemical forms of the same radionuclide. There are also differences in radionuclide uptake between plant species growing on the same soil type.

There will probably be a decrease with time in the activity concentrations of most radionuclides in plants after a short-duration release of radionuclides into the environment due to the gradual fixation by soils (and sediments) discussed above.

After the initial emergency exposure situation of a few months to a year, the dominant processes determining radionuclide movement in farming systems change. The extent to which radionuclides transfer from soil into agricultural products during the later planned or existing exposure situation depends not only on the density of contamination but also on soil type, moisture regime, texture, agrochemical properties and the plant species. The impact of differing radioecological sensitivities of soils is often more important in explaining spatial variation in transfer of radionuclides in agricultural systems. Therefore, identification of radioecologically sensitive areas for animals and animal products is based on both the deposition density of different radionuclides and their mobility within different types of soil.

In terrestrial systems, wind action and rain “splash” on the soil can reintroduce radionuclides to the air where they can be ingested (if deposited on vegetation

surfaces) or inhaled by animals. Such resuspension and soil adhesion are influenced by the height and type of the plant canopy as well as weather (wind, rain), soil type and animal trampling. Grazed plants are likely to include radionuclides associated with soil adhered to the plant, as well as being incorporated within the plant itself. For radionuclides with a low transfer from soil to plant, the soil adhered on the surface of pasture grass may be the major source of radionuclide ingested by grazing ruminants. For example, root uptake of plutonium is negligible compared to direct contamination of leaves via adhered soil from rain splash or resuspension, so most ingested plutonium will be associated with adhered soil, especially for pastures with a low plant biomass.

5.2.5 Quantification of Radionuclide Transfer to Plants and Fodder Crops

The transfer from soil to plants is commonly quantified using the concentration ratio (CR) (also called a transfer factor (TF)), which is equal to the plant mass activity concentration (often in Bq/kg dw), divided by soil activity concentration, Bq/kg (dw). Available CR transfer parameter values for a wide range of radionuclides and crops for different soil types are available free in the downloadable TECDOC 1616 (IAEA 2009) and TRS 472 (IAEA 2010).

5.2.6 Intake and Absorption of Radionuclides by Animals

The transfer of radionuclides from plants (and soil) to herbivores occurs mainly by ingestion, although uptake via water can contribute to intake in the emergency phase if water sources have become contaminated after the deposition of radionuclides.

Animal products can be contaminated within a few hours of radionuclide release, mainly by the consumption of contaminated food and, to a lesser extent, water. Contamination through the skin is infrequent and absorption by inhalation is marginal for most radionuclides. The most radioecologically sensitive scenario is that of animals grazing outdoors that are directly consuming contaminated plants which have intercepted radionuclides on their surfaces.

For radionuclides that are not readily taken up by plants, soil adhesion can represent the most important route of intake especially since topsoil tends to be much more contaminated than plant material (IAEA 1994). In some instances, soil ingestion by animals may be deliberate (e.g. to obtain essential minerals), but soil can also be ingested by licking or preening of fur, feathers or offspring (Whicker and Schultz 1982). Radionuclides that are adsorbed to soil matrices may be less bioavailable than when incorporated into plant material for transfer into animal products.

Animals that are housed in pens and barns and given previously stored food (as long as that is protected from fallout) will not be significantly affected although the source of water would need to be identified. Surface water systems can be initially directly contaminated by deposited radionuclides, but dilution in water bodies normally greatly reduces the radionuclide activity concentrations in water.

5.2.7 Gastrointestinal Absorption

Absorption of radionuclides from the gastrointestinal tract (GI tract) of animals depends on, amongst other factors, the physico-chemical form of the radionuclide, the composition of the feed and the nutritional status of the animal.

Although absorption can occur through the skin and lungs, oral ingestion of radionuclides in feed, and subsequent absorption through the GI tract, is the major route of entry of radionuclides. The absorbed fraction (F_a) is defined as the fraction of that ingested by animals that is transferred through the GI tract and is a key factor determining the extent of radionuclide contamination of animal tissues and milk. The absorbed fraction depends on many different factors including metabolic status (e.g. age, lactation state, physiological condition), chemical and physical speciation of the radionuclide and the presence of competing ions.

The method of determination of GI tract absorption is important. An apparent absorption is derived from information on the whole-body intake and excretion of the radionuclide. A true absorption value is measured in a metabolic study that involves injection of a tracer which enables determination of endogenous faecal excretion (i.e. direct transfer from blood to the intestine). Endogenous secretion from tissues into the gut occurs for the key radionuclide, radiocaesium, so it is important to distinguish whether reported values refer to an apparent or true absorption value.

Available information on the fractional absorption values for radionuclides in ruminants is available in the TECDOC 1616 and TRS 472 (IAEA 2009, 2010). Fractional absorption values for the most well-studied radionuclide elements are given in Table 5.2. The number of data available on F_a in ruminants for different radionuclides varies, and, therefore, so does the confidence attributable to each

Table 5.2 Range in fractional GI tract absorption values (F_a) for different elements in domestic ruminants

Fractional absorption magnitude	Radionuclide
0.1–1	I, Cl, Na, Cs, P, Se, Ca, Te, Zn, Sr, Fe
0.01–0.1	Ag, Ba, Co, Pb, U
0.001–0.01	Mn, Ru, Cd, Y
0.0001–0.001	Zr, Ce, Pm, Am, Nb
0.00001–0.0001	Pu

Howard et al. (2016a)

Table 5.3 Fractional absorption values for adult humans

Radionuclide	Fractional absorption
H, C, Cs, S, Mo, I	1
Se	0.8
Zn, Tc, Po	0.5
Te, Sr, Ca	0.3
Ba, Ra, Pb	0.2
Co, Fe, Sb	0.1
Ru, Ni, Ag	0.05
U	0.02
Zr, Nb	0.01
Ce, Th, Np, Pu, Am, Cm	0.0005

ICRP (2006)

value. The F_a values vary from almost negligible, in the case of actinides such as plutonium, to 100% for radioiodine (Howard et al. 2009a, 2016a). Data on F_a for iodine, caesium and strontium are considered in more detail in Sect. 6.4.

The compiled ruminant F_a values for radionuclides or stable elements are similar to those reported by the International Commission on Radiological Protection (ICRP) shown in Table 5.3 for humans (and relevant for other monogastric animals such as pigs). Therefore, if ruminant-specific F_a values are not available, those given for humans may be used instead.

After absorption, radionuclides circulate in the blood to different tissues as discussed below.

5.2.8 *Quantification of Radionuclide Transfer to Animal Products*

To quantify the transfer of radionuclides to milk and meat, two types of parameter values are commonly used: transfer coefficients (F_m for milk and F_f for other tissues) and concentration ratios (CR) as follows:

Transfer coefficient (d/kg or d/L)

$$F_m \text{ or } F_f = \frac{\text{Equilibrium activity concentration in food product } \left(Bq / \text{kg } fw\right)}{\text{Daily intake of radionuclide } \left(Bq / \text{d}\right)}$$

Concentration ratio

$$CR = \frac{\text{Equilibrium activity concentration in food product } \left(Bq / \text{kg } fw\right)}{\text{Radionuclide activity concentration in feed } \left(Bq / \text{kg } dw\right)}$$

Transfer coefficient values can be derived by dividing a CR value by the daily dietary intake (in kg/d), and, conversely, CR values can be derived by multiplying the transfer coefficient value by the daily dietary intake (in kg/d). Over the last 40 years following the introduction of the transfer coefficient concept, many studies have been conducted to determine values for a range of radionuclide – animal product combinations.

To accurately estimate intake, both the dietary composition and relative contamination of each component (in *Bq*/kg dw) need to be quantified. Estimates of the feed intake of animals are more accurate in experimental studies under controlled conditions, whereas in field studies the intake is often not measured, which can lead to variability in reported F_f and F_m values.

The typical diet of agricultural animals varies between and within countries, and with the season according to feeding regimes (including whether the animals graze outdoors or are kept indoors), and is related to live weight, maintenance requirements and milk production rates. Regional data on animal nutrition requirements relevant to the region and farming system being considered can be used to derive dietary intake information. Preferably feed intake estimates would either be based on agricultural production criteria or acquired directly from the farming community. Grassy vegetation tends to be much more highly contaminated than other components of the diet, so all radionuclide intake can be assumed to come from this part of the diet when animals are consuming grass-based fodder. In published international cow milk datasets, some F_m and F_f values are based on estimated daily dry matter intake (DMI) many of which are best estimates or recommended values that do not take account of changes in the factors discussed above. Although the lack of measured daily DMI introduces uncertainty, it is unlikely to change derived F_m values by more than a factor of 3.

Transfer coefficients of radionuclides to milk and meat are generally lower for large animals, such as cattle, than for small animals, such as sheep, goats and hens. However, this is a side effect of the definition of the F_m and F_f because transfer coefficients incorporate daily DMI which increases with animal size. A higher F_m or F_f value does not mean that animal products from small animals will be more highly contaminated than those from larger animals, as was mistakenly reported in the past.

An alternative, simpler, approach to quantify transfer is to remove the dietary intake used in the estimation of F_m and calculate the CR – the equilibrium ratio between the radionuclide activity concentration in the animal food product (Bq/kg fw) divided by the radionuclide activity concentration in the feedstuff ingested (Bq/kg dw) (Howard et al. 2009a, b, 2016b; Smith and Beresford 2005). For most radionuclides, the compiled CR data gives similar values between different livestock species; therefore those derived for one species could be applied to another, providing a more generic parameter than the transfer coefficient. The advantage, especially for field studies, is that daily DMI does not need to be calculated or a value assumed. To apply CR values when a number of different feed types are consumed suggests that the relative proportions of each dietary component need to be known. However, if the grassy component is the main source of radionuclide

contamination (which is normally the case), then the intake from other components, especially if imported, can be discounted.

Tables of available CR and Tag values for various animal products are provided for radionuclides in TRS 472 (IAEA 2010) and are discussed in more detail in TECDOC 1616 (IAEA 2009). More recent analysis of transfer parameters for goat and cow milk is provided in Howard et al. (2016b, 2017). Using available CR geometric mean values given in these two papers, the predicted radionuclide activity concentrations at equilibrium have been calculated for feed that contains 1000 Bg/kg dw. The figures show the considerable difference in transfer to milk and meat for different radionuclides. For cow milk (Fig. 5.2), the relatively high transfer of I, Cs and Sr is evident, and U is also high although there are only seven reported values for this radionuclide and therefore less confidence in the value. For meat the transfer of Cs and I is also relatively high. There is no data for Sr probably because transfer to these products is low and not a cause for concern (Fig. 5.3). Furthermore, other radionuclides may be important for meat, notably S, U and Co. Notably, Po, which is an alpha emitter, has mid-range CR values for both milk and meat although based on relatively few data.

The aggregated transfer coefficient is often used to quantify radionuclide transfer in non-intensive systems (termed a Tag, with units of m^2/kg) especially for animals and animal products. Tag is equal to the plant mass or animal tissue activity concentration (Bq/kg dw or fw) per unit area deposition density in the soil (Bq/m^2). Tag values are easier to apply in the emergency response and the transition phases after a NRE as authorities will probably initially report contamination in deposition density units of Bq/m^2. Tag were first proposed as more suitable for game animals after the Chernobyl NRE (Howard et al. 1991, 1996a, b). The determination of the

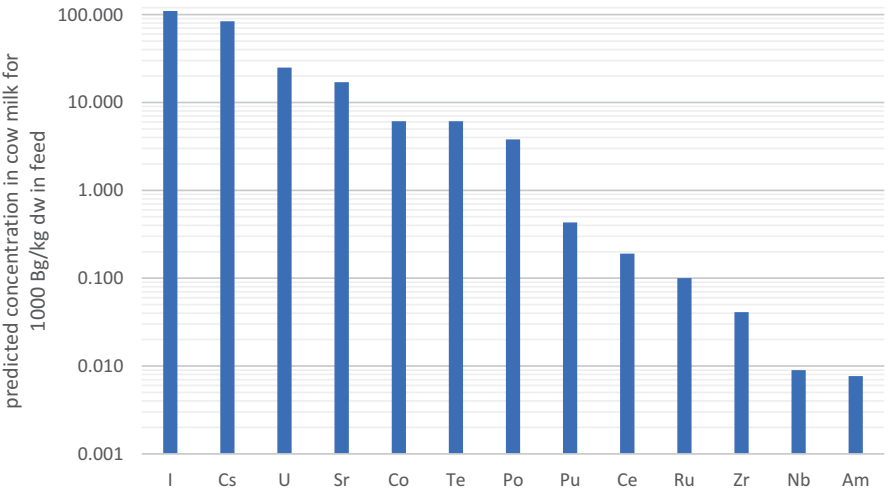

Fig. 5.2 Predicted activity concentrations of some radionuclides in cow milk from dairy cows given feed that contains 1000 Bg/kg dw. Note the plot uses a logarithmic axis

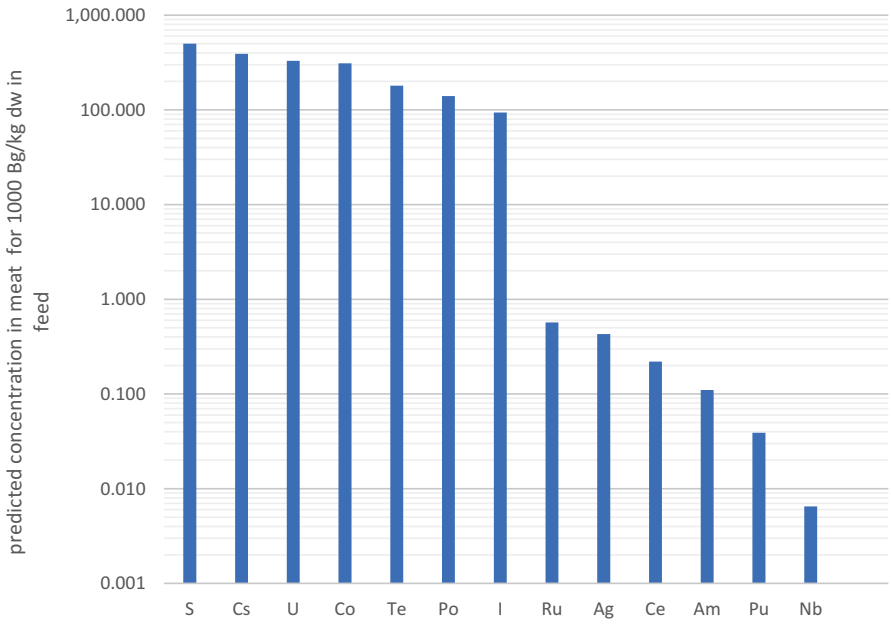

Fig. 5.3 Predicted activity concentrations of some radionuclides in meat for animals given feed that contains 1000 Bg/kg dw. Note the plot uses a logarithmic axis

underlying data for the deposition to soil needed to estimate aggregated transfer coefficients (Tag) (Howard et al. 1991, 1996a, b) is a key component in the use of the Tag value. The spatial resolution of the data is limited, and the animals considered have different sizes of home range from which they derive their food, which introduces an averaging effect but unavoidably includes uncertainties.

The use of Tag amalgamates a large number of underlying processes and is inevitably less precise than other measures described above that can be used if dietary intake is known or can be reliably estimated. Tag values rather than CR values are commonly used for free-ranging animals and for game animals in forested areas. Tag values are provided for some radionuclides in TRS 472 (IAEA 2010) and are discussed in more detail in TECDOC 1616 (IAEA 2009).

5.2.9 Quantification of the Time Dependency of Radionuclide Activity Concentrations in Animal Products

Assessments of the transfer of radionuclides via the human food chain are often based on equilibrium models using the parameter values given above. Such parameter values have limitations as they are not directly applicable to dynamic situations such as that which occurs after a NRE when radionuclide activity concentrations can change rapidly in the first few days or weeks. Once the release of radionuclides ceases, radionuclide activity concentrations in animals and animal products decline

with time. Models that simulate the dynamic accumulation and excretion of radionuclides in farm animals and animal products often use biological half-lives $\left(T_{1/2}^{b}\right)$ combined with F_f, F_m or CR values to estimate the change with time (IAEA 2009; Brown and Simmonds 1995).

5.2.10 Biological Half-Life ($T_{1/2}^{b}$) in Animal Tissues

It is important to have some knowledge of the rate of loss from animals of ingested (or inhaled) radionuclides released after NREs. $T_{1/2}^{b}$values are used to quantify how quickly agricultural or other animals will become decontaminated if they are fed uncontaminated feed or removed from the contaminated area. $T_{1/2}^{b}$is defined as the time it takes for a given activity concentration in a tissue or an animal product, such as muscle, thyroid or milk, to reduce to half of its original activity concentration by processes excluding physical decay. $T_{1/2}^{b}$values have been compiled in tables for different animal products by Fesenko et al. (2015).

$T_{1/2}^{b}$ for milk are normally described using a single exponential function. For cow milk, $T_{1/2}^{b}$ values for different radionuclides are similar at about 2 days after a single administration (Fesenko et al. 2015). For all radionuclides considered, the $T_{1/2}^{b}$varied within a narrow range of 0.6–3.5 days with the shortest values for ^{131}I and ^{132}Te. The key message is that if grazing animals are removed from contaminated areas, or given uncontaminated (clean) feed, the radionuclide contamination of the milk will rapidly decline. If animals have been eating contaminated feed for a number of weeks, the rate of reduction in milk may be slower due to release and redistribution of radionuclides retained in different tissues.

There is variation in $T_{1/2}^{b}$ values due to age, species and tissues. Some differences occur because metabolic rate decreases with increasing body size. The $T_{1/2}^{b}$ tends to be longer for larger animals. For example, ^{137}Cs loss from muscle is faster for small ruminants such as sheep and goats than for larger ruminants such as cattle. Compiled $T_{1/2}^{b}$ values for muscle of cattle reported by Fesenko et al. (2015) for isotopes of Sr, Cs and I are summarized in Table 5.4. The loss is best described by two exponential components. Data for other tissues and agricultural animals are summarized in this publication.

Table 5.4 Range of values for biological half-lives of radionuclide activity concentrations and fraction of loss of radionuclide in the first component in muscles of cattle

Radionuclide	Fraction of loss of radionuclide in the first component	Biological half lives	
		Fast loss	Slow loss
^{90}Sr	0.42–0.9	3.0–4.0	180–700
^{131}I	1.0	7.0	
^{137}Cs	0.37–0.93	3.0–22.3	36.3–81

Summarized from Fesenko et al. (2015)

Losses of radionuclides from soft tissues tend to be shorter than those from bone (Fesenko et al. 2015). The $T_{1/2}^{b}$ values are relatively short for ^{132}Te, ^{137}Cs and ^{106}Ru, whereas they are longer if the radionuclides associate with proteins or colloids (e.g. ^{144}Ce). The longest $T_{1/2}^{b}$ values are for radionuclides which are deposited in bone, notably plutonium, americium and ^{90}Sr with half-life of 600–3100 days in cattle. Animals and animal products often have fast and slow components of retention in tissues that are described by double exponential functions.

Some tissues which accumulate certain elements (and their radioisotopes) for metabolic requirements need to retain the elements and, consequently, have long $T_{1/2}^{b}$ values. Key examples are thyroid which accumulates iodine (and, therefore, radioisotopes of iodine such as ^{131}I) and bone which accumulates Ca and its analogue ^{90}Sr.

5.2.11 Ecological and Effective Half-Lives

The long-term time-dependent behaviour of radionuclides in animal tissues can also be quantified using ecological or effective half-lives which integrates all biological, environmental and ecological processes that cause a decrease of radionuclide activity concentrations in an animal product.

The ecological half-life, $T_{1/2}^{eco}$, describes the reduction of amount of radionuclide (Bq) or activity concentration (Bq/kg) in a specific environmental medium. The ecological half-life for animal products is equal to the time required for the radionuclide activity concentration in a target specific animal tissue (or milk) to decrease by a factor of 2. It does not include the effects of physical radioactive decay of an isotope. Instead of estimating, $T_{1/2}^{eco}$, from radionuclide activity concentrations, the analysis can also be applied to transfer parameters described above such as the CR or the Tag.

Effective half-lives are derived when the reduction in activity concentration, CR or Tag due to physical decay has been considered in the data. The effective half-life ($T_{1/2}^{eff}$) is defined as the time required to lose half of the radionuclide activity concentration (or the value of a transfer parameter) in the target (such as an animal tissue) and is a result of the interrelation between the physical ($T_{1/2}^{p}$) and biological ($T_{1/2}^{b}$) half-lives. The $T_{1/2}^{eff}$ can be calculated according to the following equation:

$$T_{1/2}^{eff} = \left(T_{1/2}^{p} \times T_{1/2}^{b}\right) / \left(T_{1/2}^{p} + T_{1/2}^{b}\right)$$

For ^{131}I which has a $T_{1/2}^{p}$ of 8 days and, for example, a $T_{1/2}^{b}$ of 138 days, the $T_{1/2}^{eff}$ can be calculated as:

$$T_{1/2}^{eff} = \left(8 \times 138\right) / \left(8 + 138\right) = 1104 / 146 = 7.6 \text{days}.$$

Long-term time series data of radiocaesium and radiostrontium activity concentrations in animal products can be used to provide such values. The data for changes with time are fitted with either a single or double exponential giving either a single

$T_{1/2}^{eff}$ or two $T_{1/2}^{eff}$ with an estimate of the proportion of loss that can be attributed to each component.

There are three prime sources of information on radionuclide half-lives in animal products: the Kyshtym and Chernobyl NREs and global fallout.

After the Chernobyl NPP NRE, there was a short-duration release with well-known characteristics, high contamination levels and varying environmental characteristics (such as soil and climate). As a result, extensive data on the changes with time of ^{137}Cs in animals have been obtained. Although the Fukushima NRE was also a relatively short-pulse release, there were few data for animals and animal products reported due to the disruption caused by the tsunami and earthquake and the relatively low importance of animal products because many agricultural animals were housed.

Global fallout represented a variable source term of radionuclides for the environment, as deposition of radionuclides occurred over a number of years, with maximum deposition observed in 1962–1964. A decade after the peak deposition period, when external contamination of plants was no longer occurring, long-term monitoring data provided an opportunity for deriving long-term effective half-lives for ^{90}Sr and ^{137}Cs.

5.3 Monitoring Animal Food Products

Monitoring the presence of radioactivity entering the food chain is of prime importance to ensure the safety of animal products reaching the human consumer. Milk is a major constituent of the diet for children, and the presence of ^{90}Sr, ^{131}I and ^{137}Cs needs to be carefully assessed. Regular examination of dairy and agricultural produce has been an important role of the veterinary and relevant authorities in many countries for many years. For example, milk in Europe is routinely analysed from the vicinity of nuclear sites to assess the exposure from ingested foodstuffs to the local population. The NREs at Chernobyl and Fukushima Daiichi intensified surveillance globally.

After NREs, national monitoring programmes have been implemented and maps of the deposition of radioactive contamination prepared. The strategies for monitoring need to adapt to the changing characteristics of contamination that occur with time. Initially, ^{131}I is potentially the major hazard in milk, after which monitoring for ^{137}Cs in milk and meat is more likely to dominate. Therefore, sampling of milk from contaminated areas is given a high priority. Fortunately, collection and analysis of milk is much easier for ^{131}I and radiocaesium than for other animal products. Analysis of milk from individual farms will give detailed information about the extent and character of the contamination. However, there is also some advantage in sampling milk from bulk sources such as tankers, which gives data representing several hundred cows sourced from a wide area.

If the radionuclide activity concentration in an animal product is above the intervention level, management options such as decontamination by clean feeding, or administration of Cs binders, which reduce its absorption in the gut, can be used to

lower the activity concentration before slaughter (see management options and datasheets). The time period needed to do this can be assessed based on measured radionuclide activity concentrations in muscle and the corresponding radiation safety standard (intervention level), utilizing knowledge of $T_{1/2}^{b}$.

The use of live monitoring reduces the need to condemn meat and provides important information on the effectiveness of options which aim to reduce contamination of animals. Live monitoring has been used extensively after the Chernobyl NRE in both the USSR (subsequently termed the former Soviet Union (fSU) countries) and Western Europe to measure radiocaesium in a wide range of live ruminants and also for carcasses of wild animals to inform hunters of the contamination levels in the meat. The advantage of live monitoring is that estimates of radiocaesium activity concentrations can be made without the need to slaughter the animal. Live monitoring was less widely used after the Fukushima NRE due to the relatively low radiocaesium activity concentrations. Blood sampling and analysis was also used to assess animal product contamination.

5.4 Radionuclide Transfer to Intensively Farmed Agricultural Animals

Although many different radionuclides may be released in a NRE, only a few present potentially serious health hazards to humans and animals. There are three key radionuclides: radioiodine, radiocaesium and radiostrontium, which are environmentally mobile in many production systems and which transfer readily to animal products. Because of their importance, specific text on these three radionuclides is included for each subsection describing environmental transfer rates below.

This section describes various factors which influence radionuclide transfer in intensively managed systems which are normally fertilized, and where the farm animals are in a good condition with high milk and meat production rates. Data for CR are provided in tables for different radionuclides and animal products based on compilations that were published by the IAEA (which used the term Transfer factor) in IAEA (2009, 2010).

5.4.1 Soil and Plant Aspects

Soil is the main terrestrial sink of long-lived radionuclides deposited on the landscape, so the interaction between radionuclides and different soil characteristics is particularly important after the initial phase. In some cases, a substantial proportion of the radionuclide may become strongly associated with soil components and thereby becomes less mobile.

5.4.1.1 Radioiodine

The geochemistry of iodine is dominated by its volatility. The volatilization of organo-iodine compounds and elemental iodine from biological and non-biological sources in the oceans is a major component of its global cycle. Iodine is strongly enriched in soils 50–80 km inland from marine systems. Some wetland soils also form terrestrial sources of volatilized iodine. The dominant species of iodine in the aerobic soil environment are I^-, IO_3^- and I_2.

Stable ^{127}I is normally present in soils at an average concentration of 5 mg/kg dw. Typically, terrestrial plants and food crops contain from 0.07 to 10 mg/kg dw of stable I (^{127}I). There is another natural isotope of iodine, ^{129}I, that is much less abundant and which can be released during some nuclear activities, including NREs, but has a much lower radiological impact than ^{131}I.

Radioiodine dissolves in water and moves easily from the atmosphere into different components of the environment. However, it readily absorbs to various soil components such as organic matter and soil minerals which limits the uptake of iodine through the plant root system. The two naturally occurring isotopes usually behave similarly although soil to plant uptake rates have been shown to differ in some soils (IAEA 2009).

The importance of soil to plant transfer for short-lived radioiodine isotopes, especially ^{131}I, is generally thought to be negligible because of the short physical half-life of the iodine isotopes of relevance for internal dose to humans. After NRE, the interception by plants of the short-lived ^{131}I in the emergency and transition phase is important, but in the longer term, accumulation of iodine in plants is only relevant for ^{129}I.

The transfer of radioiodine from soil to plant in the emergency phase after NREs has received little attention from the research and radiation protection community. There are few compiled data for iodine transfer to plants (Table 5.5) with CR values varying from 0.1 to 5.0 for vegetative plant mass. No CR values for iodine are given for soil to grass species in TRS 472 (IAEA 2010). CR values for iodine are low for soils with a high cation exchange capacity and organic matter content. For grain (rye and wheat), which can be components of animals' diet, iodine CR values vary from 5×10^{-4} to 8×10^{-3}.

Table 5.5 Soil to plant transfer factors for I (IAEA 2009, 2010)[a]

Plant group	Plant compartment	Soil group	N	GM	Minimum	Maximum
Cereal	Grain	All	13	6.3×10^{-4}	1.0×10^{-4}	1.1×10^{-2}
		Clay	6	5.7×10^{-4}	2.0×10^{-4}	1.6×10^{-3}
		Loam	5	3.6×10^{-4}	1.0×10^{-4}	1.2×10^{-3}
		Sand	2		1.0×10^{-3}	1.1×10^{-2}
Leafy vegetables	Leaves	All	12	6.5×10^{-3}	1.1×10^{-3}	1.0×10^{-1}
		Clay	2	4.6×10^{-3}	1.6×10^{-3}	1.3×10^{-2}
		Loam	8	4.1×10^{-3}	1.1×10^{-3}	8.0×10^{-3}
		Sand	1			

(continued)

Table 5.5 (continued)

Plant group	Plant compartment	Soil group	N	GM	Minimum	Maximum
Nonleafy vegetables	Head, berries, buds	All	1	1.0×10^{-1}		
Leguminous vegetables	Seeds and pod	All	23	8.5×10^{-3}	2.0×10^{-4}	1.4×10^{-1}
		Clay	2		2.0×10^{-4}	3.0×10^{-4}
		Loam	3	4.4×10^{-4}	3.0×10^{-4}	7.0×10^{-4}
		Sand	2		3.3×10^{-3}	3.7×10^{-3}
Root crops	Root	All	28	7.7×10^{-3}	1.4×10^{-3}	4.7×10^{-2}
		Clay	7	4.5×10^{-3}	1.4×10^{-3}	2.8×10^{-2}
		Loam	12	4.7×10^{-3}	1.5×10^{-3}	1.6×10^{-2}
		Sand	9	2.3×10^{-2}	1.2×10^{-2}	4.7×10^{-2}
Tubers	Tuber	All	1			
Pasture	Stems, leaves	All	12	3.7×10^{-3}	9.0×10^{-4}	5.0×10^{-1}
		Clay	2		8.4×10^{-3}	9.0×10^{-3}
		Sand	9	1.8×10^{-3}	9.0×10^{-4}	8.5×10^{-3}
Cereal	Stems, leaves	All	16	5.2×10^{-2}	7.0×10^{-3}	7.5×10^{-1}
		Clay	7	4.5×10^{-2}	1.0×10^{-2}	1.9×10^{-1}
		Loam	7	3.6×10^{-2}	7.0×10^{-3}	2.0×10^{-1}
		Sand	2		1.1×10^{-1}	7.5×10^{-1}

[a] N - sample number, GM - goemetric mean - The mean is a geometric mean except where the number of data values (N) is less than 3, in which case it is an arithmetic mean. Further statistical information is given for a wider range of radionuclides in TECDOC 1616 and TRS 472 (IAEA 2009, 2010)

5.4.1.2 Radiocaesium

Radiocaesium has a high biological and ecological mobility as stable caesium is an alkali element, which is a chemical analogue of the biologically important element, potassium. Stable caesium exists in the environment in the 1+ oxidation state with concentrations ranging between 0.3 and 25 mg/kg dw. Radiocaesium is highly mobile in soils of both agricultural and free-ranging farming and harvesting systems in the emergency phase after NRE deposition.

In the transition phase and the subsequent existing exposure situation, after radiocaesium has been lost from the surfaces of plants, root uptake of radiocaesium from soil dominates. During the year following the Chernobyl NRE, the ^{137}Cs activity concentration in plants declined by a factor of between 3 and 100 as root uptake from different soil types became the dominant contamination route. The most important process controlling plant root uptake of radiocaesium is the interaction between soil matrix and soil solution which depends primarily on the cation exchange capacity of the soil. For mineral soils, this is influenced by the concentrations and types of clay minerals and the concentrations of competitive major cations, especially potassium and ammonium. The extent of selective, irreversible absorption differs for different clay minerals. Sorption of caesium to organic colloids and dissolved organic matter is not important in most (but not all) soils, so caesium is relatively more mobile in peaty and sandy soils. Organic soils often contain sufficient illitic clay minerals to immobilize radiocaesium present in organic soils, but the organic matter holds the clay in an expanded state, thereby maintaining availability of radiocaesium for plant uptake (Hird et al. 1995).

Accumulation of radiocaesium into crops and pasture is related to soil texture. On sandy soils, uptake of radiocaesium by plants is approximately twice as high as on loam soils mainly due to the lower concentrations of potassium in sand. Radiocaesium uptake from poor, often unfertilized, soils tends to exceed that of plants grown on fertile agricultural soils by several orders of magnitude. The highest ^{137}Cs uptake by roots from soil to plants occurs in poor highly organic, boggy soils, which are one to two orders of magnitude higher than in sandy soils. Agricultural practices often reduce the transfer of radionuclides from soils to plant by physical dilution (e.g. ploughing) or by adding competitive elements during normal fertilization procedures. For radiocaesium, application of its analogue, potassium, is highly effective in reducing transfer to crops.

In TRS 472 (IAEA 2010), CR values for caesium have been given for a wide range of different plant groups (Table 5.6). Caesium uptake from soil by a single crop is less than 0.1% of the soil's content (Menzel 1963). CR values vary considerably from about 10^{-3} up to about 1.0. Variations in the accumulation of ^{137}Cs by plants due to differences in soil properties are up to a factor of 100, and the effect of biological features of plants causes up to a further tenfold variation (Alexakhin and Korneyev 1991). Mean caesium CR values are a factor of 2–10 lower than those of

Table 5.6 Soil to plant for Cs (IAEA 2009, 2010)[a]

Plant group	Plant compartment	Soil group	N	GM	Minimum	Maximum
Cereal	Grain	All	470	2.9×10^{-2}	2.0×10^{-4}	9.0×10^{-1}
		Clay	110	1.1×10^{-2}	2.0×10^{-4}	9.0×10^{-2}
		Loam	158	2.0×10^{-2}	8.0×10^{-4}	2.0×10^{-1}
		Sand	156	3.9×10^{-2}	2.0×10^{-3}	6.6×10^{-1}
		Organic	28	4.3×10^{-2}	1.0×10^{-2}	7.3×10^{-1}
Maize	Grain	All	67	3.3×10^{-2}	3.0×10^{-3}	2.6×10^{-1}
		Clay	11	1.2×10^{-2}	3.0×10^{-3}	7.0×10^{-2}
		Loam	14	1.6×10^{-2}	3.2×10^{-3}	7.0×10^{-2}
		Sand	47	4.9×10^{-2}	8.0×10^{-3}	2.6×10^{-1}
Leafy vegetables	Leaves	All	290	6.0×10^{-2}	3.0×10^{-4}	9.8×10^{-1}
		Clay	67	1.8×10^{-2}	5.0×10^{-4}	7.2×10^{-1}
		Loam	119	7.4×10^{-2}	3.0×10^{-4}	7.3×10^{-1}
		Sand	96	1.2×10^{-1}	2.1×10^{-3}	9.8×10^{-1}
		Organic	7	2.25×10^{-2}	4.0×10^{-3}	4.6×10^{-1}
Nonleafy vegetables	Head, berries, buds	All	38	2.1×10^{-2}	7.0×10^{-4}	7.3×10^{-1}
		Clay	14	9.1×10^{-3}	7.0×10^{-4}	1.6×10^{-2}
		Loam	5	3.3×10^{-2}	6.3×10^{-3}	3.0×10^{-1}
		Sand	17	3.5×10^{-2}	1.2×10^{-2}	7.3×10^{-1}
Leguminous vegetables	Seeds and pod	All	126	4.0×10^{-2}	1.0×10^{-3}	7.1×10^{-1}
		Clay	18	1.3×10^{-2}	2.0×10^{-3}	8.1×10^{-2}
		Loam	42	2.0×10^{-2}	1.0×10^{-3}	4.2×10^{-1}
		Sand	66	8.7×10^{-2}	3.5×10^{-3}	7.1×10^{-1}

(continued)

Table 5.6 (continued)

Plant group	Plant compartment	Soil group	N	GM	Minimum	Maximum
Root crops	Root	All	81	4.2×10^{-2}	1.0×10^{-3}	8.8×10^{-1}
		Clay	17	2.4×10^{-2}	5.0×10^{-3}	6.0×10^{-2}
		Loam	21	3.0×10^{-2}	1.0×10^{-3}	1.6×10^{-1}
		Sand	37	6.2×10^{-2}	8.0×10^{-3}	4.0×10^{-1}
		Organic	5	5.9×10^{-2}	1.6×10^{-2}	8.8×10^{-1}
Tubers	Tuber	All	138	5.6×10^{-2}	4.0×10^{-3}	6.0×10^{-1}
		Clay	21	2.5×10^{-2}	5.0×10^{-3}	9.0×10^{-2}
		Loam	40	3.5×10^{-2}	4.8×10^{-3}	1.4×10^{-1}
		Sand	69	9.3×10^{-2}	4.0×10^{-3}	6.0×10^{-1}
		Organic	7	5.8×10^{-2}	1.610^{-2}	5.4×10^{-1}
Grasses	Stems, leaves	All	64	6.3×10^{-2}	4.8×10^{-3}	9.9×10^{-1}
		Clay	9	1.2×10^{-2}	4.8×10^{-3}	4.3×10^{-2}
		Loam	10	4.8×10^{-2}	1.2×10^{-2}	2.1×10^{-1}
		Sand	41	8.4×10^{-2}	1.0×10^{-2}	9.9×10^{-1}
		Organic	4	2.8×10^{-1}	2.1×10^{-1}	3.4×10^{-1}
Fodder leguminous	Stems, leaves	All	85	1.6×10^{-1}	1.0×10^{-2}	1.8
		Clay	4	4.6×10^{-2}	1.3×10^{-2}	3.0×10^{-1}
		Loam	51	1.5×10^{-1}	1.0×10^{-2}	1.2
		Sand	29	2.4×10^{-1}	1.8×10^{-2}	1.8
Pasture	Stems, leaves	All	401	2.5×10^{-1}	1.0×10^{-2}	5.0
		Clay	75	1.8×10^{-1}	1.0×10^{-2}	1.2
		Loam	124	1.9×10^{-1}	1.0×10^{-2}	2.6
		Sand	169	2.9×10^{-1}	1.0×10^{-2}	4.8
		Organic	31	7.6×10^{-1}	3.0×10^{-1}	5.0
Herbs	Stems, leaves	All	4	6.6×10^{-2}	4.8×10^{-3}	2.8
Other crops		All	9	3.1×10^{-1}	3.6×10^{-2}	2.2
Cereal	Stems, leaves	All	130	1.5×10^{-1}	4.3×10^{-3}	3.7
		Clay	37	5.6×10^{-2}	4.3×10^{-3}	5.3×10^{-1}
		Loam	36	1.1×10^{-1}	6.5×10^{-3}	1.5
		Sand	35	2.1×10^{-1}	4.1×10^{-2}	1.9
Maize	Stems, leaves	All	101	7.3×10^{-2}	3.0×10^{-3}	4.9×10^{-1}
		Clay	11	2.2×10^{-2}	7.8×10^{-3}	6.0×10^{-2}
		Loam	10	1.5×10^{-2}	3.0×10^{-3}	5.2×10^{-2}
		Sand	77	1.0×10^{-1}	1.4×10^{-2}	4.9×10^{-1}
		Organic	3	1.4×10^{-1}	1.0×10^{-1}	1.6×10^{-1}
Root crops	Leaves	All	12	3.5×10^{-2}	6.0×10^{-3}	4.5×10^{-1}
		Clay	7	2.6×10^{-2}	6.0×10^{-3}	4.7×10^{-2}
		Loam	2		9.0×10^{-3}	4.3×10^{-2}
		Sand	3	1.1×10^{-1}	5.1×10^{-2}	4.5×10^{-1}

[a] The mean is a geometric mean except where the number of data values (N) is less than 3, in which case it is an arithmetic mean. Further statistical information is given for a wider range of radionuclides in TECDOC 1616 and TRS 472 (IAEA 2009, 2010)

Table 5.7 Radioecological sensitivity for soil-plant transfer of $^{137}Cs/^{134}Cs$

Sensitivity	Soil characteristic	Mechanism	Example
High	– Low nutrient content – Very low fraction of clay minerals – High organic content	– Little competition with potassium and ammonium in root uptake	Peat soils
Medium	– Poor nutrient status, consisting of minerals including some clays	– Limited competition with potassium and ammonium during root uptake	Podzol, other sandy soils
Low	– High nutrient status – High fraction of clay minerals	– Radiocaesium strongly bound to clay minerals – Strong competition with potassium and ammonium during root uptake	Chernozems Clay and loam soils (used for intensive agriculture)

strontium in most soils. The radioecological sensitivity of soils for radiocaesium can be broadly divided into the categories listed in Table 5.7.

A substantial proportion of the radiocaesium in soil gradually becomes less available for plant uptake as it becomes irreversibly bound by clay minerals. Differences in radioecological sensitivities of soils after the first few years can have a significant impact on animal production contamination after an NRE. In some areas with low radiocaesium deposition densities and highly radioecologically sensitive soils after the Chernobyl accident, there were high radiocaesium activity concentrations in plants, and hence animals, which persisted for decades. Conversely, some areas of high deposition with soils of low radioecological sensitivity for radiocaesium had only low to moderate radiocaesium activity concentrations in plants and animals.

5.4.1.3 Radiostrontium

Natural strontium consists of 4 stable isotopes with mass numbers of 84, 86, 87 and 88. The content of stable Sr in the Earth's crust is about 3×10^{-2}%. The chemical properties of strontium are determined by its position in group 2 of the periodic system and are typical for alkali-earth elements. Strontium is a close analogue of calcium and its behaviour in soils and transfer to plants are highly influenced by the status of calcium in soils. Strontium is a highly mobile and bioavailable element that exists in the environment in the Sr(II) oxidation state at concentrations in soils that range between 50 and 1000 mg/kg dw. Strontium is usually present in the surface environment as a carbonate or a sulphate mineral. The dominant aqueous strontium species in natural waters over a broad pH range (2–9) is the free divalent Sr^{2+}. Cation exchange is the key mechanism of absorption of Sr in soil.

Strontium is one of the most biologically mobile elements. Plant crops take up about 0.2% to 3% of the strontium in the soil (Menzel 1963). The Kyshtym NRE

was the first instance where large areas were contaminated by radionuclides, and ^{90}Sr was one of the most important radionuclides released. Therefore, there is a large amount of available information on the behaviour of radiostrontium in soils. The uptake of ^{90}Sr from soil to plants is affected by presence of both stable strontium and stable calcium (Gulyakin and Yudintseva 1962, Arkhipov et al. 1969). The interaction with these two stable elements is one of the main contributors to variability in Sr CR values. Strontium uptake by plants is generally highest from soils of low calcium content and, in many cases, of high organic matter content.

A large number of CR values are reported for Sr in TRS 472 (IAEA 2010) which are summarized in Table 5.8. Strontium CR values differ by more than a factor of 100, depending on soil properties and biological features of plants. Most of the variation in CR values of ^{90}Sr can be attributed to the stable strontium concentrations in soil and its interaction with calcium. These two factors largely account for the low CR values, and also the large variability reported between individual plant

Table 5.8 Soil to plant transfer factors for Sr[a]

Plant group	Plant compartment	Soil group	N	GM	Minimum	Maximum
Cereal	Grain	All	282	1.1×10^{-1}	3.6×10^{-3}	1.0
		Clay	72	7.8×10^{-2}	5.3×10^{-3}	7.1×10^{-1}
		Loam	71	1.1×10^{-1}	1.6×10^{-2}	7.2×10^{-1}
		Sand	123	1.4×10^{-1}	3.6×10^{-3}	1.0
		Organic	10	9.7×10^{-2}	1.2×10^{-2}	3.6×10^{-1}
Maize	Grain	All	39	3.2×10^{-1}	2.0×10^{-3}	2.6
		Clay	7	6.9×10^{-2}	2.0×10^{-3}	3.9×10^{-1}
		Loam	13	3.6×10^{-1}	1.5×10^{-1}	8.6×10^{-1}
		Sand	19	5.2×10^{-1}	4.0×10^{-2}	2.6
Leafy vegetables	Leaves	All	217	7.6×10^{-1}	3.9×10^{-3}	7.8
		Clay	54	1.5×10^{-1}	3.9×10^{-3}	2.2
		Loam	84	1.2	4.1×10^{-2}	5.0
		Sand	72	1.7	6.4×10^{-2}	7.8
		Organic	6	2.1×10^{-1}	1.5×10^{-1}	3.0×10^{-1}
Nonleafy vegetables	Head, berries, buds	All	19	3.6×10^{-1}	7.1×10^{-3}	7.9
		Clay	8	1.3×10^{-1}	7.1×10^{-3}	8.6×10^{-1}
		Loam	3	1.4	9.0×10^{-1}	2.3
		Sand	5	8.7×10^{-1}	2.0×10^{-1}	7.9
		Organic	2	2.2×10^{-1}	1.9×10^{-1}	2.5×10^{-1}
Leguminous vegetables	Seeds and pod	All	148	1.4	1.3×10^{-1}	6.0
		Clay	25	6.2×10^{-1}	1.3×10^{-1}	2.6
		Loam	68	1.3	1.7×10^{-1}	4.6
		Sand	55	2.2	3.0×10^{-1}	6.0
Root crops	Root	All	56	7.2×10^{-1}	3.0×10^{-2}	4.8
		Clay	13	4.1×10^{-1}	5.2×10^{-2}	3.9
		Loam	16	6.1×10^{-1}	4.4×10^{-2}	4.5
		Sand	26	1.1	3.0×10^{-2}	4.8

(continued)

Table 5.8 (continued)

Plant group	Plant compartment	Soil group	N	GM	Minimum	Maximum
Tubers	Tuber	All	106	1.6×10^{-1}	7.4×10^{-3}	1.6
		Clay	21	1.3×10^{-1}	2.6×10^{-2}	6.7×10^{-1}
		Loam	41	1.3×10^{-1}	7.4×10^{-3}	4.5×10^{-1}
		Sand	39	2.2×10^{-1}	2.6×10^{-2}	1.6
		Organic	4	5.8×10^{-2}	8.0×10^{-3}	2.3×10^{-1}
Grasses	Stems, leaves	All	50	9.1×10^{-1}	2.5×10^{-1}	2.8
		Clay	7	7.9×10^{-1}	4.8×10^{-1}	9.7×10^{-1}
		Loam	6	6.0×10^{-1}	2.9×10^{-1}	2.0
		Sand	34	1.1	2.6×10^{-1}	2.8
		Organic	3	2.6×10^{-1}	2.5×10^{-1}	2.8×10^{-1}
Fodder leguminous	Stems, leaves	All	35	3.7	1.3	1.8×10
		Clay	10	2.8	1.3	5.8
		Loam	11	3.3	1.4	9.8
		Sand	14	4.9	1.3	1.8×10
		Organic	1	3.9×10^{-1}		
Pasture	Stems, leaves	All	172	1.3	5.6×10^{-2}	7.3
		Clay	22	8.0×10^{-1}	9.0×10^{-2}	2.8
		Loam	58	1.1	3.7×10^{-1}	2.6
		Sand	87	1.7	9.8×10^{-2}	7.3
		Organic	4	3.5×10^{-1}	5.6×10^{-2}	1.2
Herbs	Stems, leaves	All	1	4.5		
Other crops		All	9	8.8×10^{-1}	2.0×10^{-2}	8.2
Cereal	Stems, leaves	All	37	1.1	1.5×10^{-1}	9.8
		Clay	20	7.5×10^{-1}	1.5×10^{-1}	2.8
		Loam	3	1.8	7.2×10^{-1}	3.6
		Sand	11	2.1	9.3×10^{-1}	9.8
Maize	Stems, leaves	All	36	7.3×10^{-1}	1.2×10^{-1}	3.0
		Clay	6	5.0×10^{-1}	1.8×10^{-1}	1.1
		Loam	7	7.0×10^{-1}	2.8×10^{-1}	1.4
		Sand	23	8.2×10^{-1}	1.2×10^{-1}	3.0

[a] The mean is a geometric mean except where the number of data values (N) is less than 3, in which case it is an arithmetic mean. Further statistical information is given for a wider range of radionuclides in TECDOC 1616 and TRS 472 (IAEA 2009, 2010)

types, which are affected by the need and ability to accumulate calcium. The radioecological sensitivity of soils for radiostrontium can be broadly divided into two categories listed in Table 5.9.

Decrease in exchangeable strontium in soil occurs very slowly, so the availability of soil ^{90}Sr to plants decreases only slightly with time. Relatively higher rates of ^{90}Sr vertical migration occur in sandy soils and lower rates in peat soils.

Table 5.9 Radioecological sensitivity for soil-plant transfer of ^{90}Sr

Sensitivity	Soil characteristic	Mechanism	Example
High	Low nutrient status Low organic matter content	Limited competition with calcium in root uptake	Podzol sandy soils
Low	High nutrient status Medium to high organic matter content	Strong competition with calcium in root uptake	Umbric gley soils, peaty soils

5.4.1.4 Other Radionuclides

Brief information is provided here on the other radionuclides of potential concern after an NRE based on text from IAEA TRS 472 (IAEA 2010) and TECDOC 1616 (IAEA 2009). Further, more detailed information, including CR values, can be accessed in these publications.

Transuranic elements (Am, Cm, Pu, Np) exhibit a complex soil chemistry, because of various degrees of oxidation, absence of stable carriers and high tendencies to complexation and hydrolysis. CR values for transuranic elements vary from about 100 to about 10^{-6}. Due to these relatively low CR values, the activity concentrations of these radionuclides in fruits and grains are 10–1000 times lower than in the vegetative parts of plants. Accumulation of these elements decreases in the order Np > Am > Cm > Pu. Hydrolysis is a major factor influencing the behaviour of Am and Cm in soils. The mobility of Pu depends on its valency form and decreases in the order Pu (V) > Pu (VI) > Pu (III) > Pu (IV).

The fission products (^{89}Sr, ^{90}Sr, ^{134}Cs, ^{137}Cs, ^{129}I, ^{131}I, ^{95}Zr, ^{95}Nb, ^{103}Ru, ^{106}Ru, ^{141}Ce, ^{144}Ce) include a diverse class of elements. Of these radionuclides, ^{95}Zr, ^{95}Nb, ^{103}Ru, ^{106}Ru and ^{141}Ce, ^{144}Ce are poorly accumulated by agricultural plants because of their strong sorption in soil, leading to low CR values. Soil pH and organic matter content are the most significant soil characteristics that influence the behaviour of these radionuclides. Up to 99% of the plant uptake of these radionuclides is retained in the roots, so there is little transfer to above-ground plant parts that may be consumed by animals. CR values vary by factors of 10–30 for different soils, with the lowest plant uptake for ^{95}Zr and ^{141}Ce, ^{144}Ce.

The activation products (^{60}Co, ^{65}Zn) are radioisotopes of biologically important microelements. They have high mobility in soil-plant systems and, therefore, relatively high CR values. In particular, ^{65}Zn has CR values from 1.0 to 15.0, but it is not likely to be released in large quantities after a NRE.

The behaviour of other radionuclides not mentioned above depends on the oxidation-reduction potential of the soil, the acidity of soil solution and the organic matter content.

5.4.2 Dairy Production

The consumption of milk contaminated by ^{131}I, ^{90}Sr and ^{137}Cs is potentially one of the main contributors to the internal dose to humans after a NRE.

The highest contamination levels in plants are normally reached during the urgent response phase when radionuclides are intercepted by plants and before they are lost from the plant surfaces. At the time of the Chernobyl NPP NRE, vegetation was at different growth stages in different countries that were affected depending on latitude and elevation. In the first few weeks, interception on plant leaves of dry deposition and atmospheric washout with precipitation were the main pathways of contamination. Because radionuclides were released over a period of 10 days, and plant growth had commenced in the adjacent areas (as it was late April and early May), radionuclides were intercepted by plant surfaces including pasture grass. In contrast, because the Fukushima Daiichi NRE occurred in mid-March, there was much less plant biomass present that could intercept the radionuclides in the atmosphere. Therefore, in the prevailing intensive farming systems, the initial extent of contamination of most plants was much lower than that after the Chernobyl NRE.

In the USSR, the food-production systems at the time of the Chernobyl NRE were largely collective farms and small private subsistence farms. The collective farms had an intensive farming approach using land rotation combined with ploughing and fertilization to improve productivity. In contrast, the traditional small subsistence or "private" farms usually had privately owned livestock which often grazed in forest clearings to which they applied manure to improve yield instead of artificial fertilizers. Root uptake of radiocaesium becomes the key transfer route to milk after the emergency response phase and the early part of the transition phase. The highest activity concentrations of radionuclides in most agricultural animal product foodstuffs occurred in the growing season of 1986. In many regions of the USSR, as well as in Germany, France and Southern Europe, dairy animals were already grazing outdoors, so some contamination of cow, goat and sheep milk occurred. In contrast, in Northern Europe, in the early spring, most dairy cows, sheep and goats were not yet on pasture; therefore, there was little milk contamination.

The extent of transfer of radionuclides into cow, sheep and goat milk has been reported as both F_m and CR values in the IAEA publications TECDOC 1616 and TRS 472 (IAEA 2009, 2010). The data for cow and goat milk has recently been updated during the IAEA MODARIA programme (Howard et al. 2016a, b, 2017). F_m and CR values for selected radionuclide elements that are most relevant for NRE in the MODARIA tables are shown in Tables 5.10 and 5.11 for cow milk and Tables 5.12 and 5.13 for goat milk, respectively. Available parameter values for other radionuclides/elements can be found in Howard et al. (Howard et al. 2016a, b, 2017).

For some radionuclides released from previous NREs, there are few data, notably for ^{210}Po and ^{95}Zr. Also, data for transuranic elements such as plutonium, americium

Table 5.10 Transfer coefficients (F_m, d/kg) for radionuclides relevant for NREs for cow milk

Element	N	GM	Minimum	Maximum
Am	3	1.6×10^{-6}	3.0×10^{-7}	3.0×10^{-5}
Ce	8	1.5×10^{-5}	1.0×10^{-6}	1.3×10^{-4}
Co	16	3.2×10^{-4}	2.2×10^{-5}	1.0×10^{-2}
Cs	289	4.9×10^{-3}	6.0×10^{-4}	5.7×10^{-2}
I	105	6.0×10^{-3}	4.0×10^{-4}	4.4×10^{-2}
Nb	1	4.1×10^{-7}		
Po	4	2.4×10^{-4}	1.2×10^{-4}	3.0×10^{-4}
Pu	3	3.6×10^{-5}	7.5×10^{-6}	5.0×10^{-4}
Ru	6	9.4×10^{-6}	6.7×10^{-7}	1.4×10^{-4}
Sr	118	1.3×10^{-3}	1.5×10^{-5}	4.3×10^{-3}
Te	11	3.2×10^{-4}	7.8×10^{-5}	1.0×10^{-3}
U	7	2.5×10^{-3}	5.0×10^{-4}	6.1×10^{-3}
Zr	6	3.6×10^{-6}	5.5×10^{-5}	1.7×10^{-5}

Howard et al. (2017)

Table 5.11 Concentration ratios (CR, kg/L) for radionuclides relevant for NREs for cow milk

Element	N	GM	Minimum	Maximum
Am	3	7.7×10^{-6}	6.2×10^{-6}	6.2×10^{-4}
Ce	8	1.9×10^{-4}	1.0×10^{-5}	3.2×10^{-3}
Co	16	6.1×10^{-3}	4.5×10^{-4}	2.4×10^{-1}
Cs	289	8.4×10^{-2}	3.6×10^{-3}	9×10^{-1}
I	105	1.1×10^{-1}	3.0×10^{-3}	1.1×10^{-1}
Nb	1	9.0×10^{-6}		
Po	4	3.8×10^{-3}	2.4×10^{-3}	5.4×10^{-3}
Pu	3	4.3×10^{-4}	5.8×10^{-5}	5.0×10^{-3}
Ru	6	1.0×10^{-4}	1.0×10^{-5}	1.4×10^{-3}
Sr	118	1.7×10^{-2}	5.6×10^{-4}	1.4×10^{-1}
Te	11	6.1×10^{-3}	1.4×10^{-3}	1.1×10^{-2}
U	7	2.5×10^{-2}	5.0×10^{-3}	6.1×10^{-2}
Zr	6	4.1×10^{-5}	1.0×10^{-5}	$1.7x10^{-4}$

Howard et al. (2017)

and uranium are sparse. However, there are a large number of data for the most important radionuclides, ^{134}Cs, ^{137}Cs, ^{90}Sr and ^{131}I, and, therefore, there is more confidence in these transfer parameter values. Various factors that lead to the variability in the transfer values, such as the effect of the intake of a close stable element analogue to a radionuclide, is discussed for the three most important radionuclide elements below.

Table 5.12 Transfer coefficients (F_m, d/kg) for radionuclides relevant for NREs for goat milk*

Element	N	AM	ASD	GM	GSD	Minimum	Maximum
Am	2	2.8×10^{-5}				3.7×10^{-6}	5.2×10^{-5}
Ce	1	4.0×10^{-5}					
Cs	27	1.4×10^{-1}	7.9×10^{-2}	1.1×10^{-1}	2.1	9.0×10^{-3}	3.3×10^{-1}
I	23	3.2×10^{-1}	2.3×10^{-1}	2.1×10^{-1}	3.0	2.7×10^{-2}	7.7×10^{-1}
Po	2	2.3×10^{-3}				1.8×10^{-3}	2.7×10^{-3}
S	12	4.7×10^{-2}	1.9×10^{-2}	3.8×10^{-2}	1.7	1.6×10^{-2}	6.8×10^{-2}
Sr	21	2.0×10^{-2}	1.9×10^{-2}	1.5×10^{-2}	2.0	5.8×10^{-3}	8.1×10^{-2}
Te	1	4.4×10^{-3}					
U	1	1.4×10^{-3}					
Zr	1	5.5×10^{-6}					

N Sample size, *AM* arithmetic mean, *ASD* arithmetic standard deviation, *GM* geometric mean, *GSD* geometric standard deviation,
Howard et al. (2016a, b)
*The mean is a geometric mean except where the number of data values (N) is less than 3, in which case it is a n arithmetic mean. Further statistical information is given for a wider range of radionuclides in TECDOC 1616 and TRS 472 (IAEA 2009, 2010)

Table 5.13 Concentration ratios (CR, kg/L) for radionuclides relevant for NREs for goat milk*

Element	N	AM	ASD	GM	GSD	Minimum	Maximum
Am	2	4.4×10^{-5}				4.4×10^{-6}	8.4×10^{-5}
Ce	1	6.4×10^{-5}					
Cs	26	2.2×10^{-2}	9.8×10^{-2}	2.0×10^{-1}	1.7	4.9×10^{-2}	4.3×10^{-1}
I	21	5.3×10^{-1}	4.0×10^{-1}	3.2×10^{-1}	3.1	4.4×10^{-2}	1.2×100
Po	2	3.6×10^{-3}				2.9×10^{-3}	4.3×10^{-3}
S	12	8.3×10^{-2}	3.9×10^{-2}	7.3×10^{-2}	1.7	3.4×10^{-2}	1.3×10^{-1}
Sr	21	3.4×10^{-2}	3.2×10^{-2}	2.6×10^{-2}	2.1	9.3×10^{-3}	1.3×10^{-1}
Te	1	1.3×10^{-2}					
U	1	4.8×10^{-4}					
Zr	1	1.7×10^{-5}					

N Sample size, *AM* arithmetic mean, *ASD* arithmetic standard deviation, *GM* geometric mean, *GSD* geometric standard deviation,
Howard et al. (2016a, b)
*The mean is a geometric mean except where the number of data values (N) is less than 3, in which case it is a n arithmetic mean. Further statistical information is given for a wider range of radionuclides in TECDOC 1616 and TRS 472 (IAEA 2009, 2010)

5.4.2.1 Radioiodine

The deposition of atmospheric iodine (mainly from marine sources) onto the aerial parts of plants is an important contributor to stable iodine (^{127}I) in plants and is a major source for grazing animals. Iodine intake by agricultural animals is also enhanced by consumption of cattle feed fortified with iodine and the use of iodine-containing sterilants in the dairy industry.

Unlike many of the other radionuclides that affect the food chain, stable iodine is essential for normal growth and development in animals (including humans). It

accumulates in various organs and tissues of the body, notably the thyroid. The major function of the thyroid gland is to produce the thyroid hormones, T4 (thyroxine) and the more active T3 (triiodothyronine), so it accumulates iodine from the plasma to produce these compounds.

Raw milk is one of the foods that are most likely rapidly to become contaminated by radioiodine as livestock feeds on grass which has been contaminated by deposited radioiodine. Radioiodine isotopes intercepted by pasture vegetation ingested by grazing animals such as dairy cows, goats and sheep are quickly and completely absorbed through the gut (Howard et al. 1996a, b; Vandecasteele et al. 2000). The consumption of different physico-chemical forms of iodine does not change the extent of true absorption which is consistently complete (i.e. F_a is 1) (Howard et al. 1996a, b; Vandecasteele et al. 2000). Furthermore, there is no reduction in gut absorption of radioiodine isotopes due to enhanced stable iodine intake. Iodine is rapidly absorbed into the blood plasma where it circulates as an iodide and from which it is subsequently accumulated in the thyroid. Radioiodine is also transferred into the mammary gland and excreted via milk. It is also excreted via urine.

The capacity of the thyroid to concentrate iodine magnifies the hazard imposed by ^{131}I as it is accumulated in a similar manner to stable iodine. Therefore, it accumulates in the thyroid and also rapidly transfers into the milk within 30 min of introduction into the body (Thorell 1964). Peak radioiodine activity concentrations will be reached in 6–12 h. Radioactive iodine can also be absorbed via the lung into the plasma.

Goat's milk and sheep's milk contain approximately tenfold higher radioiodine activity concentration than cow's milk. For cows the milk/plasma ratio has been reported as 0.6–5.5, whereas for sheep and goats, it was 2–24 (Lengemann 1970).

In a controlled feeding experiment, using herbage recently contaminated by fallout from the Chernobyl NRE, the transfer coefficient of ^{131}I to sheep milk was 0.3 ± 0.017 d/L (Howard et al. 1993). These data are similar to F_m values reported for iodine for sheep milk in TRS 472 (IAEA 2010) of 0.23 d/L (geometric mean) and varied from 0.03 to 0.9 d/L. Similar values of F_m (range 0.015–0.020 d/L) after the Chernobyl NRE were reported for stable iodine in dairy cows by Vandecasteele et al. (2000). The daily proportion of ^{131}I intake which was secreted in sheep milk was 5.6 ± 0.035% which is an order of magnitude higher than for cattle and agrees with the higher transfer of stable iodine from plasma to milk which occurs in sheep and goats. The lactation phase does not seem to have a significant effect on iodine transfer to milk (Vandecasteele et al. 2000).

As for humans, it is important to establish the effect of stable iodine intake for dairy animals. In controlled experiments, Vandecasteele et al. (2000) reported that the mean F_m values for oral radioiodine to milk increased from 0.020 d/L for a low stable iodine intake to 0.024 d/L for a moderate stable iodine rate. There was a significant decrease in the transfer to milk for the high stable dietary iodine intake rate (mean F_m of 0.018 d/L) compared with the moderate treatment. The differences for the three stable iodine treatments were due to differential affinities and saturation levels of the thyroid and milk pathways competing for the available iodine.

Associated modelling studies confirmed that the stable iodine intake may affect the partitioning of iodine between thyroid, milk and excreta (Crout et al. 2000). The

model was used to predict the effects of variation in stable iodine intake and the extent of consequent chemical contamination of milk by stable iodine. The predicted time taken for radioiodine to reach peak concentrations in milk following a deposition event varied significantly (ca. 2 days) over a range of stable iodine intakes. Administration of low amounts of stable iodine of <100 mg/d to dairy animals could increase F_m, whereas >150 mg/d stable iodine would reduce radioiodine transfer to milk. However, administration of sufficient stable iodine to reduce the radioiodine transfer to milk would result in stable iodine concentrations in milk that were greatly in excess of internationally advised limits. Therefore, increased stable iodine supplementation should not be used as a countermeasure to reduce radioiodine transfer to milk due to the elevated stable iodine in milk (Howard et al. 1996a, b).

The $T_{1/2}^{b}$ of ^{131}I measured in ewes that were moved from contaminated pasture to housing and then fed an ^{131}I-free diet was 1 day, accounting for 97.4% of the reduction in the ^{131}I activity concentration in milk. Data on $T_{1/2}^{b}$ in cow, goat and sheep milk show consistently fast reduction at 1–2 days (Howard et al. 1993; Fesenko et al. 2015), and it is longer in various organs, e.g. thyroid, 100 days; bone, 14 days; and kidney, spleen and reproductive organs, 7 days.

Radioiodine in milk was an important contributor to internal dose in the emergency response phase and the initial part of the transition phase after the Chernobyl NRE. The ingested radioiodine was completely absorbed in the gut and rapidly transferred to the animals' thyroid and milk (within about 1 day). Throughout the contaminated areas of the USSR and parts of Eastern and Western Europe, peak ^{131}I activity concentrations in milk occurred rapidly after deposition in late April or early May 1986 depending on when the radioactive contamination reached each county. Therefore, transfer of ^{131}I to milk was the initial priority.

The ^{131}I activity concentration in milk after the Chernobyl NRE decreased with an $T_{1/2}^{eff}$ of 4–5 days due to its short physical half-life and the reduction in iodine activity concentrations on plants due to various removal processes from leaf surfaces. The removal rate, measured as a mean weathering half-life on grass, was about 9 days for radioiodine and 11 days for radiocaesium (Kirchner 1994).

5.4.2.2 Radiocaesium

Radiocaesium can be ingested or inhaled. The most important isotope with a physical half-life of 30 years is ^{137}Cs. Cs-134 has a shorter physical half-life of ~2 years, so its relative importance declines much faster than that of ^{137}Cs.

After the Chernobyl NRE, from June 1986, radiocaesium was the dominant radionuclide in most environmental samples and in food products contributing to the human food chain. The contamination of milk with radiocaesium decreased during spring 1986 with an $T_{1/2}^{eff}$ of about 2 weeks due to weathering, biomass growth and other natural processes. The amount and type of feed ingested by dairy cattle changes considerably during the course of lactation and with season leading to temporal variations in radiocaesium transfer to milk. Radiocaesium activity concentrations increased in many countries during winter 1986/1987 due to cows being fed with contaminated hay harvested in spring/summer 1986.

The physical and chemical form in which radiocaesium is ingested substantially affects the extent of absorption across the gut and the subsequent radiocaesium activity concentrations in animals and animal products. Radiocaesium absorption varies over a 50-fold range, depending upon dietary source (Beresford et al. 2000). Radiocaesium recently deposited after the Chernobyl NRE onto leaf surfaces was initially less available for gut absorption (F_a of 0.24) than that when it was plant-incorporated (Howard et al. 1989; Beresford et al. 2000). Once radiocaesium is incorporated into the internal plant structure through leaf absorption or root uptake, it is more highly absorbed in the GI tract (F_a of 0.8–1.0). The absorption of sediment- or soil-associated radiocaesium may be lower than that in plant-incorporated form and will vary for different types of soil (as does plant uptake) (Beresford et al. 2000). The availability for biological uptake of radionuclides associated with fuel particles that were deposited mostly within a 50 km radius of the Chernobyl NPP was lower than for plant-incorporated sources.

There were differing rates of ^{137}Cs transfer to milk in areas with different soil types. The transfer to milk declines in the order as follows: peat bog > sandy and sandy loam > chernozem and grey forest soils.

The $T^b_{1/2}$ of radiocaesium in milk is fast at 1–2 days (Fesenko et al. 2015) so the ^{137}Cs or ^{134}Cs activity concentrations in milk from dairy cows removed from contaminated areas declined rapidly. The long-term time trend of radiocaesium activity concentrations in milk (and meat) roughly follows that for vegetation (with a time lag) and can be divided into two time periods (Fesenko et al. 1997). For the first 4–6 years after deposition of Chernobyl NRE radiocaesium, there was an initial fast decrease with an ecological half-life between 0.8 and 1.2 years. Later, the rate of decline was slower and varied with soil type (Fesenko et al. 1997).

5.4.2.3 Radiostrontium

The behaviour of strontium in all organisms is strongly influenced by the presence of its analogue, calcium. The calcium requirement of an animal varies due to factors such as milk yield and stage of pregnancy (Howard et al. 1997). In response to these requirements, the calcium intake of dairy animals changes throughout the year. Typically, the calcium intake by dairy goats will range from 15 to 30 g/d, whilst that of cows will be 70–150 g/d (Beresford et al. 1998).

The gastrointestinal absorption of radiostrontium is less dependent upon dietary source than that of radiocaesium. Calcium status is generally the controlling influence on strontium absorption. The absorption of calcium is homeostatically controlled, and the extent of absorption is determined by animals' requirement for growth, milk production, etc. When calcium intake is in excess of requirement than for all sources, the F_a for Sr is 0.1–0.3. For a given calcium requirement, Ca absorption is inversely proportional to dietary Ca intake. Hence, Sr absorption should also be inversely proportional (Comar 1966). Collated data from experiments after the Kyshtym NRE, which included a number of data with relatively low ratios of calcium intake to requirement, and other data reported during the period of global weapons fallout, showed a clear reduction in the F_m of ^{90}Sr with an increasing ratio of intake/requirement for calcium (Beresford et al. 1998).

The use of the reported mean F_m for radiostrontium in Howard et al. (2016b, 2017) is only appropriate for productive agricultural systems where calcium is readily available (Comar 1966; Howard et al. 1997). F_m may be higher in low-productivity regions with low calcium intakes.

The $T^b_{1/2}$ of ^{90}Sr in milk is fast at 1–2 days (Fesenko et al. 2015), so the ^{90}Sr activity concentrations in milk from dairy cows that are removed from contaminated areas will decline rapidly.

5.4.3 Meat and Offal Production

Different radionuclides are accumulated in different tissues. The most important tissue for the food chain of many countries is muscle for which the data is much more extensive than that for other accumulating tissues.

5.4.3.1 Transfer of Radionuclides to Meat

Within a few weeks of the Chernobyl NRE, there were high reported ^{137}Cs and ^{134}Cs activity concentrations in the muscle of ruminants, resulting in intensive monitoring of meat from cattle, goats, sheep, reindeer, game and fish. Data on the transfer of radiocaesium to different animals has been reported from many countries after the Chernobyl NRE; there are much more data available for cow meat than for any other agricultural animal. The transfer of radiocaesium to meat is higher than that to milk. The extent of transfer of radionuclides into the meat of different types of animals is given as both F_f and CR values in TRS 472; selected relevant values for F_f are shown in Tables 5.14, 5.15, 5.16, 5.17 and 5.18 and for CR in Table 5.19.

Table 5.14 Transfer coefficients for radionuclides relevant for NREs to cow meat d/kg

Element	N	Reference value	Minimum	Maximum
Am	1	5.0×10^{-4}		
Co	4	4.3×10^{-4}	1.3×10^{-4}	8.4×10^{-4}
Cs	58	2.2×10^{-2}	4.7×10^{-3}	9.6×10^{-2}
I	5	6.7×10^{-3}	2.0×10^{-3}	3.8×10^{-2}
Nb	1	2.6×10^{-7}		
Pu	5	1.1×10^{-6}	8.8×10^{-8}	3.0×10^{-4}
Ru	3	3.3×10^{-3}	2.2×10^{-3}	6.4×10^{-3}
Sr	35	1.3×10^{-3}	2.0×10^{-4}	9.2×10^{-3}
Te	1	7.0×10^{-3}		
U	3	3.9×10^{-4}	2.5×10^{-4}	6.3×10^{-4}
Zr	1	1.2×10^{-6}		

IAEA (2010)

Table 5.15 Transfer coefficients for radionuclides relevant for NREs to sheep meat d/kg

Element	N	Reference value	Minimum	Maximum
Ag	1	4.8×10^{-4}		
Am	1	1.1×10^{-4}		
Ce	1	2.5×10^{-4}		
Co	2	1.2×10^{-2}	8.0×10^{-3}	1.6×10^{-2}
Cs	41	1.9×10^{-1}	5.3×10^{-2}	1.3
I	1	3.0×10^{-2}		
Pu	2	5.3×10^{-5}	2.0×10^{-5}	8.5×10^{-5}
Ru	2	2.1×10^{-3}	6.3×10^{-4}	3.6×10^{-3}
S	3	1.7	1.2	2.1
Sr	25	1.5×10^{-3}	3.0×10^{-4}	4.0×10^{-3}

IAEA (2010)

Table 5.16 Transfer coefficients for radionuclides relevant for NREs to goat meat d/kg

Element	N	Reference value	Minimum	Maximum
Cs	11	3.2×10^{-1}	1.2×10^{-1}	1.9
Nb	1	6.0×10^{-5}		
Sr	8	2.9×10^{-3}	2.0×10^{-3}	3.7×10^{-3}
Te	1	2.4×10^{-3}		
Zr	1	2.0×10^{-5}		

IAEA (2010)

Table 5.17 Transfer coefficients for radionuclides relevant for NREs to pig meat d/kg

Element	N	Reference value	Minimum	Maximum
Cs	22	2.0×10^{-1}	1.2×10^{-1}	4.0×10^{-1}
I	2	4.1×10^{-2}	1.5×10^{-2}	6.6×10^{-2}
Ru	1	3.0×10^{-3}		
Sr	12	2.5×10^{-3}	5.0×10^{-4}	8.0×10^{-3}
U	2	4.4×10^{-2}	2.6×10^{-2}	6.2×10^{-2}

IAEA (2010)

Table 5.18 Transfer coefficients for radionuclides relevant for NREs to poultry meat d/kg

Element	N	Reference value	Minimum	Maximum
Co	2	9.7×10^{-1}	3.0×10^{-2}	1.9
Cs	13	2.7	1.2	5.6
I	3	8.7×10^{-3}	4.0×10^{-3}	1.5×10^{-2}
Nb	1	3.0×10^{-4}		
Po	1	2.4		
Sr	7	2.0×10^{-2}	7.0×10^{-3}	4.1×10^{-2}
Te	1	6.0×10^{-1}		
U	2	7.5×10^{-1}	3.0×10^{-1}	1.2
Zr	1	6.0×10^{-5}		

IAEA (2010)

Table 5.19 Concentration ratios for radionuclides relevant for NREs to the meat of different animals

	Cattle				Sheep				Pork				Generic
Element	CR	Minimum	Maximum	N	CR	Minimum	Maximum	N	CR	Minimum	Maximum	N	
Ag					4.3×10^{-4}			1					4.3×10^{-4}
Am					1.1×10^{-4}			1					1.1×10^{-4}
Ce					2.2×10^{-4}			1					2.2×10^{-4}
Co	3.9×10^{-1}	7.2×10^{-3}	7.8×10^{-1}	2	2.3×10^{-1}								3.1×10^{-1}
Cs	2.3×10^{-1}	2.2×10^{-2}	7.3×10^{-1}	17	6.4×10^{-1}	5.3×10^{-2}	7.5	51	9.2×10^{-2}	8.3×10^{-3}	2.4×10^{-1}	4	3.9×10^{-1}[a]
I	9.5×10^{-2}	3.2×10^{-2}	1.9×10^{-1}	3				1	9.3×10^{-2}	3.5×10^{-2}	1.5×10^{-1}	2	9.4×10^{-2}
Nb	6.5×10^{-6}			1									6.5×10^{-6}
Po	1.4×10^{-1}	3.7×10^{-2}	4.1×10^{-1}	7									1.4×10^{-1}
Pu					3.9×10^{-5}	1.5×10^{-5}	6.3×10^{-5}	3					3.9×10^{-5}
Ru					5.7×10^{-4}			1					5.7×10^{-4}
S					5.0×10^{-1}								5.0×10^{-1}
Te	1.8×10^{-1}			1									1.8×10^{-1}
U	3.3×10^{-1}	3.0×10^{-3}	1.7	8									3.3×10^{-1}
Zn	1.7	4.7×10^{-1}	3.2	9	2.1	1.3	2.9	2					1.9

[a]Goat value of 6.2×10^{-1} for Cs (n = 4) is included in the generic value

IAEA (2010)

5.4.3.2 Other Accumulating Tissues

The transfer of radionuclides to eggs is high compared with meat. Transfer parameter values for eggs are listed in Table 5.20 and are largely based on data from chickens. There are F_f values reported by a number of sources for the three key elements, I, Cs and Sr, but few values for most other elements.

5.4.3.3 Target Tissues for Different Radionuclides

Some radionuclides accumulate in specific organs. The key accumulating organs in animals for radionuclides released during NREs is shown in Table 5.21. The table is largely based on a review of Russian language literature which reported F_f values

Table 5.20 Transfer coefficients for radionuclides relevant for NREs to egg contents d/kg

Element	N	GM	Minimum	Maximum
Am	1	3.0×10^{-3}		
Ce	1	3.1×10^{-3}		
Co	2	3.3×10^{-2}	2.6×10^{-2}	4.0×10^{-2}
Cs	11	4.0×10^{-1}	1.6×10^{-1}	7.1×10^{-1}
I	4	2.4	1.9	3.2
Nb	1	1.0×10^{-3}		
Po	1	3.1		
Pu	2	1.2×10^{-3}	9.9 × 10–6	2.3×10^{-3}
Ru	1	4.0×10^{-3}		
Sr	9	4.9×10^{-1}	2.5×10^{-1}	4.8
Te	1	5.1		
U	2	1.1	9.2×10^{-1}	1.2
Zr	1	2.0×10^{-4}		

IAEA (2010)

Table 5.21 Accumulating organs for different radionuclides

Radionuclide	Accumulating organs
Ag	Liver
Am	Bone and liver
Ce	Bone, kidney and liver
Co	Liver and kidney
Cs	All soft tissue except adipose tissue
I	Thyroid and milk
Pu	Bone and liver
Ru	Kidney and liver
Sb	Liver
Sr	Bone
Tc	Thyroid, liver and stomach wall

Fesenko et al. (2018)

for various organs consumed by humans (Fesenko et al. 2018). Radiocaesium is present at similar activity concentration in most soft tissue (and tends to be higher in the kidney, but not consistently) with lower accumulation in bone and adipose tissue. Many heavy metal radionuclides accumulate in the liver. No relevant transfer parameter data for ^{210}Po or ^{95}Zr have been identified.

5.5 Radionuclide Transfer in Non-intensive Animal Production

The Chernobyl fallout contaminated large areas of the terrestrial environment with a major impact on animal production on unimproved land. Depending on the weather patterns for the first 2 weeks after the NRE, parts of Eastern and Western Europe were contaminated, especially where the passage of the contaminated fall-out in the atmosphere coincided with heavy rainfall. These areas included upland areas and clearings within, or bordering woodland. They are collectively termed here as non-intensive systems (but also called seminatural, extensive systems or free-ranging systems). In these areas, unfertilized, highly organic soils are often used for extensive agricultural production of animal products, mainly for grazing by ruminants, such as sheep, goats, reindeer and cattle, on alpine meadows and upland regions. Therefore, problems with animal products were widely experienced not only within the USSR but also in many other countries in Europe.

The initial impact of radionuclide deposition on these systems, as for intensive systems, depended on the extent of interception by plants consumed by the animals. Thereafter, soil to plant to animal transfer dominated. These systems are potentially important after NREs due to the prevailing soil types and vegetation species which can allow relatively higher, and more prolonged, radiocaesium transfer to animals compared with intensively managed agricultural production (Howard et al. 1991, 1996a, b).

Normal agricultural practices which often reduced the transfer of radionuclides from soils to plant by physical dilution (e.g. ploughing) or by adding competitive elements (e.g. fertilizing) are generally not applied in these systems due to the low depth of soil and presence of stones and rocks. The low potassium status and high organic matter content of the soil in these often-unfertilized areas enhance the movement of radiocaesium from soil constituents into the soil solution from which it can be taken up by plants.

After the Chernobyl NRE, the high radiocaesium uptake from peaty soil in unmanaged (termed extensive) grassland was particularly important for a number of European countries where such land was used for the grazing of ruminants and the production of hay. Contamination with radiocaesium in animal food products from these radioecologically sensitive, non-intensive ecosystems often persisted for decades, even though the original deposition may not have been high (Howard et al. 2002). This is largely because there was prolonged and significant plant uptake of

radiocaesium from soil and some plant and other species consumed by animals accumulated high levels of radiocaesium, such as ericaceous species (e.g. heather) and mushrooms.

Animals kept on unimproved land had higher radiocaesium activity concentrations than those from agricultural systems after both the Chernobyl and Fukushima Daiichi NREs. Little information is available for other radionuclides and there is no current evidence of significant long-term problems with other radionuclides in these production systems.

5.5.1 Dairy Production in Low-Productivity Areas

In some countries, such as Austria and Norway, non-intensive systems are used during the growing season for dairy animals where suitable upland pastures exist and there are adequate facilities to carry out milking within a suitable distance. Some of these mountainous regions of Western European countries were amongst the most contaminated territories outside of the former USSR after the Chernobyl NRE. In these non-intensive systems, vertical migration rate of ^{137}Cs is slow, so it remains in the upper soil layer where root uptake of nutrients often occurs. A relatively high radiocaesium soil-to-vegetation transfer was reported in some of these pastures (e.g. Norway). Activity concentration of ^{137}Cs in milk in such areas rose quickly in the first 2 weeks after the dairy animals began to graze these regions (around mid-June) and remained elevated until the animals were removed in the autumn. Activity concentrations of ^{137}Cs in milk on such meadows during summertime were several orders of magnitude higher than in milk from lowland areas and valleys, where intensive agriculture occurs (IAEA 1994; Lettner et al. 2007). The ^{137}Cs activity concentration of milk would have remained above the intervention levels for many years if remediation options had not been applied. For example, ^{137}Cs activity concentrations in milk from Austrian sites remained high even 17 years after the Chernobyl NRE reflecting the persistent elevated transfer of radiocaesium from poorer soils in alpine pastures and regions with silicate bedrock.

Considerably longer ecological half-lives have been observed in cow's milk from alpine pastures than in cow's milk from lowland production sites. For the period 1988–2006, Lettner et al. (2007)) derived ecological half-lives of 0.7–1.4 years for the fast loss component and of 9.3–12.7 years for the slow loss component of ^{137}Cs activity concentrations in cow's milk. Later studies showed that the $T_{1/2}^{eff}$ and mean altitude of the alpine meadows sites were positively correlated, with higher altitude sites having significantly longer half-lives than those at lower altitudes. Depending on the site, half-lives varied from about 4 and 15 years (Lettner et al. 2009).

5.5.2 Meat Production in Low-Productivity Areas

After the Chernobyl NPP, the transfer of radiocaesium to meat of grazing stock in non-intensive areas was also higher than that in lowland regions in several countries, including Norway and the United Kingdom due to the same factors discussed for dairy animals. Free-ranging stock that graze these areas include sheep, cattle and goats; such land is also used for rearing game animals such as grouse, pheasant and partridge.

There was considerable variation in radiocaesium activity concentrations between individual animals within the same grazing areas. Reasons for the variation included individual preferences in the areas being grazed as there was considerable spatial variation on the deposition density of radiocaesium, even within a few metres, and the range of different vegetation species present.

Metabolic variation was also important. For example, there was considerable variability in the radiocaesium activity concentration of muscle between individual sheep in the same free ranging flock in contaminated upland areas of the United Kingdom (Beresford et al. 1996). Certain sheep within a flock were consistently amongst the most contaminated, whereas others were consistently the least contaminated (Beresford et al. 1995; Walters 1988). When ionic radiocaesium was orally administered to 22 sheep under controlled conditions, the F_f varied by threefold. The $T^b_{1/2}$ in muscle varied from 5 to 19 days with a mean of 9.8 days. Changes in live weight and feed intake during the study together accounted for 72% of the variation in the F_f values, and live weight change accounted for 56% of the observed variation in biological half-life. The data suggested that variation in metabolism of radiocaesium contributes to the variability in radiocaesium activity concentrations within sheep flocks in areas contaminated by Chernobyl fallout.

Contaminated animals raised for meat production cannot be sampled as easily as the milk from dairy animals. The development of equipment that was suitable for live monitoring of animals *in situ* in these areas was important in managing the situation and developing suitable remediation strategies.

5.6 Radionuclide Transfer to Game Animals

5.6.1 Forest Environments

The primary concern regarding forests from a radiological perspective is the long-term contamination of the forest environment and its products with ^{137}Cs due to its 30-year half-life. However, ^{134}Cs should not be forgotten as it may be present in large quantities and can significantly contribute to the contamination of animal products for more than a decade. The meat of game animals grazing in contaminated forests often has high radiocaesium activity concentrations.

Other radionuclides in forests such as the plutonium isotopes are of limited significance for animal products due to their low environmental mobility.

Substantial radioactive contamination of forests occurred following the Chernobyl and Fukushima Daiichi NREs. The deposition density of ^{137}Cs in Ukraine, Belarus and Russia exceeded >10 MBq/m^2 in some forested areas. In several Western European countries, such as Finland, Sweden, Norway, Germany and Austria, the deposition density of ^{137}Cs was also relatively high compared to other sources such as global fallout. After the Fukushima NRE, the extensive forest catchments in Fukushima prefecture covered about 70% of the most contaminated areas.

In many of the affected countries, the extent of game meat consumption from seminatural areas and forests by the general population was low compared with agricultural animal products. However, there were specific groups such as hunters who may consume relatively large quantities of game meat. Tree canopies, particularly at forest edges, are efficient filters of atmospheric pollutants of all kinds. The primary mechanism of tree contamination after the NREs was direct interception of radiocaesium of between 60 and 90% of the initial deposition by the tree canopy (Tikhomirov et al. 1994; Kato et al. 2012). Radionuclides on tree surfaces were gradually transferred to the upper layers of soil through natural weathering and wash-off by rainwater. Within a few years after deposition, most of the radiocaesium was transferred from the tree canopy to the underlying soil which became the major repository of radiocaesium contamination within the forest. The upper soil layers acted as a long-term sink and source of radiocaesium contamination of forest vegetation and animals.

A wide range of plants and fungi are consumed by wild animals in forests. Higher transfer of radiocaesium occurred from soil to some plants including grasses, lichens and berries, and also to mushrooms and truffles. Individual plant and fungal species differed greatly in their ability to accumulate radiocaesium, with particularly high radiocaesium activity concentrations in some mushroom species (IAEA 2010). The high levels of contamination in various mushroom species are reflected in generally high soil-mushroom Tag values which can vary by a factor of about 2000 (IAEA 2009, 2010).

Contamination of mushrooms in forests is often much higher than that of forest fruits such as bilberries. The Tag values for forest berries range from 0.02 to 0.2 m^2/kg (IAEA 2009, 2010).

The shooting of game animals or snaring of other species is often, but not always, confined to certain seasons, so the short-term impact of radionuclide deposition can initially be highly dependent on when the NRE occurs relative to the shooting season. After the transition phase, the spatial and temporal variability in contamination of game animals is affected by many different factors including:

- Highly heterogeneous deposition of radionuclides onto forests and associated terrain
- Spatial variation in soil type and therefore soil to plant transfer of radiocaesium
- Vertical migration of radiocaesium down the soil profile and out of the rooting zone
- Forest-specific differences in available edible food sources
- Seasonal variations in diet composition and feeding behaviour of game species

- Consumption of highly contaminated mushrooms and truffles
- The number of days with a heavy snow cover or ice
- $T_{1/2}^{b}$ which is longer on larger species such as moose/elk
- $T_{1/2}^{eff}$ which varies with time, species and forest characteristics

Significant variations occur in the body burden of radiocaesium in game animals due to the seasonal availability of the various components of their diet (IAEA 2009). Species-specific information on how the above factors affect some of the main species affected by radiocaesium deposition is provided in Table 5.22.

Table 5.22 General trends for radiocaesium in forest animals

Game animal	Diet	Seasonal trend in radiocaesium activity concentrations in meat
Roe deer, white-tailed deer	Winter–summer – wide variety of herbs and grasses, leaf buds and small twigs of trees and shrub Autumn – also mushrooms, lichen	Autumn peak associated with mushroom consumption
Red deer	Fibre-rich diet. Do not consume mushrooms	Not evident
Wild boar	Omnivorous diet that varies considerably with season Spring and summer – mostly herbivorous, plants Autumn – mushrooms Winter – often burrow into soil and feed on roots, tubers, larvae and earthworms and truffles with more radiocaesium than green plants; also consuming contaminated soil when burrowing. Consumption of beechnuts and acorns can reduce radiocaesium intake	The seasonal change in diet, combined with mushroom consumption during autumn and winter, can lead to an up to twofold increase in winter than in the spring and summer. However, diet intake can be highly variable in the different seasons, increasing with mushroom, truffle and soil consumption
Moose or elk	Herbivore – consumes many types of terrestrial vegetation, mainly consisting of forbs and other non-grasses, and fresh shoots from trees such as willow and birch. Therefore, soil type is a key variable	Higher in winter than in summer when moose often have access to pastures
Reindeer and caribou	Summer – a wide range of plants Autumn – consumption of mushrooms increases the radiocaesium intake Winter – consumption of lichens, which retain a high proportion of deposited radiocaesium and have a low K content. The change in diet is accompanied by a two- to threefold increase in the biological half-life of radiocaesium from about 7 to about 20 days	Highest in winter During summer and early autumn, only 10–20% (or less) of that in winter Higher in autumn than in the summer

Based on Skuterud et al. (2004), Strebl and Tataruch (2007), IAEA (2009, 2010)

Table 5.23 Comparison of Tag values for game animals obtained within 5 years after the Fukushima Daiichi and Chernobyl NREs

Species or group	Range of GM Tag values (m^2/kg fm)	
	Fukushima NRE (Tagami et al. 2016)	Chernobyl NRE (IAEA 2009)
Deer	$5.1 \times 10^{-3} – 7.2 \times 10^{-3}$	$7.6 \times 10^{-3} – 9.4 \times 10^{-2}$ $2.8 \times 10^{-2} – 5.0 \times 10^{-2}$
Wild boar	$2.6 \times 10^{-3} – 6.8 \times 10^{-3}$	$4.0 \times 10^{-3} – 6.7 \times 10^{-2}$
Bear	$2.8 \times 10^{-3} – 5.2 \times 10^{-3}$	$4.3 \times 10^{-2} – 7.1 \times 10^{-2}$
Pheasant	$1.6 \times 10^{-3} – 4.8 \times 10^{-3}$ $1.0 \times 10^{-4} – 8.9 \times 10^{-4}$	3.2×10^{-4}
Wild duck	$2.2 \times 10^{-4} – 8.7 \times 10^{-4}$	$2.4 \times 10^{-3} – 1.3 \times 10^{-2}$

Tagami et al. (2016), IAEA (2009)

Tag values have been reported in numerous publications, but it is difficult to identify generally applicable trends due to the wide variation in spatial and temporal trends. Tag values are often higher for wild boar than other species and the difference seems to increase with time. Also Tag values for the larger ruminants such as red deer and moose are often lower than for small deer and wild boar. Tag values compiled for the first 5 years after the Fukushima Daiichi accident, for three species, are compared with the equivalent period for Chernobyl NRE in Table 5.23.

Since the NREs, the natural decontamination of forest plants and, therefore, animals has been much slower than that in agricultural areas. Wild ruminants with access to agricultural land often have lower radiocaesium concentrations than those grazing inside forests (Kiefer et al. 1996).

The prevailing conditions in many forests, with often low potassium contents and high organic matter contents in the upper soil layers, and consequently high uptake of radiocaesium by some plants and mushrooms, lead to long $T_{1/2}^{eff}$ of radiocaesium in game animals. After the Chernobyl NRE, the $T_{1/2}^{eff}$ of ^{137}Cs in game meat varied from about 3 to 10 years. Over several decades, the physical decay rate of ^{137}Cs has been the key factor determining the rate of reduction in ^{137}Cs activity concentrations in some forest game animals.

5.7 Impacts on the Health of Livestock Exposed to Nuclear Contamination

A key feature of both the Kyshtym and Chernobyl NREs was the difference in the impact on the health of livestock between the emergency response phase, when there was an initial, intensive short-term radiation impact, and the subsequent transition phase, with a slow decline in the dose rate. Doses from radioactivity that may endanger the health and well-being of livestock are only likely to occur in the immediate vicinity of a major NRE involving a nuclear reactor.

To reliably estimate the impact of post-NRE doses to farm animals, information needs to be collected soon after the NRE for animals remaining in these areas. The limited data available for the period after NRE have been reviewed by Fesenko (2019) for the Kyshtym NRE and Geras'kin et al. (2008) and other sources given below who focused on the Chernobyl NRE.

The exposure routes for animals remaining in areas that have been highly contaminated include:

- External exposure from highly contaminated surfaces such as contaminated soil and surfaces of trees
- Internal exposure from consumption of highly contaminated plant material leading to direct irradiation of the digestive tract
- Internal exposure due to the absorption of radionuclides through the gut and accumulation into the tissues

There are considerable challenges associated with collecting relevant data for agricultural animals after a NRE. It is difficult to accurately estimate the doses received which vary greatly with location and with time. Some problems experienced after the Chernobyl NRE given by Geras'kin et al. (2008) include:

- Extreme small- and large-scale heterogeneity in the extent of radioactive contamination in affected areas due to the prolonged period of intensive radionuclide releases and variable meteorological conditions, combined with the wide spectrum of deposited radionuclides.
- High uncertainty in the estimation of doses received for observed biological effects. In the emergency response phase, radiation monitoring will inevitably be insufficient to allow a robust, reliable estimation of the consequent biological effects. Rapid changes of doses to agricultural animals occur due to the decay of short-lived radionuclides, radionuclide redistribution in the environment, changes in contribution of different radionuclides to different exposure pathways and the presence of highly contaminated particles.
- Difficulty in estimation of radiation effects due to the lack of verified methods for reconstruction of absorbed doses to living organisms in the complex emergency response phase.
- Changes in the sensitivity of animals to radiation doses during the different stages of growth, which can vary by orders of magnitude.

Dose Estimation After the Kyshtym NRE Information from the Kyshtym NRE is summarized here based on a recent review by Fesenko (2019). In contrast to the Chernobyl NRE, the Kyshtym NRE did not release short-lived radioiodine isotopes. Domesticated cattle and sheep were the most exposed agricultural animals after the Kyshtym NRE with initial radiation effects for domesticated animals being observed shortly after the NRE. The decision to evacuate both the public and animals living in the most affected areas was taken 12 days after the NRE. During that time the animals were grazing pasture with a total contamination density (combining all radionuclides released) of around 900-1000 MBq m^{-2} and received estimated external doses of 1.4–3.0 Gy. The corresponding doses to the GI tract were higher and

reached 4–24 Gy. The radiation doses resulted in a high mortality rate of exposed cattle with symptoms that could be attributed to acute radiation sickness, including bleeding of mucous membranes and leucopoenia.

The cattle grazing slightly further away from the most contaminated area received lower external doses of about 0.1 Gy and doses to the GI tract of 1.0–2.0 Gy. These animals survived although some detrimental changes occurred in the blood-producing metabolic systems that produce blood components over the first 6 months.

Similar effects were observed for highly contaminated sheep. Sheep grazing on sites close to the source of the release received external doses of 1.4–3 Gy and absorbed doses to the GI tract of 8–54 Gy during the first 12 days after the NRE and before evacuation. As for the cattle, the doses caused symptoms of acute radiation sickness and death in most of the animals.

No substantial radiation effects were observed in sheep at less contaminated sites (100–200 MBq m^{-2} of total radioactivity). For these sheep, the calculated doses during the first 12 days after the NRE were 0.1–0.2 Gy, and the GI tract doses were 2–4 Gy. Over the next few months, temporary changes in the blood-producing system of these animals occurred after evacuation.

An absorbed dose of around 1 Gy to the GI tract of large herbivores led to a reduction in wild game populations. Some reduction in the number of moose and roe deer occurred in 1957–1958 in areas where the GI tract doses would have been 10–30 Gy. However, increased mortality of large animals was not documented due to the difficulty in locating animals. At sites with a lower ^{90}Sr deposition density of 37 MBq m^{-2}, animals could have received an additional external dose of 2–3 Gy. At such doses, early radiation effects and even death of some animals may have occurred.

Dose Estimation After the Chernobyl NRE Appraisals of the effects of radiation on livestock inhabiting the area immediately surrounding the nuclear power plant at Chernobyl have been reported in the last decade (Fesenko et al. 2005; Geras'kin et al. 2008). Initially, there was an acute phase of radiation exposure of approximately 3–4 weeks that was due to the short-lived radionuclides, including ^{131}I deposited on vegetation and the ground surface. High exposure of the thyroids of vertebrates occurred due to inhalation and ingestion of radioiodine isotopes. Approximately 80% of the total radiation dose accumulated by animals were received within the first three months after the NRE, mostly due to ß-radiation. A second phase of exposure followed in the autumn of 1986 when the short-lived radionuclides had decayed, due to environmental pathways that transported various longer-lived radionuclides. The third stage of radiation exposure, continuing to the present day, is chronic exposure due mainly to ^{137}Cs.

A review of radiation doses and effects by Geras'kin et al. (2008) for the Chernobyl NRE has been used as the source of much of the information summarized here. The large-scale and heterogeneous radioactive contamination of the affected areas led to a variety of responses at different levels of molecular and cellular biological organization. The most affected livestock were within the 30 km

Chernobyl NPP zone when the highest exposures occurred during the first 10–20 days after the NRE. The major contributors to the absorbed dose in this period were short-lived radionuclides.

Radiation damage to agricultural animals was largely caused by the accumulation of various radioiodine isotopes in the thyroid. In the first 240 days after the NRE, the ratio of absorbed doses from all sources of exposure between the thyroid, GI tract mucosa and whole body was 230:1.2:1 (Alexakhin et al. 1992).

Doses received by farm animals depended on the deposition density of radionuclides at their locations and their residence time in the contaminated regions. Doses to the GI tract mucosa in a few cattle grazing in the 30 km zone reached 10 Gy over the first month after the NRE. The doses were about 7 Gy to tens of thousands of evacuated animals and about 1 Gy in the remaining livestock (Alexakhin et al. 2004). There was a 69% and 82% reduction in thyroid function in cattle associated with an estimated thyroid dose of 50 Gy and 280 Gy, respectively (Astasheva et al. 1991).

Animals that remained in the exclusion zone for several months had impaired immune responses, lowered body temperatures and cardiovascular disorders. Increased lethality was observed in evacuated cows 5–8 months after the NRE. Damage included partial atrophy or total destruction of the thyroid, liver degeneration, increased amount of visceral fat, gall bladder and spleen enlargement and myocardium dystrophy (Alexakhin et al. 2004).

Changes in the concentration of thyroid hormones and adenylyl cyclase activity in cattle in the first year after the NRE were reversible. This response indicated that there was a compensatory mechanism for the activation of cyclic AMP system in animals with reduced secretion of thyroid hormones in case of thyroid damage (Shevchenko et al. 1990). Concentrations of thyroid hormone were also low during lactation.

The offspring of exposed cows had reduced live weight, but reproductive capacity returned to normal by 1989 (Astasheva et al. 1991). There was no evidence of an increased occurrence of congenital malformations in offspring of cows that were evacuated from the 30 km zone.

The severity of radiation damage to the thyroid was linked with the stable iodine content in the animal's diet. In sheep from the Belarusian Poliessie, a reduced level of iodine nutrition (that commonly occurred in this area) led to the thyroid accumulating a relatively large proportion of the absorbed radioiodine and 2–2.5-fold higher doses to the thyroid than in controls (Budarkov et al. 1992).

Five months after the Chernobyl NRE, many sheep evacuated from the 30 km zone developed serious haematological alterations in the peripheral circulation (Alexakhin et al. 2004). Leucopenia was reported in 89% of animals and lymphopenia in 90%. Also 54% of sheep exhibited initial and marked anaemia and 34% had serious inhibition of haemopoiesis. Offspring of highly exposed cows had reduced weight, decreased daily live weight gains and disruptions to their hormonal status (Astasheva et al. 1991). Reproduction returned to normal in the spring of 1989. No valid data on an increased occurrence of teratogenesis in offspring of the evacuated from the 30 km zone animals was recorded.

Chronic radiation damage was still detected in sheep and horses that had been in a highly contaminated area nearly 2 years after they had been removed. They were generally in poor condition and emaciated and had decreased thyroid hormone levels.

5.8 Routes of Radionuclide Intake via Aquatic Pathways

Radionuclides released after a NRE enter the aquatic environment via a number of routes. When released into the atmosphere, radionuclides will be deposited onto catchments from which there will be an initial transfer through the catchment via runoff, especially if deposition is associated with rainfall, into streams and rivers which will ultimately be discharged into coastal and open ocean marine systems. After the initial period of radionuclide deposition during the emergency response phase, subsequent transfer from catchments occurs through processes such as runoff, erosion, decontamination activities and forestry practices. The rate of loss of radionuclides from catchments may also be enhanced during heavy rainfall events such as typhoons.

After the Chernobyl NRE, long-lived ^{137}Cs and ^{90}Sr formed the major component of contamination of aquatic ecosystems. Fractions of many radionuclides in sediments in aquatic environments may remain in mobile (or exchangeable) states and may transfer from the sediment compartment to the water column (Boyer et al. 2018). The fraction of a particular radionuclide present in these exchangeable phases will depend on numerous factors including, amongst others, the sediment or soil characteristics, the presence of competing ions, pH and redox conditions.

During the first few weeks after the NRE, activity concentrations in river waters rapidly decline, because of the physical decay of short-lived isotopes and as radionuclide deposits gradually became absorbed to soils and bottom sediments. In rivers, due to the constant throughflow of water, there is less contamination in the longer term, since contaminated upper layers of bottom sediments tend to be replaced, particularly in flood conditions.

The reduction in ^{90}Sr and ^{137}Cs activity concentrations occurred at a similar rate for different rivers in the vicinity of Chernobyl and in rivers in Western Europe (Monte 1995). In small catchments, highly organic soils such as saturated peat soils released up to an order of magnitude more radiocaesium to surface waters than occurred where there were mineral soils present (Smith et al. 2004).

In some lakes radiocaesium activity concentrations in water remained relatively high due to continuing inputs of runoff from organic soils in the catchment. In addition, internal cycling of radiocaesium in lakes with little inflow and outflow of water led to much higher activity concentrations in their water and aquatic biota than were typically seen in open lakes and rivers with higher amounts of water inflow and outflow. Radionuclide activity concentrations in water declined rapidly in reservoirs and lakes with significant inflow and outflow of water.

Radionuclides deposited onto lakes or reservoirs are also removed from the water by the sedimentation of particulate material, leading to the long-term removal

of radionuclides from the surface layers to bottom sediments. Radiocaesium activity concentrations in lakes decline relatively rapidly during the first months after fallout followed by slower declines over a period of years as radiocaesium became more strongly absorbed to soils and river bed sediments.

In lakes where the radiocaesium originated from organic soil catchments, the contamination was approximately an order of magnitude higher than in nearby lakes with mineral soil catchments (Hilton et al. 1993). Some lakes in Western Europe with organic catchments had radiocaesium activity concentrations in water and fish that were similar to those in some lakes in the more highly contaminated areas in Ukraine and Belarus. Long-term contamination can also be caused by remobilization of radionuclides from bed sediments. In shallow "closed" lakes where there were no significant surface inflow and outflow of water, the bed sediments played a major role in determining radionuclide activity concentration in the water.

5.8.1 Radionuclides in Freshwater Fish

The principal route of accumulation of radionuclides for aquatic animals is via food, but some radionuclides can be directly absorbed from the water. Radionuclide uptake from freshwater is influenced by the ambient chemistry.

Radionuclide activity concentrations in fish vary considerably in different species and depend on physiological features such as mass, dietary preferences and preferred habitat within the water column.

There are only limited data on uptake of ^{131}I in fish. After the Chernobyl NRE, ^{131}I was rapidly absorbed by fish reaching as high as 6000 Bq/kg fw soon after the contamination of water bodies but within approximately 1 month fell to only 50 Bq/kg fw (IAEA 2006a). This represents a rate of decline similar to that of its physical decay. The ^{131}I activity concentrations in fish became insignificant a few months after the NRE.

There have been many studies on radiocaesium contamination of freshwater fish. Because of its chemical similarity to caesium, the potassium concentration of lake or river water influences the rate of accumulation of radiocaesium in fish. Strong inverse relationships were reported between the potassium concentration in water and that of ^{137}Cs in fish (Smith et al. 2002). Bioaccumulation factors in lakes with low potassium concentrations could be one order of magnitude higher than that in lakes with high potassium concentration. Thus, fish from lakes in agricultural areas where runoff of potassium fertilizer is significant had lower bioaccumulation factors than fish from lakes in seminatural areas (Smith et al. 2002).

After the Chernobyl NRE, the accumulation of radiocaesium resulted in activity concentrations in some fish that were above intervention levels for consumption. The elevated levels persisted for many years in some areas in both the most affected regions of the USSR and parts of Western Europe (Jonsson et al. 1999).

There are relatively high transfer and retention of radiocaesium by some fish species, despite low radiocaesium activity concentrations in water. Uptake of radiocaesium in small fish was relatively rapid, with the maximum activity concentrations occurring a few weeks after a NRE (Jonsson et al. 1999; Zibold et al. 2002). Due to the slower uptake rates of radiocaesium in large predatory fish (e.g. pike, eel), maximum activity concentrations took up to a year after the NRE to be established.

In shallow closed lakes, ^{137}Cs activity concentrations in fish declined slowly in comparison with fish in rivers and open lake systems, due to the slow decline in radionuclide activity concentrations noted above. In the long term, ^{137}Cs activity concentrations in predatory fish were significantly higher than non-predatory fish, and large fish tended to have higher activity concentrations than small. The increase in activity concentration in large fish is termed the "size effect" and is due to metabolic and dietary differences. Radiocaesium activity concentration in large predatory fish could be five to ten times higher than in non-predatory fish.

After the Chernobyl NRE, there was a focus on collecting data for radiocaesium from some of the many lakes in Finland. The concentration of ^{137}Cs in pike tissues peaked after only 2 years. Over a 10-year study period, the $T_{1/2}^{eff}$of strontium was 15 years for pike and perch and 9 years for vendace (Saxen 2004). However, site-specific characteristics of the lakes led to considerable variation in $T_{1/2}^{eff}$ in individual lakes ranging from 7 to 29 years for pike, 11 to 30 years for perch and 7 to 11 years for vendace. Activity concentrations of ^{137}Cs in 20 different species of fish varied considerably even 15 years after initial contamination, ranging from 16 to 6400 Bq/kg (Saxén and Sundell 2006).

In a contaminated, closed lake in Russia, the ^{137}Cs activity concentration was two orders of magnitude higher than in fish in rivers or flow-through lakes in the same region (Travnikova et al. 2004).

Chernobyl fallout ^{90}Sr entered water courses via runoff and remained in the water phase rather than depositing in sediments as rapidly as ^{137}Cs (Outola et al. 2009). Nevertheless, ^{90}Sr activity concentrations in fish in Finland were much lower than those of ^{137}Cs. Stable strontium and ^{90}Sr behave in a similar chemical and biological manner to calcium in freshwater systems. The ^{90}Sr activity concentration in fish depended on the water chemistry with higher accumulation associated with (i) low calcium concentration in the water (i.e. "soft water") and (ii) low electrical conductivity. Radiostrontium accumulated in calcium-containing organs such as the skin, bones, fins and head of the fish (Kaglyan et al. 2008). Depending on the pattern of deposition of radioactive fallout, there were differences in the concentrations in fish from different lakes. In 15 lakes the average ^{90}Sr activity concentration in fish muscle was 20 and 60 times higher, respectively, in vendace (a non-predator species) and perch (mixed habit) than in pike (a predator). After the initial deposition from Chernobyl, it took 3 years for ^{90}Sr activity concentrations to reach a peak in pike. After this, concentrations decreased sharply to pre-Chernobyl levels. In contrast, in non-predatory vendace, ^{90}Sr activity concentrations were highest 1–2 years after contamination (Outola et al. 2009).

5.9 The Risk for Public Health (Placement on the Market for Human Consumption)

5.9.1 Radioiodine

After the onset of the NRE, the most immediate and important potential source of internal exposure to radioactivity is the short-lived radioiodine isotopes such as ^{131}I. Radioactive caesium (^{134}Cs and ^{137}Cs), in contrast to radioactive iodine, has a long half-life (^{134}Cs, 2 years; ^{137}Cs, 30 years).

The role of iodine in human health and the importance of iodine sources have been reviewed by Fuge and Johnson (2015); some of the main points from the review are briefly described here. Iodine is an essential element in the human diet, and a deficiency can lead to a number of health outcomes collectively termed iodine deficiency disorders (IDD). Human intake of iodine is mainly from food with some populations also obtaining appreciable quantities of iodine from drinking water. Plant-derived dietary iodine is generally insufficient alone. Seafood is an important source of iodine, but other inputs are mainly from sources such as the use of iodized salt and dairy produce.

Radioactive iodine (particularly ^{131}I) in food is of immediate concern due to its rapid transfer to milk from contaminated feed and its accumulation in the thyroid gland. I-131 has a relatively short half-life (8 days), so it will naturally decay over a short time frame. If radioactive iodine is breathed in or swallowed, it will concentrate in the thyroid gland and increase the risk of thyroid cancer.

The uptake of radioactive iodine into the thyroid gland can be decreased or prevented by ingestion of stable iodine in the form of potassium iodide pills. Once the thyroid is saturated with iodine, no further iodine can be incorporated. Iodized table salt should not be used as an alternative to potassium iodide pills as it does not contain sufficient iodine to saturate the thyroid. Furthermore, high salt intake may have adverse health effects.

After the Chernobyl NRE, the ^{131}I activity concentrations in milk were particularly high in privately owned dairy cows which were grazing forest clearings and unimproved land in contaminated areas. Initially, information regarding the need to stop the cows grazing such pasture, and to avoid consuming the milk, was less effective for subsistence households. Consequently, people in these households received relatively high radioiodine doses, leading to elevated rates of thyroid cancers in these areas (IAEA 2006a, b). The impact of ^{131}I consumption was enhanced by the deficiency of iodine in the diet of some of the more contaminated areas around the NPP.

5.9.2 Radiocaesium

In contrast to short-lived radioiodine isotopes, radiocaesium (^{134}Cs and ^{137}Cs) has a long half-life (^{134}Cs, 2 years; ^{137}Cs, 30 years).

Over time, radiocaesium can be accumulated in various terrestrial animals, or into rivers, lakes and the sea where fish and other seafood could take up the radionuclides. Animal products from the wild, such as game meat, may continue to be a radiological problem for a long time. Fish and aquatic microflora may bioconcentrate certain radionuclides, but due to the high dilution of radionuclides in water, contamination tends to be confined relatively locally.

Radiocaesium can stay in the environment for many years and could continue to present a long-term problem for food, and food production, and as a threat to human health. If radiocaesium enters the body, it is distributed uniformly throughout the body's soft tissues, resulting in exposure of those tissues. Compared to some other radionuclides, ^{137}Cs remains in the body for a relatively short time.

5.9.3 Other Radionuclides

Other radionuclides could be of concern, depending on the nature of the NRE and release of specific isotopes.

References

Akleyev, A. V., Krestinina, L. Y., Degteva, M. O., & Tolstykh, E. I. (2017). Consequences of the radiation accident at the Mayak production association in 1957 (the 'Kyshtym accident'). *Journal of Radiological Protection, 37*, R19–R42.

Alexakhin, R. M., & Korneyev, N. A. (Eds.). (1991). *Agricultural radiology*. Moscow: Ecology. (in Russian).

Alexakhin, R. M., Sarapultsev, I., Spirin, E., & Udalov, D. (1992). Formation of dose burdens to farm animals after the Chernobyl accident and influence of their evacuation on the absorbed doses. *Proceedings of the USSR Academy of Sciences, 323*, 576–579. (in Russian).

Alexakhin, R. M., Buldakov, L., Gubanov, V., Drozhko, Y., Ilyin, L., Kryshev, I., et al. (2004). *Large radiation accidents: Consequences and protective countermeasures*. Moscow: Izdat Publisher. (in Russian).

Arkhipov, N. P., Egorov, A. V., & Klechlovskey, V. M. (1969). Estimation of ^{90}Sr transfer from soil to plant and its accumulation in crop. *Dokladu VASHNIL, 1*, 2–4. (in Russian).

Astasheva, N., Lazarev, N., Khramtsova, L., Drozdenko, V., Chmirev, M., & Zigarenko, V. (1991). *Influence of radiation released during the Chernobyl NPP accident on clinical and physiological status of agricultural animals. Problems of agricultural radiology* (pp. 176–180). Kiev: UIAR. (in Russian).

Beresford, N. A., Barnett, C. L., Mayes, R. W., Lamb, C. S., Howard, B. J., & Wilson, P. (1995). *Radiocaesium variability within sheep flocks in the restricted area of Cumbria. Final report to the Ministry of Agriculture, fisheries and food* (TFS project TO7051jl). Grange-over-Sands: Institute of Terrestrial Ecology.

Beresford, N. A., Barnett, C. L., Crout, N. M. J., & Morris, C. C. (1996). Radiocaesium variability within sheep flocks: Relationships between the ^{137}Cs activity concentrations of individual ewes within a flock and between ewes and their progeny. *The Science of the Total Environment, 177*, 85–96.

Beresford, N. A., et al. (1998). Generic relationship between calcium intake and radiostrontium transfer to the milk of dairy ruminants. *Radiation and Environmental Biophysics, 37*, 129–131.

Beresford, N. A., Mayes, R. W. M., Cooke, A. I., Barnett, C. L., Howard, B. J., Lamb, C. S., & Naylor, G. P. L. (2000). The importance of source-dependent bioavailability in determining the transfer of ingested radionuclides to ruminant-derived food products. *Environmental Science & Technology, 34*, 4455–4462.

Boyer, P., Wells, C., & Howard, B. J. (2018). Extended K_d distributions for freshwater environment. *Journal of Environmental Radioactivity, 192*, 128–142.

Brown, J., & Simmonds, J. R. (1995). *Farmland: A dynamic model for the transfer of radionuclides through terrestrial foodchains (NRPB-R275)*. Chilton: National Radiological Protection Board. 76 pp.

Budarkov, V., Zenkin, A., Arkhipov, N., Junosova, R., Mayakov, E., Amirkhanian, A., et al. (1992). Effects of ^{131}I on sheep dependent on the stable iodine content in the ration, (in Russian). *Radiobiology, 32*, 451–458.

Comar, C. (1966). Transfer of strontium-90 into animal produce. In R. Russell (Ed.), *Radioactivity and human diet* (pp. 247–275). Oxford: Pergamon Press.

Crout, N. M. J., Beresford, N. A., Mayes, R. W., Maceachern, P. J., Barnett, C. L., Lamb, C. S., & Howard, B. J. (2000). A model of radioiodine transfer to goat milk incorporating the influence of stable iodine. *Radiation and Environmental Biophysics, 39*, 59–65.

Fesenko, S. (2019). Review of effects of non-human species in areas affected by the Kyshtym accident. *Journal of Radiological Protection, 39*, R1–R17.

Fesenko, S. V., Colgan, P. A., Lissianski, K. B., Vazquez, C., & Guardans, R. (1997). The dynamics of the transfer of ^{137}Cs to animal fodder in areas of Russia affected by the Chernobyl accident and doses resulting from the consumption of milk and milk products. *Radiation Protection Dosimetry, 69*(4), 289–299.

Fesenko, S. V., Alexakhin, R. M., Geras'kin, S. A., Sanzharova, N. I., Spirin, Y. V., Spiridonov, S. I., et al. (2005). Comparative radiation impact on biota and man in the area affected by the accident at the Chernobyl nuclear power plant. *Journal of Environmental Radioactivity, 80*, 1–25.

Fesenko, S., Isamov, N., Barnett, C. L., Beresford, N. A., Howard, B. J., Sanzharova, N., & Fesenko, E. (2015). Review of Russian language studies on radionuclide behaviour in agricultural animals: Biological half-lives. *Journal of Environmental Radioactivity, 142*, 136–151.

Fesenko, S., Isamova, N., Howard, B. J., Sanzharova, N., & Wells, C. (2018). Review of Russian language studies on radionuclide behaviour in agricultural animals: Transfer to animal tissues. *Journal of Environmental Radioactivity, 192*, 233–249.

Fuge, R., & Johnson, C. C. (2015). Iodine and human health, the role of environmental geochemistry and diet, a review. *Applied Geochemistry, 63*, 282–302.

Garland, J. A., & Wakeford, R. (2007). Atmospheric emissions from the Windscale accident of October 1957. *Atmospheric Environment, 41*, 3904–3920.

Geras'kin, S. A., Fesenko, S. V., & Alexakhin, R. M. (2008). Effects of non-human species irradiation after the Chernobyl NPP accident. *Environment International, 34*, 880–897.

Gulyakin, I. V., & Yudintseva, E. B. (1962). *Radioactive fission products in soils and plants*. M. Gosatomizdat (in Russian).

Heinemann, K., & Vogt, K. J. (1980). Measurements of deposition of iodine onto vegetation and of the biological half-life of iodine on vegetation. *Health Physics, 39*, 463–474.

Hilton, J., Livens, F. R., Spezzano, P., & Leonard, D. R. P. (1993). Retention of radioactive Cs by different soils in the catchment of a small lake. *Science of the Total Environment, 129*, 253–266.

Hird, A. B., Rimmer, D. L., & Livens, F. R. (1995). Total caesium fixing potential of acid organic soils. *Journal of Environmental Radioactivity, 26*, 103–118.

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

Howard, B. J., Mayes, R. W., Beresford, N. A., & Lamb, C. S. (1989). Transfer of radiocesium from different environmental sources to ewes and suckling lambs. *Health Physics, 57*, 579–586.

Howard, B. J., Beresford, N. A., & Hove, K. (1991). Transfer of radiocesium to ruminants in unimproved natural and semi-natural ecosystems and appropriate countermeasures. *Health Physics, 61*, 715–725.

Howard, B. J., Beresford, N. A., Mayes, R. W., & C.S. (1993). Lamb transfer of ^{131}I to sheep milk from vegetation contaminated by Chernobyl fallout. *Journal of Environmental Radioactivity, 19*, 155–161.

Howard, B. J., Johanson, K. J., Linsley, G. S., Hove, K., Prohl, G., & Horyna, J. (1996a). Transfer of radionuclides by terrestrial food products from semi-natural ecosystems. In *Second report of the VAMP Terrestrial Working Group* (IAEA-Tecdoc-857) (pp. 49–79). Vienna: IAEA.

Howard, B. J., Voigt, G., Segal, M. G., & Ward, G. M. (1996b). A review of countermeasures to reduce radioiodine in milk of dairy animals. *Health Physics, 71*, 661–673.

Howard, B. J., Beresford, N., Mayes, R. W., Hansen, H. S., Crout, N. M. J., & Hove, K. (1997). The use of dietary calcium intake of dairy ruminants to predict the transfer coefficient of radiostrontium to milk. *Radiation and Environmental Biophysics, 36*, 39–43.

Howard, B. J., Strand, P., Assimakopoulos, P., Brechignac, F., Gasco, C., Metivier, H., Moberg, L., Smith, J. T., Tamponnet, C., Trueba, C., Voigt, G., & Wright, S. M. (2002). Estimation of radioecological sensitivity. *Radioprotection – Collogues, 37*, C1167–C1173.

Howard, B. J., Beresford, N. A., Barnett, C. L., & Fesenko, S. (2009a). Radionuclide transfer to animals: Revised recommended transfer coefficient values. *Journal of Environmental Radioactivity, 100*, 263–273.

Howard, B. J., Beresford, N. A., Barnett, C. L., & Fesenko, S. (2009b). Quantifying the transfer of radionuclides to food products from domestic farm animals. *Journal of Environmental Radioactivity, 100*, 767–773.

Howard, B. J., Wells, C., Barnett, C. L., & Sheppard, S. C. (2016a). How knowledge of the gastrointestinal absorption of elements could be used to predict transfer to milk. *Scientific Reports, 6*, 37041. https://doi.org/10.1038/srep37041.

Howard, B. J., Wells, C., & Barnett, C. L. (2016b). Improving the quantity, quality and transparency of data used to derive radionuclide transfer parameters for animal products. 1. Goat milk. *Journal of Environmental Radioactivity, 154*, 34–42.

Howard, B. J., Wells, C., Barnett, C. L., & Howard, D. C. (2017). Improving the quantity, quality and transparency of data used to derive radionuclide transfer parameters for animal products. 2. Cow milk. *Journal of Environmental Radioactivity, 167*, 254–268.

International Atomic Energy Agency. (1994). *Handbook of parameter values for the prediction of radionuclide transfer in temperate environments* (Technical report series N°364). Vienna: IAEA.

International Atomic Energy Agency. (2006a). *Environmental consequences of the Chernobyl accident and their remediation: Twenty years of experience* (Report of the UN Chernobyl Forum Expert Group "Environment". STI/PUB/1239). Vienna: IAEA.

International Atomic Energy Agency. (2006b). *Chernobyl's legacy: Health, environmental and socio-economic impacts and recommendations to the Governments of Belarus, the Russian Federation and Ukraine* (The Chernobyl Forum: 2003–2005 second revised version). Vienna: IAEA.

International Atomic Energy Agency. (2009). *Quantification of radionuclide transfer in terrestrial and freshwater environments for radiological assessments* (TECDOC 1616). Vienna: IAEA.

International Atomic Energy Agency. (2010). *Handbook of parameter values for the prediction of radionuclide transfer in terrestrial and freshwater environment* (Technical reports series no. 472). Vienna: IAEA.

International Atomic Energy Agency. (2015). *Fukushima Report. Radiological Consequences* (Technical Volume 4/5). Vienna: IAEA.

International Commission on Radiological Protection (ICRP). (2006). *Human alimentary tract model for radiological protection*. Annals of the ICRP, Publication 100.

Jonsson, B., Forseth, T., & Ugedal, O. (1999). Chernobyl radioactivity persists in fish. *Nature, 400*, 417.

Kaglyan, O., Gudkov, D., Klenus, V., Shyroka, Z., & Nazarov, O. (2008). Strontium-90 in fish from the lakes of the Chernobyl exclusion zone. In P. Strand, J. Brown, & T. Jølle (Eds.), *Proceedings: Posters – Part 2. International Conference on Radioecology & Environmental Radioactivity 2008 Jun 15–20, Bergen, Norway* (pp. 176–179). Østerås: Norwegian Radiation Protection Authority.

Kato, H., Onda, Y., & Gomi, T. (2012). Interception of the Fukushima reactor accident-derived ^{137}Cs, ^{134}Cs and ^{131}I by coniferous forest canopies. *Geophysical Research Letters, 39*(20), 1.

Kiefer, P., Pröhl, G., Müller, G., Lindner, G., Drissner, J., & Zibold, G. (1996). Factors affecting the transfer of radiocaesium from soil to roe deer in forest ecosystems of southern Germany. *Science of the Total Environment, 192*, 49–61.

Kinnersley, R. P., Goddard, A. J. M., Minski, M. J., & Shaw, G. (1997). Interception of Caesium-contaminated rain by vegetation. *Atmospheric Environment, 31*, 1137–1145.

Kirchner, G. (1994). Transport of iodine and caesium via the grass-cow-milk pathway after the Chernobyl accident. *Health Physics, 66*, 653–665.

Lengemann, F. W. (1970). Metabolism of radioiodide by lactating goats given iodine^{-1} 31 for extended periods. *Journal of Dairy Science, 53*, 165–170.

Lettner, H., Hubmer, A., Bossew, P., Strebl, F., & Steinhäusler, F. (2007). Effective and ecological half-lives of ^{137}Cs in cow's milk in alpine agriculture. *Radiation and Environmental Biophysics, 48*, 47–56.

Lettner, H., Hubmer, A., Bossew, P., & Strebl, F. (2009). ^{137}Cs and ^{90}Sr transfer to milk in Austrian alpine agriculture. *Journal of Environmental Radioactivity, 98*, 69–84.

Menzel, R. G. (1963). Factors influencing the biological availability of radionuclides for plants. *Federation Proceedings. Federation of American Societies for Experimental Biology, 22*, 1398–1401.

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

Outola, I., Saxén, R. L., & Heinävaara, S. (2009). Transfer of ^{90}Sr into fish in Finnish lakes. *Journal of Environmental Radioactivity, 100*, 657–664.

Saxen, R. (2004). Long-term behaviour of ^{137}Cs in Finnish lakes. In *Book of extended synopses, IAEA-CN-118* (pp. 54–55). Vienna: IAEA.

Saxén, R., & Sundell, J. (2006). ^{137}Cs in freshwater fish in Finland since 1986-A statistical analysis with multivariate linear regression models. *Journal of Environmental Radioactivity, 87*, 62–76.

Shevchenko, A. S., Vakulenko, A. D., & Isamov, N. N. (1990). Increase in activation by prostaglandin E1 of adenilatcyclase in blood cells of animals found in the Chernobyl affected region. *Reports of All-Union Academy of Agricultural Sciences, 11*, 55–58. (in Russian).

Skuterud, L., Pedersen, Ø., Staaland, H., Røed, K. H., Salbu, B., Liken, A., & Hove, K. (2004). Absorption, retention and tissue distribution of radiocaesium in reindeer: Effects of diet and radiocaesium source. *Radiation and Environmental Biophysics, 43*, 293–301.

Smith, J. T., & Beresford, N. A. (2005). *Chernobyl catastrophe and consequences*. Berlin: Springer.

Smith, J. T., Kudelsky, A. V., Ryabov, I. N., Daire, S. E., Boyer, L., Blust, R. J., Fernandez, J. A., Hadderingh, R. H., & Voitsekhovitch, O. V. (2002). Uptake and elimination of radiocaesium in fish and the "size effect". *Journal of Environmental Radioactivity, 62*, 145–164.

Smith, J. T., Wright, S. M., Cross, M. A., Monte, I., Kudelsky, A. V., Saxén, R., Vakulovsky, S. M., & Timms, D. N. (2004). Global analysis of the riverine transport of ^{90}Sr and ^{137}Cs. *Environmental Science and Technology, 38*, 850–857.

Strebl, F., & Tataruch, F. (2007). Time trends (1986–2003) of radiocaesium transfer to roe deer and wild boar in two Austrian forest regions. *Journal of Environmental Radioactivity, 98*, 137–152.

Tagami, K., Howard, B. J., & Uchida, S. (2016). The time-dependent transfer factor of radiocesium from soil to game animals in Japan after the Fukushima Daiichi nuclear accident. *Environmental Science & Technology, 50*, 9424–9431.

Thorell, C. B. (1964). Secretion of radioiodine in milk following a single oral administration in the cow. *Acta Veterinaria Scandinavia, 5*, 217–233.

Tikhomirov, F. A., Shcheglov, A. I., & Sidorov, V. P. (1994). Forests and forestry: Radiation protection measures with special reference to the Chernobyl accident zone. *Science of the Total Environment, 137*(1–3), 289.

Travnikova, I. G., Bazjukin, A. N., Bruk, G. J., Shutov, V. N., Balonov, M. I., Skuterud, L., Mehli, H., & Strand, P. (2004). Lake fish as the main contributor of internal dose to lakeshore residents in the Chernobyl contaminated area. *Journal of Environmental Radioactivity, 77*, 63–75.

United Nations Scientific Committee on the Effects of Atomic Radiation. (2011). *UNSCEAR 2008 report to the General Assembly with scientific annexes*. New York: United Nations.

Vandecasteele, C. M., Van Hees, M., Hardeman, F., Voigt, G. M., & Howard, B. J. (2000). The true absorption of I-131, and its transfer to milk in cows given different stable iodine diets. *Journal of Environmental Radioactivity, 47*, 301–317.

Walters, B. (1988). Chernobyl derived activity in sheep: Variation within a single flock and with time. *Journal of Environmental Radioactivity, 7*(2), 99–106.

Whicker, F. W., & Schultz, V. (1982). *Radioecology: Nuclear energy and the environment* (Vol. 1). Boca Raton: CRC Press, Inc.

Zibold, G., Kaminski, S., Klemt, E., & Smith, J. T. (2002). Time-dependency of the ^{137}Cs activity concentration in freshwater lakes, measurement and prediction. *Radioprotection – Colloques, 37*, 75–80.